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REMEDIAL INVESTIGATION REPORT FOR THE 3163 BUFFALO AVENUE SITE NIAGARA FALLS, NEW YORK

Volume I

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Prepared for:

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EXECUTIVE SUMMARY

This Remedial Investigation (RI) has been completed at 3163 Buffalo Avenue in the City of Niagara Falls, New York. The RI was performed at the request of the New York State Department of Environmental Conservation (NYSDEC) under the Stipulation and Order, Civil No. 83-1301 C. The RI was designed to assess the extent of chemicals in unsaturated soils and in groundwater on the site, and to extrapolate levels of chemicals in groundwater off site. A risk assessment has also been performed to determine if potential risks exist to receptors in the vicinity of the site.

Major contaminants of concern at the site are benzene and its derivatives including chlorobenzene, 1,2-chlorobenzene, 1,4-dichlorobenzene, and 1,2,4-trichlorobenzene. Arsenic, barium, cadmium, chromium, lead, manganese, zinc, and cyanide were also found at elevated concentrations. Relative to the benzenes, lesser concentrations of chlorinated organics were found on site. These compounds, which include trichloroethene, tetrachloroethene, total-1,2-dichloroethene and vinyl chloride, were most prevalent in the bedrock aquifer as compared to the saturated and unsaturated overburden.

Soil sampling from the RI has shown that two areas of high soil contamination (greater than 100,000 microgram/kilogram [µg/kg]) exist at ppb the site. These areas are primarily in the southeast and southwest site quadrants. Benzene, chlorobenzene, and the various derivatives of dichlorobenzene and trichlorobenzene were the most significant chemicals found in these two areas. Eleveated concentrations of metals, such as arsenic, barium, cadmium, chromium, manganese, zinc, and cyanide, were also found in these areas.

Sixteen monitoring wells were installed in the saturated A-Zone and in the B- through F-Zones of the Lockport Dolomite primarily to characterize groundwater quality on the site. These wells substantiated that the southwest area of the site was a source area for contaminants. This is best evidenced by the fact that very high levels (greater than 100,000 µg/L) of benzene and benzene-related compounds were found in monitoring wells in the southwest area. Dense Non-Aqueous Phase Liquids (NAPLs) were expected to exist at two well locations--MW-1B and MW-2B. Phase separation was observed in the sample from monitoring well 2B. In addition, this sample yielded a concentration of chlorobenzene of 190,000,000 µg/L which exceeds this compound's solubility by 236 times. Groundwater from monitoring wells immediately downgradient of the southeast area yielded concentrations one to two orders of magnitude lower than groundwater from monitoring wells in the southwest. Therefore, while the southeast area must contribute contaminants to the groundwater, its impact is not as significant as the southwest area.

Groundwater in the A- and B-Zones flows across the site toward the north, northwest and west. The CD-Zone also flows to the northwest, and the C- and F-Zones flow toward the northeast. It should be noted, however, that groundwater flow directions in C-, CD-, and F-Zones were determined using only two or three wells per zone and thus should be regarded as approximate. Contaminants in the groundwater (chiefly benzenes and related benzene compounds) flow in the direction of groundwater transport, which is principally toward north, northwest, and west. Moderately high levels (greater than 1,000 µg/L) were detected in monitoring well nest MW-5 and monitoring well nest MW-4, which are located at the north and northwest site boundaries, respectively.

The ultimate discharge source for groundwater is the Niagara River within the Niagara Gorge. Groundwater may flow to the river by movement within the bedrock aquifer or may also be intercepted by the Falls Street Tunnel, which then discharges to the Niagara River to the immediate southwest of the Rainbow Bridge.

For the risk assessment, all site groundwater contaminants were assumed to be intercepted by the Falls Street Tunnel, which discharges to the Niagara River near the Rainbow Bridge. In actuality, the flow is partially diverted to the Niagara Falls Wastewater Treatment Plant during low flow periods. However, a conservative approach has been adopted, which has assumed the entire flow discharges to the Niagara River. Three potential exposure pathways were evaluated.

- Inhalation of site-derived contaminants by workers in the gorge near the Rainbow Bridge;
- Exposure of recreational fishermen using the Niagara River near the Power Authority generating station; and
- Use of Niagara River water at the mouth of the river for residential drinking water purposes.

The estimated cancer risks due to the mobile, site-derived contaminants were about 30 times less than 1 x 10^{-6}, the minimum risk level considered to be of concern by the EPA and many state regulatory agencies. The total estimated risk due to all of the site groundwater contaminants just reached the 1 x 10^{-6} level. However, because of the conservative approach used in estimating contaminant loadings to the Niagara River, in particular, and in the risk assessment process as a whole, this estimate probably substantially overestimates the actual risks posed by site groundwater contaminants via the exposure pathways evaluated. The estimated noncarcinogenic risks for all of the site groundwater contaminants were about 3,000 times less than the threshold level above which the potential for adverse noncarcinogenic effects begins to be of concern. To summarize, it is unlikely that all of the site groundwater contaminants taken together would pose a significant threat to human health by the exposure pathways evaluated, and even less likely that the mobile, site-derived contaminants would pose such a risk.

1. INTRODUCTION

An RI was performed at 3163 Buffalo Avenue, which is located in the City of Niagara Falls, New York (see Figure 1-1). The RI is being completed at the request of NYSDEC under a Stipulation and Order, Civil No. 83-1401 C. Ecology and Environment, Inc. (E & E) was retained by a group of Participating Parties (PPs), including ICC Industries, Inc., the Solvent Chemical Company, Mader Capital Corporation, E.I. Dupont deNemours and Company, Occidental Chemical Corporation, and the United States of America.

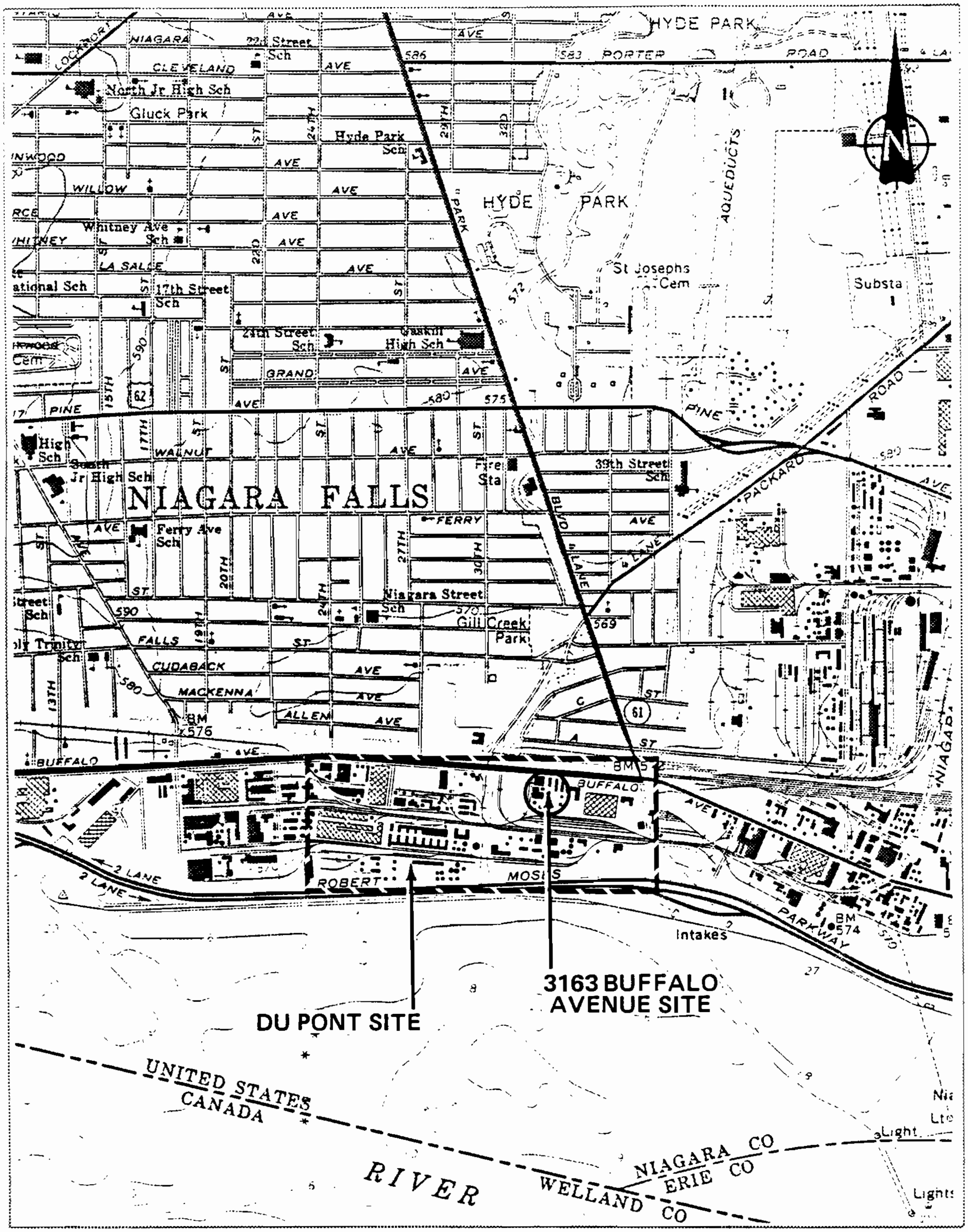
The purpose of the RI at the 3163 Buffalo Avenue site was to:

- Assess the extent of contamination in subsurface soils and groundwater that exist on site;
- Characterize the site hydrogeology and extrapolate contaminant movement in groundwater off site; and
- Perform a Risk Assessment.

Fieldwork for the RI was performed as per the specifications in the document, Final Work Plan for Remedial Investigation of the 3163 Buffalo Avenue Site, December 1989, negotiated and agreed to by PPs and NYSDEC. The Work Plan and the RI were developed using the guidelines of the Interim Final October 1988 United States Environmental Protection Agency (EPA) document, Guidance for Conducting Remedial Investigations and Feasibility Studies Under CERCLA.

A site map appears in Figure 1-2. To date, three investigations have been performed at the site. These include:

- A 1980 investigation by Wehran Engineering in association with Recra Research;



SOURCE: USGS 7.5 Minute Series (Topographic) Quadrangle: Niagara Falls, NY-ONT, 1980.

Scale: in feet

0 1000 2000 3000 4000 5000 6000 7000

Figure 1—1 REGIONAL LOCATION OF 3163 BUFFALO AVENUE SITE

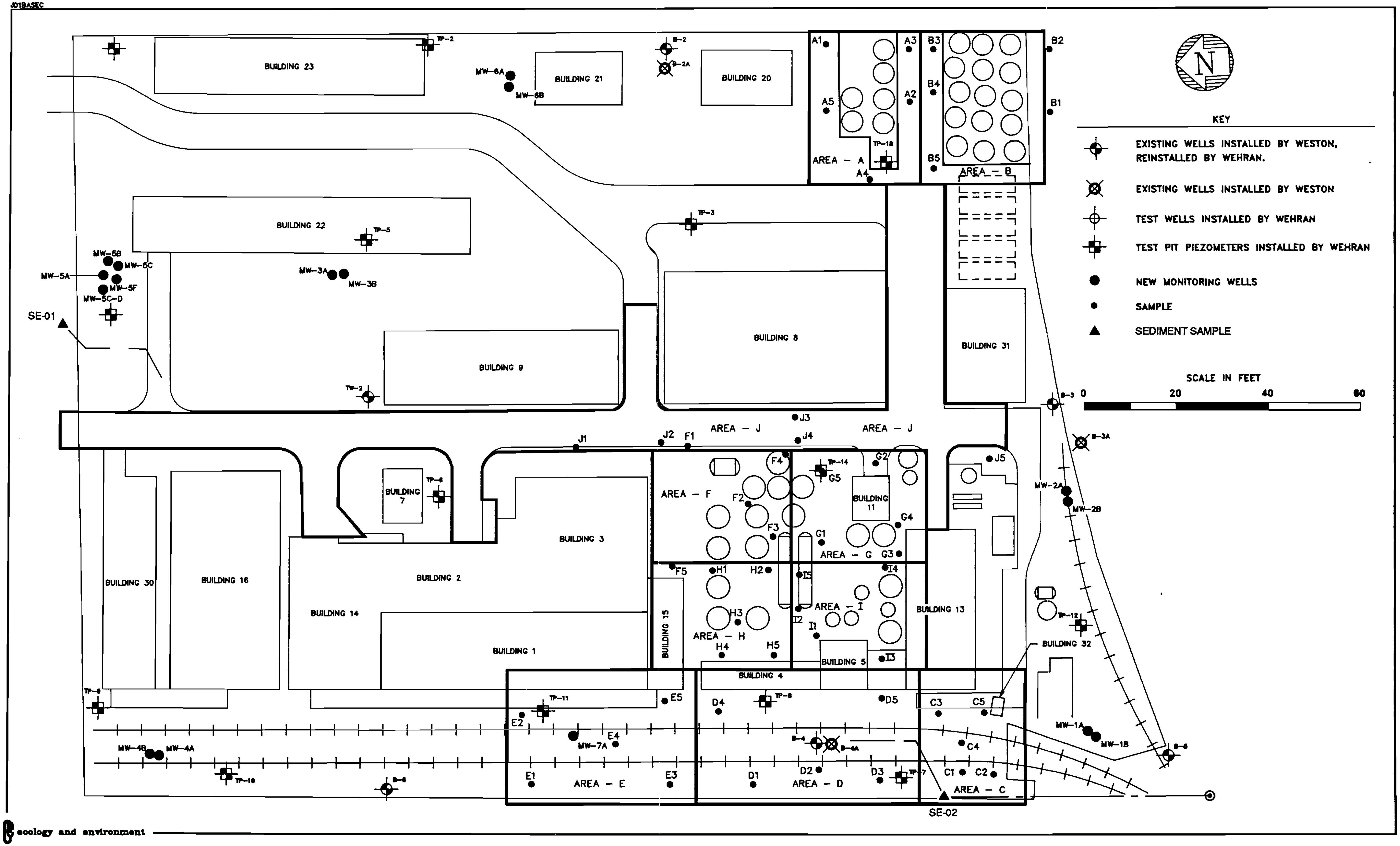


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Figure 1-2
BASE MAP SOIL SAMPLE LOCATIONS
3163 BUFFALO AVENUE

- A 1983 Phase I Investigation for NYSDEC by Engineering Science Inc. (ES) in association with Dames and Moore; and
- A 1985 Phase II Investigation for NYSDEC performed again by ES and Dames and Moore.

Other hydrogeological investigations have also been performed in the vicinity of the 3163 Buffalo Avenue site; most notably is work performed by Woodward Clyde Consultants at the Dupont Plant, which is adjacent to 3163 Buffalo Avenue.

The RI focuses on the integration of site data, data from other local sites, and regional information to develop a coherent interpretation of the extent of contamination and the important environmental media that may transport contaminants off site. Section 2 discusses physical characteristics of the study area. Section 3 recounts the methods that were employed during the field investigation. Section 4 discusses the types and extent of contaminants found on site and the various mechanisms that may transport these contaminants off site. Section 5 presents the potential risk to human health and the environment. Section 6 highlights the major conclusions from the investigation.

2. PHYSICAL CHARACTERISTICS OF THE STUDY AREA

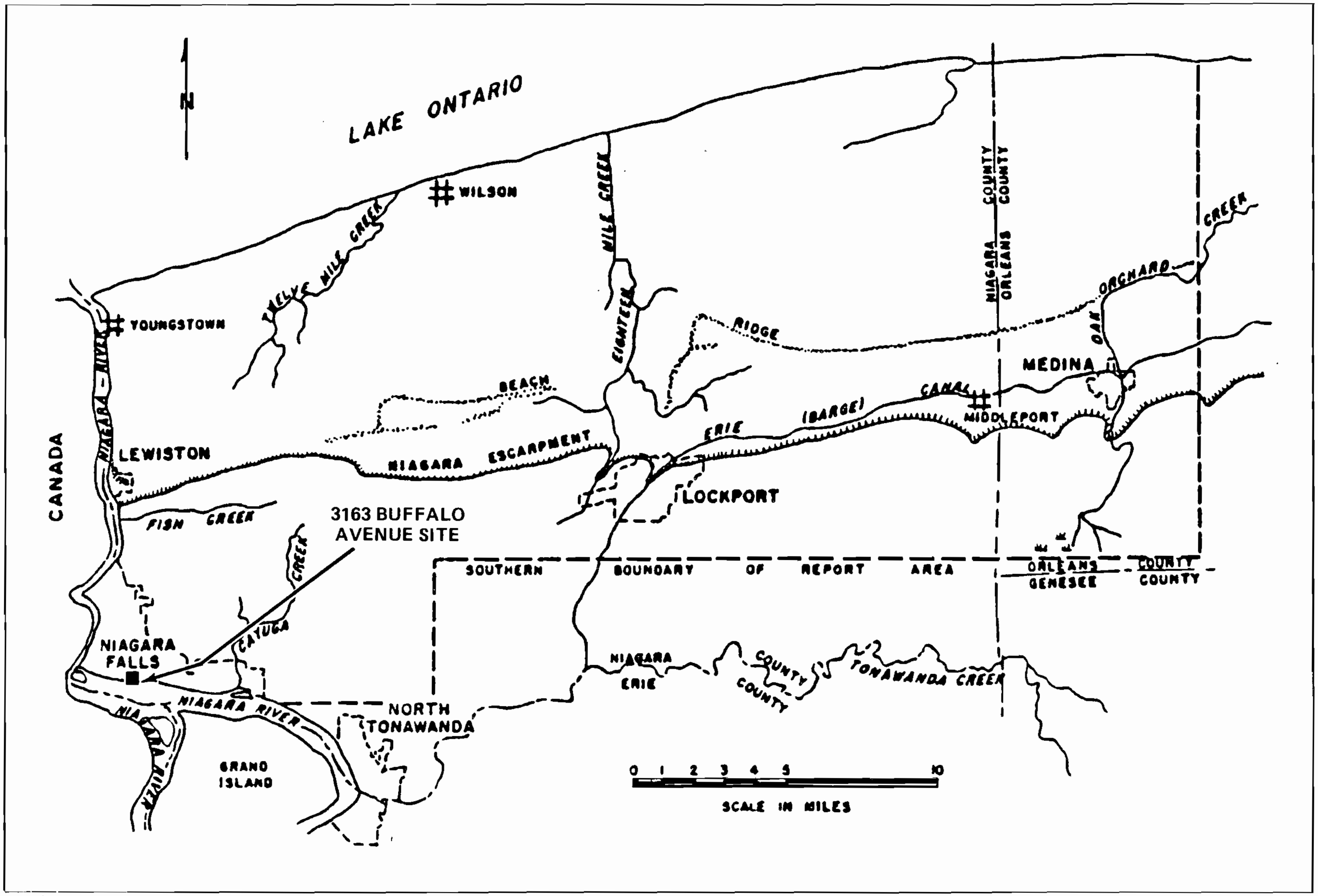
2.1 INTRODUCTION

This section summarizes the regional physical characteristics of the Niagara Falls area in order to provide a background for the field investigation. Site-specific details of the geology, hydrogeology and chemicals in environmental media are presented in Section 4, Nature and Extent of Contamination.

2.2 SURFACE TOPOGRAPHY AND DRAINAGE

The 3163 Buffalo Avenue site is located along Buffalo Avenue in the City of Niagara Falls, New York (see Figure 1-1). The coordinates of this site are: 43°04′51.5" North Latitude, 79°01′44" West Longitude. The principal topographic features of this region are the gorge of the Niagara River and the east-west trending Niagara escarpment (see Figure 2-1). At the river, the escarpment is a 200-foot high cliff that gradually diminishes to the east into a broad, gentle incline. The plain to the north of the escarpment slopes almost imperceptibly toward Lake Ontario.

The Lake Tonawanda Plain, which lies south of the Niagara escarpment, is also a fairly flat-lying area. A topographic low trends east-west through the plain and is drained along its axis by Tonawanda Creek. The 3163 Buffalo Avenue site lies approximately 1500 feet north of the Niagara River, south of the Niagara escarpment, and 600 feet east of Gill Creek (see Figure 1-1). Locally, the topography slopes less than 1° west before encountering Gill Creek, which is tributary of the Niagara River.



SOURCE: Adapted from Johnson 1964.

Figure 2–1 PHYSICAL FEATURES OF THE NIAGARA FALLS AREA

2.3 REGIONAL GEOLOGY

The surface of the Niagara Falls area is covered by a thin layer of unconsolidated deposits laid down during the closing phases of the Great Ice Age, or Pleistocene Epoch. In the wake of the receding glaciers, terminal moraines, shallow lakes, and small streams were formed. This resulted in the deposition of overburden material composed of glacial till, lake clay, silt, fine sand, and localized stream-deposited sands and gravels.

The bedrock of the Niagara Falls area consists of nearly flat-lying Paleozoic sedimentary rocks. These sediments were deposited during the Middle Silurian, a period characterized by extensive shallow seas. During the Late Silurian, restricted seas caused the precipitation of salt, gypsum, and anhydrite deposits. At the close of the Silurian Period, the sea withdrew from Western New York but persisted in the eastern part of the state. The remainder of the Paleozoic Era, as well as the Mesozoic and most of the Cenozoic Eras, have left no geologic record in the Niagara Falls area of Western New York (Buehler and Tesmer 1963).

The Silurian bedrock units beneath the Niagara Falls area are composed of dolomite, shale, limestone, and sandstone (see Figure 2-2). The bedrock dips to the south at approximately 30 feet per mile. The depth to bedrock is between 5 and 12 feet in the vicinity of the Buffalo Avenue site and the first bedrock formation encountered is the Silurian-aged Lockport Dolomite.

The member of the Lockport Dolomite investigated beneath the Buffalo Avenue site is the Oak Orchard. The Oak Orchard comprises over half of the mass of the Lockport formation in Western New York and is approximately 60 feet thick. Minerals such as anhydrite, sphalerite, fluorite, pale yellow calcite, and galena are often found recrystallized in the vugs and fractures within the Lockport Dolomite. Hemispherical masses called stromatolites and disorganized fossiliferous masses called biostromes have been found in several areas within the Oak Orchard member. These features are generally 4 to 10 feet across (Zenger 1965), but have also been reported as large as 50 feet across and 10 to 20 feet thick (Johnson 1964). Underlying the Lockport Dolomite is the Rochester

Table 1

WATER-BEARING CHARACTERISTICS OF UNCONSOLIDATED DEPOSITS AND BEDROCK IN THE NIAGARA FALLS AREA

Age			Water-bearing Unit	Thickness (feet)	Lithologic Description	Yield of wells (gallons per minute)	Water-bearing Characteristics
Quaternary	Unconsolidated Deposits		Sand and gravel	0 - 70	Fine- to medium-grained sand interbedded with coarse sand and pebbles.	1-200	Highly permeable. Sand and gravel generally occur in thin deposits in isolated hills and in the "beach ridge". Thinness of the deposits and occurrence as topographic highs prevent development of large supplies. Wells are almost always adequate for domestic use and locally are adequate for small commercial supplies.
			Lake clay, silt and fine sand	0 - 90	Laminated clay and silt with thin beds of fine sand.	Generally less than 1	Clays and silts have very low permeability and yield little water. Water is found principally in the fine sand. Wells penetrating sand beds are generally adequate for domestic use.
			Glacial till ("hardpan")	0 - 20	Mixture of boulders and pebbles in a matrix of sand, silt, and clay.	Generally less than 1	Water occurs principally in thin sand lenses in till and a "washed zone" at the top of bedrock. The washed zone often directly overlies a fractured zone in the bedrock forming a continuous aquifer. Wells penetrating the washed zone are usually adequate for domestic use.
Silurian	Bedrock	Lockport Dolomite	Upper part and middle part (Oak Orchard Member)	110	Dark-gray to grayish-brown dolomite, massive to thin bedded, locally containing algal reefs and small masses of gypsum.	Major part of area 2 - 110 (average 31)	Groundwater occurs principally in four water-bearing zones parallel to bedding which are much more permeable than the surrounding rock. Vertical joints and small cavities formed by solution of gypsum yield small amounts of water, particularly in the upper 10 to 15 feet of rock. Wells are usually adequate for domestic and small commercial uses.
						Small area adjacent to upper Niagara River 50 - 2,200 (average 800)	Same as above except that the bedding joints, which comprise the principal water-bearing zones, and the vertical joints are hydraulically connected to the Niagara River. This permits the development of large supplies, much of which infiltrates from the Niagara River. Yields of individual wells vary greatly because of differences in the degree to which bedding joints have been widened by solution and variations in the number and spacing of vertical joints.
			Lower part (Eramosa, Goat Island, Gasport and DeCew members)	50	Gray to brown dolomite, locally containing gypsum. Light gray coarse-grained limestone and shaly dolomite at the base.	1/2 - 20 (average 7)	Groundwater occurs principally in three water-bearing zones parallel to bedding. These three zones are less permeable than those in the upper and middle Lockport, causing wells to have lower yields. Wells are usually adequate for domestic uses except immediately adjacent to the Niagara escarpment where part of the Lockport is dewatered.
			Clinton and Albion groups (Rochester, Grimsby, Whirlpool)	200	Light gray limestone grading downward into shale, alternating beds of shale and sandstone, and white sandstone at the base.	1/2 - 5 (average 2 1/2)	Groundwater occurs principally in bedding joints and vertical joints within the sandstones and limestones. the limestone near the top (Irondequoit) and basal sandstone (Whirlpool) yield most of the water. The Rochester Shale at the top is almost impermeable and acts as a confining bed to the limestones and sandstones below.
Ordovician			Queenston Shale	1,200	Red sandy to clayey shale.	7 (in fractured or weathered zone)	Groundwater occurs principally in a fractured zone in the top few feet of shale. The remainder of the formation is almost impermeable. Wells not obtaining water from the fractured zone are usually inadequate for domestic use.

SOURCE: Adapted from Johnson 1964.

Figure 2-2
STRATIGRAPHY AND WATER-BEARING CHARACTERISTICS OF THE UNCONSOLIDATED DEPOSITS AND BEDROCK IN THE NIAGARA FALLS AREA

Shale. Groundwater in the Lower Lockport and Rochester Shale was not considered in this study.

2.4 REGIONAL HYDROGEOLOGY

Groundwater in the Niagara Falls area occurs in both the unconsolidated deposits and bedrock. However, the majority of the overburden is composed of lake silt and clay deposits, which are not transmissive enough to yield adequate water for domestic or small agricultural needs (see Figure 2-2 for a summary of the water-bearing characteristics of these sediments). In general, the glacial till deposits within the overburden also do not produce much water. In some locations within the Niagara Falls area, the sand and gravel deposits within the overburden do produce enough water to satisfy domestic and small farm demands. However, these sand and gravel deposits are very discontinuous and, consequently, readily dewater when pumped heavily.

The Lockport Dolomite is the primary aquifer in the Niagara Falls area. Groundwater occurs within the bedrock in secondary openings in the form of fractures due to tectonic stresses, isostatic rebound, and near-surface weathering. A more minor cause of secondary openings are cavities formed by solution of minerals. Vertical fractures occur predominately in the upper 10 to 20 feet of bedrock. A conjugate northeast-southwest (N 65°E) and northwest-southeast (N 30°W) fracture system has been reported in this area (Yaeger and Kappel 1987).

The primary conduits of groundwater flowing through the bedrock are fractures parallel to bedding. The highest flow rates are obtained in these fractures, which have been further widened by secondary solution of the rock. These fractures may pinch out laterally and be replaced by other fractures in a complex network. In general, the upper 10 to 20 feet of the Lockport bedrock has more interconnected fractures than the deeper, thickly bedded intervals and, consequently, is more transmissive.

A thorough description of the fracture system within the bedrock would provide the most comprehensive evaluation of the hydrologic media of the Niagara Falls area. However, the fractured bedrock of the Lockport is very complex, particularly when focusing on small-scale detail. Bedrock cores, slug tests, and well log data from the surrounding Dupont facility provide some of the site-specific detail of fracture characteristics. Nomenclature to describe fracture zones within the Oak Orchard member of the Lockport Dolomite has been developed from hydrogeologic studies at the nearby DuPont facility. The names of these fracture zones are identified by their relative depth beneath the surface as follows:

B-Zone -	Weathered zone immediately beneath the overburden; averages 10 to 15 feet thick. This is the most transmissive zone of Lockport Dolomite.
C-Zone -	First significant fracture zone beneath B-Zone; average depth is 20 to 30 feet beneath ground surface.
CD- and D-Zones -	Next significant Fracture Zones beneath C-Zone; averages 35 to 70 feet beneath ground surface.
F-Zone -	Fracture Zone beneath D-Zone; averages 90 to 100 feet beneath ground surface.

These zones and their effect on site hydrogeology are described in detail in Section 4.

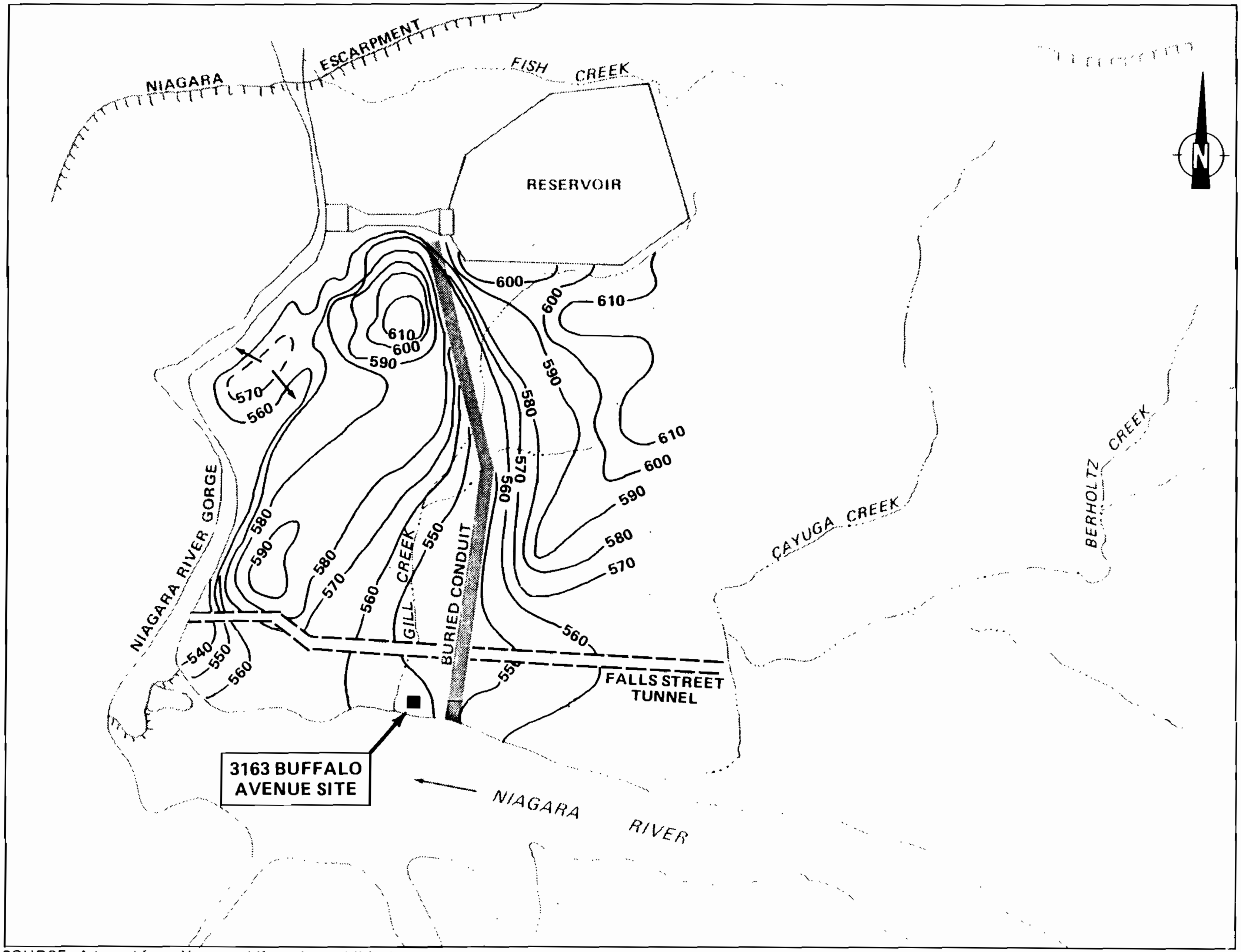
Average relative estimates of the hydraulic properties of the Niagara Falls aquifers were reported by Johnson (1964). Based on isolated measurements at a limited number of locations, the silts and clays in the overburden were calculated as having a hydraulic conductivity of approximately 0.04 gallon per day per square foot (gpd/ft^2). The hydraulic conductivity of the glacial tills was estimated as less than or equal to 23 gpd/ft^2. The majority of the bedrock aquifer wells produce 1 to 110 gallons per minute (gpm) except for some exceptionally high-yielding wells close to the Niagara River. Regional hydraulic conductivities in the bedrock were reported to range from 8 to 500 gpd/ft^2.

The natural direction of groundwater movement in the bedrock of the site area was thought to be toward the west (Yager and Kappel, unpublished). Under this flow regime, the upper Niagara River was considered to be a recharge zone for groundwater and the lower Niagara River within the gorge was considered a discharge area. However, regional groundwater flow maps for groundwater movement in the upper Lockport Dolomite

(Yager and Kappel, unpublished) (see Figure 2-3) suggests groundwater passing adjacent to the Buffalo Avenue site generally flows to the northeast. In addition, the results of a study conducted in 1980 by Wehran Engineering in association with Recra Research (untitled but cited in the Engineering Science/Dames and Moore Phase II Investigation, July 1985) suggested that both the overburden and bedrock aquifers have a hydraulic gradient and, hence flow, to the northeast. This direction of groundwater flow evidently is artificially induced by the drains that surround two water conduits cut deep into the Lockport Dolomite and located 1,500 feet east of the site. This has been confirmed independently by the Woodward-Clyde study of the adjacent DuPont plant site (Supplemental Geohydrologic Investigations, Niagara Plant, Niagara Falls, New York, October 1984). The conduits, constructed in 1959 and 1960, carry water from the Niagara River to the New York Power Authority's Niagara Hydroelectric Power Plant. The drains surrounding the conduits provide a groundwater sink and cause the groundwater to flow toward the conduits. A major discharge point for water in the conduit drains is the Falls Street Tunnel where it crosses the conduits. This intersection is approximately 2,000 feet northeast of the site and is apparently the discharge point for the Lockport aquifers under the site. The Falls Street Tunnel is oriented east to west, and the majority of the groundwater discharges into the Lower Niagara River close to the Rainbow Bridge (Miller and Kappel 1987; USGS Water Resources Investigations 86-4130 "Effect of Niagara Power Projection Groundwater Flow in the Upper Part of the Lockport Dolomite, Niagara Falls, New York"). During low flow, the discharge is diverted to the Niagara Falls Wastewater Treatment Plant. Most likely, the Falls Street Tunnel also receives groundwater along its entire course between its intersection with the conduits and its discharge point near the Rainbow Bridge.

2.5 REGIONAL CLIMATE

Climate in the Niagara Falls area can be characterized as a humid, changeable, continental type, modified considerably by the Great Lakes. Both Lake Erie and Lake Ontario have a direct effect on the area's climate as the prevailing westerly winds crossing these waters moderate winter cold and summer heat.



SOURCE: Adapted from Yager and Kappel unpublished.

Figure 2–3 REGIONAL GROUNDWATER FLOW IN THE SHALLOW LOCKPORT DOLOMITE

Winters are long and rather cloudy in this area. Snowfall averages 50 to 60 inches, about 40 inches of which can be expected from general or synoptic storms; the remainder are due to the "lake effect" phenomenon. These lake effect snows, caused by cold air crossing the warmer Lake Erie waters, can affect Niagara County from time to time, but usually remain to the south of the area in Erie County. Only rarely does Lake Ontario induce such activity here. Average daily temperatures fall below freezing from mid-December through mid-March, but thaws occur frequently. Subzero temperatures are quite rare, averaging three occurrences per winter. The ground is snow-covered for about 60 days in an average year, but periods of bare ground are not unusual even in mid-winter. Ground frost reaches a maximum depth of 3 feet.

The least precipitation frequently occurs in the months of April and May in Western New York. Typically, groundwater levels remain at some of their highest levels in March because of snowmelt and begin to drop in mid-April and May. The season's last frost usually occurs in late April near the lakeshore and early May inland. Sunshine increases noticeably in spring as the cooler lakes become a stabilizing influence on the area's climate.

Summer begins rather abruptly in mid-June. The prevailing southwest winds off Lake Erie keep temperatures from rising over 90° during all but the most intense heat waves. June and July often have extended dry periods, with soil moisture deficits usually evident by late June. Rainfall increases somewhat in August.

Autumns are dry and mild, at least through October, but cloudiness increases markedly in November and the season's first snowfall can be expected in mid-November.

Niagara County's total precipitation averages 30 to 35 inches and is rather evenly distributed throughout the year. Its frequency and intensity varies from season to season; however, summer rains are intense but of short duration, while winter precipitation is light but frequent.

3. INVESTIGATION METHODOLOGY

3.1 INTRODUCTION

Four investigations have been conducted to assess the nature and extent of contamination at the 3163 Buffalo Avenue site. These include: an investigation completed in 1980 by Wehran Engineering in association with Recra Research; the Phase I and Phase II investigations conducted in 1983 and 1985, respectively, by Engineering Science, Inc., in association with Dames and Moore, for NYSDEC; and the RI performed by E & E in 1990. The following discussion presents primarily the methods employed during E & E's investigation. A chronology of the field work performed at the site during the RI appears in Table 3-1.

Prior to the RI, the most comprehensive contaminant study involving the site was completed in 1980 by Recra Research and Wehran Engineering. Water, soil, and sludge samples were collected from 26 shallow wells and test pits, as well as from sewers. As a result of this study, dichlorobenzene was detected in groundwater, soil, and sewer samples in concentrations ranging from 9 parts per billion (ppb) to 480,000 ppb. Benzene was detected in groundwater samples in concentrations ranging from 3,300 to 110,000 ppb. Low concentrations (5 to 570 ppb) of trichlorobenzene also were present in the groundwater. The highest concentrations of every contaminant were recorded in samples obtained from the southwest corner of the site.

During the 1985 Phase II study, four surface water samples were collected from Gill Creek, located approximately 600 feet to the west of the site. Detectable levels of methylene chloride, aldrin, benzene hexachloride (BHC), cadmium, copper, mercury, and zinc were identified in the surface water samples. The concentrations of these substances were relatively low and based on the 1985 study could not be attributed to the site.

Table 3-1

FIELD WORK CHRONOLOGY

Task	Dates Work Performed
Site Reconnaissance/Geophysical Surveys	November 1, 1989 to November 2, 1989
Monitoring Well Installation/Development	November 3, 1989 to January 9, 1990
Environmental Sampling:	
Soil from Monitoring Wells	November 3, 1989 to January 9, 1990
Sewer/Storm Drain Sediments	January 8, 1990 and March 21, 1990
Subsurface Soil Composites (Trenching)	March 13, 1990 to March 15, 1990
Groundwater Sampling	February 6, 1990 to February 13, 1990
Nonaqueous Phase Liquid (NAPL)	March 16, 1990 and May 15, 1990
Aquifer Testing	March 16, 1990 to March 21, 1990
Monitoring Well 1B Repairs/Development	May 2, 1990
Water Level Measurements	February 2, 1990 and May 15, 1990

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In addition to the foregoing studies, Occidental Chemical Corporation (Occidental) is conducting ongoing investigations and remediation at its S-Area landfill, located 1/2-mile from the site; and DuPont and Olin Chemical have conducted investigations on their property, which is located immediately adjacent to the site. Of particular significance is the fact that DuPont has installed and sampled three bedrock well clusters near the site (designated by Du Pont as Nos. 10, 17 and 18). The location of each well is shown in Figure 3-1.

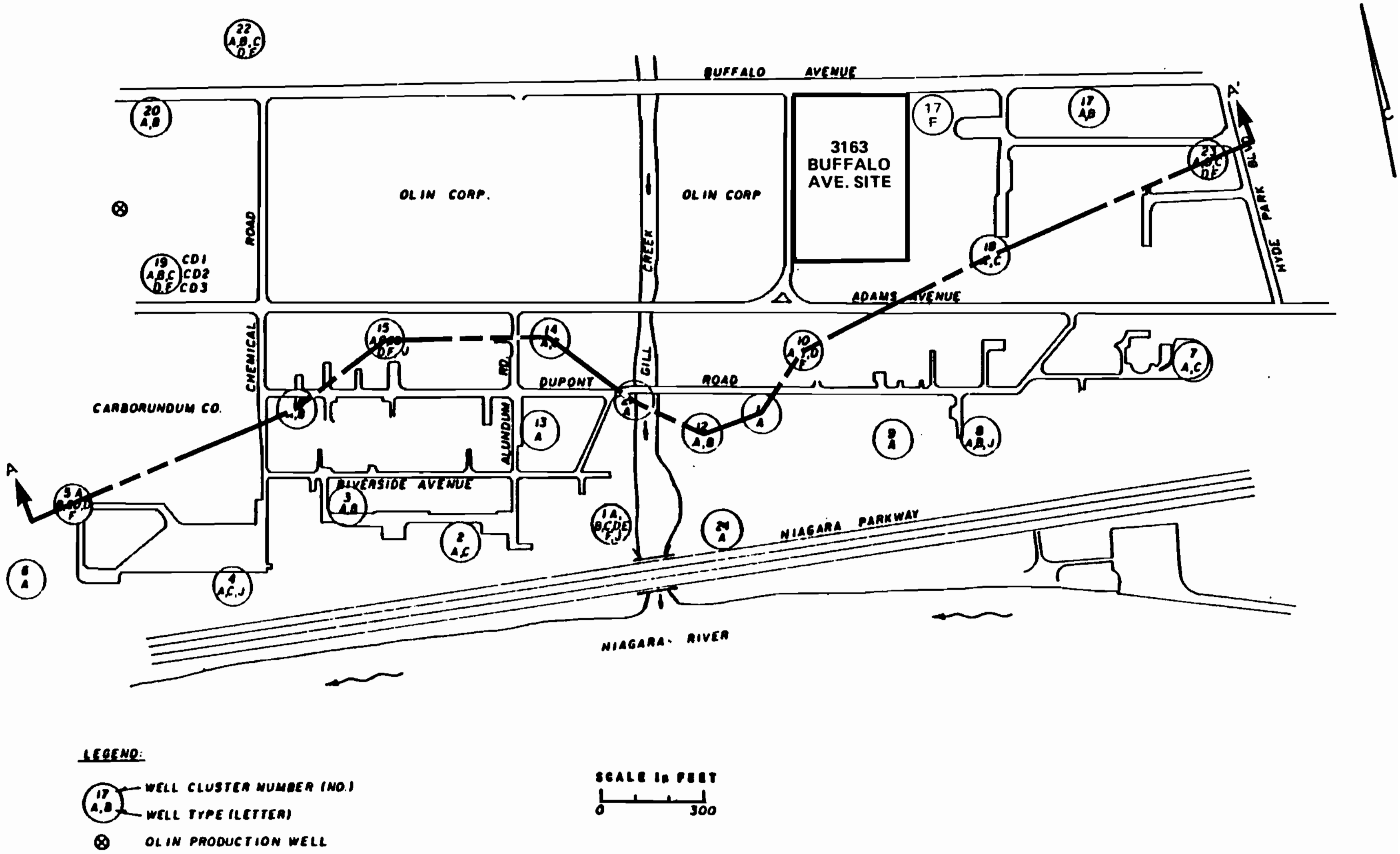
The purpose of this section is to describe the field investigation tasks that have been performed at the 3163 Buffalo Avenue site and verify that the work performed meets NYSDEC quality control (QC) standards. Interpretations of relevant data that have been generated from these tasks will be presented and discussed in Section 4. The various tasks performed during the course of the investigation include:

- Electromagnetic conductivity (EM31) and magnetometer surveys;
- Monitoring well installation and development;
- Environmental sampling:
  - groundwater
  - soil
  - sewer/storm drain
  - nonaqueous phase liquid (NAPL) sampling; and
- Slug testing.

For a complete chronology of field work see Table 3-1.

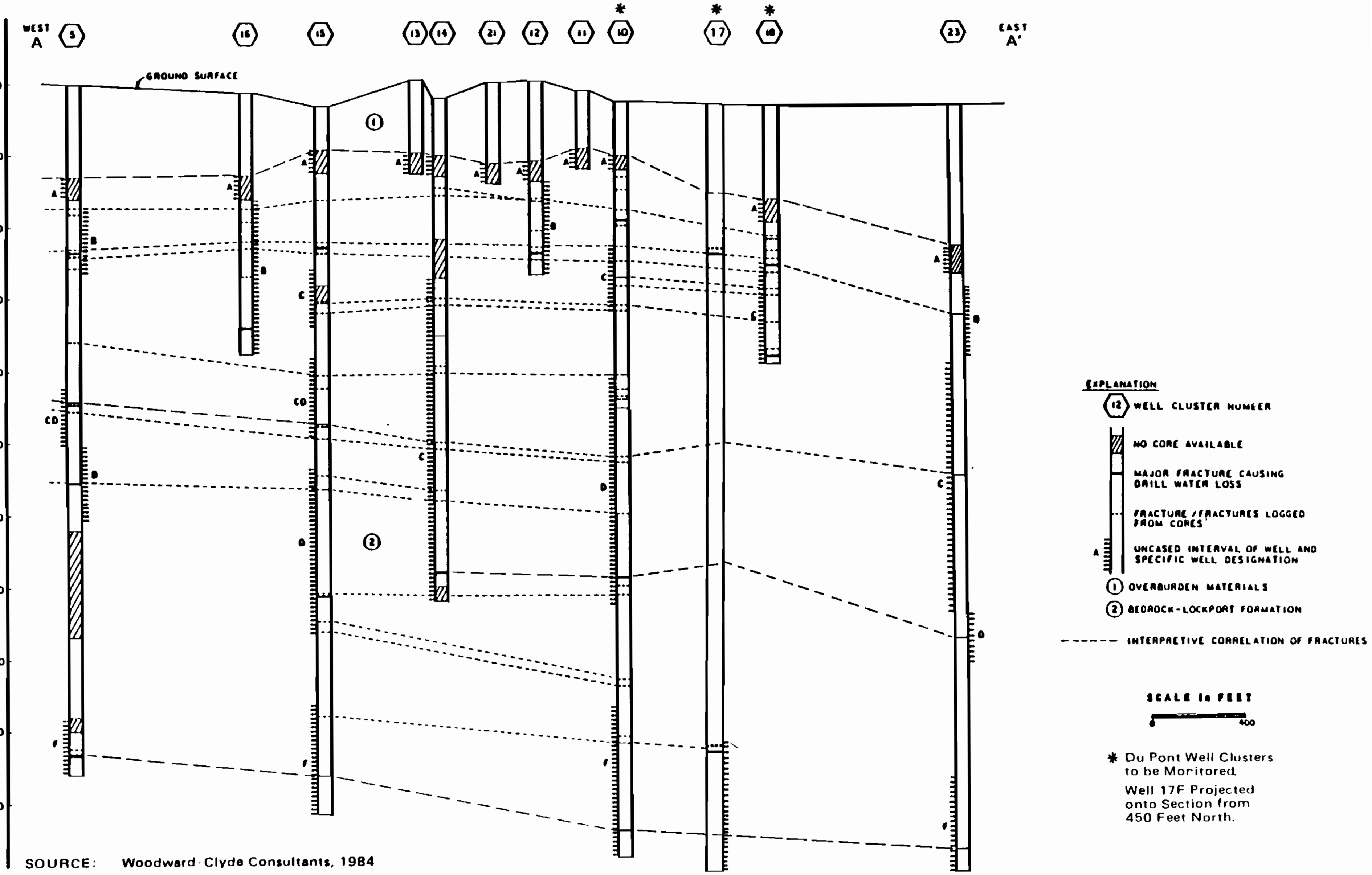
3.2 GEOPHYSICAL SURVEYS

A geophysical investigation was conducted by E & E at the 3163 Buffalo Avenue site on November 1 and 2, 1989. The geophysical investigation consisted of an EM31 (electromagnetic terrain conductivity) survey and a portable proton magnetometer (total earth field magnetics) survey. The main objective of the geophysical survey used was to ensure that no subsurface obstructions were present at monitoring well locations.



SOURCE: Woodward-Clyde Consultants, 1984

Figure 3-1
LOCATIONS OF DU PONT MONITORING WELLS AND CROSS SECTION (See Figure 3-2)
NEAR 3163 BUFFALO AVENUE SITE



SOURCE: Woodward-Clyde Consultants, 1984

Figure 3-2
EAST-WEST CROSS SECTION OF DU PONT SITE SHOWING FRACTURE ZONES AND AQUIFERS IN MONITORING WELL CLUSTERS IN THE LOCKPORT DOLOMITE

Survey grids were established at each of the prospective groundwater monitoring well cluster locations. The X and Y axes of each survey grid were oriented east-west and north-south, respectively. A total of seven survey grids were established at the site. Survey grid coordinate 0,0 is located in the southwest corner of each grid. Semi-permanent wooden stakes were used to mark the proposed drilling locations, prior to any actual drilling.

Survey grids 1 through 7 included the locations of sixteen monitoring wells (see Table 3-2). The dimensions vary from grid to grid due to physical obstacles (buildings, debris, etc.), but the station spacing (10 feet) is identical for each survey grid on site. Both vertical and horizontal dipole readings were recorded at each survey grid station while performing the electromagnetic ground conductivity survey using the Geonics, Ltd., EM31 Ground Conductivity Meter. The effective depths of penetration provided by the EM31 in the vertical and horizontal dipole modes are about 18 feet and 9 feet, respectively. These depths were adequate to delineate any buried materials that would have been encountered while drilling. Magnetometer readings were recorded at each station in both north-south and east-west orientations using the EG&G Unimag II (Model G-846) Portable Proton Magnetometer.

All conductivity and geomagnetic field data were recorded in a separate logbook dedicated to this investigation along with the photo log.

Based upon the interpretation of the data collected, the locations originally proposed for the majority of the monitoring wells was acceptable. The only wells that needed to be moved were MW-1A and MW-1B, and the MW-5 cluster. Monitoring wells MW-1A and MW-1B were relocated approximately 10 feet due east of their original location, in order to avoid drilling into a sanitary/storm sewer buried in the area. Monitoring well cluster MW-5 was moved approximately 15 feet north of the proposed location to avoid drilling into a buried water line.

3.3 GROUNDWATER MONITORING WELLS

3.3.1 Objectives

E & E's monitoring well program was designed to characterize the site and meet the following objectives:

Table 3-2

GEOPHYSICAL SURVEY GRID

Geophysical Survey Grid No.	Monitoring Wells Located Within Grid	Grid Dimensions* (X axis) x (Y axis) (in feet)	Comments
1	MW-1A MW-1B	50′ x 30′	
2	MW-2A MW-2B	30′ x 30′	Interference from large amounts of surface metal debris and nearby chain-link fence
3	MW-3A MW-3B	40′ x 30′	Two chain-link fences join with coordinate 0,0, the southwest corner of the site
4	MW-4A MW-4B	30′ x 40′	
5	MW-5A MW-5B MW-5C MW-5CD MW-5F	20′ x 40′	
6	MW-6A MW-6B	30′ x 40′	
7	MW-7A	20′ x 40′	

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Key:

A = Indicates shallow, overburden well
B-F = Indicates progressively deeper bedrock wells

*Grid dimensions vary due to physical restrictions (buildings, etc.)

- Determine the degree and extent of contamination on site in both the overburden and the Lockport Dolomite down through Zone F;

- Measure the hydraulic gradients across the site in both the overburden and the Lockport Dolomite;

- Measure the vertical gradients between the overburden and upper bedrock and within the bedrock down to Zone F;

- Determine the hydraulic conductivity of the overburden;

- Determine the hydraulic conductivity of the upper part of the bedrock and individual bedrock zones down to Zone F; and

- Estimate the contaminant loading and migration within each aquifer.

To accomplish all of these objectives, a total of 16 wells were installed. Twelve small-diameter wells have been placed in sets of two, each containing a 2-inch diameter shallow well completed in the overburden (maximum depth 15 feet) and a 4-inch diameter deeper well completed in the uppermost aquifer of the Lockport Dolomite (Zone B). Another overburden well was installed about midway along the west side of the site (see Figure 1-2). The three deeper bedrock wells (5C, 5CD and 5F) were completed in Zones C, C-D, and F (see Figure 1-2). The location of the deep well cluster was selected to provide information on groundwater and contaminant flow in conjunction with the existing deep well clusters installed by DuPont. DuPont clusters 10, 17 and 18 were used for this purpose.

Well drilling, installation, and construction specifications are described in the following subsection. Well logs appear in Appendix A.

A-Zone Wells

A-Zone wells were installed as follows (see Figure 3-3):

- An 8-inch borehole was augered in the overburden with 6.25-inch internal diameter (I.D.) hollow-stem augers until weathered bedrock was encountered. In general, the depth of bedrock ranged from 5 to 12 feet below the surface. Continuous split spoon samples were obtained from MW-2A, -3A, -4A, -6A and -7A. Discrete samples from 2 to 4 feet and composite samples over the entire boring were also

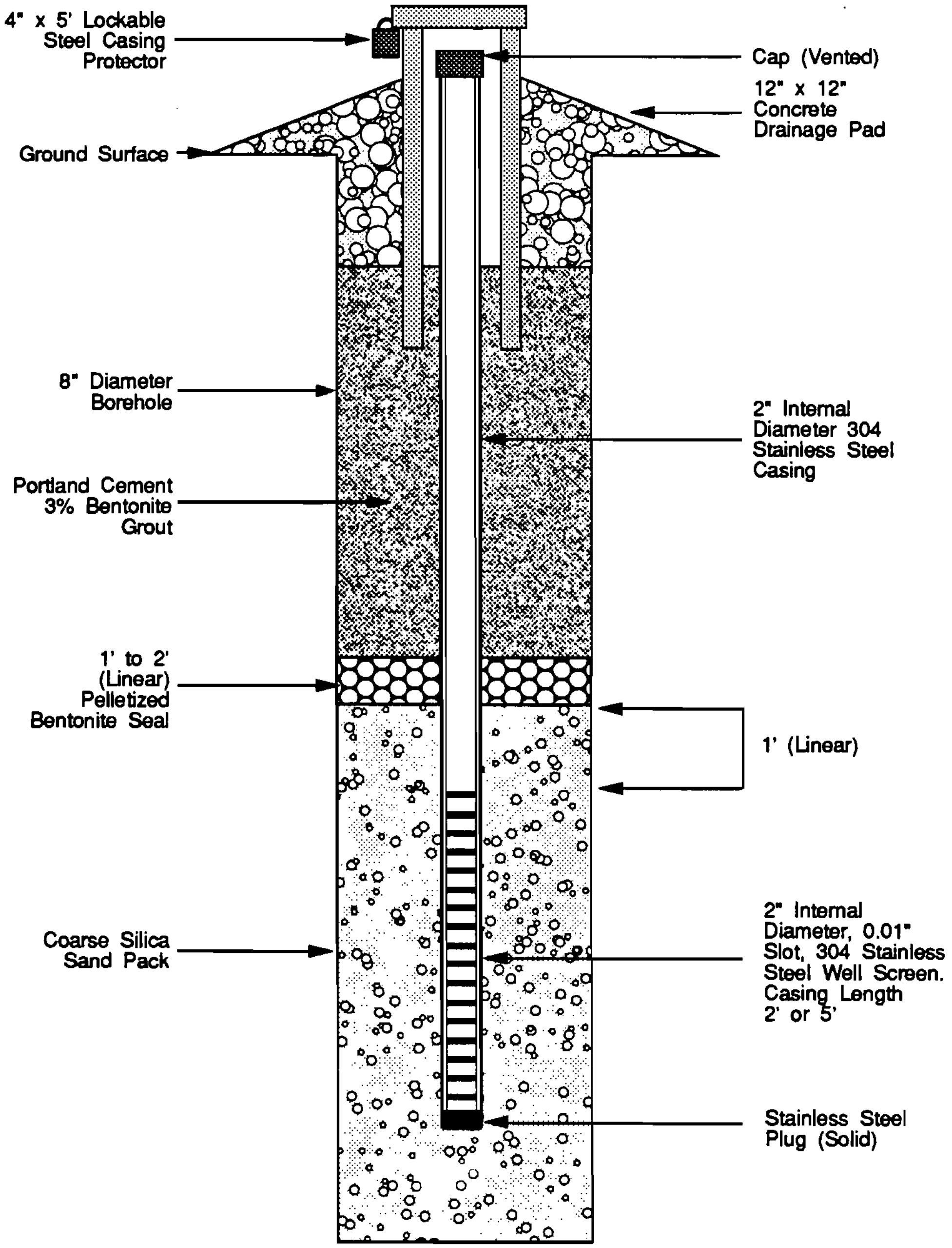

Figure 3-3
MONITORING WELL CONSTRUCTION TYPE A

obtained from these wells. Only A-Zone wells 1A and 5A were not logged. (The adjacent B wells were logged and sampled.)

- Upon reaching weathered bedrock (depth of auger refusal), the augers were progressively extracted and a 2-inch I.D. flush joint (type 304), stainless-steel monitoring well was constructed. When possible, the length of screen was designed to extend from the bottom of the well to 1 foot above the potentiometric surface to allow for detection of light NAPL. A 2- or 5-foot screen was placed in each of the A wells. A coarse silica sandpack was placed in the well annulus from the bottom of the boring to 1 foot above the top of the screen. The sandpack also included a 6-inch base beneath the bottom of the screen, when possible. A 1-foot bentonite seal was placed above the sandpack. The balance of the well was then grouted with 3% bentonite grout. A 6-inch diameter locking, surface casing with a 2- to 3-foot stick-up and a 1-foot diameter sloping concrete pad was then installed at the surface.

B-Zone Wells

B-Zone wells were also installed with 6.25-inch I.D. hollow-stem augers in the soil overburden (see Figure 3-4). Continuous split-spoon samples were collected and described by an on-site geologist for Wells 1B and 5B only. Well installation proceeded as follows:

- Augering proceeded until weathered bedrock was encountered. A 6-inch diameter hole was then drilled 1 foot into bedrock with a tricone roller bit using the water-rotary method. Prior to drilling into bedrock, a bentonite slurry was circulated in the auger hole to prevent water loss during subsequent drilling.

- Following drilling, a 1-foot bedrock rock-socket, 4-inch diameter (type 304), flush-joint stainless-steel casing was placed in the borehole. The casing was grouted with a 3% bentonite cement grout as the augers were progressively extracted from the boring. The grout was allowed to set for a minimum of 12 hours before bedrock coring was initiated.

- Continuous NX (1.87-inch diameter) cores were then obtained from the bottom of casing to 5 feet below the first water loss fracture or 10 feet into bedrock, whichever was less. Cores were described by the on-site geologist. Characteristics included are attitude of fractures, fracture frequency, lithology, and evidence of secondary mineralization. The final well design consisted of: a 4-inch I.D. stainless-steel casing to 1 foot beneath the bedrock overburden interface; a 2.96-inch diameter open

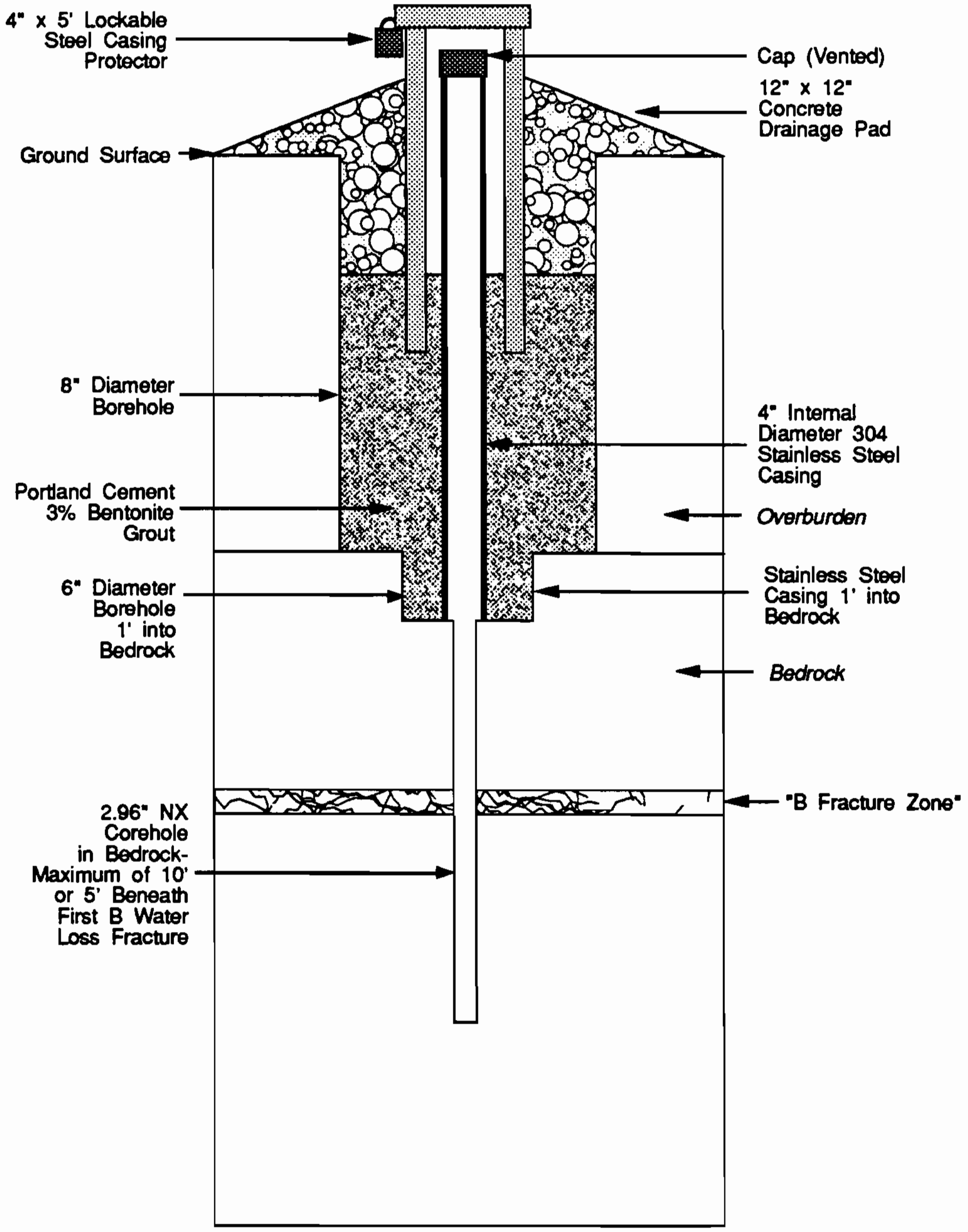

Figure 3-4
MONITORING WELL CONSTRUCTION TYPE B

borehole a maximum of 10 feet into bedrock; and 6-inch diameter locking surface casing and a 1-foot square sloping concrete pad at the surface.

Monitoring Well MW-1B, Re-Installation

During the early stages of the development of monitoring well MW-1B, a stainless-steel bailer became lodged just beneath the 4-inch stainless steel casing of the well. After numerous attempts at retrieving the bailer, it became apparent that the well had in fact collapsed around the bailer. Monitoring well MW-1B was then re-drilled/installed (see Figure 3-5) as follows:

- o Drilled over bailer using NX core barrel, and retrieved the lost bailer;
- o Enlarged the NX open hole to a four-inch diameter hole using a 3 7/8-inch tri-cone drill bit;
- o Installed 10 feet of 2-inch I.D. stainless-steel .010 slot continuous wrap screen, with 2-inch I.D. stainless-steel riser; and
- o The sandpack, bentonite and grout were then placed as shown in Figure 3-5.

C-Zone Well

Overburden soils were augered as in the B-Zone wells. No split-spoon samples or overburden logging were conducted in this well as the B-Zone well in this cluster (MW-5B) had previously been logged. The augers were progressively withdrawn and replaced with 12 feet of 6-inch I.D. black iron-steel casing to a depth of 11 feet below ground surface (BGS). The 6-inch I.D. steel casing was then grouted with a 3% bentonite grout. The grout was allowed to set for a minimum of 12 hours before drilling was resumed (see Figure 3-6). In addition:

- o A 6-inch diameter bedrock hole was drilled with a roller bit using the water rotary method to 19 feet, which is 2 feet above the bottom of the adjacent B-Zone well. A 7.5-foot core was obtained from the base of the B-Zone from 19 feet to 26.5 feet. This resulted in an overlap of cores between the C-Zone monitoring well and the adjacent B-Zone monitoring well. The cored interval was then reamed to a 6-inch diameter and a 4-inch stainless-steel casing was installed. The casing was grouted using the casing method that consists of grouting from the inside of casing with a

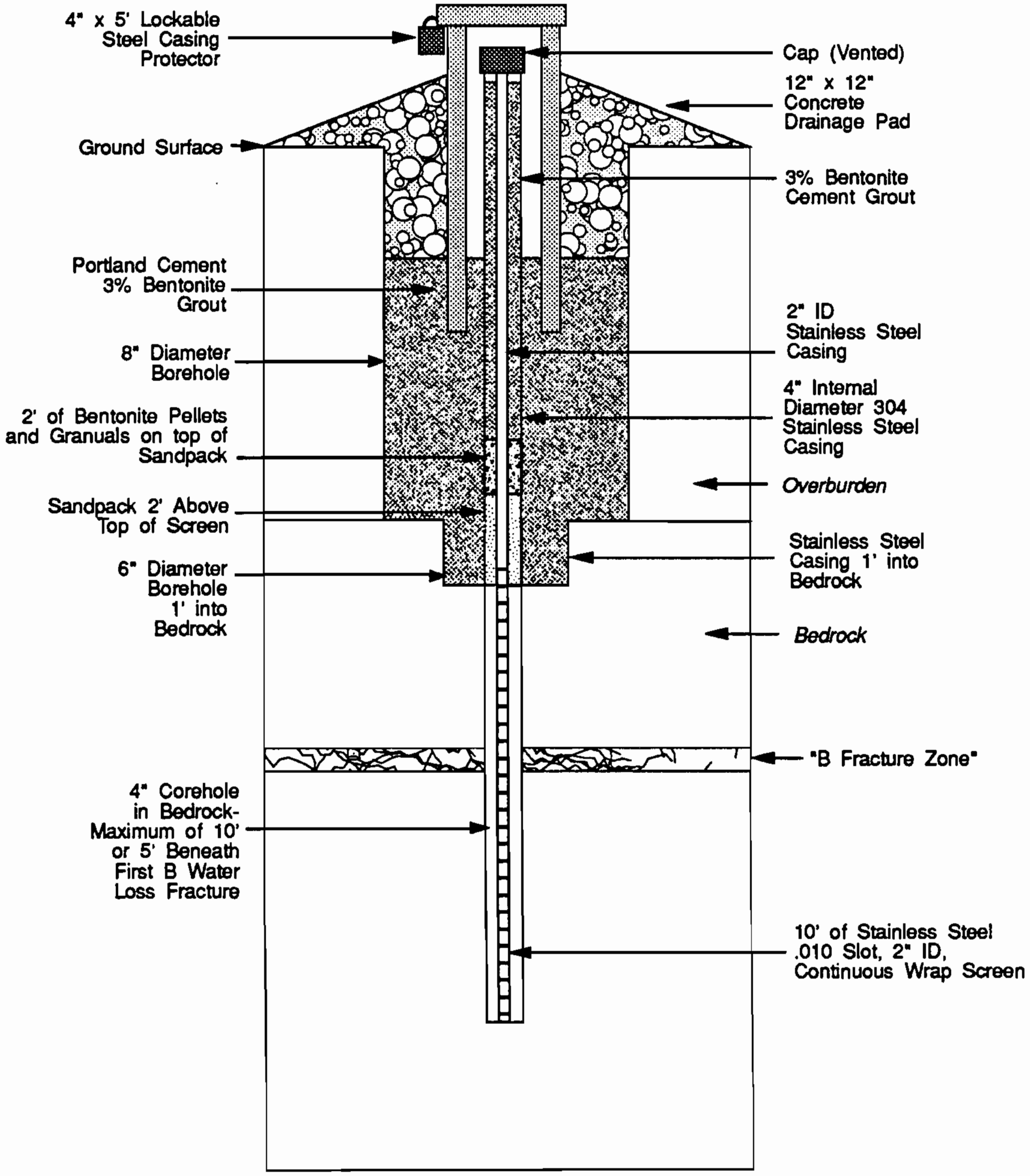

Figure 3-5
MW-1B MONITORING WELL CONSTRUCTION

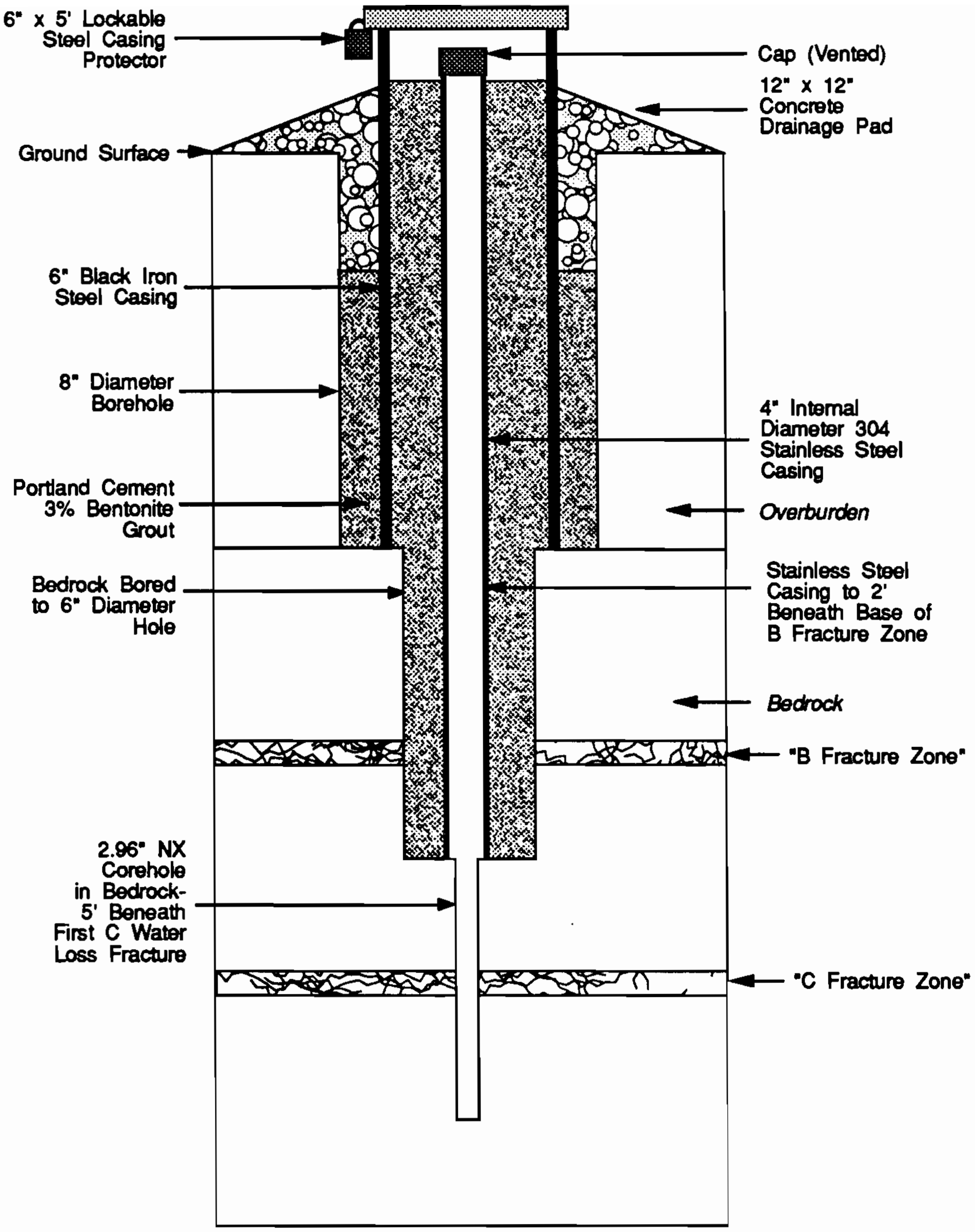

Figure 3-6
MONITORING WELL CONSTRUCTION TYPE C

tremmie pipe. A specially designed packer at the bottom of the 4-inch casing was used to ensure that grout did not migrate into the casing annulus. The grout was allowed to set for a minimum of 12 hours.

- Continuous NX cores were collected from the bottom of casing (26.5 feet) to 37.3 feet. The bedrock core was described as specified for the B-Zone wells. The length of hole open to the bedrock was 10.8 feet. The final well design consisted of: a 6-inch black-steel overburden casing set at 11 feet BGS with a 1-foot stickup, and a locking cap; a sloping concrete pad at the surface; 4-inch I.D. stainless-steel casing, which penetrates to 26.5 feet below surface; and a 2.96-inch open bedrock corehole which penetrates to 3.75 feet beneath the first water loss fracture in the C-Zone. Throughout the bedrock drilling a total of approximately 1,775 gallons of drill water was lost down the hole at depths between 21 feet to 21.6 feet, 22 feet to 24 feet, and 32.75 feet to 37.3 feet.

CD-Zone Well

The CD-Zone well (see Figure 3-7) was installed the same as the C-Zone well except that the 4-inch I.D. stainless-steel casing was extended to 39.5 feet BGS. The CD-Zone well drilling and installation was as follows:

- Set 6-inch I.D. black steel through overburden to 10.9 feet BGS;
- Reamed with 6-inch tri-cone roller bit to 34.5 feet;
- NX cored from 34.5 feet to 39.5 feet;
- Reamed with 6-inch tri-cone to 39.5 feet and set a 4-inch stainless-steel casing to 39.5 feet BGS; and
- The well was then finished by NX coring from 39.5 feet to 51.5 feet, and completed as a 2.96-inch (NX) open bedrock corehole within the CD-fracture zone.

During the course of drilling and coring, MW-5CD water loss into the bedrock fractures between 16.5 feet BGS and 39.5 feet BGS became a problem. A 4-inch stainless-steel casing was set at as shallow a depth as was possible to prevent further water loss during drilling. However, approximately 700 gallons of water were lost in the formation during drilling.

F-Zone Well

In the F-Zone well, steel casing was set at 11 feet BGS; the 4-inch stainless steel casing was set at 25.5 feet due to excessive water loss during drilling from 15 to 25 feet. After the 4-inch stainless-steel casing was set and grouted, a hole was drilled to 49.5 feet. Coring was resumed at 49.5 feet which was two feet shallower than the MW-5C well. Coring and reaming continued to 77 feet. The well was then cored from 77 feet into the F-Zone fractures to a total depth of 98 feet. Two-inch type 304, flush-joint stainless-steel casing was then grouted in from surface to 77 feet. A 3 5/8-inch stainless-steel flange was welded to the base of the casing to provide an appropriate seal. Prior to grouting, 2 feet of bentonite pellets were placed on top of the flange. The well annulus was then grouted with a tremmie pipe between the casing and the borehole. The grout contained 3% bentonite and calcium chloride accelerator. Approximately 4,000 gallons of water were lost in the well during the coring and reaming from depths of 49.5 feet to 98 feet, during which no water returned to the surface.

Grout contamination was observed in MW-5C following the completion of MW-5F. It was suspected that grout migrated through a horizontal fracture between the two wells during the completion of MW-5F. Upon discovery of this problem, MW-5C was reamed and produced for an extended period. The well achieved a sustained production rate of 5 gallons per minute and turbidity was lowered to below 50 nephelometric units; the pH was also lowered significantly from 12.0 to 9.95. The pH had decreased to an average 8.9 at the time of groundwater sampling.

3.3.2 Monitoring Well Development

All wells were developed at the facility between December 18, 1989 and February 13, 1990, following the completion of the drilling program. All A- and B-Zone wells were developed by using stainless-steel, bottom-loading bailers. Water was bailed from varying depths within the water column, ensuring that all static water was purged. During the course of the development, readings for pH, temperature, conductivity, and turbidity were measured periodically (see Table 3-3). In general, well development continued until pH and conductivity stabilized. An attempt was also made to reduce turbidity levels to the NYSDEC goal of 50

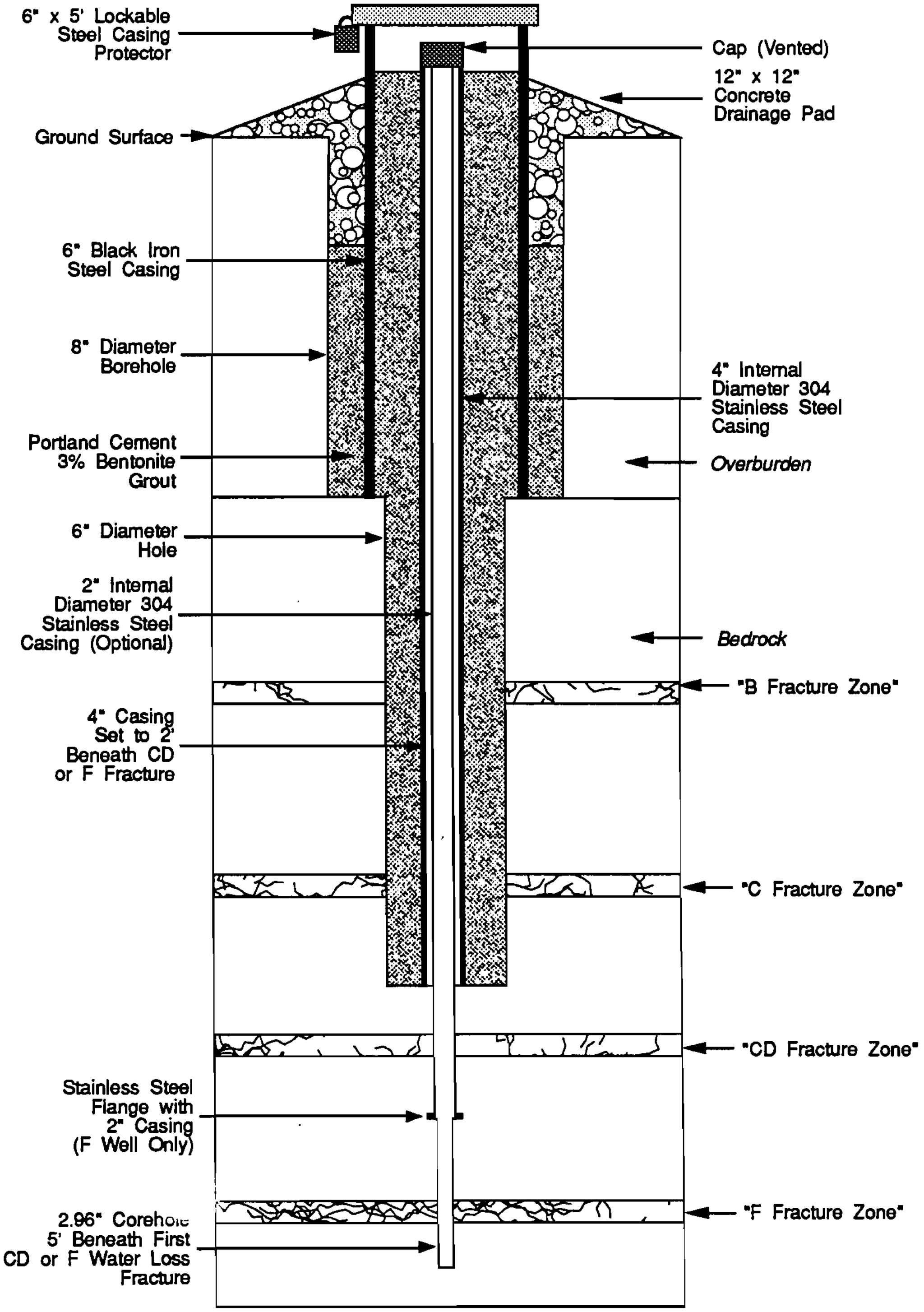

Figure 3-7
MONITORING WELL CONSTRUCTION TYPE CD AND F

Table 3-3

WELL DEVELOPMENT

Monitoring Well No.	Date	Development Method	Gallons Removed	Turbidity (NTUs)	pH	Conductivity (millimhos/meter)	Temp (°C)	Comments
MW-1A	12-28-89	Bailed	2	>200	6.85	3,800	5	Bailed dry, silty
	1-4-90	Bailed	1.5	>1000	6.35	4,025	6	Bailed dry, silty
MW-1B	12-28-89	Bailed	8	--	7.30	3,300	9	Bailer became stuck in well
	1-9-90	Bailed	4.5	100	7.85	3,800	9.5	Bailer still in well
	5-2-90	Bailed	6	>1000	6.222	15,960	14.4	Bailer removed, 2" stainless-steel well installed
MW-2A	12-27-89	Bailed	5	--	7.45	4,000	6	Dark gray, bailed dry
	1-4-90	Bailed	4	325	7.50	4,100	7.5	Bailed dry
	2-7-90	Bailed	4	--	4.16	--	5	Bailed dry
MW-2B	12-27-89	Bailed	14	--	7.35	2,400	7	Dark gray, did not clear up
	12-29-89	Bailed	20	180	6.20	1,600	7	--
	2-7-90	Bailed	13	--	4.14	--	6	--
MW-3A	12-27-89	Bailed	6	--	6.55	1,700	6	Light brown, cloudy
	1-5-90	Bailed	6	>1000	6.90	1,400	5.5	Cloudy brown
	2-7-90	Bailed	4	--	7.04	8,770	6	--
MW-3B	12-27-89	Bailed	10	--	7.50	2,100	9	Light brown, cloudy
	12-29-89	Bailed	16	190	6.45	650	7	--
	2-7-90	Bailed	4	--	7.05	15,490	8	--
MW-4A	12-28-89	Bailed	3	--	7.10	1,000	6	Light brown-gray, bailed dry
	12-29-89	Bailed	4	30	5.65	1,900	5	--
	1-4-90	Bailed	4	>1000	6.35	2,900	4	--
	2-8-90	Bailed	4	--	6.65	15,270	7	--
MW-4B	12-28-89	Bailed	11	--	8.35	2,000	8	Cloudy with slight oil sheen
	12-29-89	Bailed	20	600	5.70	1,500	7	--
	2-8-90	Bailed	6.25	--	7.72	13,260	7.5	--
MW-5A	12-28-89	Bailed	3	--	6.90	1,000	7	--
	1-5-90	Bailed	5	<1000	6.95	1,000	8	Cloudy brown
	2-8-90	Bailed	2.5	--	6.37	8,890	9.7	--
MW-5B	12-28-89	Bailed	7	--	7.15	1,900	7	--
	12-29-89	Bailed	18	420	6.15	1,900	10	--
	2-8-90	Bailed	2.73	--	6.38	13,080	12	--

Table 3-3 (Cont.)

Monitoring Well No.	Date	Development Method	Gallons Removed	Turbidity (NTUs)	pH	Conductivity (millimhos/meter)	Temp (°C)	Comments
MW-5C	1-4-90	Pumped	20	3.4	12.05	4,800	9	Clear
	1-5-90	Pumped	10	>1000	12.90	8,400	10.5	Grout particulates, quickly separates
	1-8-90	Pumped	25	54	11.45	2,700	9.5	--
	1-9-90	Pumped	65	48	9.95	3,800	10	Gray becoming clear
	2-9-90	Pumped	20	--	8.92	—	6.8	Clear
MW-5CD	1-8-90	Double tube reverse airlift	105	200	7.75	4,000	15	Cloudy
	2-9-90	Bailed	33	--	6.63	--	7	--
MW-5F	1-9-90	Double tube reverse airlift	110	0.4	8.10	2,800	9.5	Clear
	2-13-90	Bailed	93	--	7.62	--	12.2	--
MW-6A	12-18-89	Bailed	5	--	6.85	2,750	7.5	Cloudy gray
	12-28-89	Bailed	1	--	6.80	2,300	6	--
	1-5-90	Bailed	5	82	7.85	2,800	5.2	Bailed dry, cloudy gray
	2-7-90	Bailed	3.5	--	5.3	--	7.8	Bailed dry
MW-6B	12-28-89	Bailed	10	--	10.85	9,500	7	Cloudy dark gray
	12-29-89	Bailed	18	210	9.0	6,600	8	--
	1-5-90	Bailed	8	68	11.45	6,000	10	—
	2-7-90	Bailed	14.5	--	5.3	--	9	—
MW-7A	12-28-89	Bailed	5	—	7.45	2,900	3	Cloudy dark gray with oil sheen
	1-4-90	Bailed	5	>1000	6.85	2,600	4	Black
	2-8-90	Bailed	3	--	6.99	14,890	5.2	—

nephelometric turbidity units (NTUs). However, only well MW-5C and MW-5F actually had turbidity levels below 50 NTU. All wells with high turbidities were developed on three or four separate occasions in an attempt to reduce turbidities for the 50 NTU goal.

The two deepest wells (MW-5CD and MW-5F) were developed using a double tube reverse airlift method, which successfully isolated the desired fracture zones by using two inflatable packers, above and beneath the fracture zone. Compressed air was pumped under pressure down through one tube, which brought the water and air to the surface through the other tube.

Groundwater from monitoring wells 2A, 2B, 6A, and 6B had relatively low pH values during purging on February 7, 1990. pH levels in these wells were 4.16 and 4.14 in 2A and 2B, and 5.3 in both 6A and 6B. It is possible that the pH meter was not functioning properly during this period.

3.4 ENVIRONMENTAL SAMPLING

For the RI of the 3163 Buffalo Avenue site, specific sampling procedures were followed for each type of sample collected, including soil samples, groundwater samples, and sewer/storm drain sediment samples. Analytical data appear in Appendix B.

All procedures used were consistent with NYSDEC and EPA protocol.

3.4.1 Groundwater Sampling

Sampling of the newly installed monitoring wells consisted of the following activities:

- Surveying of monitoring wells;
- Measurement of depth-to-water level and total depth of the well (to calculate well volume);
- Evacuation of static water (purging); and
- Collection of the sample.

These activities are described below.

3.4.1.1 Surveying

All wells were surveyed to the nearest 0.01 foot and water level measurements were taken as simultaneously as possible on February 2 through February 9, 1990, March 16 through March 20, 1990, and on May 15, 1990, to determine the directions of groundwater flow. For the deeper zones, E & E measured the water levels in DuPont well clusters No. 10, 17, and 18 during the March and May sampling periods. On-site wells were surveyed to the same benchmark used for DuPont wells.

3.4.1.2 Measurement of Water Level and Well Volume

Prior to sampling, the static water level and total depth of each monitoring well was measured with a calibrated water level indicator. Care was taken to decontaminate equipment between each use to avoid cross contamination of wells, and when possible two water level indicators were used which allowed one to be thoroughly decontaminated while the other was in use. The number of linear feet of static water (difference between static water level and total depth of well) was calculated.

The static volume was calculated using the formula:

$$V = Tr^2(0.163)$$

where

V = Static volume of well in gallons;

T = Depth of water in the well, measured in feet;

r = Inside radius of well casing in inches; and

0.163 = A constant conversion factor which compensates for $\pi r^2 h$ factor for the conversion of the casing radius from inches to feet, the conversion of cubic feet to gallons, and π (pi).

3.4.1.3 Purging Static Water

A minimum of three static water volumes was purged from each monitoring well prior to collecting the samples, or the well was bailed dry at least twice. Purging of the A, B, and C wells was performed using dedicated stainless-steel bailers, and purging of the deeper CD and F wells was performed using a double tube reverse airlift system.

Because the water removed from the well during the purging process could contain hazardous materials, the purge water was contained and not discharged on the ground.

3.4.1.4 Groundwater Sample Collection

Samples were collected as follows:

- A dedicated, decontaminated stainless-steel bailer was used to collect the groundwater samples;
- When transferring water from the bailer to sample containers, care was taken to avoid agitating the sample to prevent the loss of volatile constituents;
- Samples to be analyzed for metals were preserved with nitric acid prior to shipment for analysis;
- Samples to be analyzed for total cyanide were preserved with NaOH;
- Any observable physical characteristics of the groundwater (e.g., color, sheen, odor, turbidity) were recorded at the time of sampling;
- NAPL samples were collected from wells MW-1B and MW-2B, and analyzed using EPA methods 5030, 8080, 8240, and 8270. The samples were collected using the dedicated stainless-steel bailers;
- Groundwater samples were collected from each monitoring well along with the necessary quality assurance/quality control (QA/QC) samples, and analyzed for the parameters as listed in Table 3-4; and
- Municipal water which was used for decontamination and drilling (i.e., field water source) was sampled and analyzed prior to use. For the sample analysis, the same parameters were used as indicated for the groundwater sampling subtask described in Table 3-4.

3.4.2 Soil Sampling

Three types of soil samples were collected at the 3163 Buffalo Avenue site (see Table 3-5). One set was collected from between two to four feet from the surface using the drill rig at each well nest location. Along with the two- to four-foot discrete sample, a composite sample consisting of continuous split-spoon samples from ground surface to the top of bedrock was collected. This sample was made up of equal

Table 3-4

SUMMARY OF SAMPLING AND ANALYSES

Sample Type	Number of Samples[1]	TCL[2] Organics and TOC	CLP[3] Metals	Total Organic Halogens	Cyanide (Total)	Method[4] 8240	Method[4] 8270
Groundwater	24	X	X	X	X	--	--
Soil	29	X	X	--	X	--	--
NAPL	2	--	--	--	--	X	X
Sewer/Storm Drain Sediment	2	X	X	--	X	--	--
Decontamination Water	1	X	X	X	X	--	--
Field Water Source (Municipal water)	1	X	X	X	X	--	--

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[1] Total number of samples; includes field blanks, duplicates, and QA/QC.

[2] NYSDEC Target Compounds List.

[3] EPA Contract Laboratory Program metals.

[4] Method 8240 for volatile organics. Method 8270 for semi-volatiles. Extract for Method 8240 by Method 5030.

Table 3-5

SOIL SAMPLE SUMMARY

Sample Location	Soil Sample No.	Depth and Type of Sample
MW-1B	BA-SB-1B$_2$ BA-SB-1B$_c$ BA-SB-1B-NA	2'-4' grab 0'-6' composite 4'-6' grab, methods 8240, 5030, 8270
MW-2A	BA-SB-2A$_2$ BA-SB-2A$_c$	2'-4' grab 0'-9.3' composite
MW-3A	BA-SB-3A$_2$ BA-SB-3A$_c$	2'-4' grab 0'-9.8' composite
MW-4A	BA-SB-4A$_2$ BA-SB-4A$_c$	2'-4' grab 0'-8' composite
MW-5B	BA-SB-5B$_2$ BA-SB-5B$_c$ BA-DU-SB-5B$_c$	2'-4' grab and MS/MSD 0'-10' composite 0'-10' composite duplicate
MW-6A	BA-SB-6A$_2$ BA-SB-6A$_c$ BA-BL-6A	2'-4' grab 0'-9.6' composite and MS/MSD Trip blank
MW-7A	BA-SB-7A$_2$ BA-SB-7A$_c$	2'-4' grab 0'-8' composite
West side of site	BA-SE-01	Sewer sample
North side of site	BA-SE-02	Sewer sample
Area A	BA-SB-A	2'-4', area composite
Area B	BA-SB-B	2'-4', area composite
Area C	BA-SB-C	2'-4', area composite
Area D	BA-SB-D	2'-4', area composite
Area E	BA-SB-E	2'-4', area composite
Area F	BA-SB-F	2'-4', area composite
Area G	BA-SB-G	2'-4', area composite
Area H	BA-SB-H	2'-4', area composite
Area I	BA-SB-I BA-SB-I MS/MSD	2'-4', area composite Matrix spike/matrix spike duplicate
Area J	BA-SB-J BA-SB-J-D	2'-4', area composite 2'-4', area composite duplicate

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Note: One soil sample was also taken from each well nest to be analyzed for grain size and one for Atterburg limits. These samples were taken from a variety of depths to aid in characterizing the site's overburden.

portions of each split-spoon, which were homogenized in a stainless-steel bowl, prior to being placed into the sample containers. The third type of soil sample collected was a composite of five equal aliquots taken from depths of 2 to 4 feet in each of 10 areas of known or suspected contamination. The 10 areas of concern were labeled A through J as shown on the site map (see Figure 1-2). Due to the nature of the ground surface (fill materials), it was necessary to use a backhoe to obtain the samples from areas A through J.

A total of 39 soil samples were taken at the 3163 Buffalo Avenue site as follows:

- Seven 2-foot to 4-foot grab samples, one from each monitoring well nest (one double volume for matrix spike/matrix spike duplicate [MS/MSD]);
- Eight surface-to-bedrock composites, one from each monitoring well nest, and one duplicate (one double volume for MS/MSD);
- One 4-foot to 6-foot specific sample from MW-1B, analyzed using EPA methods 8240, 5030, and 8270;
- One trip blank;
- Two sewer sediment samples;
- Twelve 2-foot to 4-foot area composite samples including one MS/MSD and one duplicate; and
- Seven for grain size analysis (various depths), one from each well nest.

All equipment that came into contact with the samples was decontaminated before and after taking each sample in accordance with NYSDEC-approved procedures.

All samples were screened in the field using an Organic Vapor Analyzer (OVA) portable gas detector in the survey mode, or an HNu Systems photoionization detector, in order to identify any sample with volatile or semivolatile organic contamination.

All soil samples were then analyzed for the same parameters as for groundwater (see Table 3-4), with the exception of TOX (Total Organic Halogen).

As it was being sampled, all observable physical characteristics of the soil (e.g., color, physical state, grain size, wetness, odor) were recorded to the extent possible, and any sign of NAPL was noted.

3.4.3 Sewer/Storm Drain Sediment Sampling

As called for in the work plan, an attempt was made to sample and visually verify that both the main Buffalo Avenue sewer and a storm drain leaving the southwest corner of the site had been plugged. Field personnel however, were unable to sample or inspect the sewer/storm drain running from the site to the main Buffalo Avenue sewer due to an abandoned construction trailer parked on top of the sewer cover. Therefore, a sample was collected outside the fence along Buffalo Avenue. The drain was inspected and confirmed to be plugged. The drain leaving the site on the southwest side towards Gill Creek was also inspected and sampled. However, the field crew was unable to verify that the sewer on the southwest side was plugged without physically entering the manhole. This verification was not completed as the Site Work Plan specifically stated, "To the extent feasible without entry into either of the lines, the consultant will visually verify that the lines have been plugged and remain sealed; ...".

Samples from both sewers were taken from manholes with decontaminated stainless steel spoons. Samples were then analyzed as shown in Table 3-4.

3.5 SLUG TESTS

Slug tests were performed at the site between March 16 and March 21, 1990.

When possible, rising head tests were measured using the following technique:

- Initially, the static water level of the well was measured using an electronic water level indicator;
- The well's water level was then bailed as low as was physically possible, and remeasured;
- As the well recharged, it was continuously monitored at 15 second intervals with the water level indicator; and

- o Both the time and the water level were recorded until the water level had reached 95% recovery.

Rising head tests were not used for all the wells on site, due to the very large amounts of water produced by many of the wells. When a well produced a large volume of water, it became necessary to conduct falling head tests on the remaining wells. Falling head tests were initially carried out as follows:

- o First, the static water level was measured and recorded;
- o One or both of two stainless-steel rods (3.6 feet long with a 1.5-inch diameter, and 7.1 feet long with a 1.5 inch diameter) were dropped into the well; and
- o The new water level was then measured, and readings were recorded until the water level stabilized at or near the original static level.

When slug testing with the stainless-steel rods was inadequate due to either extremely shallow water in the wells or extremely large volumes of water in the wells, it became necessary to perform the slug tests differently. The tests were carried out the same as mentioned above, except instead of using stainless-steel rods, either 7.5 or 10.0 gallons of water were poured into the wells. The water was obtained from a fire hydrant, as was all other field/drill water used on site. Hydraulic conductivity for slug tests was determined by matching raw data with computer-driven type curves after Bouwer and Rice 1976. All data obtained during the aquifer testing are listed in Appendix C.

3.6 DECONTAMINATION

All decontamination conducted during the 3163 Buffalo Avenue RI was performed in accordance with NYSDEC-approved procedures. Sampling methods and equipment were chosen to minimize decontamination requirements and the possibility of cross contamination. Any sample tubing, rope, rods, etc., were dedicated equipment or disposed of after sampling. Sampling equipment used on more than one location was steam cleaned between locations or decontaminated in accordance with the following procedure:

- Low phosphate detergent wash;
- Tap water rinse;
- 10% nitric acid rinse (metals only);
- Tap water rinse;
- Acetone or methanol rinse (pesticide-grade);
- Hexane rinse (pesticide-grade);
- Deionized water rinse; and
- Air dry.

All protective clothing, drill cuttings, well purge water, and decontamination water was stored on site awaiting receipt of water and soil analytical results. Thereafter, they will be disposed of in an appropriate manner consistent with state and federal regulations.

3.7 QUALITY ASSURANCE/QUALITY CONTROL (QA/QC)

Throughout the environmental sampling program carried out at the 3163 Buffalo Avenue site, QA/QC samples were taken. In general, QA/QC samples are used to validate samples for the various sample media. They include:

- Field duplicates equivalent to 10% of all samples taken;
- Field and trip blanks equivalent to 10% of all samples taken; and
- MS/MSD samples.

Fieldwork and analytical work conducted for the RI met the major objectives outlined in the QA/QC Plan. These objectives include: deviation, accuracy, completeness, validity, representative sampling, and comparability. Due to the very high levels of chemicals in the various media at the site, a fair degree of difficulty was encountered when attempting to identify Tentatively Identified Compounds (TICs).

E & E followed the Contract Laboratory Program (CLP) protocols. Additionally, in the case of all 3163 Buffalo Avenue samples, attempts were made to identify non-Target Compound List (TCL) sample components at levels lower than that required by NYSDEC CLP protocols. In many cases identification of non-TCL compounds was prevented by the high concentration of TCL compounds present in the sample. Methodology requires that the sample be diluted to the point where it falls within the calibrated range of the instrument. Therefore, if non-TCL compounds were present at a much lower levels than the TCL compounds they would have been diluted to a level beyond detection.

Additionally attempts were made to identify the compounds specified at page 2-7 of the work plan by producing extracted ion profiles (EIPs) in some of the more contaminated samples. This process involved having the GC/MS data system reconstruct the sample analysis using information specific to those special interest compounds. Unfortunately, this process may have removed some of the other sample information that may have provided information regarding the presence or absence of non-TCL compounds.

In summary, all required analyses were conducted and all technically practical means were utilized to identify the specific compounds in accordance with required CLP protocols. However, because of the inherent limitations of CLP protocols, no additional compounds could be identified.

3.7.1 Sample Containers

The volumes and containers required for the environmental sampling activities are listed in Tables 3-6 and 3-7. Prewashed sample containers were provided by E & E's ASC and prepared in accordance with EPA bottle-washing procedures. These procedures are incorporated in E & E's Laboratory and Field Personnel Chain-of-Custody Documentation and Quality Assurance/Quality Control Procedures Manual, April 1987.

Table 3-6

SAMPLE CONTAINERS, VOLUMES, PRESERVATION, AND HOLDING TIMES FOR WATER SAMPLES

Type of Analysis	Type and Size of Container	Number of Containers and Sample Volume (per sample)	Preservation	Maximum Holding Time
Purgeable (Volatile) Organics	40-ml glass vial with Teflon-backed septum	Two (2); fill completely, no air space	Cool to 4°C (ice in cooler)	7 days
Extractable Organics, PCBs, Pesticides	1/2-gallon bottles with Teflon-lined caps	Two (2); total volume approx. 1 gallon; fill completely	Cool to 4°C (ice in cooler)	Must be extracted within 5 days; analyzed within 30 days
Metals	1-liter polyethylene bottle with polyethylene-lined caps	One (1); fill 7/8 full	Nitric acid to below pH 2 (approx. 1.5 ml Con HNO_3 per liter)	6 months
Cyanides	1-liter polyethylene bottle with polyethylene-lined caps	One (1); fill completely	Sodium hydroxide to pH 12 and cool to 4°C (ice in cooler)	14 days or 24 hours, if sulfide present

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Note: All sample bottles will be prepared in accordance with EPA bottle-washing procedures. These procedures are incorporated in E & E's Laboratory and Field Personnel Chain-of-Custody Documentation and Quality Assurance/Quality Control Procedures Manual, April 1987.

Table 3-7

SAMPLE CONTAINERS, VOLUMES, PRESERVATION, AND HOLDING TIMES FOR SOIL SAMPLES

Type of Analysis	Type and Size of Container	Number of Containers and Sample Volume (per sample)	Preservation	Maximum Holding Time
Purgeable (Volatile) Organics	40-ml glass vial with Teflon-backed septum	Two (2); fill completely, no air space	Cool to 4°C (ice in cooler)	10 days
Extractable Organics, PCBs, Pesticides	8-oz. glass jar with Teflon-lined cap	One (1); fill completely	Cool to 4°C (ice in cooler)	Must be extracted within 10 days; analyzed within 30 days
Metals	8-oz. glass jar with Teflon-lined cap	One (1); fill half-full	Cool to 4°C (ice in cooler)	6 months
Cyanides	8-oz. glass jar with Teflon-lined cap	One (1); fill completely	Cool to 4°C (ice in cooler)	14 days or 24 hours, if sulfide present
Total Organic Carbon	125-ml polyethylene	One (1); fill completely	Sulfuric acid to pH below 2 and cool to 4°C (ice in cooler)	7 days
Total Organic Halogen (TOX)	1-liter amber glass bottle	One (1); fill completely, no air space	Cool to 4°C (ice in cooler)	--

02[IL]JD1900:D3048/1177/20

Note: All sample bottles will be prepared in accordance with EPA bottle-washing procedures. These procedures are incorporated in E & E's Laboratory and Field Personnel Chain-of-Custody Documentation and Quality Assurance/Quality Control Procedures Manual, April 1987.

4. NATURE AND EXTENT OF CONTAMINATION

4.1 INTRODUCTION

The 3163 Buffalo Avenue site is situated in a heavily industrialized area. Olin Chemical Corporation is located to the west, and the DuPont chemical plant is located south and east of the site. From 1941 through 1945, DuPont designed, constructed and operated an Warfare Sevices Chemical plant on this site. The plant produced Impregnite during this time, after which it was put on standby. From 1951 to 1953, Hooker Electrochemical Company resumed manufacture of the product, and in 1953 the plant was once again put on standby. In 1972, the City of Niagara Falls purchased the site and resold it to the Solvent Chemical Company. From 1973 to 1978, the site was used for the production of chlorinated hydrocarbons, zinc chloride, and zinc ammonia chloride.

Phase I (1983) and Phase II (1985) investigations performed by Engineering Science, Inc., in conjunction with Dames & Moore, indicated that leaks and accidental spills from storage drums and tanks on site were probable sources of contamination. It was concluded that the potential for off-site contamination through groundwater migration existed, and in response to Phase II Investigation recommendations, the performance of the Remedial Investigation was initiated.

Previous to the 1983 and 1985 investigations, a hydrogeologic study of this site was conducted by Wehran Engineering (1980). Results of this study, which were substantiated by the subsequent Phase I and II investigations, revealed dichlorobenzene, benzene, and trichlorobenzene in the groundwater. These compounds are considered to have originated from former industrial activities at the site.

This section discusses the physiographic aspects that govern contaminant migration, including overburden and bedrock geology, surface hydrology, and hydrogeology. The concentrations and the aerial and vertical extent of selected compounds are discussed with respect to soil and groundwater. These parameters are then assimilated into an assessment of the fate and transport of those compounds which may pose a threat to human health or the environment.

4.2 SITE GEOLOGY

4.2.1 Introduction

The 3163 Buffalo Avenue property is underlain by a downward sequence of fill, glacial sediment, and sedimentary rocks of Silurian age. The bedrock encountered directly beneath the glacial sediment is the Lockport Dolomite, which is about 120 feet thick in the site area. All bedrock wells on site terminate in the upper to middle portion of the Lockport Dolomite, which is the Oak Orchard Member. Beneath the Lockport Dolomite lies the almost impermeable Rochester Shale.

The characterization of the on-site geology is necessary in order to determine the hydrogeology of the area, and thus evaluate local groundwater movement. Geologic and hydrogeologic assessments were based on results from the installation and hydrologic testing of 16 monitoring wells and the analysis of split-spoon and core samples taken from those wells. Phase I and II investigations, 1980 Wehran Engineering report, as well as the Woodward-Clyde Consultants' geohydrologic reports on the nearby DuPont facility were reviewed, and the pertinent data were assimilated into the RI's geologic/hydrogeologic interpretations.

4.2.2 Characteristics of the Overburden

The overburden on site is composed of a layer of fill over unconsolidated glacial sediment, extending to an average depth of 8.8 feet. The fill ranges from 1.5 to 3.5 feet in thickness, and is composed of dolomitic cobbles, broken roadbed, and industrial debris in a silty sand matrix. It is likely that pre-existing structures on the site were demolished, mixed with cinders and ash, and graded to the present surface level, creating the heterogeneity of fill material.

The glacial sediment encountered beneath the fill is native to the site, having been laid down by receding ice sheets and in proglacial lakes during the most recent (Pleistocene) glaciated interval in this area. These deposits vary in thickness from 6 to 25 feet regionally, and are observed on the site to be 2 to 9 feet thick. Thicknesses of original overburden material have been altered at 3163 Buffalo Avenue due to previous cut and fill operations. In its entirety, the overburden is called the A-zone, after the Woodward-Clyde system used at the DuPont plant.

Grading downward from sandy silt to sandy clay to a thick, massive, plastic clay, the glacial deposits show little variation in boreholes across the site. With the exception of monitoring well 7A, every overburden well terminates in saturated glacial clay. It can be assumed to extend across the entire site. These lake bottom sediments act as an aquitard to the Lockport Dolomite below.

4.2.3 Characteristics of the Bedrock

As noted above, the bedrock underlying the site is the Lockport Dolomite, which is composed of the following members: Oak Orchard, Eramosa, Goat Island, Gasport, and the basal DeCew Member. The Oak Orchard makes up the upper and middle portions of the Lockport Dolomite. All on-site bedrock wells terminate in this member. In areas adjacent to the Niagara Falls area, the Oak Orchard is capable of producing large volumes of water. The lower members of the Lockport Dolomite are less permeable than the upper and middle Lockport. Underlying the Gasport is the Rochester Shale of the Clinton and Albion groups, which is so impermeable that it acts as a confining bed to the sedimentary units below (see Section 2 for a discussion of regional bedrock characteristics).

The primary focus of this section is to investigate those portions of the overburden and the shallower bedrock that are capable of transmitting appreciable portions of groundwater and thus, soluble chemicals off site. The deepest well drilled into the Lockport Dolomite ended 98 feet below ground surface in a deep fracture zone of the Lockport. Four water-bearing fractures zones (B-Zone through F-Zone) have been identified by Woodward-Clyde Consultants in the Lockport Dolomite beneath the DuPont facility. In general, the fractures follow the bedding planes of the dolomite, with each zone having characteristic weathered and/or heavily fractured regions. Vertical and high angle fractures were observed to a lesser extent than the horizontal fracturing. However, these vertical fractures may provide hydraulic connection with the four fracture zones. For the purposes of this study, the Woodward-Clyde Consultants' nomenclature for the A-Zone to the F-Zone has been used at the Buffalo Avenue site. Descriptions of the water-bearing zones which follow are derived from drilling logs which appear in Appendix A.

A-Zone - Overburden material. See previous discussion of the overburden.

B-Zone - This zone includes the highly weathered top of bedrock, as well as multiple horizontal fracture horizons. High angle (60° to 75°) fractures are observed, often showing secondary mineralization (calcite), which is indicative of groundwater movement through these areas. Open fractures in borehole cores are noted at numerous depths in the interval, which span a depth of 6.3 feet (top of rock) to 21.5 feet. Two distinct highly fractured zones were observed between the approximate depths of 15 to 16 feet and 19 to 20 feet. Loss of drill water return between 20.35 feet and 20.5 feet implies well-connected fracturing in this region.

C-Zone - This interval extends from 26.5 feet to 36.5 feet BGS, exhibiting one vertical and one 45° fracture in the core. Horizontal bedding plane fractures are observed at numerous locations in the interval, with some secondary mineralization, and an area of drilling water loss between 32 and 33 feet.

CD-Zone - This zone (39.5 feet to 51.5 feet BGS) generally shows tight, horizontal bedding plane fractures with secondary calcite occurring along fractures from 40.1 feet to 42.9 feet. One large, open fracture in the core was observed at 36.7 feet to 36.8 feet with massive secondary calcite. Loss of drill water was observed between 40.8 feet and 41.0 feet.

F-Zone - This zone exhibits highly fractured segments and a weathered horizon. Horizontal bedding plane fractures are frequent. Solution features such as vugs and secondary calcium and gypsum are prevalent in the upper parts of the zone. Numerous large open fractures, as well as vertical fractures are observed throughout this interval (77 to 98 feet).

Fractures in all horizons are likely to be the result of the reduction of loading on the strata, which occurred as the glaciers made a final retreat through the area. Isostatic rebound, the strata's response to unloading, causes horizontal and vertical fracturing due to geologically rapid pressure release.

4.3 SITE HYDROGEOLOGY

4.3.1 Introduction

Groundwater occurs in both the unconsolidated overburden and in the bedrock beneath the 3163 Buffalo Avenue site. In this investigation, Zone A refers to the water-bearing region within the overburden, and Zones B, C, CD, and F refer to the water-bearing (fractured) regions of the Lockport Dolomite. In order to characterize the site hydrogeology, a total of seven wells were drilled into the overburden; six into zone B; and one each in Zones C, CD, and F (Table 4-1). Wells from the neighboring DuPont facility were surveyed for comparison data in zones C, CD, and F.

Regionally, groundwater flow direction is to the northwest, from the upper Niagara River toward the lower Niagara River, discharging in the Niagara Gorge. East of Gill Creek, the upper Niagara River serves as recharge to the Lockport Dolomite, and to a lesser degree, to the overburden in this area. The regional discharge effect of the Niagara Gorge is due to its lower hydraulic (elevation) head.

Locally, two man-made features influence groundwater flow. The Falls Street Tunnel, due north of the site, and the New York State Power Authority's (NYPA) intake tunnels, to the east, act as groundwater sinks. The drains surrounding the tunnels cause a line sink with a resultant decrease in hydraulic head along the length of the tunnels. The effect of the Falls Street Tunnel is to accentuate local groundwater gradients, increasing northerly flow. The NYPA's intake tunnels cause a local divergence of groundwater flow to the northeast, which was noted in Zones C and F on site. Wells at the neighboring DuPont plant were observed to have been affected by the NYPA's intake tunnels in Zones B through F.

The extrapolation of groundwater elevation data indicate that Gill Creek, which discharges south to the Niagara River and to the west of 3163 Buffalo Avenue, may be a discharge boundary for both the overburden and Zone B of the Lockport Dolomite. However, as previously noted by Woodward-Clyde Consultants, Gill Creek appears to be a recharge boundary

Table 4-1

GROUNDWATER ELEVATIONS IN MONITORING WELLS

Number	Inner Casing Elevation (feet)	Total Depth (feet)	Screened or Open Interval (feet)	Depth to Water (February)	Groundwater Elevation (February)	Depth to Water (May)	Groundwater Elevation (May)
MW1A	571.81	6.0	4.0 - 6.0	5.05	566.76	5.2	566.61
MW1B	572.63	17.15	6.3 - 17.15	11.49	561.14	11.8	560.83
MW2A	571.11	8.2	4.0 - 8.2	3.11	568.0	3.2	567.91
MW2B	571.60	20.0	8.2 - 20.0	9.48	562.12	9.3	562.3
MW3A	573.17	8.5	4.5 - 8.5	6.42	566.75	6.53	566.6
MW3B	572.44	21.2	9.6 - 21.2	15.94	556.5	15.96	556.5
MW4A	570.83	8.0	3.0 - 8.0	4.11	566.72	4.65	566.2
MW4B	570.54	22.5	8.0 - 22.5	10.70	559.84	11.05	559.5
MW5A	573.22	12.5	4.5 - 10.5	7.84	565.38	8.61	564.6
MW5B	573.28	21.0	12.5 - 21.0	17.47	555.81	17.67	555.6
MW5C	571.71	36.5	26.5 - 36.5	18.82	552.89	--	--
MW5C-D	571.29	42.9	39.5 - 42.9	22.93	548.78	22.23	549.06
MW5F	572.48	98.0	77.0 - 98.0	16.54	555.94	14.35	558.13
MW6A	574.02	9.6	3.5 - 9.6	5.86	568.16	5.97	568.05
MW6B	571.93	20.0	9.6 - 20.0	9.78	562.15	8.65	563.28
MW7A	571.84	7.5	2.5 - 7.5	4.35	567.49	4.68	567.16

-- Indicates water level not available.

for the overburden and bedrock in the vicinity of the east DuPont plant. Groundwater from the site will ultimately discharge into the Niagara River by some means. Most groundwater discharges via the Fall Street Tunnel/NYPA conduit system or from flow through bedrock. However, based on the extrapolation of site data, it is possible for some groundwater to discharge to the Niagara River via Gill Creek.

Monitoring well elevations were measured with an electronic water level indicator in February and May 1990 (see Table 4-1). These gauging events revealed minor seasonal groundwater flow variations including slightly steeper May groundwater gradients and a relative decrease in elevation head at several of the monitoring wells. These differences are discussed further in the following sections.

The gauging events indicated that groundwater flow from the site in both seasons is primarily to the north and west. Wells screened at different zones at the same location indicate decreasing groundwater elevation with increasing depth below the surface for Zones A thorough C-D (i.e., a downward vertical hydraulic gradient). Wells screened in the F-Zone display a greater groundwater elevation than D-Zone screened wells indicating an upward vertical gradient at that horizon.

4.3.2 Hydrogeology of the Overburden

The saturated thickness in the overburden is an average 5.2 feet, with groundwater occurring in both fill material and glacial sediments. The hydraulic conductivities obtained from slug testing in this zone result from the combined effect of the properties of both materials. Wells 1A through 6A, which intersect both fill and glacial sediment, exhibit hydraulic conductivities of 0.4 to 2.5 gpd/ft^2. Well 7A, which terminates in fill material (above the glacial deposits), shows a hydraulic conductivity of 18.63 gpd/ft^2 (see Table 4-2). This disparity between conductivity values has an obvious influence on the average hydraulic conductivity for the entire overburden (4.8 gpd/ft^2), being heavily weighted by the fill.

The saturated glacial sediment in this regime functions as an aquitard, capable of storing groundwater but not transmitting it quickly. This is evidenced by the results of the well development procedure, in which four of the six wells intersecting the sediment were

Table 4-2

AQUIFER TESTING SUMMARY TABLE
3163 BUFFALO AVENUE

				Bouwer-Rice Method	
Well	Aq.Type	Screen (ft)	Water Level (ft)	Hydr. Cond (gal/day/ft^2)	Transmissivity (gal/day/ft)
Shallow Wells					
MW1A	Unconfined	4.0 - 6.0	5.16	1.4	2.81
MW2A	Unconfined	4.0 - 8.2	7.06	1.6	8.05
MW3A	Unconfined	4.5 - 8.5	6.12	1.2	7.12
MW4A	Unconfined	4.0 - 8.0	4.2	0.4	1.97
MW5A	Unconfined	6.0 - 12.5	7.87	2.5	18.96
MW6A	Unconfined	4.0 - 9.6	5.93	1.26	7.08
MW7A	Unconfined	3.5 - 7.5	4.49	18.63	93.17
Intermediate Wells					
MW1B	Unconfined	6.3 - 17.15	11.57	38.45	221.9
MW2B	Unconfined	8.2 - 20.0	9.24	3.75	45.02
MW3B	Unconfined	9.6 - 21.2	16.05	0.8	9.34
MW4B	Unconfined	8.0 - 22.5	10.62	--*	--*
MW5B	Unconfined	12.5 - 21.0	18.04	1.04	8.82
MW6B	Unconfined	9.6 - 20.0	9.6	8.72	90.69
Deep Wells					
MW5C	Unconfined	26.5 - 36.5	19.32	19.17	191.72
MW5CD	Unconfined	39.5 - 51.5	23.83	24.77	297.24
MW5F	Unconfined	77.0 - 98.0	17.92	11.31	237.51

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*Inadequate data.

Testing Dates: March 16, 1990 - March 21, 1990.

Average Transmissivities:

Shallow Wells: 19.88 gpd/ft (heavily weighted by T value for SC7A)
Intermediate wells: 75.15 gpd/ft (heavily weighted by T value for SC1B)
Deep Wells: 242.16 gpd/ft

bailed dry. Compounds will percolate rapidly through the highly conductive fill, remaining longer in the saturated glacial sediment.

Seasonal fluctuation in groundwater elevation is not pronounced. Individual wells in this zone showed differences of only 0.09 to 0.77 feet from February to May (Table 4-1). The average lateral hydraulic gradient varies only slightly from 0.0189 ft/ft in February to 0.021 ft/ft in May, as expected from the small changes observed in water level data. The direction of flow is to the north and west, with all six wells exhibiting a downward vertical gradient with respect to B-Zone wells (see Figures 4-1 through 4-4).

Water level elevations were measured with an electronic water level indicator. High contaminant concentrations in the site groundwater caused the water level indicator to fail during the February 6, 1990 gauging event. The February gauging event was completed on February 9, 1990. Data shown on Figures 4-1 and 4-3 was collected on either February 6 or February 9, 1990. The general groundwater trends observed during the February gauging events are very similar to the groundwater trends observed during the May gauging event; therefore, the February data was used to prepare Figures 4-1 and 4-3.

Particle size analyses performed on samples from Wells MW-1A, MW-5A, and MW-7A provided a second means of estimation for hydraulic conductivity values. After each sediment sample was classified according to its particle size distribution, standard hydraulic conductivities from literature were applied and compared to field tests results.

Well MW-1A yielded medium to fine sand sized sediment and a corresponding range of 21.2 to 0.212 gpd/ft^2 for hydraulic conductivity. These values are in agreement with the slug test result of 1.4 gpd/ft^2. Well 7A produced coarse gravel to medium sand sized fragments, with a hydraulic conductivity range of 2,120 gpd/ft^2 to 21.2 gpd/ft^2. The slug test result of 19.93 gpd/ft^2 falls near the lower range of this estimation (conductivity estimations based on Freeze & Cherry 1979).

Groundwater velocities were estimated for the A horizon using effective porosity values of 0.01 and 0.1 (Table 4-3). For the A-Zone, February velocities range from 0.122 ft/day with a maximum effective porosity of 0.1, to 1.22 ft/day using an effective porosity of 0.01. May velocities vary from 0.135 ft/day (effective porosity = 0.1) to 1.35

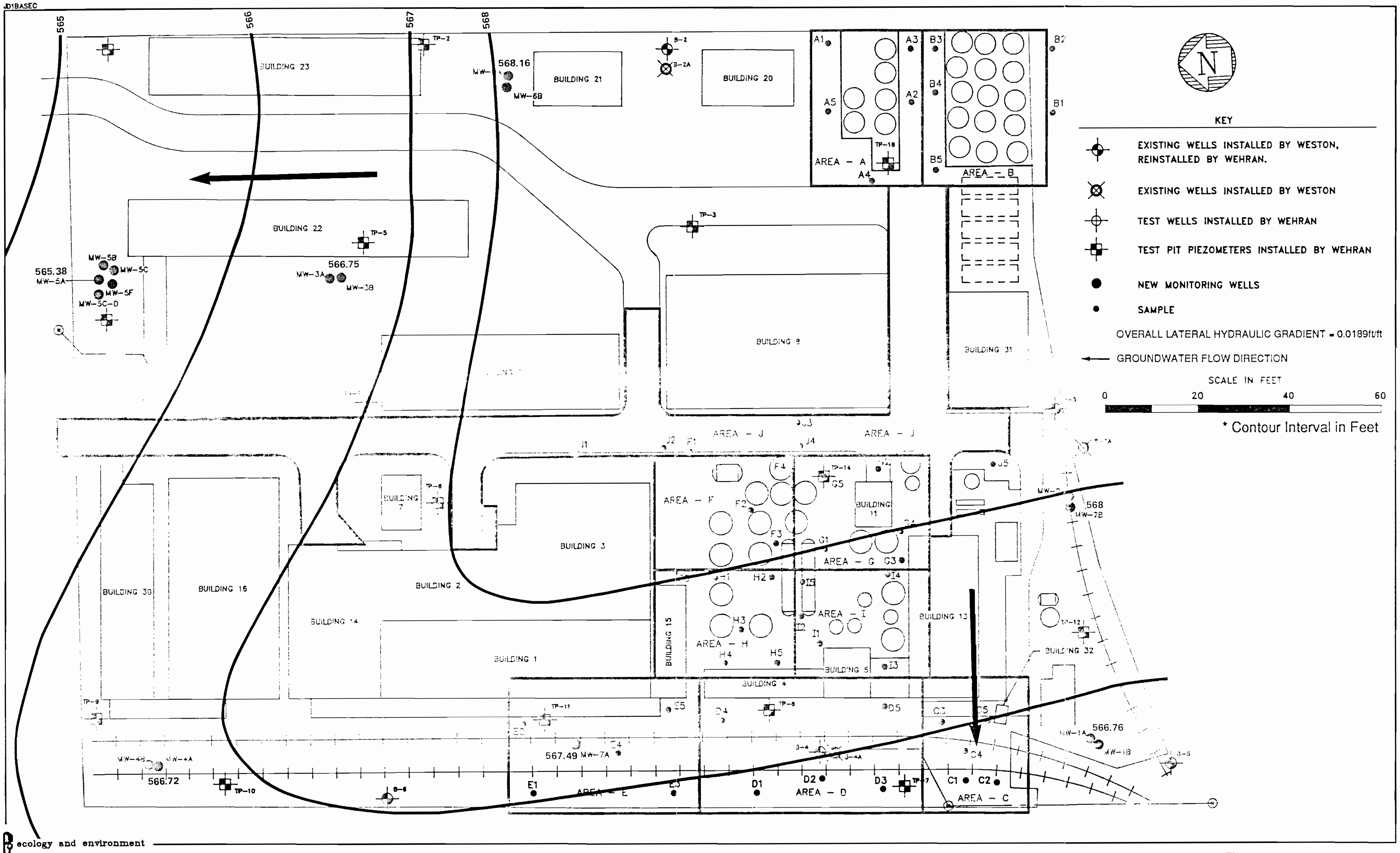


Figure 4-1
GROUNDWATER ELEVATIONS IN A ZONE
(READINGS TAKEN 2/6/90 AND 2/9/90)

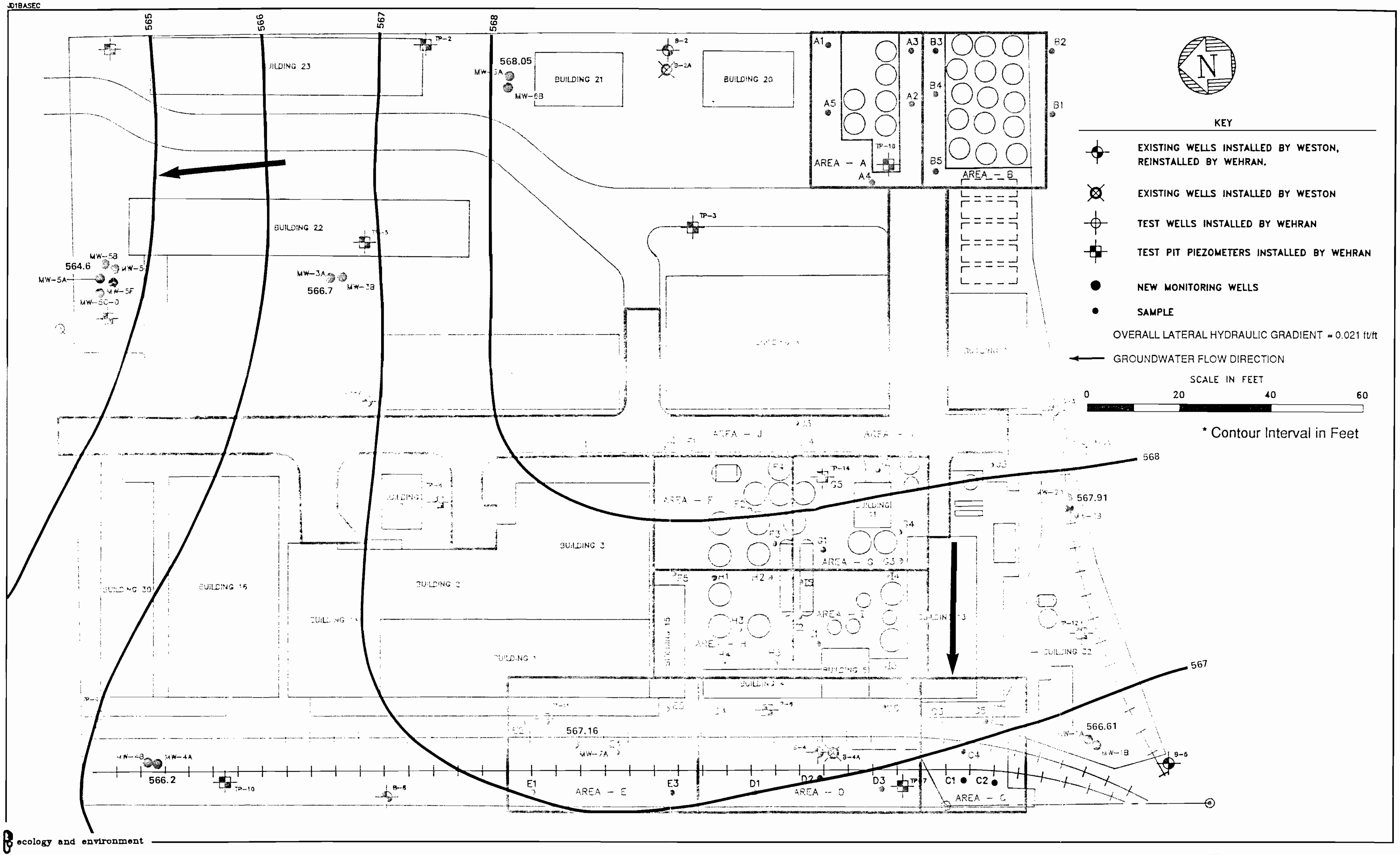


Figure 4-2
GROUNDWATER ELEVATIONS IN A ZONE
(READINGS TAKEN 5/15/90)

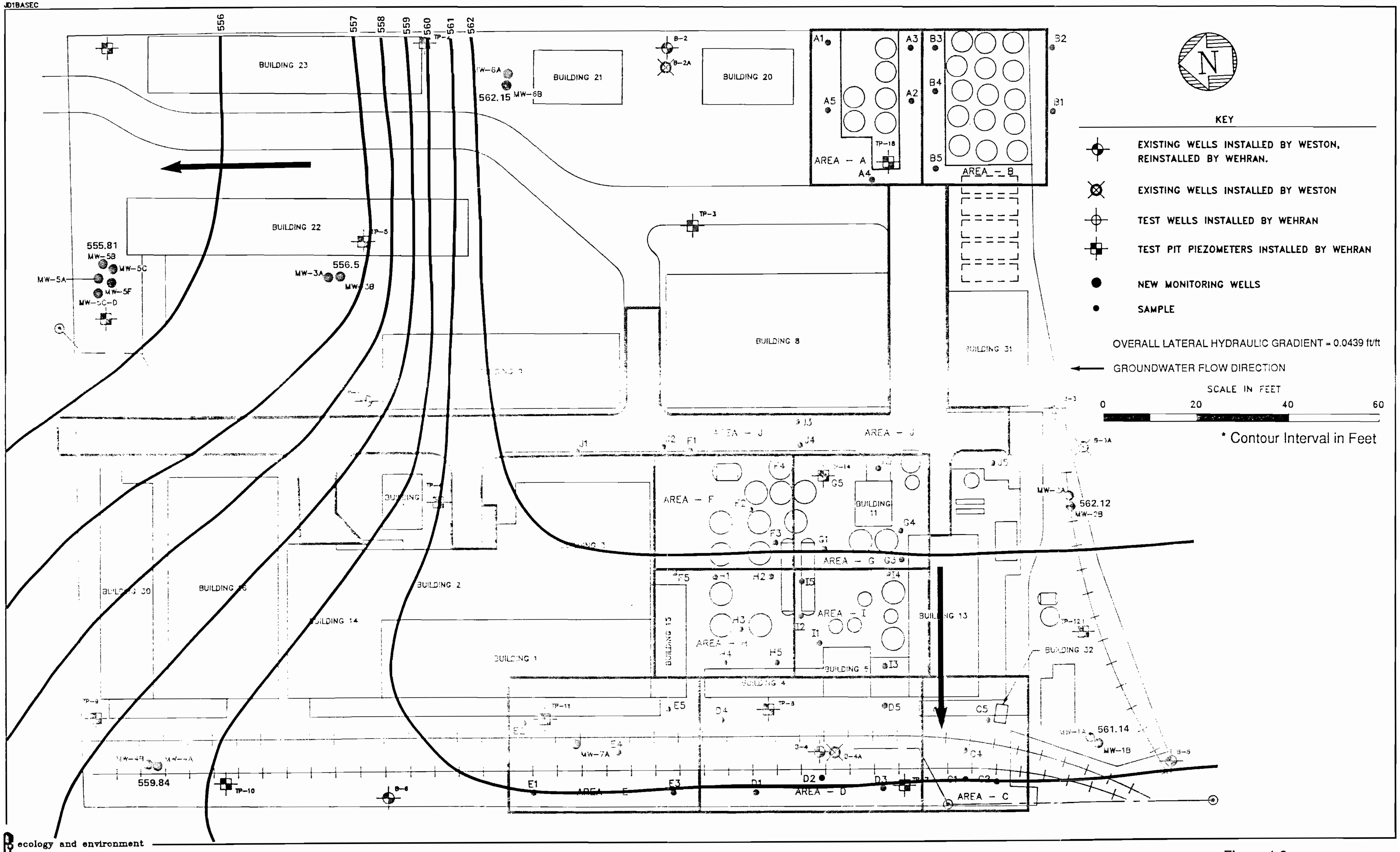


Figure 4-3
GROUNDWATER ELEVATIONS IN B ZONE
(READINGS TAKEN 2/6/90 AND 2/9/90)

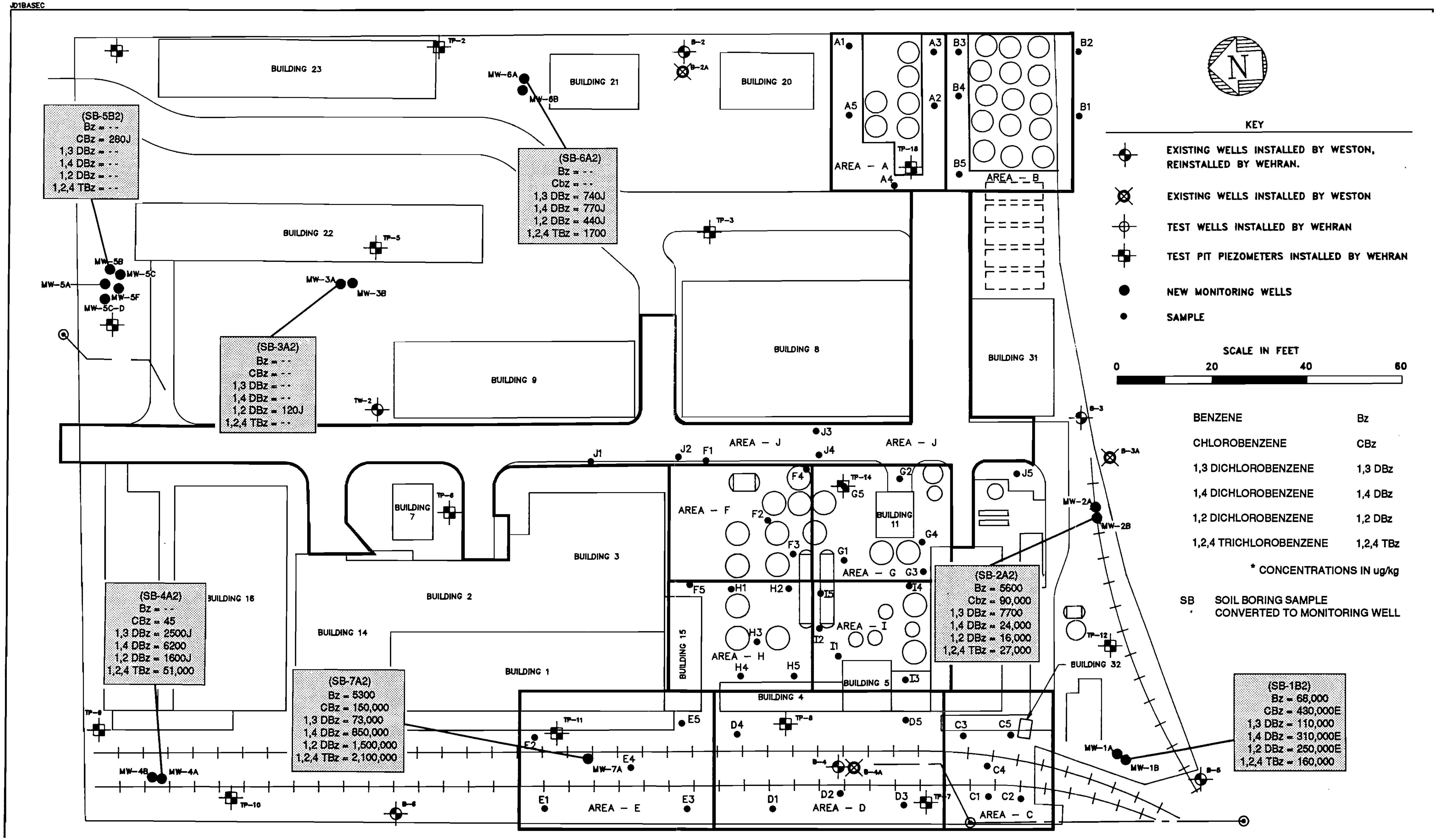


Figure 4-5
SIGNIFICANT VOLATILES AND SEMIVOLATILES IN SUBSURFACE SOIL (2.0' TO 4.0')

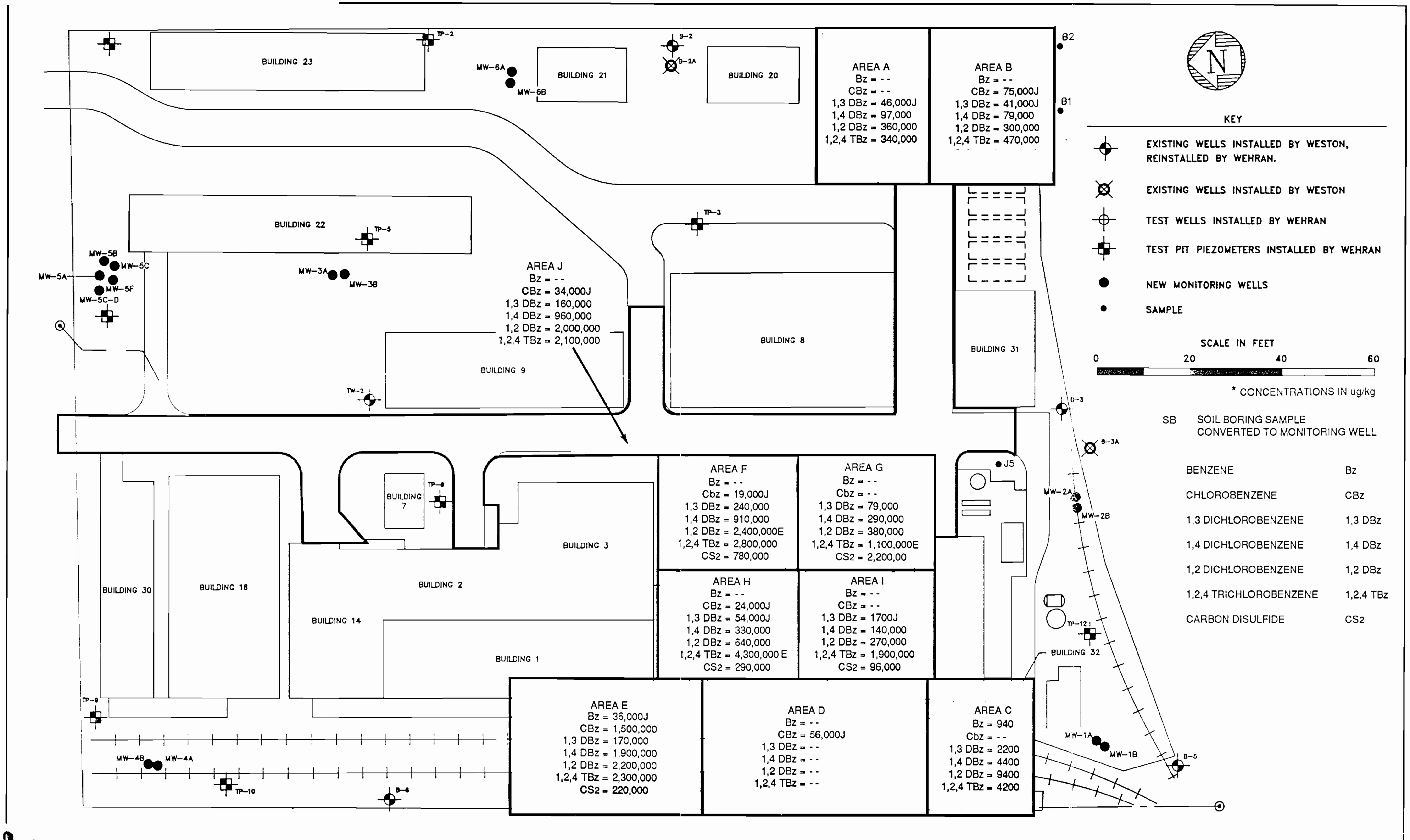


Figure 4-7
SIGNIFICANT VOLATILES AND SEMIVOLATILES
IN SUBSURFACE COMPOSITE SOIL SAMPLES
(2.0' TO 4.0')

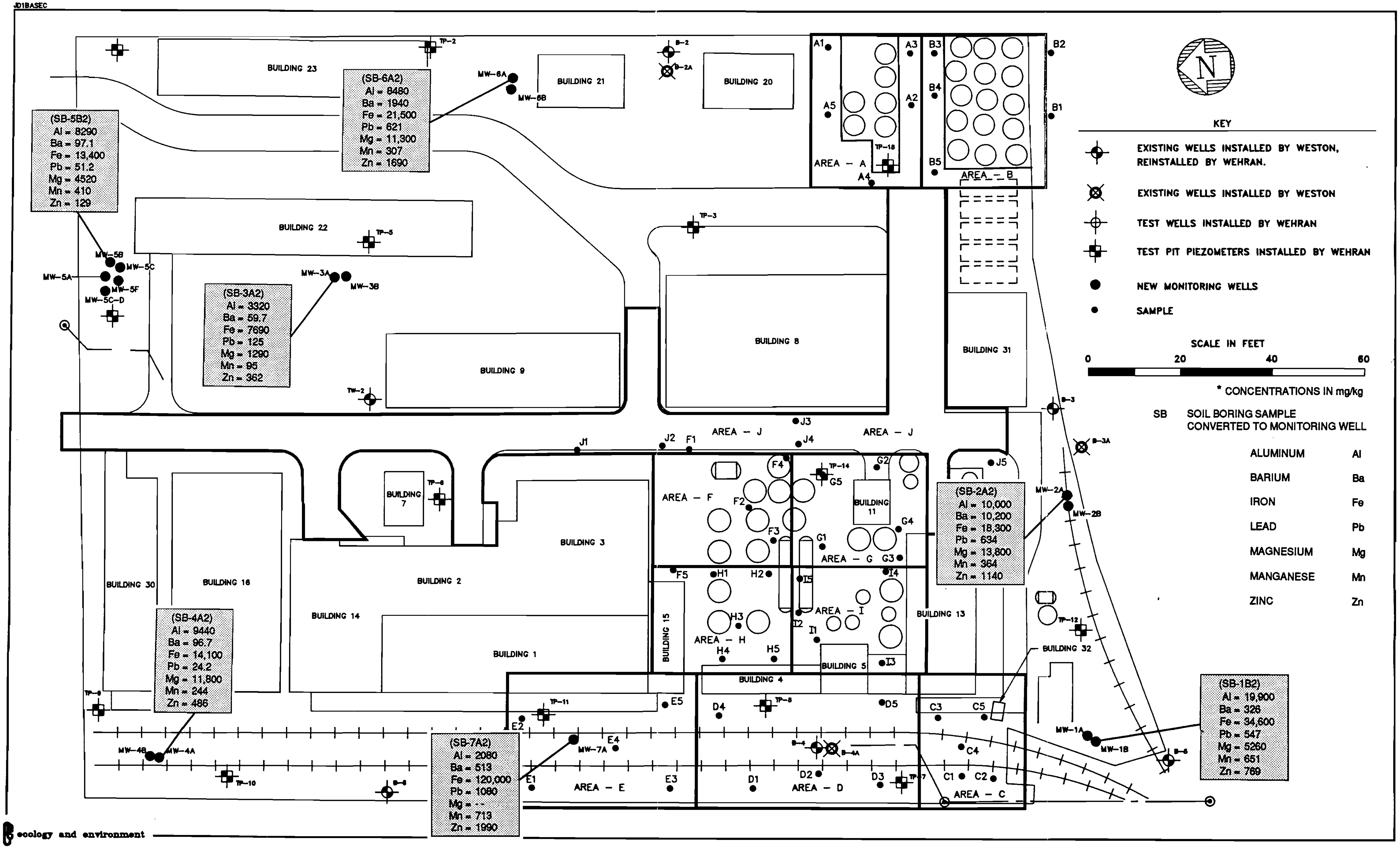
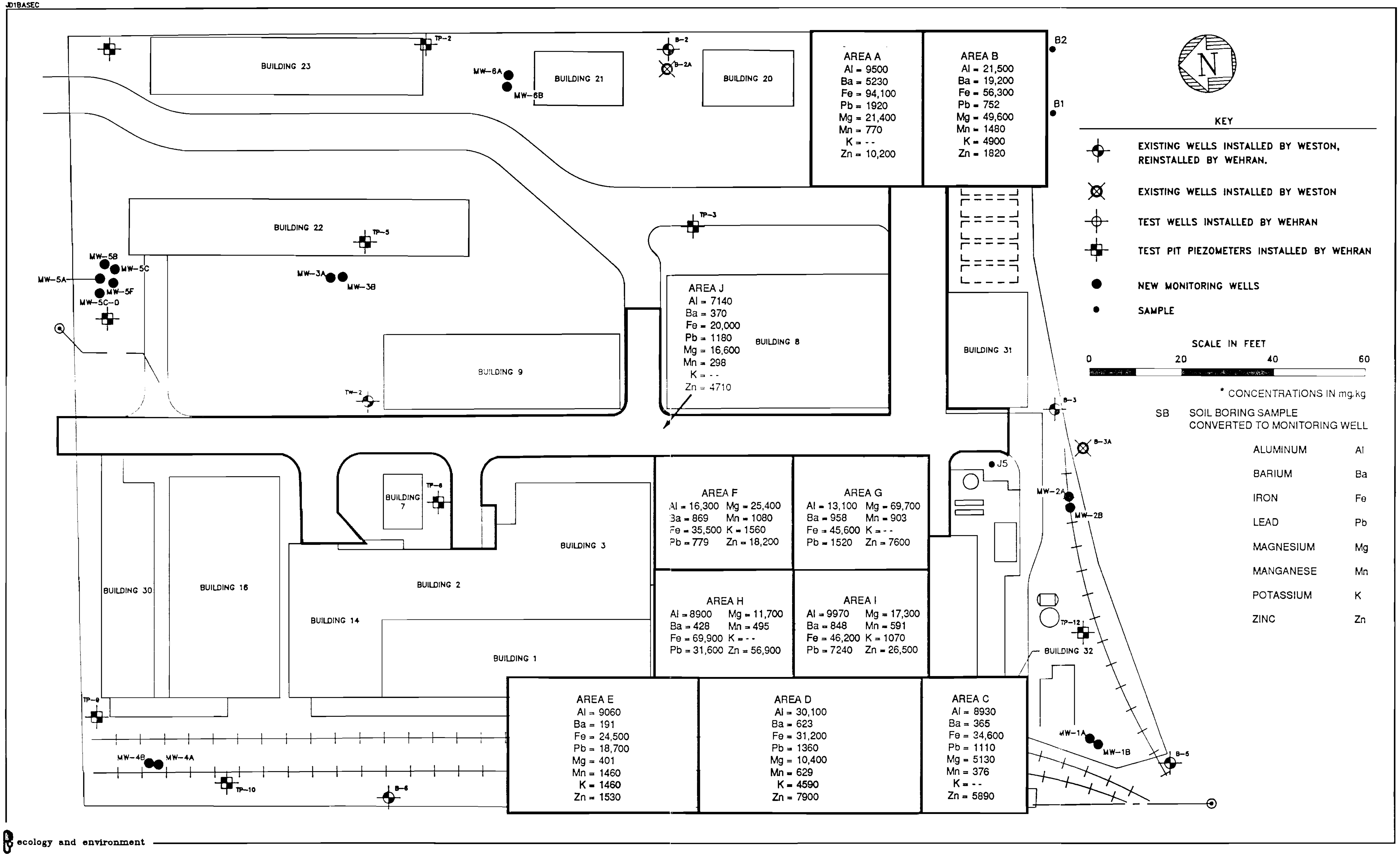


Figure 4-8
SIGNIFICANT METALS IN SUBSURFACE SOILS (2.0' TO 4.0')



ecology and environment

Figure 4-10
SIGNIFICANT METALS IN SUBSURFACE
COMPOSITE SOIL SAMPLES (2.0' TO 4.0')

ft/day (effective porosity = 0.01). Groundwater velocities for this zone most likely approach the estimations for a net effective porosity of 0.1, which are 0.122 ft/day and 0.135 ft/day.

4.3.3 Hydrogeology of the Bedrock

The bedrock underneath the 3163 Buffalo Avenue site includes water-bearing (fracture) Zones B, C, CD, and F. Hydraulic properties vary with the differing geologic characteristics of each zone. Each horizon will therefore, be discussed individually, and an overview of the bedrock hydrology regime will then be provided.

B-Zone - This zone includes the highly weathered top of bedrock, and is situated directly beneath the glacial aquitard. The average hydraulic conductivity is 10.55 gpd/ft^2, which is heavily weighted by a high value of 38.45 gpd/ft^2 for Well 1B (see Table 4-2). The average saturated thickness is 10.46 feet.

The transmissivity is an average 75.09 gpd/ft for this zone, which, when compared to the overburden average (excluding Well 7A), is an order of magnitude more transmissive. Compounds that reach groundwater in this zone are much more likely to be able to be transported effectively off site than compounds in the overburden.

Seasonal variation in groundwater elevation exhibited an increase in two wells from February to May, with an overall average decrease of only 0.35 feet across the site. Lateral hydraulic gradient values varied from 0.0439 in February to 0.0614 in May. The direction of flow is to the north and west (see Figures 4-3 and 4-4).

February groundwater velocities range from 66.4 ft/day using an effective porosity of 0.001, to 0.66 ft/day using an effective porosity of 0.1. May velocities vary from 92.9 ft/day to 0.93 ft/day, using effective porosity values of 0.001 to 0.1, respectively. Most likely, the effective porosity of the B-Zone is about 0.1, producing a groundwater velocity in the range of 0.5 to 1 ft/day.

C-Zone - One monitoring well was installed in each of the C-, CD-, and F-Zones. Gradients were calculated with data from the 3163 Buffalo Avenue wells and the nearby DuPont facility wells. Hydraulic conductivity values are reported for 3163 Buffalo Avenue wells only.

Table 4-3

GROUNDWATER VELOCITIES

Zone	Month	Effective Porosity	Velocity (Ft/D)
A	February	0.01	1.22
	February	0.1	0.12
	May	0.01	1.35
	May	0.1	0.14
B	February	0.001	66.41
	February	0.01	6.64
	February	0.1	0.66
	May	0.001	92.9
	May	0.01	9.3
	May	0.1	0.93
C	March	0.001	64.03
	March	0.01	6.4
	March	0.1	0.64
CD	May	0.001	123.06
	May	0.01	12.31
	May	0.1	1.23
F	May	0.001	228.98
	May	0.01	22.9
	May	0.1	2.3

02[IL]JD1900:D3048/1190/33

The C-Zone is the second water-bearing (fracture) zone encountered in the Lockport Dolomite. It has a saturated thickness of 10 feet, and a hydraulic conductivity of 19.17 gpd/ft^2. Seasonal fluctuation in groundwater elevation data were not available for this zone. The lateral hydraulic gradient in March was 0.0255 ft/ft. In conjunction with DuPont's Well 18, this zone demonstrates a northeasterly flow direction. Velocities for this zone vary from 64.03 ft/day to 0.64 ft/day, using effective porosity values of 0.001 to 0.1. Using a net effective porosity of 0.01, the most likely groundwater velocity would be about 6 ft/day.

CD-Zone - This zone has a saturated thickness of 12 feet and a hydraulic conductivity of 24.77 gpd/ft^2. The groundwater elevation increased from February to May by 0.07 foot, and displayed a lateral hydraulic gradient in May of 0.0228 ft/ft. Using DuPont Well 10D with Well MW-5CD, the groundwater flow direction is to the northwest, with a May velocity of 123.06 ft/day to 1.23 ft/day. Effective porosity values used for this calculation ranged from 0.001 to 0.1. The most likely effective porosity of about 0.01 would produce a groundwater velocity of about 12 ft/day.

F-Zone - The deepest zone that was intersected by a monitoring well, this horizon has a saturated thicknesses of 21 feet. The hydraulic conductivity value obtained from slug testing was 11.31 gpd/ft.^{2}. The seasonal variation in groundwater elevation showed an increase of 2.19 feet from February to May. The lateral hydraulic gradient in May was 0.048 ft/ft with a northeasterly direction of flow. Velocities in May ranged from 228.98 ft/day to 2.3 ft/day using an effective porosity range of 0.001 to 0.1. The best estimate for hydraulic conductivity of 22.9 ft/day results from using an effective porosity of 0.01.

At the 3163 Buffalo Avenue site, the Lockport Dolomite appears to exhibit a general increase in hydraulic conductivity and groundwater velocity downward. In the monitoring well nest 5, a downward vertical hydraulic gradient is seen from Zones B through CD. F-Zone reverses this trend, having an upward vertical hydraulic gradient and maintaining a potentiometric surface equal to that in the B-Zone. Consequently, chemicals in solution in the Lockport Dolomite would be transported horizontally to the north and west and downward until encountering the F-Zone.

4.3.4 Conclusions

The Buffalo Avenue site hydrogeology can be summarized by the following characteristics:

- The direction of groundwater flow in the overburden and bedrock is to the north, northwest, west and northeast. The predominance of flow, however, is to the north and northwest. This suggests that the primary groundwater discharge for the site area is the Niagara Gorge. However, groundwater may also flow into the Fall Street Tunnel, which discharges to the Niagara Gorge near the Rainbow Bridge;
- Groundwater flow in the C- and CD-Zones must be regarded as speculative because only two wells (one DuPont and one Buffalo Avenue) were available to determine the direction of groundwater flow in this zone. (At least three wells are necessary to estimate flow in a particular zone);
- The overburden has an average 5.16 feet of saturated thickness and is composed of highly conductive fill overlying saturated, low conductivity glacial deposits;
- The glacial sediment constitutes an aquitard, and thus should reduce migration rates through the overburden;
- The Lockport Dolomite at 3163 Buffalo Avenue has four important water-bearing fracture zones, with groundwater velocity and hydraulic conductivity values increasing with depth;
- Groundwater flow through bedrock is primarily through bedding plane fractures, but vertical and high angle fractures provide hydraulic communication between zones;
- The horizontal gradient within the A- through F-Zones varies seasonally with groundwater elevations, as does groundwater velocity; and
- The vertical hydraulic gradient is downward from A-Zone through CD-Zone, and reverses upward from F-Zone.

4.4 SITE HYDROLOGY

The 3163 Buffalo Avenue site is relatively flat, having only a 0.1% southwest slope toward Gill Creek. Although this grade does not encourage surface water run-off, local receptors may include Gill Creek to the west, and the Niagara River to the south. A road embankment separates the site from the upper Niagara River. Gill Creek is a south-flowing tributary to the Niagara River, and receives run-off all along its course. Surface water sampling was not considered in this investigation.

Groundwater elevation data suggest local discharge to Gill Creek may occur, and regional discharge to the lower Niagara River in the gorge area is likely. The upper Niagara River serves as a recharge boundary for the overburden and the B-Zone of the Lockport Dolomite in this regime. Regardless of its local course, groundwater will eventually flow to the Niagara River, and discharge into Lake Ontario.

4.5 OBSERVED ZONES OF CONTAMINATION

4.5.1 Introduction

Chlorinated aliphatic hydrocarbons, monocyclic aromatics and various metals were detected in both the soil and the groundwater at the 3163 Buffalo Avenue site. To a lesser extent, carbon disulfide and cyanide were also detected in overburden soils. Carbon disulfide was also found in one NAPL sample. The areas of highest concentration are located in the southwest corner. In these areas, the monocyclic aromatics were the most abundant compounds found with greatest detection in surface soils and the groundwater of the B-Zone of the Lockport Dolomite.

Of the monocyclic aromatics, benzene (unchlorinated), chlorobenzene, 1,2-dichlorobenzene, 1,3-dichlorobenzene, 1,4-dichlorobenzene, and 1,2,4-trichlorobenzene (all chlorinated) had the most significant concentrations and were the most ubiquitous to the site. Aluminum, iron, and magnesium were among the metals present in high concentrations.

4.5.2 Contamination in the Unsaturated Zone

The primary method of movement of compounds in the unsaturated zone is downward percolation. Lateral movement of compounds in this horizon is generally negligible. Once the compound has reached the saturated zone, however, lateral migration due to groundwater movement occurs according to the hydrogeologic nature of the saturated material. Thus, contaminants originating at the surface of this site move through the unsaturated zone to the saturated zone, and then migrate off site in groundwater.

Three types of soil samples were taken at the site: discrete samples from 2 to 4 feet deep in each monitoring well nest; composite soil to bedrock in each well nest; and subsurface soil area composites in Areas A-J. With the exception of MW-2A, the 2- to 4-foot samples were taken within the unsaturated zone and act as indicators of disposal practices during facility operations. Composite samples are designed to provide a spectrum of contaminants over the entire overburden interval. However, actual contaminant concentrations may be decreased, particularly for volatile organics, due to the nature of sample compositing. Sampling data for all soil samples appear in Tables 4-4 through 4-13 and in Figures 4-5 through 4-10.

In soils taken from monitoring wells at 2- to 4-foot depths, significant concentrations of benzene, chlorobenzene, 1,3-dichlorobenzene, 1,4-dichlorobenzene, 1,2-dichlorobenzene, as well as 1,2,4-trichlorobenzene were detected in southwest corner wells MW-1B, MW-2A, and MW-7A (see Figure 4-5 and Table 4-4). The highest level of benzene detected in these samples was 68,000 µg/kg from MW-1B. Other examples of high concentrations include 430,000 µg/kg of chlorobenzene in MW-1B; 1,500,000 µg/kg of 1,2-dichlorobenzene in MW-7A; and 2,100,000 µg/kg of 1,2,4-trichlorobenzene in MW-7A (see Figure 4-5 and Tables 4-4 and 4-5). Contamination in composite bedrock samples is also highest in southwest portion of the site (see Figure 4-6 and Tables 4-6 and 4-7). Notable examples of high concentration of volatiles and semivolatiles are as follows: levels of benzene of up to 120,000 µg/kg (MW-1B); levels of chlorobenzene of up to 480,000 µg/kg (MW-7A); and levels of 1,2,4-trichlorobenzene of up to 730,000 µg/kg (MW-7A). Levels of

Table 4-4

VOLATILE ORGANICS IN OVERBURDEN SOILS
(2.0' - 4.0' MONITORING WELLS)

Results in µg/kg (ppb)

Compound	SB1B2	SB2A2	SB3A2	SB4A2	SB5B2	SB6A2	SB7A2
Vinyl Chloride	--	--	--	--	--	--	--
Total 1,2-Dichloroethene	--	--	--	--	--	--	--
1,2-Dichloroethane	--	--	--	--	--	--	--
Trichloroethene	--	--	--	--	--	--	--
1,1,2-Trichloroethane	--	--	--	5J	--	--	--
Benzene	68,000	5,600	--	--	--	--	5,300
Tetrachloroethene	--	--	--	--	--	--	--
Toluene	--	--	--	--	280J	--	--
Chlorobenzene	430,000E	90,000	--	45	280J	--	150,000
Total Xylenes	--	--	--	4J	--	1,100J	--

02[IL]JD1900:D3048/1191/23

E = Exceeds quantifiable range; estimated
J = Below quantifiable range; estimated
-- Below detectable limits

Table 4-5

SEMIVOLATILE ORGANICS IN OVERBURDEN SOILS
(2.0' - 4.0' MONITORING WELLS)

Results in µg/kg (ppb)

Compound	SB1B2	SB2A2	SB3A2	SB4A2	SB5B2	SB6A2	SB7A2
Phenol	6,100	--	--	--	--	180^{J}	--
1,3 Dichlorobenzene	110,000	7,700	--	2,500^{J}	--	740^{J}	73,000
1,4 Dichlorobenzene	310,000^{E}	24,000	--	6,200	--	770^{J}	850,000
1,2 Dichlorobenzene	250,000^{E}	16,000	120^{J}	1,600^{J}	--	440^{J}	1,500,000
1,2,4 Trichlorobenzene	160,000	27,000	--	51,000	--	1,700	2,100,000
Hexachlorobutadiene	98^{J}	350^{J}	--	--	--	--	--
2-Chlorophenol	850^{J}	--	--	--	--	--	--
4-Methylphenol	--	--	--	--	--	62^{J}	--
Isophorone	--	--	3,100	--	--	--	--
Naphthalene	2,600	420^{J}	--	--	--	510^{J}	930^{J}
2-Methylnaphthalene	2,300	400^{J}	--	--	--	180^{J}	--
Acenaphthylene	71^{J}	83^{J}	--	--	--	44^{J}	--

02[IL]JD1900:D3048/1236/15

E = Exceeds quantifiable range; estimated
J = Below quantifiable range; estimated
-- Below detectable limits

Table 4-6

VOLATILE ORGANICS IN OVERBURDEN SOILS
(MONITORING WELL COMPOSITE SAMPLES)

Results in µg/kg (ppb)

Compound	SB1BC	SB2AC	SB3AC	SB4AC	SB5BC	SB6AC	SB7AC
Vinyl Chloride	--	--	--	--	--	--	--
Total 1,2-Dichloroethene	--	--	--	--	--	--	--
1,2-Dichloroethane	--	--	--	--	--	--	--
Trichloroethene	--	--	--	--	260^J	--	--
Benzene	120,000	50,000	--	--	--	--	11,000
Tetrachloroethene	--	--	--	--	--	--	--
Toluene	--	--	--	--	290^J	--	--
Chlorobenzene	280,000	190,000	--	--	--	560^J	480,000
Total Xylenes	--	--	--	--	--	210^J	--

02[IL]JD1900:D3048/1193/23

E = Exceeds quantifiable range; estimated
J = Below quantifiable range; estimated
-- Below detectable limits

Table 4-7

SEMIVOLATILE ORGANICS IN OVERBURDEN SOILS
(MONITORING WELL COMPOSITE SAMPLES)

Results in µg/kg (ppb)

Compound	SB1BC	SB2AC	SB3AC	SB4AC	SB5BC	SB6AC	SB7AC
Phenol	$2,200^J$	360^J	210^J	--	--	190^J	--
1,3-Dichlorobenzene	39,000	15,000	60^J	600^J	--	180^J	$36,000^J$
1,4-Dichlorobenzene	150,000	65,000	160^J	$2,700^J$	--	200^J	500,000
1,2-Dichlorobenzene	98,000	61,000	1,200	32,000	--	110^J	650,000
1,2,4-Trichlorobenzene	72,000	$52,000^E$	200^J	340,000	--	480^J	730,000
2-Chlorophenol	--	290^J	--	--	--	--	--
4-Methylphenol	--	--	75^J	--	--	--	--
Isophorone	--	--	--	--	100^J	--	--
2,4-Dimethylphenol	--	--	64^J	--	--	--	--
Naphthalene	--	180^J	--	920^J	--	100^J	--
2-Methylnaphthalene	--	430^J	--	--	--	--	--

02[IL]JD1900:D3048/1196/21

E = Exceeds quantifiable range; estimated
J = Below quantifiable range; estimated
-- Below detectable limits

Table 4-8

VOLATILE ORGANICS IN OVERBURDEN SOILS
(SUBSURFACE SOIL AREA COMPOSITE SAMPLES)

Results in µg/kg (ppb)

Compound	SB-A	SB-B	SB-C	SB-D	SB-E	SB-F	SB-G	SB-H	SB-I	SB-J
Vinyl Chloride	--	--	--	--	--	--	--	--	--	--
Total 1,2-Dichloroethene	--	--	--	--	--	--	--	--	--	--
1,2-Dichloroethane	--	--	--	--	--	--	--	--	--	--
Trichloroethene	--	--	--	--	--	--	--	--	--	--
Benzene	--	--	940	--	36,000^J	--	--	--	--	--
Tetrachloroethene	--	--	--	--	--	--	--	--	--	--
Chlorobenzene	--	75,000^J	--	56,000^J	1,500,000	19,000^J	--	24,000^J	--	34,000^J
Carbon Disulfide	--	--	--	--	220,000	780,000	2,200,000	290,000	96,000	--

02[IL]JD1900:D3048/1194/2

E = Exceeds quantifiable range; estimated
J = Below quantifiable range; estimated
-- Below detectable limits

Table 4-9

SEMIVOLATILE ORGANICS IN OVERBURDEN SOILS (SUBSURFACE SOIL AREA COMPOSITE SAMPLES)

Results in μg/kg (ppb)

Compound	SB-A	SB-B	SB-C	SB-D	SB-E	SB-F	SB-G	SB-H	SB-I	SB-J
1,3-Dichlorobenzene	46,000^{J}	41,000^{J}	2,200	--	170,000	240,000	79,000	54,000^{J}	1,700^{J}	160,000
1,4-Dichlorobenzene	97,000	79,000	4,400	--	1,900,000	910,000	290,000	330,000	140,000	960,000
1,2-Dichlorobenzene	360,000	300,000	9,400	--	2,200,000	2,400,000^{E}	380,000	640,000	270,000	2,000,000
1,2,4-Trichlorobenzene	340,000	470,000	4,200	--	2,300,000	2,800,000	1,100,000^{E}	4,300,000^{E}	1,900,000	2,100,000
2-Methylphenol	--	--	640	--	--	--	--	--	--	--
4-Methylphenol	--	--	920	--	--	--	--	--	--	--
4-Dimethylphenol	--	--	640	--	--	--	--	--	--	--
Naphthalene	--	--	170^{J}	--	--	--	--	--	--	--

02[IL]JD1900:D3048/1197/3

E = Exceeds quantifiable range; estimated
J = Below quantifiable range; estimated
-- Below detectable limits

Table 4-10

METALS IN OVERBURDEN SOILS
(2.0' - 4.0' MONITORING WELLS)

Results in mg/kg (ppm)

Metal	SB1B2	SB2A2	SB3A2	SB4A2	SB5B2	SB6A2	SB7A2
Aluminum	19,900	10,000	3,320	9,440	8,290	8,480	2,080
Arsenic	5.3	23.6	3.1	2.5	4.4	6.6	4.9
Barium	326	10,200	59.7	96.7	97.1	1,940	513
Cadmium	3.6	5.6	--	1.8	1.4	11.8	14.7
Calcium	68,700	70,800	6,730	53,700	16,400	93,800	34,200
Chromium	112	37.2	27.5	14.2	14.8	44.8	71.9
Cobalt	29.9	12.9	--	--	--	--	--
Copper	60.9	1,870	23.1	29.8	19.4	643	643
Iron	34,600	18,300	7,690	14,100	13,400	21,500	120,000
Lead	547	634	125	24.2	51.2	621	1,080
Magnesium	5,260	13,800	1,290	11,800	4,520	11,300	--
Manganese	651	364	95	244	410	307	713
Mercury	443	4.4	0.92	6.4	0.68	0.44	0.26
Nickel	435	98.3	--	13.7	14.8	79.4	36.6
Zinc	769	1,140	362	486	129	1,690	1,990
Sodium	5,260	1,420	--	--	--	--	--
Vanadium	30.1	33.4	--	19.4	13.9	26.6	42.7
Cyanide	--	82.1	--	3.9	--	19.4	--
Thallium	--	--	--	--	--	0.52	--

02[IL]JD1900:D3048/1204/14

E = Exceeds quantifiable range; estimated
J = Below quantifiable range; estimated
-- Below detectable limits

Table 4-11

METALS IN OVERBURDEN SOILS
(MONITORING WELLS BEDROCK COMPOSITE SAMPLES)

Results in mg/kg (ppm)

Metal	SB1BC	SB2AC	SB3AC	SB4AC	SB5BC	SB6AC	SB7AC
Aluminum	6,750	6,310	4,810	7,180	7,180	5,110	4,960
Arsenic	4.8	13.3	3.0	3.5	3.8	6.3	3.7
Beryllium	--	0.48	--	0.47	--	--	--
Barium	188	6,340	81.4	111	84.7	2,200	104
Calcium	52,200	46,600	24,700	107,000	74,300	55,900	25,300
Cadmium	4.2	2.9	1.9	2.9	1.7	4.5	3.2
Chromium	277	28.7	30.0	12.6	14	30.6	186
Copper	142	162	31.9	28.7	28.4	333	116
Iron	38,800	14,400	12,900	12,500	17,600	15,100	20,200
Lead	3,180	225	185	38.0	60.9	386	286
Magnesium	18,000	17,400	7,540	33,400	10,600	13,800	1,690
Manganese	406	309	285	565	395	248	266
Mercury	111	15.5	2.9	1.1	0.52	38.6	0.36
Nickel	210	55.9	--	11.7	--	36.2	13.7
Zinc	394	488	267	613	93.6	1,078	952
Sodium	--	--	--	--	--	--	--
Thallium	--	--	--	--	--	0.49	--
Cyanide	--	25.5	--	--	--	3.5	--
Vanadium	13.7	17.3	12.1	15.9	18.5	15.9	13.9

02[IL]JD1900:D3048/1205/13

E = Exceeds quantifiable range; estimated
J = Below quantifiable range; estimated
-- Below detectable limits

Table 4-12

METALS IN OVERBURDEN SOILS
(SUBSURFACE SOIL AREA COMPOSITE SAMPLES)

Results in mg/kg (ppm)

Metal	SB-A	SB-B	SB-C	SB-D	SB-E	SB-F	SB-G	SB-H	SB-I	SB-J
Aluminum	9,500	21,500	8,930	30,100	9,060	16,300	13,100	8,900	9,970	7,140
Arsenic	13.1	104	26.8	13.6	18.7	13.9	19.4	39.1	14.7	14.8
Barium	5,230	19,200	365	623	191	869	958	428	848	370
Cadmium	17.2	8.7	14.4	12.7	4.0	21.0	12.6	74.8	55.0	3.7
Calcium	80,500	184,000	23,800	61,100	45,800	79,600	143,000	23,400	48,700	49,600
Chromium	91.8	1,530	33.6	49.0	24.1	54.6	200	80.4	81.4	182
Cobalt	18.4	61.6	--	--	--	--	--	--	--	--
Copper	1,020	352	285	339	644	518	468	8,180	877	248
Lead	1,920	752	1,110	1,360	627	779	1,520	31,600	7,240	1,180
Iron	94,100	56,300	34,600	31,200	24,500	35,500	45,600	69,900	46,200	20,000
Magnesium	21,400	49,600	5,130	10,400	18,700	25,400	69,700	11,700	17,300	16,600
Manganese	770	1,480	376	629	401	1,080	903	495	591	298
Mercury	23.4	22.5	19.2	23.3	3.6	1.4	1.6	3.9	2.0	2.9
Nickel	108	128	27.4	37.2	20.3	42.9	46.7	54.4	41.4	31.3
Zinc	10,200	1,820	5,890	7,900	1,530	18,200	7,600	56,900	26,500	4,710
Potassium	--	4,900	--	4,590	1,460	1,560	--	--	1,070	--
Selenium	--	--	--	--	--	--	--	13.4	--	--
Cyanide	5.6	1.8	--	--	--	--	--	--	1.5	3.2
Silver	--	--	--	--	--	--	--	16.4	--	--
Vanadium	34.5	47.6	33.7	53.3	27.5	38.0	32.6	89.4	24.3	22.7

02[IL]JD1900:D3048/1206/7

E = Exceeds quantifiable range; estimated
J = Below quantifiable range; estimated
-- Below detectable limits

Table 4-13

PESTICIDES IN OVERBURDEN SOILS
Results in $\mu g/kg$ (ppb)

Pesticide	SB-4AC	DU-5BC	SB-6A2	SB-6AC	SB-7A2	SB-7AC
Alpha BHC	74^J	7.8^J	--	--	--	--
Beta BHC	300	--	--	--	--	--
Gamma BHC (Lindane)	40^J	--	--	--	--	--
Dieldrin	--	--	$1,600^J$	480^J	--	--

02[IL]JD1900:D3048/1199/22

E = Exceeds quantifiable range; estimated
J = Below quantifiable range; estimated
-- Below detectable limits

SB**2 = 0.0' - 2.0 grab sample
SB**C = 0.0 - bedrock composite sample
DU** = Duplicate sample

volatile organics and semivolatiles in soil samples from other monitoring wells around the site are either an order of magnitude lower or not detected. Only samples from MW-4A yielded levels comparable to those in the southwest corner of the site. In particular, high levels of 1,2,4-trichlorobenzene of 51,000 µg/kg and 340,000 µg/kg were obtained from the 2- to 4-foot sample and the composite sample in MW-4A (see Figures 4-5 and 4-6, respectively).

High levels of volatile organics and semivolatiles were detected in area composite samples that were collected in nearly all the tank farm areas (see Figure 4-7 and Tables 4-8 and 4-9). Levels of 1,2,4-trichlorobenzene that are greater than 100,000 µg/kg were detected in all area composites except for those samples from Areas C and D. High levels of various derivatives of dichlorobenzene were also detected in most area composite samples. Carbon disulfide in concentration of greater than 96,000 µg/kg was detected in Areas E, F, G, H, and I.

Significant levels of metals were present throughout the site in both types of monitoring well samples and in the area composite sample (see Figures 4-8, 4-9 and 4-10 and Tables 4-10, 4-11 and 4-12). Aluminum was present in all samples with the highest concentration of 30,100 mg/kg from the composite sample from Area D. Levels of zinc were also high in most area composite samples. The highest level of zinc detected was 56,900 mg/kg from Area H (see Table 4-12 and Figure 4-10). Lead was also present in all 50 samples, but at significantly lower levels than other metals. One notable exception was the composite soil sample from Area H which contained lead at 31,600 mg/kg.

Pesticides in overburden soils are a more insignificant component of the chemicals detected in overburden soils. Dieldrin was detected in two samples from MW-6 at levels of 1,600 and 480 µg/kg.

Sediment samples taken from sewers in the southwest corner of the site (SE-01), and near the northern boundary (SE-02), follow the same overall pattern of compound concentrations, as do the soil and groundwater samples (see samples SE-01 and SE-02, Appendix B). Levels in sample SE-01, taken near the source area, are one to three orders of magnitude greater than those seen in sample SE-02. Specifically, chlorobenzene is seen at 25,000 µg/kg; 1,4-dichlorobenzene at 470,000 µg/kg; 1,2-dichlorobenzene at 840,000 µg/kg; and 1,2,4-trichlorobenzene at 2,700,000 μg/kg in sample SE-01. Comparatively, 1,4-dichlorobenzene is seen at 39,000 μg/kg; 1,2-dichlorobenzene at 52,000 μg/kg and 1,2,4-trichlorobenzene at 12,000 μg/kg in sample SE-02. Benzene and chlorobenzene were not detected in this sample. Thus, it can be concluded that contaminant migration to sewers is pronounced in the source area and is observed to a lesser extent in the near northern boundary of the site.

Numerous polycyclic aromatic hydrocarbons (PAHs) were found in soil samples. The most common occurrences of this family of compounds were in soil samples obtained from monitoring wells (see Appendix B). These compounds were present in all soil samples from borings, but were generally at least two orders of magnitude less concentrated than the previously noted volatiles and semivolatiles. The highest concentration detected in any sample was benzo(b)fluoranthene at a level of 33,000 μg/kg from SB-5BC. Levels of these PAH compounds between 1,000 to 5,000 μg/kg were detected in sampled detected from all monitoring wells on the site. This differs somewhat from volatile and semivolatiles levels in soils that were somewhat restricted to the southwest, southeast, and northeast portion of the site. The PAH concentrations, while being elevated in soil samples, are present at relatively insignificant concentrations in groundwater samples; i.e., all detections in groundwater were estimated at levels beneath the quantification range (see Appendix B).

Three volatile tentatively identified compounds (TIC) and 17 semivolatile compounds were identified in soil samples (see Appendix B). Thirteen of the 20 TICs were derivatives of benzene. The most commonly detected benzene, trichlorobenzene, was at levels of up to 1,400,000 μg/kg. Other high levels of TICs detected in soil samples were concentrations of 340,000 μg/kg and 730,000 μg/kg of the benzene tetra-chloro derivatives. Fifteen TICs were detected in sediment samples SE-1 and SE-2. As noted with soil samples, 11 of the TICs were benzene derivatives. As expected, the most common occurrences were in SE-1, which was collected in on site. Concentrations of up to 170,000 μg/kg of benzene derivatives were detected in this sample.

The nature of the contamination in the unsaturated zone can be summarized as follows:

- The major mechanism of movement of compounds in the unsaturated overburden is downward percolation;

- The succession of movement of compounds is from the unsaturated zone to the saturated overburden, to the saturated bedrock and potentially off site in groundwater;

- The most common compounds that are ubiquitous over the entire site are benzene, chlorobenzene, the various derivatives of dichlorobenzene, and 1,2,4-trichlorobenzene. The highest concentrations of these chemicals are in the tank farm area and near the abandoned railroad tracks in the southwest corner of the site. High levels of these compounds were also detected in the tank farm in the southeast corner and the area around MW-4. Most likely, all of these compounds are leaching through infiltration and groundwater fluctuation from the unsaturated to the saturated zone. Thus, these areas are considered to be source areas for a large portion of the groundwater plume which exists across the site; and

- Elevated levels of metals are present across the site in overburden soils. Of particular importance is the high level of lead detected in tank farm area H. However, relative to the benzene compounds, metals form a less significant portion of soils contamination at the site.

4.5.3 Contamination in the Saturated Zone

As noted in the previous discussion, the unsaturated overburden is the primary source, and bedrock groundwater the principal receptor of contaminants on the site. Once a compound has reached the highly fractured and weathered top of bedrock, it will migrate according to the direction of groundwater movement. Contaminants may continue to migrate downward through the B-Zone of the Lockport Dolomite, particularly since it is weathered.

Regional groundwater movement is to the north and west, with the Falls Street Tunnel and NYPA intake tunnels acting as local line sinks. The upper Niagara River is the principle source of recharge for the Lockport Dolomite, with the lower Niagara River near the gorge serving as the principal discharge. Precipitation is the primary means of recharge for the overburden enhancing the downward migration of compounds.

Volatile and semivolatile organic compounds in the saturated overburden (A-Zone) are detected in highest concentrations in the southwestern wells: MW-1A, MW-2A, and MW-7A, and the northwestern well MW-4A (see Figure 4-11 and Tables 4-14 and 4-15). Benzene is found in particularly high concentrations, 540,000 µg/L and 100,000 µg/L in wells MW-1A and MW-2A, respectively. These concentrations correspond to high levels in soil samples from MW-1A and MW-2A. Chlorobenzene, 1,4-dichlorobenzene, and 1,3-dichlorobenzene are found in significant levels which range from 2,300 µg/L to 180,000 µg/L in wells MW-2A and MW-7A. 1,2-dichlorobenzene is found at a level of 16,000 µg/L in MW-4A and 1,2,4-trichlorobenzene is detected in both MW-4A and MW-7A at concentrations of 24,000 and 10,000 µg/L, respectively. This contaminant distribution is aligned closely with the location of overburden source areas.

The levels of volatiles and semivolatiles in MW-3A, MW-5A, and MW-6A are significantly lower than the previously mentioned wells. The highest benzene compound detected in any of these three wells was a concentration of 800 µg/L of chlorobenzene in MW-3A (see Figure 4-11 and Table 4-14). Thus, it is apparent that wells in source areas (1A, 1B, 2A, 2B, and 7A) are receiving chemicals through direct vertical migration from the source areas previously discussed. Given the high concentrations in these wells, it is certainly possible the dense NAPLs may accumulate beneath source areas. Indeed small amounts of dense NAPLs were observed during installation of wells MW-1B and MW-2B. A sample of the NAPL was collected during installation of MW-1B and submitted for analysis of volatile and semivolatile parameters (sample SB-1B-NA, Appendix B). Various attempts were made to collect a dense NAPL sample after installation of MW-1B, including letting a bailer set in MW-1B for one week.

A successful sample of dense NAPL was not acquired from well MW-1B. However, a NAPL sample was retrieved from MW-2B (see Appendix B). The NAPL samples collected during the installation of MW-1B aand MW-2B both had visible phase separation from the groundwater at the time of collection. Phase separation in liquid indicates the presence of NAPL.

The NAPL samples collected during the installation of MW-1B (Sample SB-1B-NA) and from MW-2B were analyzed for volatile and semivolatile parameters (see Appendix B). The results of this analysis was compared

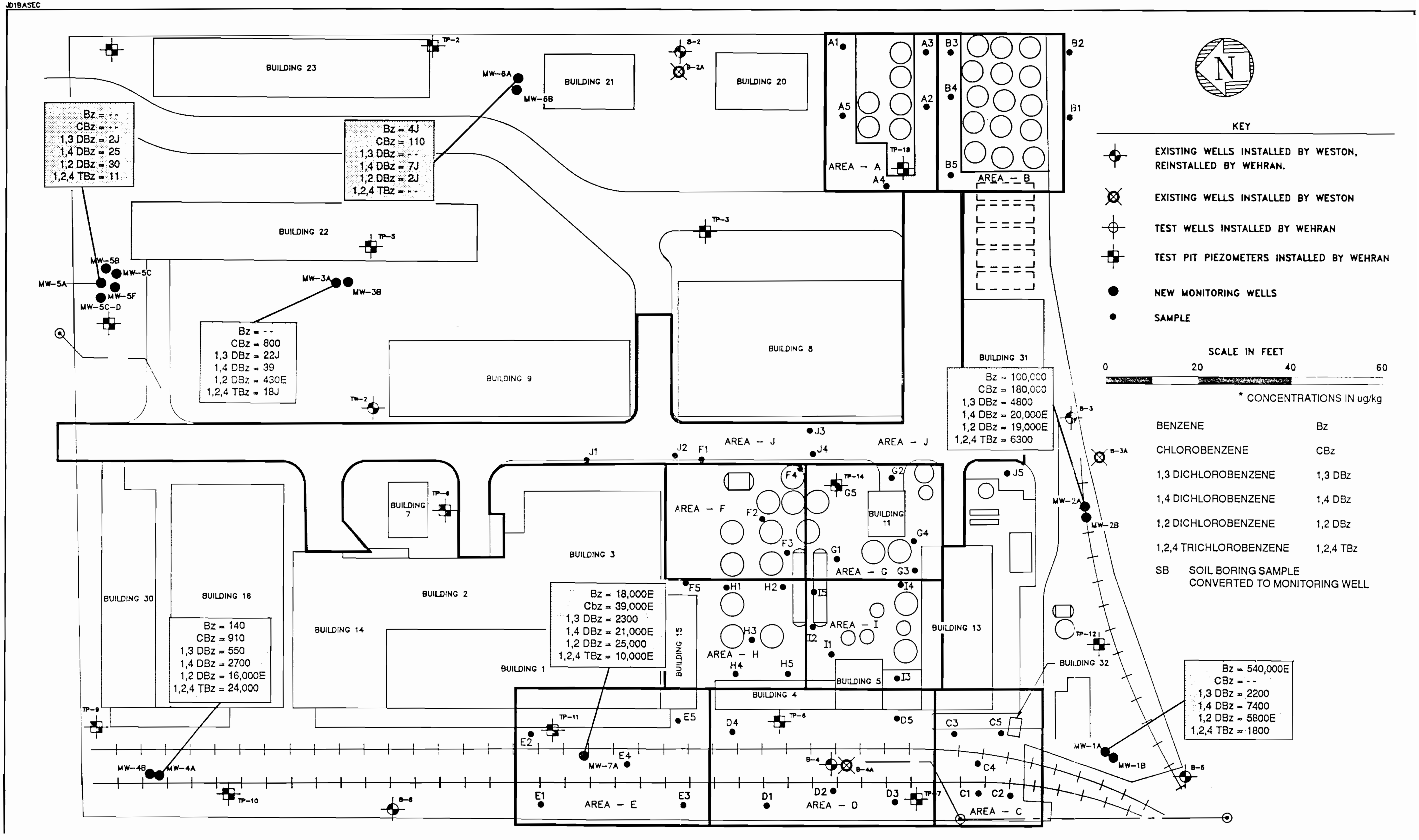


Figure 4-11
SIGNIFICANT VOLATILES AND SEMIVOLATILES
IN GROUNDWATER (A ZONE)

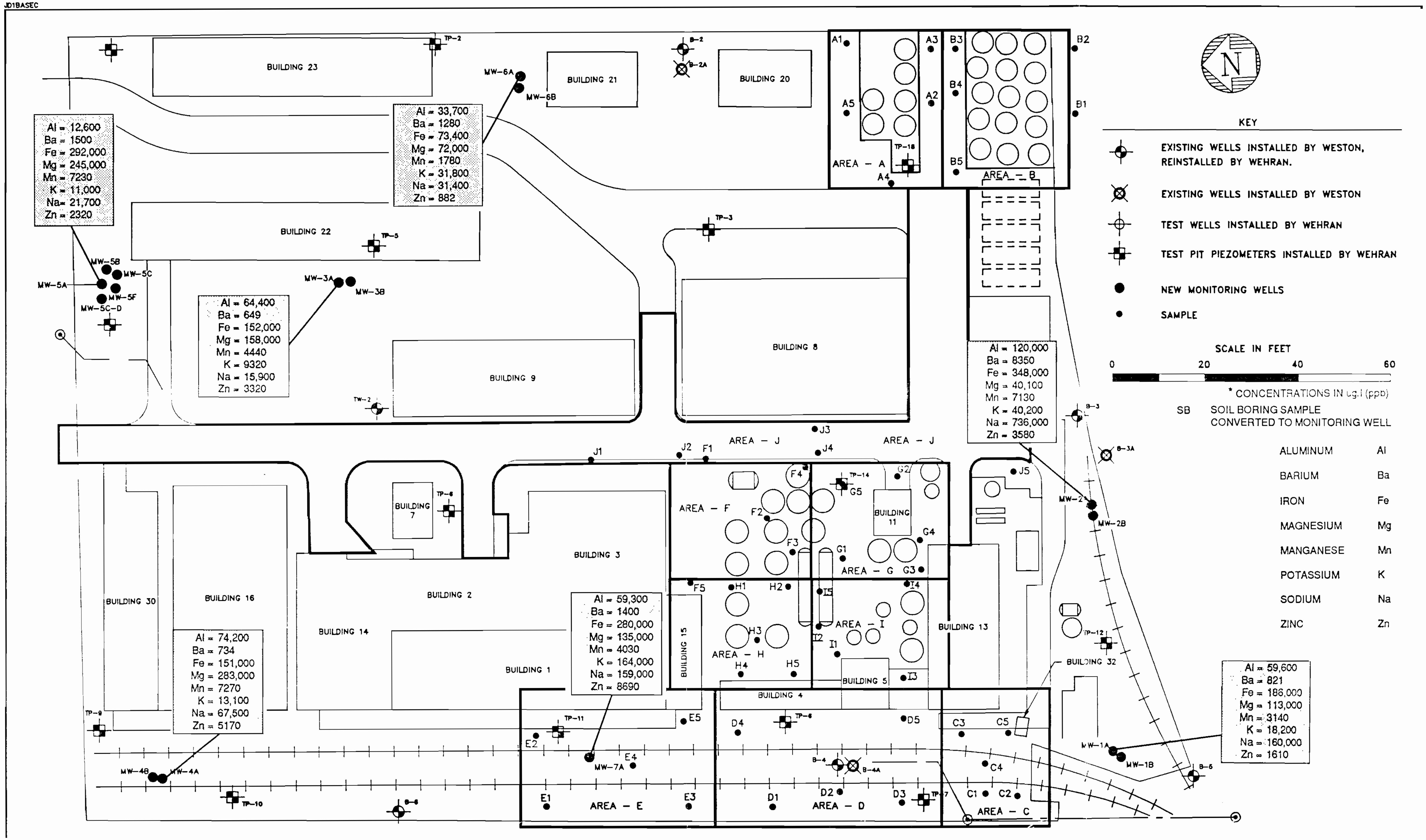


Figure 4-12
SIGNIFICANT METALS IN GROUNDWATER
(A ZONE)

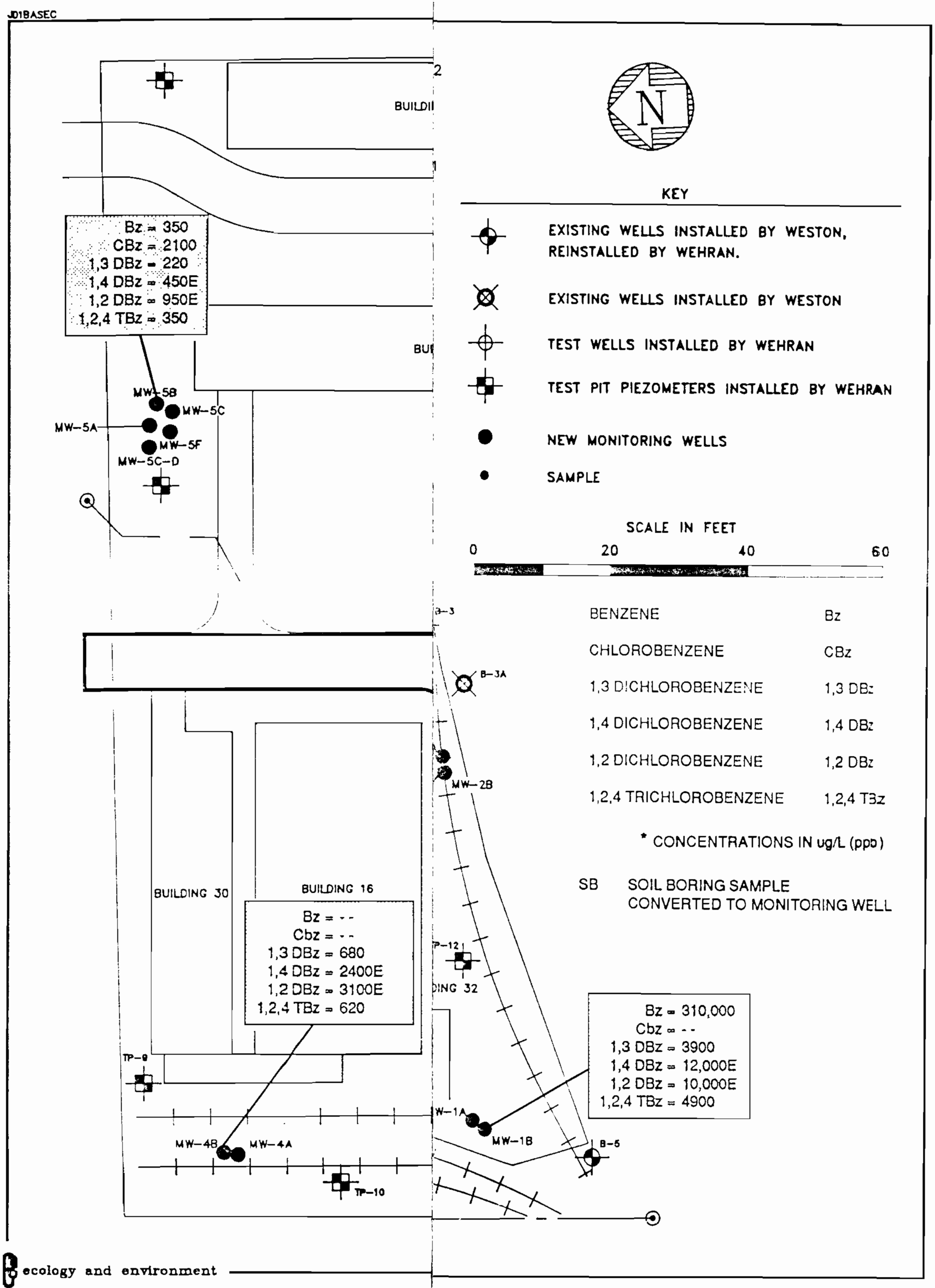


Figure 4-13
SIGNIFICANT VOLATILES AND SEMIVOLATILES IN GROUNDWATER (B ZONE)

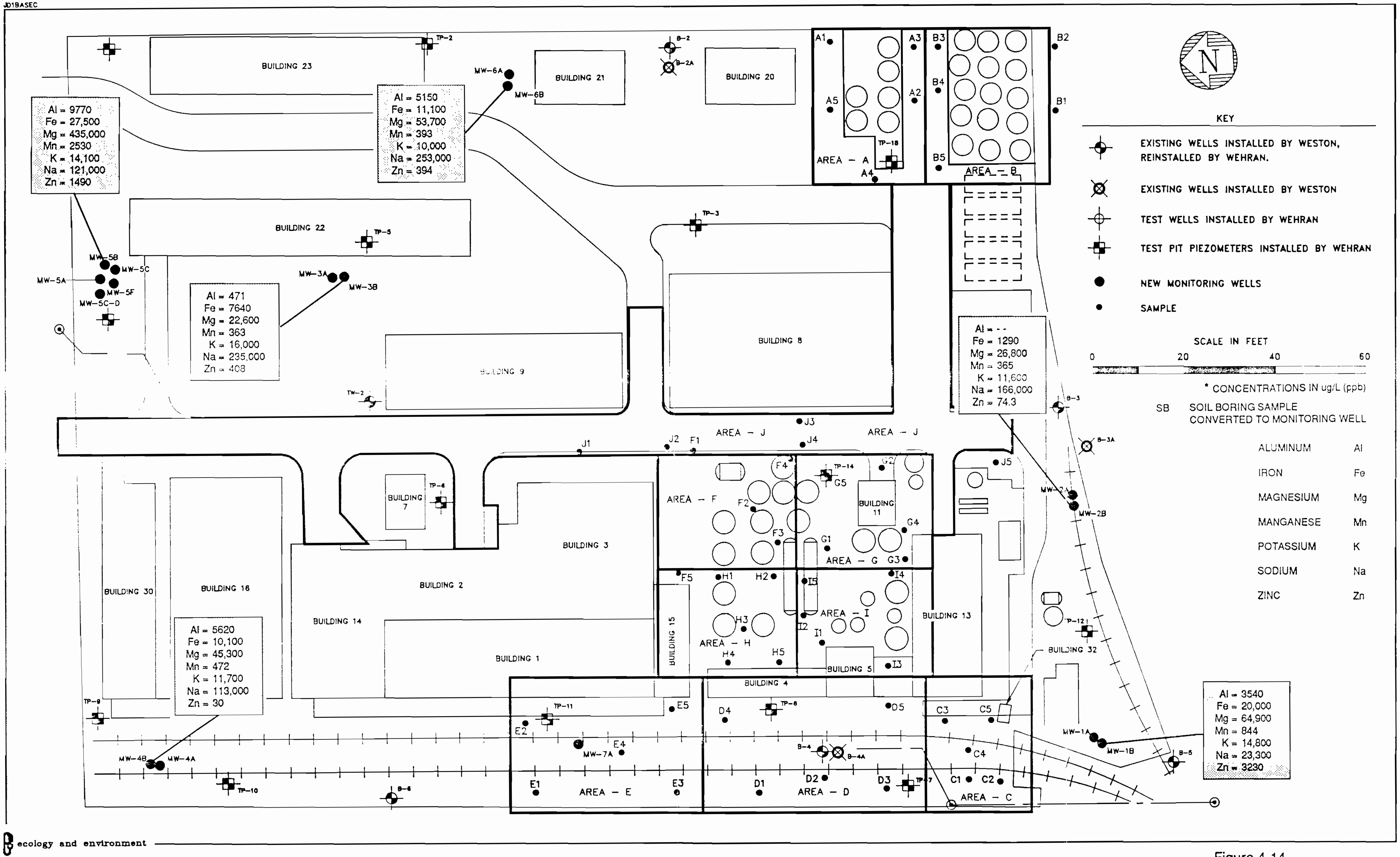


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Figure 4-14
SIGNIFICANT METALS IN GROUNDWATER
(B ZONE)

Table 4-14

VOLATILE ORGANICS IN GROUNDWATER (OVERBURDEN WELLS - HORIZON A)

Results in μg/l (ppb)

Compound	MW-1A	MW-2A	MW-3A	MW-4A	MW-5A	MW-6A	MW-7A
Vinyl Chloride	--	--	--	800	--	--	--
Methylene Chloride	--	31^J	--	--	--	--	--
Total 1,2-Dichloroethene	--	--	--	2,100	110	--	--
1,2-Dichloroethane	--	54	--	--	--	11	--
Trichloroethene	--	60	--	--	150	3^J	--
Benzene	$540,000^E$	100,000	--	140	--	4^J	$18,000^E$
4-Methyl-2-Pentone	--	380	--	--	--	--	--
Tetrachloroethene	--	510	--	--	820^E	--	--
Toluene	--	960	--	--	1^J	--	--
Chlorobenzene	--	180,000	800	910	--	110	$39,000^E$

02[IL]JD1900:D3048/1207/24

E = Exceeds quantifiable range; estimated
J = Below quantifiable range; estimated
-- Below detectable limits

Table 4-15

SEMIVOLATILE ORGANICS IN GROUNDWATER (OVERBURDEN WELLS - HORIZON A)

Results in μg/l (ppb)

Compound	MW-1A	MW-2A	MW-3A	MW-4A	MW-5A	MW-6A	MW-7A
Phenol	1,000	550	--	--	--	--	150^J
1,3-Dichlorobenzene	2,200	4,800	22^J	550	2^J	--	2,300
2-Chlorophenol	300	--	--	--	--	--	52^J
1,4-Dichlorobenzene	7,400	$20,000^E$	39	2,700	25	7^J	$21,000^E$
2,4-Dimethylphenol	--	65^J	--	--	--	--	--
1,2-Dichlorobenzene	$5,800^E$	$19,000^E$	430^E	$16,000^E$	30	2^J	25,000
2,4-Dichlorophenol	12^J	--	--	--	--	--	62^J
Naphthalene	24^J	16^J	--	--	--	--	38^J
1,2,4-Trichlorobenzene	1,800	6,300	18^J	24,000	11	--	$10,000^E$
2-Methylnaphthalene	8^J	16^J	--	--	--	--	--

02[IL]JD1900:D3048/1211/25

E = Exceeds quantifiable range; estimated
J = Below quantifiable range; estimated
-- Below detectable limits

with the analysis of the water samples collected from these locations (MW-1B, MW-2B in Tables 4-17 and 4-18) and published solubilities for the detected parameters. Analysis of the NAPL samples indicated:

- Detected parameters in both the NAPL (SB-1B-NA) and the water sample (MW-1B) at location MW-1B include: benzene (160,000 µg/l in the NAPL; 310,000 µg/l in the water), phenol (2,900 µg/l; 330 µg/l, respectively), 1,2-dichlorobenzene (28,000 µg/l; 10,000 µg/l), 1,3-dichlorobenzene (11,000 µg/l; 3,900 µg/l), 1,4-dichlorobenzene (37,000 µg/l; 12,000 µg/l), 2-chlorophenol (470 µg/l; 270 µg/l), naphthalene (83 µg/l; 25 µg/l), 1,2,4-trichlorobenzene (14,000 µg/l; 4,900 µg/l).

- The only parameters detected in the water sample (MW-1B) which were not detected in the NAPL sample (SB-1B-NA) were 2-methylnaphthalene at 26 µg/l and 2,4,6-trichlorophenol at 13 µg/l. Chlorobenzene at 290,000 µg/l; hexachlorobutadiene at 72 µg/l; phenanthrene at 90 µg/l; fluoranthene at 94 µg/l; pyrene at 76 µg/l; and benzo(b)fluoranthene at 76 µg/l were the only parameters detected in the NAPL which were not detected in the water sample collected from location MW-1B.

- Detected parameters in both the NAPL and water samples collected from MW-2B include: chlorobenzene (110,000,000 µg/l in the NAPL; 43,000 µg/l in the water), 1,2-dichlorobenzene (120,000 µg/l; 287,000 µg/l, respectively), 1,3-dichlorobenzene (45,2000 µg/l; 20,000 µg/l), 1,4-dichlorobenzene (188,000 µg/l; 75,000 µg/l), 1,2,4-trichlorobenzene (123,000 µg/l; 47,000 µg/l), 1,2-dichloroethene (31,000 µg/l; 1,400 µg/l), tetrachloroethene (210,000 µg/l; 87 µg/l), and toluene (67,000 µg/l; 58 µg/l).

- The only parameters detected in the water sample MW-2B which were not detected in the NAPL at MW-2B include hexachlorobenzene at 160 ug/l, phenanthrene at 77 ug/l, fluoranthene at 77 ug/l, and pyrene at 42 ug/l. Carbon disulfide at 1,100,000 ug/l, benzene at 1,200,000 ug/l, ethylbenzene at 16,000 ug/l, and xylenes at 36,000 ug/l were the only parameters detected in the NAPL which were not detected in the water sample collected from MW-2B.

- Generally detected parameters were reported at higher concentrations in the NAPL samples than in the water samples collected at a given location.

- Three parameters were detected in the NAPL sample collected from MW-2B at concentrations exceeding their respective solubilities in water: Chlorobenzene with a solubility of 466,000 ug/l was reported at 110,000,000 ug/l, 236 times its solubility; 1,2,4-trichlorobenzene with a solubility of

30,000 ug/l was reported at 123,000 ug/l, 4.1 times its solubility; and 1,2-dichlorobenzene with a solubility of 100,000 µg/l was reported at 287,000 ug/l, 2.87 times its solubility. These concentrations confirm the existence of a NAPL at MW-2B during the sampling event. The density of the 1.1 gm/cm^3 of the principal compound in the NAPL, chlorobenzene, suggests the NAPL was a dense phase (i.e., denser than water).

- o No parameters were reported to exist above their solubility limits in the NAPL sample SB-1B-NA collected during the installation of MW-1B.

Unfiltered water samples were submitted for total metal analysis from each of the monitoring wells. Metals adsorbed on or incorporated into the clay and silt particles suspended in unfiltered water samples will raise the concentrations of detected metals. Unfiltered water samples from the A-Zone generally had higher concentrations of metals than water samples collected from the other zones. Higher soil and clay is likely to be associated with the unconsolidated aquifer materials found in the A-Zone, rather than the bedrock associated with the wells screened at lower intervals; therefore, the higher metal concentrations in water samples collected from A-Zone is not unexpected. Aluminum, magnesium, iron, potassium, and sodium were the metals detected at the greatest concentrations (see Figure 4-12 and Table 4-16). All of these metals are common constituents of clays.

Unpublished total metals data obtained from unfiltered water samples collected from overburden wells above the Lockport are presented in Section 5. The unpublished data provides a reference for expected background metal concentrations in groundwater. The data indicate that the total metal concentrations in the unfiltered water samples at all site well locations are generally greater than expected background levels.

Benzene concentrations in the B-Zone, with the exception of well MW-1B, are reduced considerably from the A-Zone. MW-1B contains 310,000 µg/L of benzene. Benzene concentrations in groundwater from other B-Zone wells are several orders of magnitude less than the concentrations detected in Well MW-1B (see Figure 4-13 and Tables 4-17 and 4-18). Chlorobenzene is also at reduced levels in the zone, except in those concentrations found in MW-2B, and MW-3B which are 43,000 µg/L and

Table 4-16

METALS IN GROUNDWATER
(OVERBURDEN WELLS - HORIZON A)

Results in μg/L (ppb)

Metal	MW-1A	MW-2A	MW-3A	MW-4A	MW-5A	MW-6A	MW-7A
Aluminum	59,600	120,000	64,400	74,200	12,600	33,700	59,300
Arsenic	26.5	38.0	16.5	18.5	13.2	15.9	69.7
Barium	821	8,350	649	734	1,500	1,280	1,400
Cadmium	21.5	39.8	21.0	37.0	39.6	6.0	46.2
Calcium	699,000	1,070,000	840,000	1,360,000	1,290,000	75,800	496,000
Cobalt	653	131	--	62.1	112	--	60.9
Chromium	863	323	242	128	260	123	252
Copper	272	721	235	229	531	254	861
Iron	186,000	348,000	152,000	151,000	292,000	73,400	280,000
Lead	904	885	322	481	424	477	3,430
Magnesium	113,000	40,100	158,000	283,000	245,000	72,000	135,000
Mercury	216	2.9	5.4	0.92	2.1	--	3.7
Manganese	3,140	7,130	4,440	7,270	7,230	1,780	4,030
Nickel	1,280	419	178	182	339	115	271
Potassium	18,200	40,200	9,320	13,100	11,000	31,800	164,000
Sodium	160,000	736,000	15,900	67,500	21,700	31,400	159,000
Zinc	1,610	3,580	3,320	5,170	2,320	882	8,690
Vanadium	87.1	353	133	155	255	71.9	164
Cyanide	24.2	256	11.0	--	--	34.0	30.0

02[IL]JD1900:D3048/1620/24

E = Exceeds quantifiable range; estimated
J = Below quantifiable range; estimated
-- Below detectable limits

Table 4-17

VOLATILE ORGANICS IN GROUNDWATER
(FRACTURED BEDROCK WELLS - HORIZON B)

Results in μg/l (ppb)

Compound	MW-1B	MW-2B	MW-3B	MW-4B	MW-5B	MW-6B
Vinyl Chloride	--	--	23^J	--	--	2^J
Methylene Chloride	--	13^J	--	--	--	1^J
Total 1,2-Dichloroethene	--	1,400	160	--	--	2^J
1,1-Dichloroethane	--	--	--	--	--	4^J
1,2-Dichloroethane	--	--	--	--	--	--
Trichloroethene	--	--	27^J	--	--	--
Benzene	310,000	--	--	--	350	300
Toluene	--	58	95	--	--	--
Tetrachloroethene	--	87	45^J	--	--	--
Ethylbenzene	--	--	29^J	--	--	--
Chlorobenzene	--	43,000	13,000	--	2,100	1,800
Total Xylenes	--	--	150	--	--	--

02[IL]JD1900:D3048/1209/15

E = Exceeds quantifiable range; estimated
J = Below quantifiable range; estimated
-- Below detectable limits

Table 4-18

SEMIVOLATILE ORGANICS IN GROUNDWATER
(FRACTURED BEDROCK WELLS - HORIZON B)

Results in μg/l (ppb)

Compound	MW-1B	MW-2B	MW-3B	MW-4B	MW-5B	MW-6B
Phenol	330	--	--	8^J	3^J	--
1,3-Dichlorobenzene	3,900	20,000	1,800	680	220	65
2-Chlorophenol	270	--	--	14^J	6^J	--
1,4-Dichlorobenzene	$12{,}000^E$	75,000	4,900	$2{,}400^E$	450^E	94
Naphthalene	25^J	--	--	--	--	--
2-Methylnaphthalene	26^J	--	--	--	--	--
1,2-Dichlorobenzene	$10{,}000^E$	$120{,}000^E$	$12{,}000^E$	$3{,}100^E$	950^E	83
1,2,4-Trichlorobenzene	4,900	47,000	4,200	620	350	120
2,4,6-Trichlorophenol	13^J	--	--	--	--	--

02[IL]JD1900:D3048/1212/32

E = Exceeds quantifiable range; estimated
J = Below quantifiable range; estimated
-- Below detectable limits

13,000 µg/L, respectively. The majority of the remaining semivolatiles are concentrated in MW-1B and MW-2B, except 1,2-dichlorobenzene, which was detected in high levels in MW-3B (see Figure 4-13 and Table 4-18). In general high concentrations of the benzenes occurs in wells that monitor groundwater immediately beneath source areas. MW-3B, however, is a notable exception to this trend.

It is more difficult to predict the source of TCE and its two biodegradation products (1,2-dichloroethene [1,2-DCE] and vinyl chloride [VC]) than the sources of benzene and its various derivatives. Virtually none of these chlorinated organics have been found in any overburden soil samples, whether discrete or composite. Only two A wells (4A and 5A) and two B wells 2B and 3B contain any chlorinated organics. However, these compounds are quite common in the deeper bedrock monitoring wells 5C, 5-CD, and 5F. Two potential explanations exist regarding this trend. First, there is an on-site overburden source that is limited in extent and not yet adequately defined. Second, an off-site source exists, perhaps to the south upgradient from the site, which results in the deeper bedrock chlorinated organic contamination noted in 5C, 5-CD, and 5-F. Current data appears to suggest that off-site source is more likely the explanation.

Metals in the B-Zone are considerably less concentrated than A-Zone metals (see Figure 4-14 and Table 4-19). A similarly uniform distribution of metals across the site occurs in the unsaturated overburden, A-Zone and B-Zone. Magnesium, potassium, and sodium are detected at moderate to high levels in the B-Zone. Metals levels may also be biased high as unfiltered samples were collected.

The trend of dissolved contaminant flow in the B-Zone is to follow groundwater flow lines. The source areas wells, in the southwest corner of the site, exhibit predictably high levels of compounds, while downgradient areas show more moderate concentrations, indicative of contaminant migration due to groundwater flow. Downward migration of contaminants to deeper fractured horizons beneath the B-Zone by dense NAPLs is also a potential migration pathway. Evaluation of volatile and semivolatile compounds from wells MW-5C, MW-5CD, and MW-5F does not indicate a likelihood for extensive deep, dense NAPL migration. The concentrations of chlorobenzenes, dichlorobenzenes, and trichlorobenzenes in these deep wells do not exceed 5,000 µg/L, which is much

Table 4-19

METALS IN GROUNDWATER (OVERBURDEN WELLS - HORIZON B)

Results in μg/L (ppb)

Metal	MW-1B	MW-2B	MW-3B	MW-4B	MW-5B	MW-6B
Aluminum	3,540	--	471	5,620	9,770	5,150
Arsenic	11.8	14.5	14.8	--	12.5	9.4
Barium	--	--	97.7	--	--	--
Calcium	588,000	348,000	249,000	400,000	1,090,000	312,000
Cadmium	84	--	--	--	10.5	--
Cobalt	--	--	13.0	--	--	--
Chromium	51.1	--	46.4	38.5	38.2	21.1
Copper	61.4	--	--	--	40.1	--
Lead	63.2	--	14.7	51.0	524	53.9
Iron	20,000	1,290	7,640	10,100	27,500	11,100
Manganese	844	365	363	472	2,530	393
Magnesium	64,900	26,800	22,600	45,300	435,000	53,700
Mercury	11.9	--	0.24	--	--	0.29
Zinc	3,230	74.3	408	30.0	1,490	394
Nickel	--	--	--	230	--	--
Cyanide	84.0	298	1,890	--	--	480
Potassium	14,800	11,600	16,000	11,700	14,100	10,000
Sodium	23,300	166,000	235,000	113,000	121,000	253,000
Vanadium	--	--	--	367	--	--

02[IL]JD1900:D3048/1621/32

E = Exceeds quantifiable range; estimated
J = Below quantifiable range; estimated
-- Below detectable limits

Table 4-20

VOLATILE ORGANICS IN GROUNDWATER
(OVERBURDEN WELLS - HORIZON C, CD, F)

Results in µg/L (ppb)

Compound	MW-5C	MW-5CD	MW-5F
Vinyl Chloride	74^{J}	--	980
Total 1,2-Dichloroethene	820	630	2,400
1,2-Dichloroethane	--	--	--
Trichloroethene	250	--	--
Benzene	920	1,200	--
Tetrachloroethene	320	--	--
Toluene	--	98^{J}	--
Chlorobenzene	4,300	7,200	58^{J}

02[IL]JD1900:D3048/1208/33

E = Exceeds quantifiable range; estimated.
J = Below quantifiable range; estimated.
-- = Below detectable limits.

lower than the respective solubilities of these compounds (see Tables 4-20 and 4-21).

These deep wells were drilled into the Lockport Dolomite in the number 5 well nest. Wells MW-5C, MW-5CD, and MW-5F intersect the second, third, and fourth water-bearing fractures respectively. Moderate concentrations of total 1,2-DCE (range 820 to 2,400 μg/L) were detected in all three wells. The C-Zone well also contained trichloroethene (TCE) at a concentration of 250 μg/L. Wells MW-5C and MW-5F contain levels of 74 μg/L and 980 μg/L, respectively, of VC as well. Moderate to low concentrations of TCE, DCE and VC are also detected in overburden MW-4A and MW-5A, and B-Zone wells MW-2B and MW-3B (see Tables 4-14 and 4-17).

Although the concentrations of the aromatic hydrocarbons in the wells MW-5C and MW-5CD are higher than those in MW-5B, these levels are greatly reduced when compared to other B-Zone wells. It is believed that high angle fracturing has created a communication between the zones, and the anisotropic nature of the fracturing, in combination with the generally downward vertical hydraulic gradient, led to the higher concentration levels.

Metals in MW-5C, MW-5CD, and MW-5F are present at reduced levels. Sodium, magnesium, potassium, and to a lesser extent, iron showed the highest concentrations. Their distribution was, again, evenly distributed across the site (see Table 4-22).

A total of 18 tentatively identified compounds (TICs), 3 volatile and 15 semivolatile, were detected in groundwater samples at the 3163 Buffalo Avenue site (see Appendix B). The most frequently occurring TICs were volatile and semivolatile benzene derivatives. Trichlorobenzene was found most often, having been detected in 11 of 16 wells, and showed the maximum concentration value of 9,800 μg/L in MW-2B. Source area wells MW-1A and 1B, MW-2A and 2B and MW-7A most commonly exhibited a wide range of TICs, and the highest values were seen in these wells. Wells MW-5C and MW-5CD had the most frequent TIC occurrence of the lower fracture zones. Southwest corner (source area) wells showed concentrations of 23(J)μg/L to 9,800(J)μg/L, while non-source area wells had a range of 8(J)μg/L to 2,600(J)μg/L.

Table 4-21

SEMIVOLATILE ORGANICS IN GROUNDWATER
(DEEP WELLS - HORIZONS C, CD, AND F)

Results in μg/L (ppb)

Compound	MW-5C	MW-5CD	MW-5F
1,3-Dichlorobenzene	750	810	4^{J}
1,4-Dichlorobenzene	$1{,}900^{E}$	$2{,}100^{E}$	14
1,2-Dichlorobenzene	$3{,}900^{E}$	$4{,}700^{E}$	39
1,2,4-Trichlorobenzene	$2{,}800^{E}$	1,500	6^{J}

02[IL]JD1900:D3048/1213/37

E = Exceeds quantifiable range; estimated
B = Below quantifiable range; estimated
-- Below detectable limits

Table 4-22

METALS IN GROUNDWATER
(OVERBURDEN WELLS - HORIZON C, CD, F)

Results in µg/L (ppb)

Metal	MW-5C	MW5-CD	MW-5F
Aluminum	2,240	982	1,680
Arsenic	35.5	--	--
Barium	--	--	--
Calcium	121,000	630,000	563,000
Cadmium	--	--	--
Chromium	33.5	24.3	88.0
Iron	7,290	3,730	2,650
Lead	67.0	86.0	21.9
Magnesium	30,700	144,000	88,100
Manganese	218	584	93.2
Mercury	0.43	--	--
Potassium	11,800	11,200	10,200
Zinc	486	400	177
Sodium	1,380,000	255,000	196,000
Cyanide	2,540	825	--

02[IL]JD1900:D3048/1622/38

E = Exceeds quantifiable range; estimated
J = Below quantifiable range; estimated
-- Below detectable limits

Groundwater samples were also analyzed for total organic halides (TOX). TOX concentrations were compared with the total concentration of detected organic compounds (Appendix B). The comparison indicated that in most of the wells TOX concentrations exceeded the total concentration of detected organic compounds. This indicates that additional organic compounds are present in the water samples but were not specifically identified with the analytical methods used. TOX concentrations ranged from approximately one-half to more than 1,000 times the total concentration of detected organic compounds in the A zone, and from approximately one quarter to more than 70 times the total concentration of detected organic compounds in the B zone. Elevated TOX levels were also measured in Zones C, CD, and F.

Contamination in the saturated zone can be summarized by following:

- Aromatic hydrocarbons (benzenes, chlorobenzenes, dichlorobenzenes, etc.) in the A-Zone and B-Zone are found in highest concentrations in the southwest corner of the site where the primary source area exists;
- Transport of these compounds is closely aligned with the groundwater flow paths, which are from the south to the north and northwest;
- TCE and its degradation compounds were found in various wells across the site and are considered to have an origin off site, perhaps to the south; and
- Wells MW-5C and MW-5CD show elevated concentrations of aromatic hydrocarbons which may be the results of downward movement between zones or anisotropic, fractured flow paths.

4.6 FATE AND TRANSPORT

4.6.1 Introduction

This section describes the fate and transport of contaminants of concern. The major transport mechanism of compounds on site will be discussed, as well as estimates of the rate of transport of selected compounds. Contaminant loading will be addressed at the end of this section.

4.6.2 Fate and Transport of the Contaminants of Concern

Transport of chemicals in environmental media is a function of the physical and chemical properties of the chemical and of the environmental conditions at the site. The following section presents a general discussion of the chemical properties affecting mobility and chemical transformation, and summarizes transport processes most likely to affect the chemicals detected at the 3163 Buffalo Avenue site.

Water solubility is a critical property affecting the environmental transport of a chemical: highly soluble chemicals can be rapidly leached from soils or waste and are generally mobile in groundwater. For inorganic contaminants, the solubility will depend on the valence of the element and on the chemistry of the surrounding medium.

A compound's volatilization rate from water depends on its vapor pressure and water solubility. High water-soluble compounds generally have lower volatilization rates from water than compounds having a low water solubility. Vapor pressure, a measure of the volatility of chemicals in their pure state, ranges from approximately 7×10^{-9} to 7.6×10^{2} mm Hg for liquids (EPA 1986). The Henry's law constant, which is the ratio of the pressure of the solute (in atmospheres) to its concentration in solution in dilute solution (in molecules/m^3), is a more accurate measure of volatilization behavior than is vapor pressure for estimating releases to air from water. Compounds with Henry's Law constants greater than approximately 10^{-3} can be expected to volatilize readily from water. Those with values ranging from 10^{-3} to 10^{-5} volatilize less readily, while compounds with values less than 10^{-5} volatilize slowly (Lyman et. al. 1982).

The octanol-water partition coefficient (K_{ow}) is often used to estimate the extent to which an organic chemical will partition from water into lipophilic tissues of organisms, such as fish or animal fat. Log K_{ow} values generally range from -2.5 to 10.5. Chemicals with K_{ow} less than 3 are generally considered not to concentrate in animal tissues. The organic carbon partition coefficient (K_{oc}) reflects the propensity of a compound to adsorb to the organic matter found in soil. K_{oc} values for organic chemicals range from 10^0 to 10^7 (log K_{oc} - 0 to 7) with higher values indicating greater adsorption potential. Chemicals with values of K_{oc} less than 1,000 generally do not adsorb strongly enough to soils to affect overall leachability at normal soil organic content levels, which are generally below 1% (EPA 1979).

For inorganic contaminants, prediction of adsorption behavior is complex; the extent of adsorption depends on the soil content of organic matter, clay, and iron and aluminum hydroxides, as well as the pH of the surrounding medium. The affinity of a chemical for soil particles is defined as the soil water distribution coefficient, K_d, and is equated to the ratio of the concentration of the chemical in the soil to the concentration in the associated interstitial water. A value of 100 or greater is indicative of strong adsorption. Another indicator of reactivity of inorganic contaminants is the redox potential (Eh), which is a measure of the ability to transfer electrons in solution or in soils.

The organic chemicals of potential concern can be classified into categories according to their similarity in chemical structure or physical/chemical properties (i.e., factors that would influence mobility in the environment). The chemical categories and the chemicals of concern within each category are shown in Table 4-23.

4.6.2.1 Environmental Chemistry of the Organic Chemicals of Potential Concern

Monocyclic Aromatic Hydrocarbons (Chlorinated and Unchlorinated)

Benzene, chlorobenzene, 1,2-dichlorobenzene, 1,3-dichlorobenzene, 1,4-dichlorobenzene and 1,2,4-trichlorobenzene were selected as chemicals of potential concern for this risk assessment. These six compounds have similar physical-chemical properties (Table 4-23), and can be grouped together in a discussion of their fate and transport in the environment. Based on their high vapor pressures and relatively low water solubilities (Verschueren 1983), the primary fate of these monocyclic aromatic hydrocarbons in surface soils or surface water is expected to be volatilization to the atmosphere (EPA 1979). Photooxidation in the troposphere is the dominant atmospheric fate of these compounds (EPA 1979).

Aromatic hydrocarbons may leach from soils into groundwater. The K_{oc} values for the six compounds range from 83 to 9,200, indicating that

Table 4-23

PHYSICAL-CHEMICAL PROPERTIES OF ORGANIC CONTAMINANTS OF CONCERN
3163 BUFFALO AVENUE SITE

Chemical Name	CAS Number	Mole Weight (g/mole)	Physical State at 20°C	Water Solubility (mg/l)	Source	Density (g/ml)	Source	Vapor Pressure (mm Hg)	Source	Henry's Law Constant (atm-m^3/mol)	Source
Benzene	71-43-2	78.1	Liquid	1.79 E+03	E	0.88	G	9.52 E+01	E	5.43 E-03	C
Chlorobenzene	108-90-7	113.6	Liquid	4.72 E+02	A	1.11	G	1.19 E+01	A	3.46 E-03	C
1,2-Dichlorobenzene	95-50-1	147.0	Liquid	1.56 E+02	E	1.31	G	1.47 E+00	E	1.20 E-03	E
1,3-Dichlorobenzene	541-73-1	147.0	Liquid	1.11 E+02	E	1.29	G	2.35 E+100	E	1.80 E-03	E
1,4-Dichlorobenzene	106-46-7	147.0	Liquid	8.00 E+01	E	1.25	G	1.18 E+00	E	2.89 E-03	E
1,2,4-Trichlorobenzene	120-82-1	181.5	Liquid	4.88 E+01	E	1.45	G	2.90 E-01	E	4.33 E-03	E
1,2-Dichloroethane	107-06-2	99.0	Liquid	8.52 E+03	E	1.26	F	7.87 E+01	E	1.10 E-04	C
1,2-Dichloroethene (Total)	540-59-0	97.0	Liquid	3.50 E+03	E	1.28	G	2.08 E+02	A	7.50 E-03	C
1,1,2,2-Tetrachloroethane	79-34-5	167.9	Liquid	2.96 E+03	E	1.60	G	6.10 E+00	E	4.80 E-05	C
Tetrachloroethene	127-18-4	165.8	Liquid	1.50 E+03	E	1.63	G	1.85 E+0	E	1.49 E-02	E
Toluene	108-88-3	92.1	Liquid	5.35 E+02	E	0.87	G	2.84 E+01	E	5.94 E-03	E
Trichloroethene	79-01-6	131.4	Liquid	1.10 E+03	E	1.47	G	6.90 E+01	E	1.20 E-03	C
Vinyl Chloride	75-01-4	62.5	Gas	2.76 E+03	E	0.91	F	2.66 E+03	E	6.90 E-01	C
Xylenes (Total) (Mixed)	1330-20-7	106.2	Liquid	1.60 E+02	E	0.86	G	1.00 E+01	E	6.82 E-03	E

Key at end of table.

Table 4-23 (Cont.)

Chemical Name	K_{oc} (ml/g)	Source	Log K_{ow}	Source	BCF	Source
Benzene	83	D	2.13	E	6.5	L
Chlorobenzene	330	A	2.84	K	33	L
1,2-Dichlorobenzene	1,700	E	3.38	E	115	L
1,3-Dichlorobenzene	1,700	E	3.60	E	190	L
1,4-Dichlorobenzene	1,700	E	3.60	E	117	L
1,2,4-Trichlorobenzene	9,200	A	4.02	E	630	L
1,2-Dichloroethane	14	A	1.48	E	1.4	L
1,2-Dichloroethene (Total)	49	A	2.09	A	1.5	L
1,1,2,2-Tetrachloroethane	54	A	2.39	E	12	L
Tetrachloroethene	118	A	3.40	E	66	L
Toluene	300	A	2.73	E	26	L
Trichloroethene	126	A	2.42	L	13	L
Vinyl Chloride	57	E	1.38	E	0.8	L
Xylenes (Total) (Mixed)	240	A	3.20	K	70	L

02[IL]JD1900:D3048/1217/0

Key:

A = Mabey, W., et al., 1982, Aquatic Fate Process Data for Organic Priority Pollutants.
B = Jaber, H., et al., 1984, Data Acquisition for Environmental Transport and Fate Screening (EPA-600/6-84-009, 010, 011).
C = Mackay, D. and W. Shiu, 1981, J. Phys. Chem. Ref. Data, 19:1175-1179.
D = Lymon, W., et al., 1983, Handbook of Chemical Property Estimation Methods, McGraw-Hill Book Company.
E = Howard, P., 1989, Handbook of Environmental Fate and Exposure Data for Organic Chemicals, Volume 1, Lewis Publishers, Inc.
F = Windholtz, M., et al., 1976, The Merck Index, 9th Edition, Merck and Company, Inc.
G = Weast, R., ed., 1983, Handbook of Chemistry and Physics, 63rd edition, CRC Press.
H = EPA (1987a), Health Advisories for 50 Pesticides, Office of Drinking Water.
I = Callahan, M., et al., Water-Related Fate of 129 Priority Pollutants, EPA 440/4-79-029a andb.
J = Clement Associates, Inc., 1985, Chemical, Physical, and Biological Properties Present at Hazardous Waste Sites.
K = Verschueren, K., 1983, Handbook of Environmental Data on Organic Chemicals, VanNostrand Reinhold, New York.
L = Arthur D. Little, Inc., 1989, The Installation Program Toxicology Guide, Volumes 1 to 4.

sorption to organic matter in soils or sediments may occur to a limited extent. Benzene has a liquid density less than water, and may form a separate phase above the water table if present in sufficient quantity (e.g., as part of a gasoline plume). Chlorobenzene has a density greater than water, and may form a separate phase (dense NAPL) if present in sufficient quantity. Dense NAPLs have been recognized in MW-2B. Vapor phase diffusion from groundwater may be a significant transport process in unsaturated soils.

Aromatic hydrocarbons can be biologically transformed in some soils and sediments (Barker *et al*. 1987). The rate and extent of transformation is highly dependent on site-specific factors such as temperature, pH, and the microbial composition of the soil. Monocyclic aromatic hydrocarbons are not appreciably concentrated in plant or animal tissues (EPA 1979).

Halogenated Aliphatic Hydrocarbons

The halogenated aliphatic hydrocarbons detected in the site groundwater are volatile, moderately to poorly water-soluble compounds (see Table 4-23). These chemicals are liquids at 20°C, with the exception of vinyl chloride, which is a gas at this temperature. These chemicals have densities greater than water; therefore, they may form a separate phase at the bottom of an aquifer if present in sufficient volume. Halogenated aliphatic hydrocarbons tend to be poorly adsorbed to soils and persistent in groundwater.

Transformation reactions have been reported for many of the chemicals in this class. Tetrachloroethane in anaerobic soils has been shown to be slowly transformed via a series of biotic and abiotic reactions to trichloroethene (TCE), 1,1-dichloroethene, 1,2-dichloroethene, and vinyl chloride (Bouwer and McCarty 1983; Vogel and McCarty 1985). Anaerobic biotransformation of 1,1,1-trichloroethane (TCA) to 1,1-dichloroethane has also been observed (Vogel *et al*. 1987). Little or no transformation of 1,2-dichloroethane was observed in anaerobic soils after 4 months of incubation (Bouwer and McCarty 1983). Researchers have reported transformation of vinyl chloride to carbon dioxide (CO_2) by acclimated anaerobic bacteria in laboratory experiments (Vogel and McCarty 1985). However, the frequent observation of vinyl chloride accumulation in aquifers contaminated with TCE and tetrachloroethene appears to indicate that conditions conducive to transformation of vinyl chloride to CO_2 are not prevalent in the environment. The rate and extent of transformation reactions are highly dependent on site-specific factors such as nutrient availability and microbial composition of the soil.

The ultimate fate of halogenated aliphatic hydrocarbons in surface soils and surface water is generally volatilization to the atmosphere and subsequent photooxidation (EPA 1979). These compounds are not significantly bioaccumulated.

Polycyclic Aromatic Hydrocarbons (PAHs)

The term polycyclic aromatic hydrocarbons describes a diverse class of chemicals consisting of two or more fused benzene rings, which vary widely in arrangement. PAHs are chiefly formed from the combustion of organic material and are widely present in the environment due to both natural and anthropogenic activities.

The physical properties of PAHs, which affect their mobility in the environment, depend largely on the sizes of the molecules; therefore, molecular weights are key properties. Vapor pressures and solubilities vary widely within the class and can be correlated with molecular weight. Volatilization may be a significant transport mechanism for a low molecular weight PAH such as naphthalene, but much less significant for the higher molecular weight compounds, such as ideno(1,2,3-c,d)pyrene.

The relatively high water solubilities of the lower molecular weight PAHs, such as naphthalene, acenaphthylene, acenaphthene, and fluorene, can result in some migration into groundwater. The higher molecular weight PAHs such as benzo(a)pyrene tend to adsorb onto soil surfaces, especially if the soil is high in organic carbon content. For this reason, overland transport of PAHs adsorbed onto soil particles is a typical mechanism of migration. PAHs associated with creosote or petroleum may also migrate through the unsaturated zone or through an aquifer as a nonaqueous plume.

All of the PAHs are capable of undergoing photolytic transformations in the environment, and their degradation half-lives vary widely.

Photolysis is a process that is greatly affected by site-specific variables such as intensity of sunlight, turbidity of water (if in a surface water environment), depth in surface water, or nature of the adsorbent.

Biodegradation of PAHs in the environment is extremely variable across the chemical class. The process is highly complex and depends on numerous factors such as the species of microorganisms, availability of nutrients, oxygen tension, degree of acclimation, nature of the medium, concentration of the chemical, temperature, and pH. In general, the di- and tricyclic PAHs are more readily biodegraded than the tetracyclic and higher polycyclic hydrocarbons (Tabak *et al*. 1981). The concentrations of the PAHs present can affect their ability to be biodegraded. Tabak *et al*. (1981) found that fluoranthene, pyrene, and chrysene underwent significant degradation at 5 mg/L substrate levels, but were less efficiently degraded at 10 mg/L. Coover and Sims (1987) studied the persistence of PAHs at 10°C, 20°C, and 30°C in an unacclimated agricultural sandy loam soil. The authors found that increasing the soil temperature significantly improved the rate and extent of apparent loss of low molecular weight PAHs, but had little effect on the loss of five- and six-ring PAHs. Volatilization may have contributed significantly to the loss of the low molecular weight PAHs such as acenaphthene, fluorene, phenanthrene, and anthracene. Because of numerous variables involved, different investigators have measured vastly different biodegratives half-lives for the same compound.

4.6.2.2 Environmental Chemistry of Metals of Potential Concern

Barium

In soils, barium is not expected to be very mobile because it forms water-insoluble salts and does not form soluble complexes with humic and fulvic materials (EPA 1984). Under acidic conditions, however, some of the water-insoluble barium compounds may become more soluble and partitioning to groundwater may occur.

In aquatic media, barium is likely to be present as suspended particulate matter or sediments. In the absence of any other possible removal mechanism, the residence time of barium in aquatic systems would be several hundred years (EPA 1984c).

Arsenic

Arsenic (As) has four stable oxidation states, +5, +3, 0, and -3. As(V) and As(III) are the most common oxidation states in aqueous environments. The two states are readily interconverted by biological and chemical redox reactions. Arsenates (As(V)) predominate in most soils, while arsenites (As(III)) may dominate in reducing environments. As(III) species are generally more mobile than As(V) in the subsurface. There is evidence that arsenic may leach into groundwater, especially from soils with low sorptive capacity (EPA 1984). The primary processes limiting the mobility of arsenic in soils are precipitation as metal salts, coprecipitation with iron or manganese oxides, substitution for phosphorus in soil minerals, and adsorption to amorphous metal oxides.

In highly reduced soils, or in the presence of certain microorganisms and fungi, arsenic can be reduced to arsine (AsH_3). Arsine can be metabolized by soil microorganisms to dimethyl arsine or trimethyl arsine, which are volatile and highly toxic compounds (Bodek et al. 1988). Loss of arsenic by volatilization has been observed in a number of soils (Cox 1975) in laboratory experiments and in the environment. Bioaccumulation factors for arsenic in aquatic organisms were reported to range from 5,000 to 6,000, and were highest at the lower trophic levels (EPA 1979).

Cadmium

Cadmium mobility in aqueous environments is controlled by the pH of the water, through formation of hydroxides: $CdOH^+$, $Cd(OH)_2$, $Cd(OH)_3$, and $Cd(OH)_4^{-2}$. The dominant species below pH 8 is the hydrated divalent cation, Cd^{+2} (Moore and Ramamoorthy 1984). As the pH increases, cadmium hydroxides will form and remove some of the cadmium from solution. Under reducing conditions, and in the presence of sulfur, the relatively insoluble cadmium sulfide (CdS) will form, and will control cadmium solubility (EPA 1979). Organic materials such as humic acids can influence the specification of cadmium by forming cadmium-organic complexes which increase the solubility of cadmium (Gardiner 1974).

The mobility of cadmium in soils and groundwater is influenced by several processes that result in a reduction of its mobility. Although it should be noted that in spite of these interactions, cadmium is among the most mobile elements in the environment. Sorption processes have more of an influence on cadmium mobility than do precipitation reactions (Kabata-Pendias and Pendias 1984). Cadmium mobility is reduced by cation exchange reactions with clays and organic matter, sorption to clay (Korte et al. 1976) and metal oxides (EPA 1979), and complexation with organic acids at low pH (Huang et al. 1977). Sorption increases at higher pH (Frost and Griffin 1977). To summarize, although cadmium can sorb onto soils and to a lesser extent metal oxides, clays, or carbonates, these processes are only favored under specific conditions (Bodek et al. 1988).

Cadmium in soil is readily taken up by plants, and is accumulated by organisms from food or water. Bioaccumulation factors for freshwater and marine organisms range from hundreds to thousands of times the ambient water concentrations (Agency for Toxic Substances and Disease Registry [ATSDR] 1987e). Cadmium taken up by feed crops may be bioaccumulated by livestock, and thus enter the human food chain. Biological transformation of cadmium does not appear to occur (EPA 1979).

Chromium

Chromium (Cr) occurs in two oxidation states in aqueous systems: Cf(III) and Cf(VI). The chemistries of Cr(III) and Cr(VI) are very different. Trivalent chromium (Cr[III]) reacts with hydroxide ion in water to form insoluble chromium hydroxide, $Cr(OH)_3 \cdot H_2O_n$, which is rapidly removed from water by precipitation and sorption to soils or sediments (EPA 1979). Hexavalent chromium (Cr[VI]) forms soluble chromate and dichromate anions, which are not strongly adsorbed to soils or sediments, and are therefore mobile in the environment (EPA 1979). Cr(VI) and Cr(III) may be interconverted in soils or surface waters under conditions that change the redox potential of the system (EPA 1979). Cr(VI) is rapidly reduced to Cr(III) in soils having a high content of organic matter, or by contact with ferrous or sulfide ions (Bartlett and Kimble 1976). Oxidation of Cr(III) to Cr(VI) has been shown to occur in the presence of excess manganese oxide (Bartlett and James 1979).

Chromium is accumulated in freshwater and marine biota to levels ranging from approximately 100 to 4,000 times the concentration in water (EPA 1979). The element can be transferred up the food chain, but does not appear to be magnified at higher trophic levels in a food chain consisting of phytoplankton, brine shrimp, post-larval fish, and mummichog clams (EPA 1979). Chromium does not appear to undergo biological transformation reactions such as methylation, but Cr(VI) may be chemically reduced to Cr(III) upon contact with plant or animal tissue.

Copper

Copper (Cu) can be found in three oxidation states: 0, +1, and +2. Of these three, only the Cu(II) oxidation state is found in aquatic systems. In polluted environments, copper can also form complexes with cyanide, amino acids, and humic substances. In the absence of organic complexing agents, hydrolysis and precipitation dominate copper's chemistry in aqueous environments.

The interactions of copper with organic materials in natural waters have been studied extensively. Organo-copper interactions result in the increased solubility of some copper-containing minerals and the subsequent transport of the organocupric complex (Rashid and Leonard 1973). Hydrous metal oxides can sorb copper and render it immobile (Jenne 1968). This sorption process occurs in competition with binding of other metals, and competitive adsorption could result in the release of copper. Copper is not very mobile in sediments. Adsorption, precipitation, and organic complexation are also important processes in soils (Kabata-Pendias and Pendias 1984). These processes render copper one of the least mobile metals.

Lead

The geochemistry of lead is dominated by its tendency to form sparingly soluble complexes with common anions such as hydroxide, sulfide, and carbonate (EPA 1979). The divalent cation forms of lead (Pb^{+2}) are the most common in the natural environment. In unpolluted waters, lead solubility is generally controlled by the formation of lead oxide, lead sulfate, and lead carbonate solids. Lead's solubility can be increased through the formation of soluble complexes with naturally-occurring organic acids. In the presence of organic acids, lead can remain complexed to fulvic acid at pH values as low as 3, whereas, in the absence organic acids, insoluble carbonates or sulfates would be formed (Guy and Chakrabarti 1976).

The predominant fate of lead in the environment is sorption to soils and sediments. Carey et al. (1980) found that lead concentrated in surface soils either through the formation of precipitates such as lead oxide or lead carbonate or through sorption reactions. Adsorption of lead to soils is pH dependent, increasing with increasing pH. Above pH 7, essentially all lead in soil is sorbed (Huang et al. 1977). Korte et al. (1976) found that lead was virtually immobile in all but sandy soils. Thus, under most conditions, lead would not be expected to migrate to groundwater.

Lead can be taken up by a variety of plants, fish, and other organisms. Freshwater and marine plants and animals were observed to concentrate inorganic lead at levels 100 to 1,000 times above the surrounding water's concentration (EPA 1979). Lead uptake increases with decreasing pH, due to increased release of the element from sediments. Inorganic lead does not appear to be biomagnified in the food chain due to interactions with soil materials which render inorganic lead unavailable to biological organisms (EPA 1979). Lead can be methylated by microorganisms in anaerobic sediments to form the compound tetramethyl lead, which is moderately volatile and mobile in the environment (Wong et al. 1975).

Manganese

Manganese (Mn) can occur in all valence states from -3 to +7. Mn(II) is very common and forms many salts. The solubility of manganese depends upon the pH of the aquatic environment. Mn(III) and Mn(IV) are only slightly soluble, though under neutral pH conditions these species may be reduced to Mn(II) which is more soluble and more mobile. Manganese can be chelated by a variety of organic and inorganic ligands. These ligands tend to keep manganese in solution. The soluble fraction of manganese ranges from 15 to 95% of the total, and is not dependent on pH, alkalinity, specific conductivity, or concentration of humic substances in water (Laxen et al. 1984). Soil-bound manganese can be dissolved by organic acids (Pohlman and McCall 1986).

Mercury

Mercury (Hg) can be found in three oxidation states in the environment. Elemental mercury, Hg(0), is a liquid at ordinary temperatures. Mercurous mercury, Hg(I), occurs primarily as Hg_2^{2+} under environmental conditions and does not form hydroxides, oxides, or sulfides. Mercuric mercury, Hg(II), forms stable complexes with common ligands. Some of these complexes are fairly soluble while others are quite insoluble (EPA 1979). Mercuric hydroxide is not found in aquatic systems, but mercuric oxide is found and is soluble. In the presence of sulfide ion, mercuric sulfide (HgS) will precipitate from solution (Cotton and Wilkinson 1982).

Elemental mercury is the predominant species in a moderately oxidizing environment of about pH 5. Mercury becomes more soluble in the presence of chloride. Under mildly reducing conditions, HgS, which has an extremely low water solubility, will form (EPA 1979).

The dominant process controlling mercury transport in the aquatic environment is sorption to particulates and sediments. The binding capacity of the sediment is related to its organic content; pH does not affect the sorption process. Desorption does not occur readily; therefore, mercury will tend to accumulate in sediments (Ramamoorthy and Rust 1978).

Mercury can undergo microbial alkylation to methyl or dimethyl mercury in anaerobic or aerobic sediments. Organic forms of mercury exist in dynamic equilibrium with inorganic forms of mercury in natural waters. Methylation is of extreme importance in the environmental fate of mercury, because methylate mercury compounds are more water soluble, more easily absorbed through biological membranes, and bioaccumulated within animal tissues to a far greater extent than inorganic mercury.

Mercury has a high vapor pressure, and volatilization is a significant transport pathway for the metal. The rate of volatilization of mercury and its inorganic compounds from aquatic systems decreases in the order Hg > HG_2Cl_2 > $HgCl_2$ > HgS > HgO. In soils, volatilization was found to increase with temperature and alkalinity (EPA 1979).

Mercury's mobility in soils depends on the soil environment. In a study of the binding of trace metals to eleven soils, mercury was found to be moderately to highly mobile (Korte *et al*. 1976). The clay content of soils does not significantly affect sorption (Kabata-Pendias and Pendias 1984).

Nickel

Nickel is continuously transferred between air, water or groundwater, and soil by natural, chemical, and physical processes. Nickel aerosols are removed from the atmosphere primarily by wet and dry deposition. The average residence time for nickel in the atmosphere is 7 days, during which long-distance transport can occur. Nickel is extremely persistent in both water and soil.

In aquatic environments, nickel may exist in solution as hydroxide, carbonate, sulfate, and organic complexes (EPA 1979). Some of the metal in solution may be coprecipitated with hydrous metal oxides, or sorbed onto organic material.

Soils containing relatively high concentrations of iron and manganese oxides may sorb nickel significantly, while soils rich in organic matter may enhance the mobility of nickel through complexation (EPA 1985a). The average residence time of nickel in soil is estimated to be 2,400 to 3,500 years (Nriagu 1980).

Zinc

Zinc is a metal belonging to Group IIB of the periodic table. Its natural valence states are 0 and 2+. Zinc forms a variety of inorganic compounds and a number of compounds with organic ligands. The element is amphoteric and forms both acidic and basic salts.

Zinc introduced into the aquatic environment is partitioned into sediments through sorption onto hydrous iron and manganese oxides, clay minerals, and organic material; a small part may be partitioned into the aquatic phase through speciation into soluble zinc compounds. Sorption of zinc is probably the dominant fate of zinc in the aquatic environment.

Information regarding the fate of zinc in soil is inadequate. However, zinc is likely to be strongly sorbed onto soil. Soil conditions not amenable for the sorption of zinc may lead to the leaching of zinc. The tendency of zinc to be sorbed is affected by the pH and salinity of soils. Decrease of pH (<7) and increase of soil salinity favors desorption (EPA 1980).

The bioconcentration factors (BCFs) for zinc in aquatic organisms have been determined by several investigators (EPA 1980). BCFs for zinc in edible portions of aquatic organisms have been found to vary from 43 in the soft-shell clam (Mya arenaria) to 16,700 in the oyster (Crassostrea virginica) (EPA 1980).

Cyanide

Hydrogen cyanide (HCN) gas and soluble cyanide salts are the most mobile forms of cyanide in the environment (EPA 1984b). Residence times in soil and water are variable and influenced by such factors as pH, wind speed, and cyanide concentration (EPA 1979).

A pH less than 9.2 is considered more conducive to environmental transport. Volatilization is considered a major fate process from surface water and surficial soils.

A number of common plants may bioaccumulate large quantities of cyanogenic glycoside. Hydrolyzation may occur upon ingestion, forming free HCN, resulting in toxicity.

In aquatic environments, hydrogen cyanide is not expected to undergo direct photolysis, chemically hydrolyze, adsorb significantly to suspended sediments, or bioaccumulate significantly in aquatic organisms (EPA 1979). Biodegradation will occur unless cyanide concentrations are at levels toxic to microorganisms. High concentrations of cyanides in subsurface soils (i.e., landfills, spills) may leach into groundwater (EPA 1984b).

4.6.3 Transport of Contaminants of Concern

The major medium of contaminant transport at the Buffalo Avenue site is groundwater. In turn, the flow velocity and direction of the groundwater are dictated by the hydrogeologic parameters of the saturated matrix. Groundwater that migrates through the saturated overburden travels at a lower velocity than groundwater that moves through fractured rock. This effect is due to the reduction of effective porosity in the bedrock, and causes compounds to migrate more quickly. Fracture frequency and connectedness determines the directions of contaminant migration.

As stated in previous sections, the principle receptors of contaminants at Buffalo Avenue are overburden and bedrock groundwater. As the velocity of the groundwater increases with depth in the bedrock, transport through the Lockport Dolomite, and potential migration off site through this media, become of primary consideration. Velocities of groundwater through the various media are listed in Table 4-24.

Contaminant migration through the saturated media will be affected by retardation (adsorption) of the compound. The retardation factor reduces the velocity at which the compound migrates through the media by its reciprocal. For example, benzene has a retardation factor of 3.49 to 10.96 in fractured limestone. It will therefore migrate at 29% to 9.2% of the rate that groundwater flows through the bedrock. Retardation factors for significant compounds at Buffalo Avenue have been calculated based on the following equation from Freeze and Cherry (1979):

$$R = 1 + \frac{P}{\Theta} K_d$$

where:

P = bulk density of soil 1.75 gm/cm^3, assumed
Θ = effective porosity assumed to be 0.01
K_d = (organic carbon partition coefficient) (organic carbon fraction; assumed to range from 0.25% to 1%)

The retardation factors and transport velocities of selected compounds are given in Table 4-24.

4.6.4 Contaminant Loading

Rates of off-site loading were calculated for the five groundwater-bearing zones at the site. These rates are based on estimations of

Table 4-24

RETARDATION FACTORS AND TRANSPORT VELOCITIES OF SELECTED COMPOUNDS IN GROUNDWATER (WORST SEASONAL CASE ESTIMATES)

Zone	Compound	Retardation Factor	*Transport Velocity (ft/day)
A	Benzene	2.1 - 5.8	0.64 - 0.23
	Chlorobenzene	5.25 - 18.0	0.26 - 7.5 x 10^{-2}
	1,2-Dichlorobenzene	22.9 - 88.6	5.9 x 10^{-2} - 1.5 x 10^{-2}
	1,3-Dichlorobenzene	22.9 - 88.6	5.9 x 10^{-2} - 1.5 x 10^{-2}
	1,4-Dichlorobenzene	22.9 - 88.6	5.9 x 10^{-2} - 1.5 x 10^{-2}
	1,2,4-Trichlorobenzene	119.5 - 474.8	1.1 x 10^{-2} - 3 x 10^{-3}
B	Benzene	3.49 - 10.96	2.67 - 0.85
	Chlorobenzene	10.9 - 40.6	0.85 - .23
	1,2-Dichlorobenzene	52 - 205	0.18 - 4.5 x 10^{-2}
	1,3-Dichlorobenzene	52 - 205	0.18 - 4.5 x 10^{-2}
	1,4-Dichlorobenzene	52 - 205	0.18 - 4.5 x 10^{-2}
	1,2,4-Trichlorobenzene	277 - 1105	3.4 x 10^{-2} - 8 x 10^{-3}
C	Benzene	3.49 - 10.6	1.83 - 0.60
	Chlorobenzene	10.9 - 40.6	0.59 - 0.16
	1,2-Dichlorobenzene	52 - 205	0.12 - 3.1 x 10^{-2}
	1,3-Dichlorobenzene	52 - 205	0.12 - 3.1 x 10^{-2}
	1,4-Dichlorobenzene	52 - 205	0.12 - 3.1 x 10^{-2}
	1,2,4-Trichlorobenzene	277 - 1105	2.3 x 10^{-2} - 6.0 x 10^{-3}
CD	Benzene	3.49 - 10.6	3.51 - 1.14
	Chlorobenzene	10.9 - 40.6	1.13 - 0.30
	1,2-Dichlorobenzene	56 - 205	0.24 - 6.0 x 10^{-2}
	1,3-Dichlorobenzene	52 - 205	0.24 - 6.0 x 10^{-2}
	1,4-Dichlorobenzene	52 - 205	0.24 - 6.0 x 10^{-2}
	1,2,4-Trichlorobenzene	277 - 1105	4.4 x 10^{-2} - 2.4 x 10^{-2}
F	Benzene	3.49 - 10.96	6.56 - 2.1
	Chlorobenzene	10.9 - 40.6	2.1 - 0.56
	1,2-Dichlorobenzene	52 - 205	0.44 - 0.11
	1,3-Dichlorobenzene	52 - 205	0.44 - 0.11
	1,4-Dichlorobenzene	52 - 205	0.44 - 0.11
	1,2,4-Trichlorobenzene	277 - 1105	0.08 - 2.1 x 10^{-2}

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*Effective porosity value used in velocity estimates was 0.01.

seasonal groundwater gradient, average hydraulic conductivity, cross-sectional area, and concentration values of samples taken from monitoring wells. Loading rates for boundary wells were then totaled for exit areas of the site (northern and western boundaries).

The following equation was used to calculate daily loading off site from groundwater:

$$\Sigma \text{ Loading}_{gw} = (KIA)(8.345 \text{ lbs/gallon})(C)$$

where:

K = Average hydraulic conductivity of selected wells;
I = Hydraulic gradient (seasonal);
A = Cross sectional area;
C = Concentration value in ppb; and
Σ = Sum of loading from all wells considered.

Loading estimates are thus based on the specific hydraulic characteristics of the boundary wells, as well as the chemistry of samples extracted from these wells. The calculated loading rates for the five groundwater-bearing zones are listed in Appendix D.

Loading in the A-Zone follows the same trend as compound concentration values, with the highest rates occurring for benzene, chlorobenzene, the three dichlorobenzene derivatives, and 1,2,4-trichlorobenzene. The western boundary, near the source area, resulted in heaviest loading values. Metals exhibited a general loading range with no specific metal showing comparatively high loading. Total (worst case) loading for all considered compounds and metals at the western boundary is 0.289 lbs/ day. Total (worst case) loading for the same compounds and metals at the northern boundary is 0.011 lb/day. Specific compound-loading rates for all water-bearing zones are given in Appendix D.

B-Zone loading is greater comparatively than the A-Zone. Worst case, total loading in the western boundary wells is 13.92 lb/day. This result is heavily weighted by the high hydraulic conductivity value found in well MW-1B, which constitutes the western boundary in this zone. Northern boundary (worst case) loading is 0.50 lb/day. Benzene, the three dichlorobenzene derivatives, 1,2,4-trichlorobenzene, as well as manganese and zinc exhibit the heaviest individual loading.

C-, CD- and F-Zone gradients were calculated using water levels from the neighboring DuPont wells. All loading calculations were done for the MW-5 nest of wells, which are on the northern boundary of the site. Total worst case loading for these zones shows an increase with depth. The C-Zone shows 0.5724 lb/day; the CD-Zone exhibits 1.1 lb/day; and the F-Zone displays a 7.43 lb/day rate. Major chemicals in these zones include benzene, chlorobenzene, the three dichlorobenzene derivatives, and 1,2,4-trichlorobenzene. Values seen in these zones for trichloroethene and its degradation products, total 1,2-dichloroethene and vinyl chloride, probably have an off-site origin and are not considered in these calculations.

5. RISK ASSESSMENT

5.1 INTRODUCTION

This risk assessment is based upon the data generated by implementation of the PP/NYSDEC-approved work plan. It analyzes the potential environmental and public health threats from the contaminants present in groundwater at the 3163 Buffalo Avenue site. The site and surrounding area is used for industrial activities. The NYSDEC office of Regulatory Affairs and the Wildlife Resources Center were contacted for information pertaining to sensitive habitat and wetland designations for the area. Neither the site nor the surrounding areas are so designated. Therefore, the assessment focused on public health threats. Potential future land uses for the site were reviewed to evaluate potential routes of human exposure to site contaminants. Three potential exposure pathways that appeared to present the greatest potential exposure to groundwater contaminants from the site were identified and evaluated. These were:

- Inhalation of volatile contaminants in the gorge near the Falls Street Tunnel outfall, which is near the Rainbow Bridge;
- Exposure of fishermen using the lower Niagara River; and
- Use of Niagara River water emptying into Lake Ontario as a source of potable water.

This assessment has been prepared in a series of seven sequential and interacting modules necessary to evaluate a complex array of data and information (see Figure 5-1):

- Evaluation of background and environmental data;

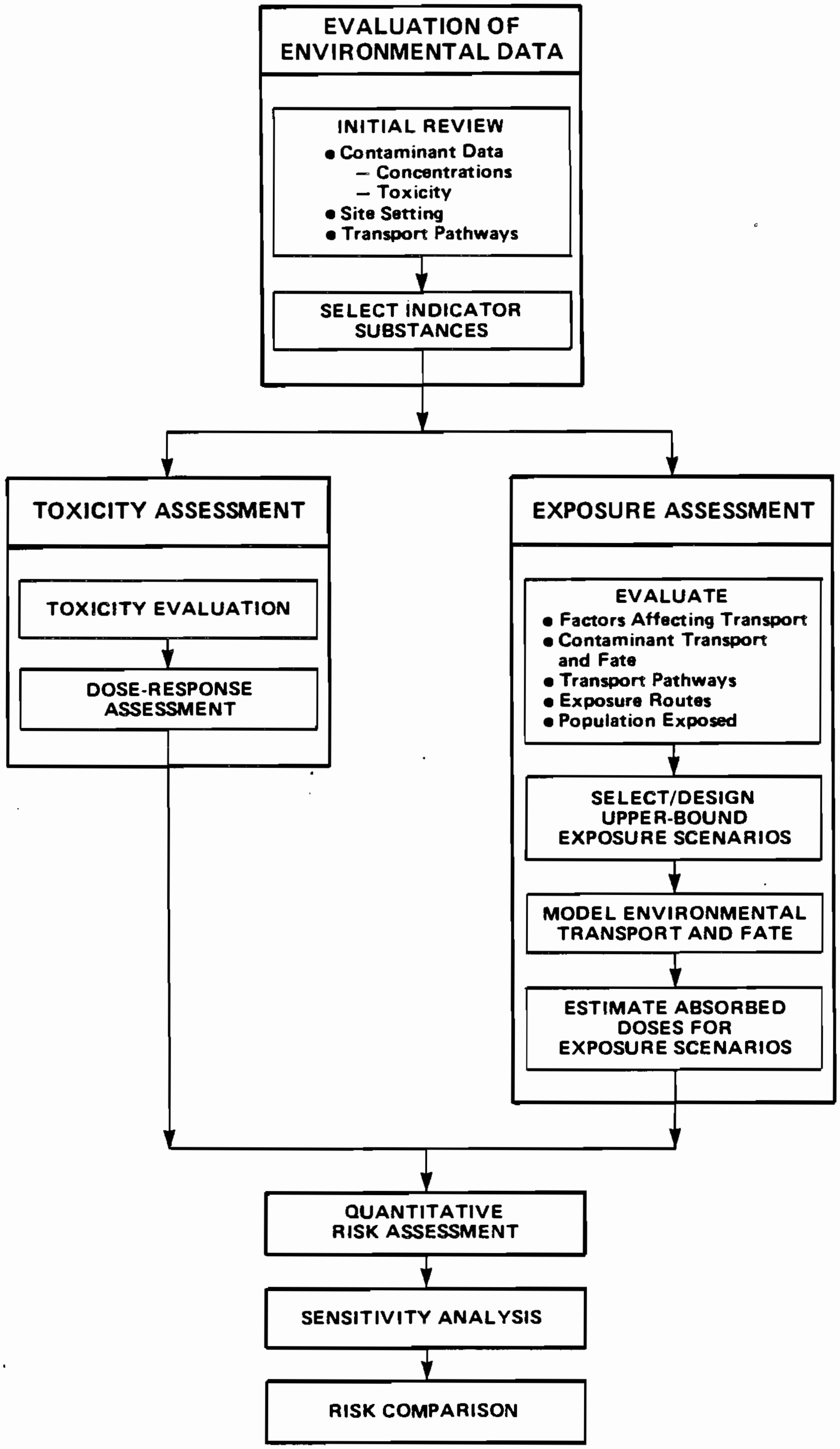

Figure 5–1 OVERVIEW OF THE PUBLIC HEALTH RISK ASSESSMENT

- Selection of indicator chemicals;
- Toxicity assessment of indicator chemicals;
- Exposure assessment;
- Quantitative risk assessment;
- Sensitivity analysis; and
- Risk comparison.

This follows the National Academy of Sciences (1983) description of the risk assessment process and is consistent with EPA guidelines. Sections 5.2 through 5.9 and Appendices E and F address each of these components in detail.

Sections 5.2 and 5.3 address the first risk assessment module, site contamination assessment, by describing:

- Extent of contamination; and
- Selection of indicator chemicals.

Two separate but parallel modules follow that assess potential exposure and toxicity. Sections 5.4 and 5.5 evaluate potential exposures to site-derived contaminants and provide an estimation of absorbed doses for input into the quantitative risk estimate (Section 5.7). Section 5.6 summarizes the toxicological properties of each indicator chemical. Further details are given in Appendix E which presents profiles for each indicator chemical. The purpose of each profile is to provide a brief but thorough overview of the toxicological parameters of each indicator chemical or class of chemicals. The profiles consider:

- Acute, chronic, and, where appropriate, subchronic toxicity;
- Reproductive and developmental effects;
- Mutagenesis;
- Weight-of-evidence carcinogenicity classification;
- Carcinogenesis; and

- Dose-response indices (e.g., reference doses [RfDs], acceptable daily intakes [ADIs], cancer slope factors [SFs], etc.).

Section 5.7 presents the fourth module, quantitative risk characterization in which the exposure and toxicity assessment developed in Sections 5.5 and 5.6 are combined.

Section 5.8 discusses the uncertainties in the risk assessment process. It reviews the major assumptions of the risk assessment and evaluates the sensitivity of the estimated risk levels (of hazard indices) to changes in the assumptions. Section 5.9 compares the estimated site-derived risks, as appropriate, to typical risks of living, and background exposures from food and environmental media.

5.2 ASSESSMENT OF SITE CONTAMINANTS

This section presents data regarding the extent and level of contamination at the 3163 Buffalo Avenue site. Two media--soils and groundwater--were analyzed for contamination, with the express purpose of defining potential source soils and the extent of existing groundwater contamination.

Tables 5-1 and 5-2 summarize the analytical results of organic and inorganic compounds detected in soil. As presented in Table 5-1, there are two classes of compounds which account for the bulk of the organic chemical contamination in soil. The first of these is composed of monocyclic aromatic hydrocarbons (MAHs) that are widely distributed in site soils subject to sampling in this investigation. Benzene, a nonchlorinated member of the class, was detected in 28% of the soil samples at concentrations ranging upwards to 120,000 µg/kg. Six chlorinated members of the class--chlorobenzene, three dichlorobenzene isomers, 1,2,4-trichlorobenzene, and hexachlorobenzene--were widely distributed in site soils sampled. Chlorobenzene was detected in 32% of the samples at concentrations ranging upwards to 1,500,000 µg/kg; whereas the three dichlorobenzene isomers were detected in 40 to 60% of the samples, with maxima ranging from 240,000 to 2,400,000 µg/kg. 1,2,4-Trichlorobenzene was detected in 64% of samples at concentrations upwards to 2,100,000 µg/kg and hexachlorobenzene in a single sample (4%) at a concentration of 1,600,000 µg/kg.

Table 5-1

ORGANIC COMPOUNDS DETECTED IN SOILS AT THE 3163 BUFFALO AVENUE SITE

Chemical	Count	Frequency of Detection	Concentration (ug/kg) Mean	Maximum
MONOCYCLIC AROMATIC HYDROCARBONS				
Benzene	7	28%	37,263	120,000
Chlorobenzene	8	32%	187,505	1,500,000
1,2-Dichlorobenzene	18	72%	631,533	2,400,000
1,3-Dichlorobenzene	10	40%	89,590	240,000
1,4-Dichlorobenzene	17	68%	443,858	1,900,000
1,2,4-Trichlorobenzene	20	80%	612,014	2,100,000
Hexachlorobenzene	1	4%	1,600,000	1,600,000
KETONES				
Acetone	1	4%	300	300
Isophorone	1	4%	3,100	3,100
PHENOLS				
2,4-Dimethylphenol	1	4%	640	640
2-Methylphenol	1	4%	640	640
4-Methylphenol	1	4%	920	920
Phenol	1	4%	6,100	6,100
CHLORINATED PESTICIDES				
Beta-Hexachlorocyclohexane	1	4%	300	300
POLYCYCLIC AROMATIC HYDROCARBONS				
Benzo(a)anthracene	4	16%	3,690	7,700
Benzo(a)pyrene	6	24%	4,292	6,300
Benzo(b)fluoranthene	8	32%	9,223	14,000
Chrysene	3	12%	4,773	8,500
Indeno(1,2,3-c,d)pyrene	6	24%	2,933	3,200
NONCARCINOGENS				
Anthracene	2	8%	2,850	4,700
Benzo(g,h,i)perylene	6	24%	2,417	5,200
Dibenzofuran	1	4%	2,100	2,100
Fluoranthene	5	20%	7,352	17,000
Fluorene	1	4%	2,600	2,600
2-Methylnaphthalene	1	4%	2,300	2,300
Naphthalene	1	4%	2,600	2,600
Phenanthrene	5	20%	6,508	29,000
Pyrene	5	20%	6,040	12,000
OTHERS				
Carbon disulfide	5	20%	678,700	2,200,000

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Source: Ecology and Environment, Inc. 1990.

Table 5-2

INORGANIC COMPOUNDS DETECTED IN SOILS AT 3163 BUFFALO AVENUE (mg/kg)

Compound	Count/26	Frequency of Detection	Concentrations		Maximum Expected Background
			Mean	Maximum	
Aluminum	26	100%	9,961	30,100	90,100
Antimony	1	3.8%	75.3	75.3	2.2
Arsenic	26	100%	15	104	22.5
Barium	26	100%	2,004	19,200	1,182
Beryllium	2	7.6%	0.48	.48	2.5
Cadmium	25	96%	11.7	74.8	10
Calcium	26	100%	59,147	143,000	21,630
Chromium	26	100%	132	1,530	159
Cobalt	4	15%	30.7	61.6	38
Copper	26	100%	676.8	8,180	71
Iron	26	100%	19,169	120,000	79,310
Lead	26	100%	2,246	31,600	420
Magnesium	25	96%	16,946	69,700	16,880
Manganese	26	100%	506	1,480	2,330
Mercury	25	96%	29	443	94
Nickel	23	88%	69.6	435	54
Potassium	5	19%	2,716	4,900	7,480
Selenium	1	3.8%	13.4	13.4	1.3
Silver	1	3.8%	16.4	16.4	3.0
Sodium	2	7.6%	3,340	5,260	30,225
Thallium	2	7.6%	.51	.52	16.3
Vanadium	25	96%	29	89.4	155
Zinc	26	100%	5,916	56,000	137
Cyanide	11	42%	14.3	82.1	NA

02[IL]JD1900:D3048/1156/19

Source: Ecology and Environment, Inc. 1990.

The second major class, PAHs were detected less extensively in site soils than the MAHs, typically in 12 to 24% of the samples. Maximum concentrations of individual PAHs ranged from 2,100 to 29,000 µg/kg.

Carbon disulfide was detected in 20% of the soil samples at concentrations ranging from 676,000 to 2,200,000 µg/kg.

Members of three other classes--two ketones, five phenols, and one chlorinated pesticides, were detected only in a single sample, concentrations of most of which were below 1,000 µg/kg.

Table 5-3 presents the mean and maximum expected concentrations of metals in soils in the eastern United States. Table 5-2 presents the concentrations of metals in 3163 Buffalo Avenue site soils. As presented in this latter table, concentrations of a number of metals exceed the upper 50th percentile for background concentrations. Most notably, the maximum concentrations of arsenic (104 mg/kg), is in 5-fold excess of background, barium (19,200 mg/kg) is in 18-fold excess of background, cadmium (74.8 mg/kg) is in five-fold excess of background, chromium (1,530 mg/kg) is in 10-fold excess of background, lead (31,600 mg/kg) is in 80-fold excess of background, mercury (443 mg/kg) is in 1,000-fold excess of background, and zinc (56,000 mg/kg) is in 40-fold excess of background. With the exception of chromium, mean concentrations of all the above metals also exceeded their maximum expected concentrations.

Tables 5-4 and 5-5 present corresponding concentrations of organic and inorganic compounds in 3163 Buffalo Avenue site groundwater. Unpublished total metal concentrations in unfiltered water samples collected from overburden wells above the Lockport in Niagara County for a NYSDEC Phase II investigation are also shown in Figure 5-5 to provide a reference for expected background metal concentrations. Primary contaminants in groundwater fall into two classes--chlorinated aliphatic hydrocarbons and monocyclic aromatics. Five chlorinated aliphatics were detected in 11.1% to 38.9% of the samples. 1,2-Dichloroethane was detected in 11.1% of the samples with a maximum of 54 µg/L; 1,2-dichloroethene was detected in 38.9% of the samples with a maximum of 2,400 µg/L. Tetrachloroethene was detected in 22.2% of the samples, trichloroethane in 16.7% of the samples, and vinyl chloride in 11.1% of the samples with maxima of 820, 250 and 980 µg/L, respectively.

Table 5-3

MEAN AND MAXIMUM EXPECTED CONCENTRATIONS OF ELEMENTS USED TO EVALUATE INORGANIC SOIL CONTAMINANTS AT THE 3163 BUFFALO AVENUE SITE

Element	Geometric Mean (mg/kg)	Geometric Deviation	Maximum Expected Concentration (mg/kg) (MxD*)
Aluminum	33,000	2.87	90,100
Antimony	0.52	2.38	2.2
Arsenic	4.8	2.56	22.5
Barium	290	2.35	1,182
Beryllium	0.55	2.53	1,182
Cadmium	<1[++]	--	2.5
Calcium	3,400	2.08	21,630
Chromium	33	2.6	159
Cobalt	5.9	2.57	38
Copper	13	2.8	7
Iron	14,000	2.87	79,312
Lead	14	1.95	42.0
Magnesium	2,100	3.55	16,880
Manganese	260	3.82	2,360
Mercury	0.081	2.52	0.4
Nickel	11	2.64	54
Potassium	12,000	0.75	7,480
Selenium	0.30	2.40	1.30
Silver	<0.5	--	3.0[+++]
Sodium	2,500	4.55	30,225
Thallium	7.7	1.58	16.3
Vanadium	43	2.51	195
Zinc	40	2.11	137

02[IL]JD1900:D3048/1155/29

[+] Unless otherwise noted, data adapted from Shacklette and Boerngen (1984).

[++] NAS (1978).

[+++] Connor and Shacklette (1975).

*(Mean)(standard deviation)(1.645), corresponding to 90th percentile of the range and used to estimate the upper-bound range of variation expected for an element in the material being studied.

Source: Ecology and Environment, Inc. 1990.

Table 5-4

ORGANIC COMPOUNDS DETECTED IN GROUNDWATER AT THE 3163 BUFFALO AVENUE SITE

			Concentration (µg/L)	
	Count	Frequency of Detection	Mean	Maximum
CHLORINATED ALIPHATIC HYDROCARBONS				
1,2-Dichloroethane	2	11.1%	33	54
1,2-Dichloroethene (total)	7	38.9%	969	2,400
Tetrachloroethene	4	22.2%	434	820
Trichloroethene	3	16.7%	153	250
Vinyl chloride	2	11.1%	890	980
MONOCYCLIC AROMATICS				
Unchlorinated				
Benzene	10	55.5%	97,123	540,000
Toluene	3	16.7%	371	960
Xylenes (total)	1	5.6%	150	150
Chlorinated				
Chlorobenzene	12	66.6%	24,510	180,000
1,2-Dichlorobenzene	17	94.4%	13,280	120,000
1,3-Dichlorobenzene	14	77.7%	2,800	20,000
1,4-Dichlorobenzene	17	94.4%	9,010	75,000
1,2,4-Trichlorobenzene	14	77.7%	7,028	47,000
KETONES				
4-Methyl-2-pentanone (MIBK)	1	5.6%	380	380
Phenols				
Phenol	3	16.7%	660	1,100
2-Chlorophenol	2	11.1%	285	300
PHTHALATES				
bis(2-Ethylhexyl)phthalate	2	11.1%	12	12
CHLORINATED PESTICIDES				
Hexachorocyclohexane				
alpha isomer	6	33.3%	.98	2.8
beta isomer	4	22.2%	.61	.86
delta isomer	1	5.6%	1.0	1.0
gamma isomer	1	5.6%	.92	.92

02[IL]JD1900:D3048/1157/18

Source: Ecology and Environment, Inc. 1990.

Table 5-5

INORGANIC COMPOUNDS DETECTED IN GROUNDWATER AT THE 3163 BUFFALO AVENUE SITE

	Detection Frequency	Number of Samples	Concentration (µg/L)		Background Levels
			Mean	Maximum	
Aluminum	17 (94%)	18	27,508	74,200	1,540
Arsenic	14 (78%)	18	22	69.7	1.2
Barium	8 (44%)	18	1,854	8,350	37.8
Cadmium	10 (56%)	18	24	46.2	5
Calcium	18 (100%)	18	634,711	1,360,000	152,000
Chromium	17 (94%)	18	156	323	10
Cobalt	6 (33%)	18	74	131	12
Copper	11 (61%)	18	295	861	15
Iron	18 (100%)	18	126,277	348,000	2,180
Lead	17 (94%)	18	490	3,430	5.6
Magnesium	18 (100%)	18	128,078	435,000	52,200
Manganese	18 (100%)	18	2,419	7,270	446
Mercury	10 (56%)	18	24	216	0.2
Nickel	9 (50%)	18	278	1,280	10
Potassium	18 (100%)	18	23,584	164,000	3,540
Sodium	18 (100%)	18	231,656	1,380,000	61,800
Vanadium	9 (50%)	18	89	367	10
Zinc	18 (100%)	18	1,895	3,580	25.2
Cyanides	13 (72%)	18	500	2,540	10

02[IL]JD1900:D3048/1158/21

Source: Ecology and Environment, Inc. 1990.

As was the case for soils, monocyclic aromatics were the dominant contaminants in groundwater. Most notable among these contaminants were benzene detected in 55.5% of the samples at concentrations ranging upwards to 540,000 µg/L and five mono- through tri-chlorinated benzenes, which were detected in 66.6 to 94.4% of the samples at maximum concentrations ranging from 20,000 to 180,000 µg/L. Members of other classes--ketones, phenols, phthalates, and various isomers of hexachlorocyclohexane (chlorinated pesticides)--were detected in significantly fewer samples and lower concentrations than members of the two dominant chemical classes. Carbon disulfide was found in a sample of dense NAPL from monitoring well 2B, but it has not been detected in aqueous phase groundwater samples, including the sample from well 2B.

The groundwater at the site is classified GA by the NYSDEC, indicating that its best potential use is as a drinking water supply source. However, the site areas and the City of Niagara Falls are served by a public water supply system. Consequently, the groundwater in the vicinity of the site is not presently used as a drinking water source, nor is it likely to be in the future. Since there is no current or expected future exposure potential, human health effects resulting from use of groundwater at the site will not be quantitatively evaluated in this assessment.

5.3 SELECTION OF CONTAMINANTS OF CONCERN

As part of the risk assessment process, contaminants which pose the highest risk to potential receptors (indicator chemicals) are identified. This selection process incorporates data from previous sections of this report, including the transport and fate and assessment of site contaminants (Section 4). Of the chemicals identified as contaminants, some exhibit a higher degree of toxicity than others, occur more frequently, occur at higher concentrations, or are more mobile. Contaminants that fall into one or more of these categories are of greater concern in regard to potential risk to human health, and therefore, become the focus of the assessment.

As discussed above, the primary purpose of the RI was to evaluate the threat of contamination of groundwater at the 3163 Buffalo Avenue site. With this purpose in mind, soil sampling was conducted and subsurficial samples composited to identify potential sources of groundwater contamination. Given the scope of work, surface soils were not evaluated separately. In light of the industrialized nature of the area in which the site is located, future use of the site property is likely to remain the same. For both these reasons, potential exposures to contaminants in soils will not be evaluated. Selection of indicator chemicals for the risk assessment will be based on contaminants detected in the site groundwater.

The contaminants of concern (or indicator chemicals) selected from groundwater contaminants are shown in Table 5-6. Also listed are the criteria upon which this selection was made. Representatives from four chemical classes, inorganics, chlorinated aliphatics, monocyclic aromatic hydrocarbons, and chlorinated pesticides were chosen.

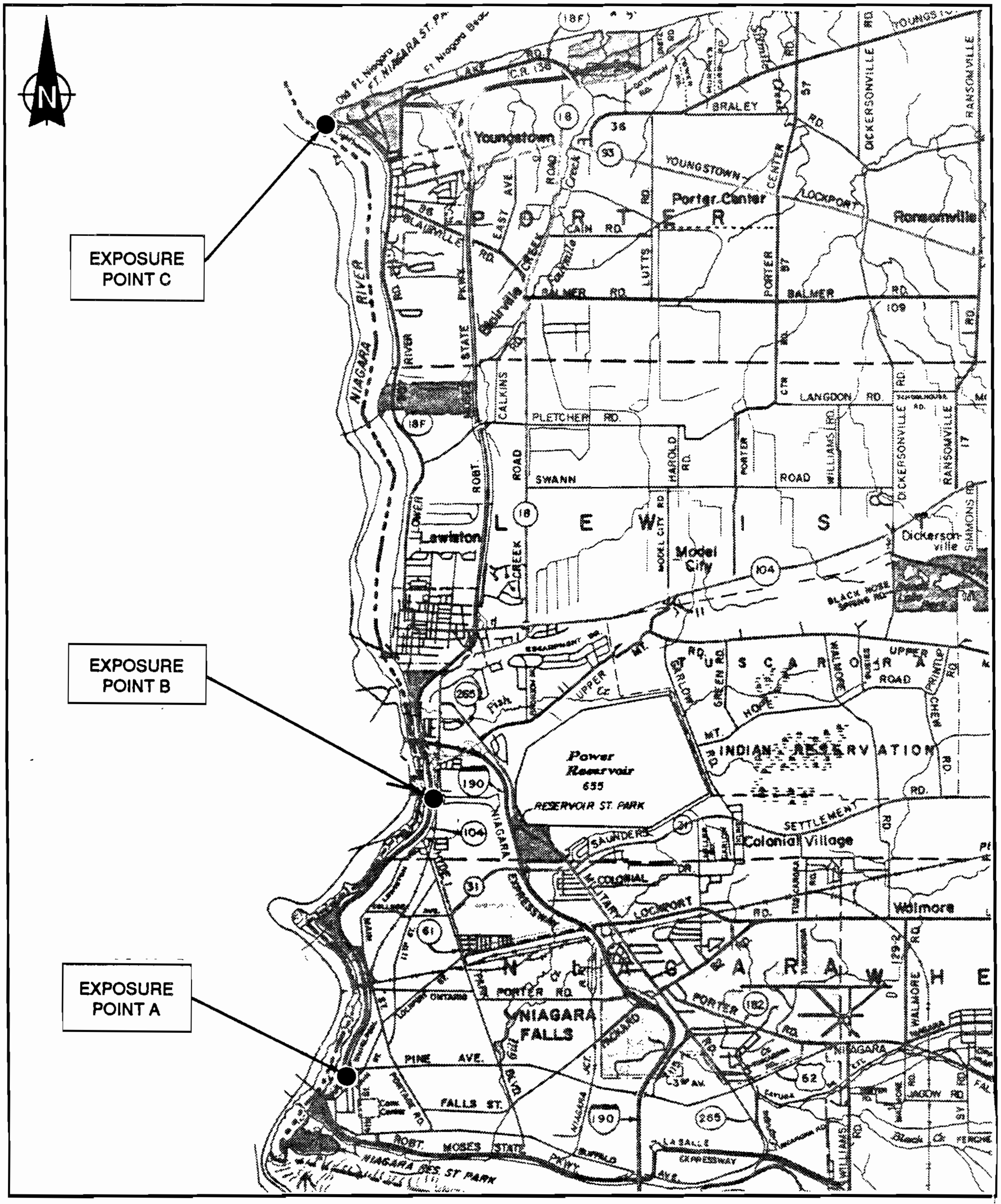
5.4 CONTAMINANT MIGRATION AND EXPOSURE PATHWAYS

5.4.1 Contaminant Migration Pathway

Subsurface soil and groundwater contaminants are expected to migrate via the groundwater to the Niagara River, then via the river to potential receptor exposure points. The groundwater flow patterns at the site and in the surrounding area, and factors affecting contaminant migration via the groundwater, have been discussed in detail in previous sections. Briefly, contaminants are believed to leave the site from its northern and western boundaries, then travel generally north and west with the regional groundwater flow until the groundwater reaches the Falls Street Sewer Tunnel. This tunnel intercepts the groundwater, which then flows along the Falls Street Tunnel, and to the Niagara River. The Falls Street Tunnel discharges to the Niagara River between the American Falls and the Rainbow Bridge. This expected contaminant migration pathway is shown in Figure 5-2.

5.4.2 Potential Exposure Pathways

The potential exposure pathways expected to result in the greatest potential exposure to possible receptors are shown schematically in Table 5-7, the conceptual site mode. The potential exposure points corresponding to these pathways are shown in Figure 5-3.

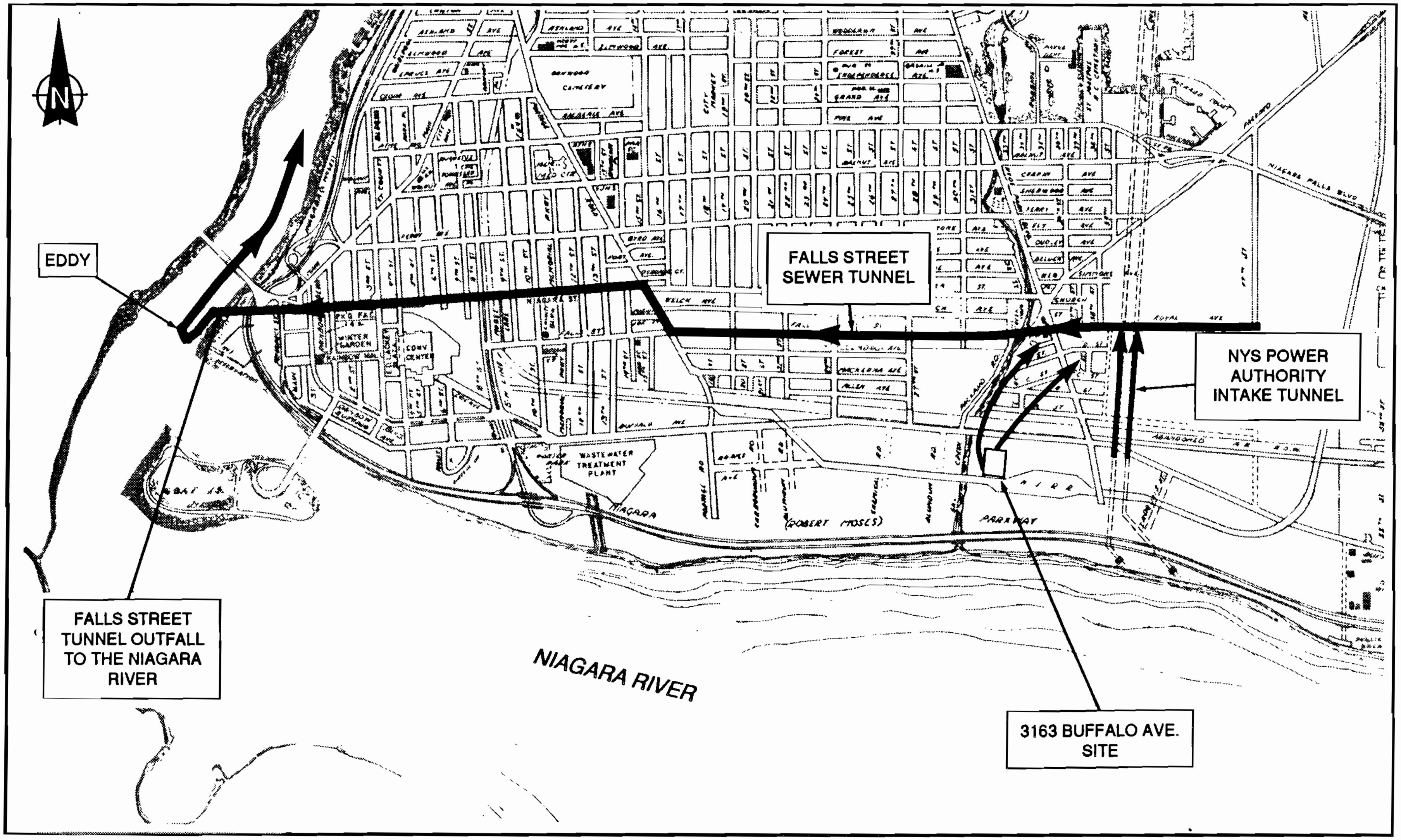


SOURCE: Niagara County Planning and Industrial Development Department, 1974, Revised 1985.

SCALE

0 1 2 3 4 Miles

Figure 5-2
LOCATIONS OF EXPOSURE POINTS FOR POTENTIAL EXPOSURE PATHWAYS EVALUATED IN THE RISK ASSESSMENT



SOURCE: Niagara County Planning and Industrial Development Department, 1974, Revised 1985.

SCALE

0 500 1000 2000 3000 feet

Figure 5-3

EXPECTED MIGRATION PATHWAY FOR GROUNDWATER CONTAMINANTS 3163 BUFFALO AVENUE SITE

Table 5-6

CRITERIA FOR SELECTING CONTAMINANTS OF CONCERN

Contaminant	Concentration[a,b]	Occurrence[a]	Toxicity
INORGANICS			
Arsenic	X	14/18	C
Barium	--	8/18	NC
Cadmium	X	10/18	NC
Chromium	--	17/18	NC
Lead	X	17/18	NC
Manganese	--	18/18	NC
Zinc	X	13/18	NC
Cyanides	X	--	--
ORGANICS			
Chlorinated Aliphatics			
1,2-Dichloroethane	X	2/17	C
1,2-Dichloroethene (total)	X	7/18	NC
Tetrachloroethene	X	4/18	C
Trichloroethene	X	3/18	C
Vinyl chloride	X	2/18	C
Monocyclic Aromatic Hydrocarbons			
Benzene	X	10/18	C
Chlorobenzene	X	12/18	NC
1,2-Dichlorobenzene	X	17/18	NC
1,3-Dichlorobenzene	X	14/18	NC
1,4-Dichlorobenzene	X	17/18	C
1,2,4-Trichlorobenzene	X	14/18	NC

02[IL]JD1900:D3048/1159/24

[a] See Tables 5-3 and 5-4.

[b] Metals considered contaminants of concern which occur above background concentration levels. Carcinogenicity concentration and frequency of detection were the driving criteria for selection of organics.

NC: Noncarcinogen by ingestion
NA: Not applicable
C: Carcinogen

Source: Ecology and Environment, Inc. 1990.

Table 5-7

CONCEPTUAL SITE MODEL FOR GROUNDWATER CONTAMINATION AT 3163 BUFFALO AVENUE

Original Source	Receiving Media	Migration Pathway	Exposure Media	Receptors and Routes of Exposure
Storage and use of chemicals →	Site soils →	Migration of contaminants to the groundwater, then migration via the groundwater to the New York State Power Authority intake tunnels and the Falls Street tunnel; then transport via the Falls Street tunnel to the Niagara River near the Rainbow Bridge. →	Niagara River water →	Use as a drinking water source: o Ingestion o Showering and bathing
			→	Ingestion and dermal contact by fishermen using the river.
			→	Exposure of fish in the river.
			Fish caught from the Niagara River →	Consumption of fish caught from the river by fishermen.
		Volatilization from the river water.	Ambient air near the river →	Inhalation by fishermen.
			→	Inhalation by workers and tourists in the gorge near the Rainbow Bridge.

02[IL]JD1900:D3048/1226/6

Source: Ecology and Environment, Inc. 1990.

Pathway 1: Inhalation of Volatile Organic Contaminants by Individuals In the Gorge Near the Falls Street Tunnel Outfall

The Falls Street Tunnel discharges to the surface of the Niagara River between the Rainbow Bridge and the American Falls. A personal communication from William M. Kappel (USGS-Ithaca) indicates that there is an eddy at the discharge point, which carries the tunnel effluent some distance upstream toward the American Falls before it mixes with the general river flow. The Maid of the Mist dock and the Prospect Point observation tower are both located in the gorge between the American Falls and the Rainbow Bridge. Tourists and workers both frequent this area of the gorge in connection with these attractions.

Since workers spend considerably more time in this area than the tourists, the workers potentially experience a substantially greater exposure; thus, the potential exposure and risks to these workers were selected for evaluation.

Pathway 2: Exposure of Fishermen Using the Lower Niagara River to Site-Derived Contaminants

The Lower Niagara River is used for recreational fishing over much of its length. One of the most popular locations and one close to the Falls Street tunnel outfall is located in the gorge near the Power Authority generating station. Fishermen could potentially be exposed to site-derived contaminants in the river by:

- o Direct dermal contact with river water while netting and landing fish and handling fishing equipment that has come in contact with river water;
- o Ingestion of river water through hand-to-mouth contact (eating, drinking, or smoking without washing in clean water after contacting river water);
- o Inhalation of vapors released from the river; and
- o Consumption of fish caught in the river.

The potential exposure of fishermen using the Lower Niagara River to site-derived contaminants via these routes of exposure was evaluated.

Pathway 3: Use of Niagara River Water as a Source of Potable Water

There are no drinking water intakes on the Lower Niagara River; nor on the south shore of Lake Ontario within 25 miles of the mouth of the Niagara River. However, both the Lower Niagara River north of the Falls and Lake Ontario are classified A-special by NYSDEC, indicating that their best potential use is as drinking water sources. Lake Ontario is used as a public water supply source by Beamsville and Grimsby, Ontario, Albion, Lyndonville, Hamlin and Rochester, New York and other communities. The Niagara River is the source of about 80% of the water entering Lake Ontario. For these reasons the risks associated with the use of Lake Ontario as a source of drinking were evaluated. The mouth of the Niagara was selected as a reasonably conservative location at which to evaluate this usage. Water discharged from the river into Lake Ontario tends to move east along the south shore of the lake; therefore, water intakes in this area receive water similar in composition to that exiting the mouth of the river. Individuals using this water for residential supply purposes could potentially be exposed to site-derived contaminants by:

- Drinking the water;
- Inhalation of vapors volatilized from the water while showering or bathing; and
- Dermal absorption of contaminants while showering or bathing.

The potential exposure of individuals by these routes was therefore evaluated.

Lake Ontario and portions of the lower Niagara River near Lake Ontario are used for swimming and water skiing. However, this potential exposure pathway was not evaluated because the average potential exposure via this pathway would be much less than the potential exposure that would result from the use of the water for domestic supply purposes, which is evaluated, because of the much lower frequency of swimming exposure.

5.5 QUANTITATIVE EXPOSURE ESTIMATES

5.5.1 Contaminant Migration Estimation Methods

The flux of contaminants leaving the site in the groundwater was estimated and described in Section 4.6.3. Contaminants leaving the site were assumed to reach the NYPA or Falls Street tunnels, then migrate to the Niagara River via the Falls Street Tunnel without loss. This is a very conservative assumption in that it neglects possible contaminant losses due to adsorption, degradation, and volatilization that could occur during this time.

On March 1, 1989, the United States District Court for the Western District of New York ordered 4.5 million gallons per day of the dry weather flow of the Falls Street Tunnel diverted to the Niagara Falls Wastewater Treatment Plant. At that time, the total dry weather flow of the Falls Street Tunnel was 11.5 million gallons per day. If the Wastewater Treatment Plant successfully removes all of the 3163 Buffalo Avenue site-derived contaminants from the wastewater, this would result in about a 40% decrease in the flux of site-derived contaminants to the Niagara River under specified flow conditions. However, this potential reduction in the flux of site-derived contaminants to the Niagara River was not factored into the risk assessment process for the following reasons:

- The removal efficiency of the Niagara Falls Wastewater Treatment Plant for the site-derived contaminants is unknown and may be considerably less than 100% for some contaminants since a number of the site-derived contaminants are resistant to biological degradation;
- The fraction of the Falls Street Tunnel flow diverted to the Niagara Falls Wastewater Treatment Plant may vary under different flow conditions (i.e., wet weather); and
- Technical factors such as process upsets may sometimes prevent compliance with the court's order.

Diversion of a portion of the Falls Street Tunnel flow to the Wastewater Treatment Plant will no doubt reduce the flux of site-derived contaminants to the the Niagara River to some degree. Therefore, neglecting the effects of this diversion in the risk estimation process may result in marginally overestimating the potential risks posed by site contaminants.

While several of the chemicals of concern found in the groundwater (particularly benzene) can biodegrade under suitable conditions, no information is available on whether biodegradation is occurring at the site, or if so, what its rate might be. Thus, biodegradation of site contaminants in the groundwater was not considered in estimating the contaminants loading rate to the river.

Many of the groundwater contaminants assumed to be discharging to the river are volatile and will tend to evaporate from the river water as a result of turbulence in the river below the falls. This process reduces the concentration of contaminants in the river water, but transfers them to the air where they can be inhaled. The rate of transfer of contaminants from the water to the air was estimated using mathematical models described in SEAM (EPA 1988a) to obtain the maximum contaminant concentrations in the river below the falls and the rate of transfer of contaminants to the air for use in estimating inhalation exposures.

Contaminant concentrations in the ambient air in the gorge near the river were estimated using a simple box air dispersion model. This model and the model used to estimate the concentrations of volatile organic contaminants in shower stall air are all described in Appendix H.

There is some question as to whether all of the contaminants of potential concern found in site groundwater originated at the site, and whether they are in a physical form that would allow them to migrate in the groundwater. The monocyclic aromatic hydrocarbons and the chlorinated pesticides were found in both the soil and groundwater, and therefore are assumed to have originated on site. The chlorinated aliphatic hydrocarbons, however, were found in the groundwater but not in the site soil; therefore, it is possible that these compounds are migrating onto the site from upgradient along with the groundwater. Soil conditions at the site made it difficult to develop the wells sufficiently to obtain water samples free of turbidity; consequently, many of the groundwater samples contained suspended sediments and, in keeping with NYSDEC recommendations, were not filtered prior to analysis.

The groundwater concentration pattern for metals, particularly the elevated concentrations of aluminum and iron, suggest that the analytical results for metals mainly reflect the metal concentrations present in the suspended sediment rather than metals in solution. Sediment particles normally are trapped in the soil matrix and do not migrate substantially with the groundwater. Therefore, metals associated with these particles would probably not leave the site.

Because of the uncertainty as to the source and mobility of some of the potential contaminants of concern, the risks associated with monocyclic aromatic hydrocarbons and chlorinated pesticides, the chlorinated aliphatic hydrocarbons and metals were calculated separately before combining them to obtain the total groundwater risk. This allows the risks associated with each group of contaminants to be considered separately, if desired.

5.5.2 Exposure Estimation Methods

The formulas used to calculate the estimated exposures by each pathway and route of exposure are given in Tables 5-8 through 5-15. The parameter values used in evaluating the equations, along with the source and rationale for their selection, are also provided. In accordance with the recommendation of EPA's Risk Assessment Guidance Manual (EPA 1989a), the parameters were selected, insofar as possible, so that the resulting exposure estimate would represent the reasonable maximum exposure (RME) that might be expected.

5.6 TOXICITY ASSESSMENT

The purpose of the toxicity assessment is to develop toxicity and carcinogenicity data for the chemicals of concern at the 3163 Buffalo Avenue site and to provide an estimate of the relationship between the extent of exposure to a contaminant and the likelihood and severity of adverse effects. The toxicity assessment will be accomplished in two steps--hazard identification and dose-response assessment.

The hazard identification is a qualitative description of the potential toxic properties of the chemicals of concern at the site. Brief descriptions of the toxic effects of the chemicals of concern are included in the summary tables of the chemicals' toxicological proper-

Table 5-8

PATHWAY 1
WORKER EXPOSURE: INHALATION OF VAPORS FROM THE NIAGARA RIVER IN THE GORGE BELOW THE FALLS

Equation:

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

where:

CA = Contaminant Concentration in Air (mg/m^3)
ABS = Fraction Absorbed (Unitless)
IR = Inhalation Rate (m^3/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)

Variable	Case	Receptor	Value (Rationale/Source)
CA	RME	Adult	Modeled value
ABS	RME	Adult	1.0 (assumed; by convention)
IR	RME	Adult	1.6m^3/hr (light to moderate activity; EPA 1989b)
ET	RME	Adult	10 hours/day (assumed)
EF	RME	Adult	250 days/year (assumed)
ED	RME	Adult	30 years (national upper bound time (90th percentile) at one residence; EPA 1989b)
BW	RME	Adult	70 kg (adult, average; EPA 1989b)
AT	RME	Adult	Pathway-specific period of exposure for noncarcinogenic effects (i.e., ED x 365 days/year) and 70 year lifetime for carcinogenic effects (i.e., 70 years x 365 days/year)

02[IL]JD1900:D3048/1218/22

RME = Reasonable Maximum Exposure

Source: Ecology and Environment, Inc. 1990.

Table 5-9

PATHWAY 2A
FISHING EXPOSURE: DERMAL CONTACT WITH CHEMICALS IN WATER

Equation:

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

where:

CW = Chemical Concentration in Water (mg/liter)
PC = Chemical-specific Dermal Permeability Constant (cm/hr)
SA = Skin Surface Area Available for Contact (cm^2)
ET = Exposure Time (hours/day)
EF = Exposure Frequency (days/year)
ED = Exposure Duration (years)
CF = Volumetric Conversion Factor for Water (1 liter/1000 cm^3)
BW = Body Weight (kg)
AT = Averaging Time (period over which exposure is averaged -- days)

Variable	Case	Receptor	Value (Rationale/Source)
CW	RME	Adult	Modeled concentration
PC	RME	Adult	Chemical-specific values used
SA	RME	Adult	3,100 cm^2 (hands and arms); EPA 1989b
ET	RME	Adult	6 hr/day; E & E 1990
EF	RME	Adult	120 days/year; E & E 1990
ED	RME	Adult	30 years (national upper-bound time (90th percentile) at one residence; EPA 1989b)
BW	RME	Adult	70 kg (average/ EPA 1989b)
AT	RME	Adult	Pathway-specific period of exposure for noncarcinogenic effects (i.e., ED x 365 days/year), and 70 year lifetime for carcinogenic effects (i.e., 70 years x 365 days/year)

02[IL]JD1900:D3048/1219/22

RME = Reasonable Maximum Exposure

Source: Ecology and Environment, Inc. 1990.

Table 5-10

PATHWAY 2B
FISHING EXPOSURE: INGESTION OF CHEMICALS IN SURFACE WATER

Equation:

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

where:

CW = Chemical Concentration in Water (mg/liter)
ABS = Fraction Absorbed (Unitless)
CR = Contact Rate (liters/event)
EF = Exposure Frequency (events/year)
ED = Exposure Duration (years)
BW = Body Weight (kg)
AT = Averaging Time (period over which exposure is averaged - days)

Variable	Case	Receptor	Value (Rationale/Source)
CW	RME	Adult	Modeled concentration
ABS	RME	Adult	1.0 (100% absorption assumed)
CR	RME	Adult	0.01 liters/event; E & E 1990
EF	RME	Adult	120 days/year; E & E 1990
ED	RME	Adult	30 years (national upper-bound time (90th percentile) at one residence: EPA 1989b)
BW	RME	Adult	70 kg (average; EPA 1989b)
AT	RME	Adult	Pathway-specific period of exposure for noncarcinogenic effects (i.e., ED x 365 days/year), and 70 year lifetime for carcinogenic effects (i.e., 70 years x 365 days/year)

02[IL]JD1900:D3048/1220/22

RME = Reasonable Maximum Exposure

Source: Ecology and Environment, Inc. 1990.

Table 5-11

PATHWAY 2C
FISHING EXPOSURE: INHALATION OF AIRBORNE (VAPOR PHASE) CHEMICALS

Equation:

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

where:

CA = Contaminant Concentration in Air (mg/m^3)
ABS = Fraction Absorbed (Unitless)
IR = Inhalation Rate (m^3/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)

Variable	Case	Receptor	Value (Rationale/Source)
CA	RME	Adult	Modeled value
ABS	RME	Adult	1.0 (assumed, by convention)
IR	RME	Adult	1.6 m^3/hr (average of light and moderate activity; EPA 1989b)
ET	RME	Adult	6 hours/day (E & E 1990)
EF	RME	Adult	120 days/year (E & E 1990)
ED	RME	Adult	30 years (national upper-bound time (90th percentile) at one residence; EPA 1989b)
BW	RME	Adult	70 kg (adult, average; EPA 1989b)
AT	RME	Adult	Pathway-specific period of exposure for noncarcinogenic effects (i.e., ED x 365 days/year and 70 year lifetime for carcinogenic effects (i.e., 70 years x 365 days/year)

02[IL]JD1900:D3048/1221/22

RME = Reasonable Maximum Exposure

Source: Ecology and Environment, Inc. 1990.

Table 5-12

PATHWAY 2D
FISHING EXPOSURE: INGESTION OF CONTAMINATED FISH

Equation:

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

where:

CW = Contaminant Concentration in Water (mg/L)
BCF = Bioconcentration Factor (L/kg)
ABS = Fraction Absorbed (Unitless)
IR = Ingestion Rate (kg/day)
FI = Fraction of Fish Consumption 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)

Variable	Case	Receptor	Value (Rationale/Source)
CW	RME	Adult	Modeled concentration
BCF	RME	Adult	Chemical-specific
ABS	RME	Adult	1.0 (assumed; by convention)
IR	RME	Adult	0.140 kg/day (90th percentile daily intake for recreational fishermen; EPA 1989b)
FI	RME	Adult	0.5 (E & E 1990)
EF	RME	Adult	120 days/year (E & E 1990)
ED	RMF	Adult	30 years (national upper bound time (90th percentile) at one residence; EPA 1989b)
BW	RME	Adult	70 kg (average; EPA 1989b)
AT	RME	Adult	Pathway-specific period of exposure for noncarcinogenic effects (i.e., ED x 365 days/year), and 70 year lifetime for carcinogenic effects (i.e., 70 years x 365 days/year)

02[IL]JD1900:D3048/1222/22

RME = Reasonable Maximum Exposure

Source: Ecology and Environment, Inc. 1990.

Table 5-13

PATHWAY 3A
RESIDENTIAL WATER USAGE: INGESTION OF CHEMICALS IN DRINKING WATER (AND BEVERAGES MADE WITH DRINKING WATER)

Equation:

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

where:

CW = Chemical Concentration in Water (mg/liter)
ABS = Fraction Absorbed (assumed equal to 1)
IR = Ingestion Rate (L/day)
EF = Exposure Frequency (days/year)
ED = Exposure Duration (years)
BW = Body Weight (kg)
AT = Averaging Time (period over which exposure is averaged - days)

Variable	Case	Receptor	Value (Rationale/Source)
CW	RME	Adult	Measured concentrations in groundwater
ABS	RME	Adult	1.0 (assumed by convention)
EF	RME	Adult	365 days/year
ED	RME	Adult	30 years (90th percentile time at one residence, EPA 1989b)
BW	RME	Adult	70 kg (EPA 1989b)
AT	RME	Adult	Pathway-specific period of exposure for noncarcinogenic effects (i.e., ED x 365 days/year), and 70 year lifetime for carcinogenic effects (i.e., 70 years x 365 days/year)

02[IL]JD1900:D3048/1225/22

RME = Reasonable Maximum Exposure

Source: Ecology and Environment, Inc. 1990.

Table 5-14

PATHWAY 3B
RESIDENTIAL WATER USAGE: INHALATION OF AIRBORNE (VAPOR PHASE) CHEMICALS DURING SHOWERING

Equation:

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

where:

CA = Contaminant Concentration in Air (mg/m3)
ABS = Fraction Absorbed (assumed equal to 1)
IR = Inhalation Rate (m^3/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)

Variable	Case	Receptor	Value (Rationale/Source)
CA	RME	Adult	Modeled value
ABS	RME	Adult	1 (assumed; by convention)
IR	RME	Adult	0.6 m^3/hr (all age groups, EPA 1989b)
ET	RME	Adult	0.2 hour/day (12 minutes; 90th percentile; EPA 1989b)
EF	RME	Adult	365 days/year
ED	RME	Adult	30 years (90th percentile time at one residence; EPA 1989b)
	RME	Adult	70 kg (EPA 1989b)
AT	RME	Adult	Pathway-specific period of exposure for noncarcinogenic effects (i.e., ED x 365 days/year), and 70 year lifetime for carcinogenic effects (i.e., 70 years x 365 days/year)

02[IL]JD1900:D3048/1224/22

RME = Reasonable Maximum Exposure

Source: Ecology and Environment, Inc. 1990.

Table 5-15

PATHWAY 3C
RESIDENTIAL WATER USAGE: DERMAL CONTACT WITH CHEMICALS DURING SHOWERING

Equation:

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

where:

CW = Chemical Concentration in Water (mg/liter)
PC = Chemical-specific Dermal Permeability Constant (cm/hr)
SA = Skin Surface Area Available for Contact (Cm^2)
ET = Exposure Time (hours/day)
EF = Exposure Frequency (days/year)
ED = Exposure Duration (years)
CF = Volumetric Conversion Factor for Water (1 liter/1000 cm^3)
BW = Body Weight (kg)
AT = Averaging Time (period over which exposure is averaged - days)

Variable	Case	Receptor	Value (Rationale/Source)
CW	RME	Adult	Measured concentrations in groundwater
PC	RME	Adult	Chemical-specific values used
SA	RME	Adult	1.8 m^2 (total body average SA for adults; EPA 1989b)
ET	RME	Adult	0.2 hour/day (12 minutes; 90th percentile; EPA 1989b)
EF	RME	Adult	365 days/year
ED	RME	Adult	30 years (90th percentile time at one residence) (EPA 1989b)
BW	RME	Adult	70 kg (EPA 1989b)
AT	RME	Adult	Pathway-specific period of exposure for noncarcinogenic effects (i.e., ED x 365 days/year), and 70 year lifetime for carcinogenic effects (i.e., 70 years x 365 days/year)

02[IL]JD1900:D3048/1223/22

RME = Reasonable Maximum Exposure

Source: Ecology and Environment, Inc. 1990.

ties included in Section 5.6.2. More complete discussions are provided in the toxicological profiles in Appendix E.

The dose response evaluation is a process that results in a quantitative estimate or index of toxicity for each contaminant at the site. For carcinogens, the index is the slope factor, and for non-carcinogens, it is the reference dose. Practices and procedures used to develop quantitative indices of toxicity and to incorporate toxicological information into the risk estimation process are described in Section 5.6.2 and the quantitative indices of toxicity are presented in Section 5.6.3.

5.6.1 QUANTITATIVE INDICES OF TOXICITY

Quantitative indices of toxicity were compiled for the dose-response assessment to be used in estimating the relationship between the extent of exposure to a contaminant and the potential increased likelihood and/or severity of adverse effects. The methods for deriving indices of toxicity and to estimate potential adverse effects are presented below. The indices of toxicity for the chemicals of concern are presented at the end of the section.

5.6.1.1 Categorization of Chemicals as Carcinogens or Noncarcinogens

For the purpose of this risk assessment, the chemicals of concern were divided into two groups: potential carcinogens and noncarcinogens. The risks posed by these two types of compounds are assessed differently because noncarcinogens are generally thought to exhibit a threshold dose below which no adverse effects occur, although scientists differ as to whether such a threshold exists for carcinogens.

As used here, the term carcinogen means any chemical for which there is sufficient evidence that exposure may result in continuing uncontrolled cell division (cancer) in humans and/or animals. Conversely, the term noncarcinogen means any chemical for which the carcinogenic evidence is negative or insufficient. These definitions are dynamic; compounds may be reclassified any time additional evidence becomes available which shifts the weight-of-evidence one way or the other.

Chemicals of concern have been classified as carcinogens or noncarcinogens based on weight-of-evidence criteria contained in the EPA

Carcinogenicity Evaluation Guidelines (EPA 1986b). Table 5-16 summarizes the five EPA weight-of-evidence categories. According to these EPA guidelines, chemicals in the first two groups--A and B (B1 or B2)--are considered human carcinogens or probable human carcinogens based on sufficient evidence and should be the subject of nonthreshold carcinogenic risk estimation procedures. Depending upon the quality of the data, Group C chemicals may also be subjected to these procedures. The remaining chemicals--in Groups D and E--are defined as noncarcinogens and should be subjected to threshold-based toxicological risk estimation procedures.

5.6.1.2 Assessment of Noncarcinogens

Risks associated with noncarcinogenic effects (e.g., organ damage, immunological effects, birth defects, and skin irritation) are usually assessed by comparing the estimated average exposure to the acceptable daily dose, now called the "reference dose" (RfD) by EPA. The RfD is selected by identifying the lowest reliable no observed or lowest observed adverse effect level (NOAEL or LOAEL) in the scientific literature, then applying a suitable uncertainty factor (usually ranging from 10 to 1,000) to allow for differences between the study conditions and the human exposure situation to which the acceptable daily dose is to be applied. NOAELs and LOAELs are usually based on laboratory experiments on animals in which relatively high doses are used. Consequently, uncertainty or safety factors are required when deriving RfDs to compensate for data limitations in the experiments and the lack of precision in extrapolating from high doses in animals to lower doses in humans. The five uncertainty factors commonly used are summarized in Table 5-17.

RfDs are generally calculated using the formula:

$$\text{RfD (in mg/kg/day)} = \frac{\text{NOAEL or LOAEL (in mg/kg/day)}}{\text{Uncertainty Factor}}$$

If the estimated exposure exceeds the estimated acceptable intake, some adverse effects are presumed to be possible and that exposure level

Table 5-16

FIVE EPA WEIGHT-OF-EVIDENCE CATEGORIES FOR CHEMICAL CARCINOGENICITY

Group	Description
A	Human Carcinogen - sufficient evidence from epidemiological studies.
B	Probable Human Carcinogen
B1	At least limited evidence of carcinogenicity to humans.
B2	Usually a combination of sufficient evidence for animals and inadequate data for humans.
C	Possible Human Carcinogen - limited evidence of carcinogenicity in animals in the absence of human data.
D	Not Classified - inadequate animal evidence of carcinogenicity.
E	No Evidence of Carcinogenicity for Humans - no evidence of carcinogenicity in at least two adequate animal tests in different species or in both epidemiological and animal studies.

02[IL]JD1900:D3048/1161/25

Source: EPA 1986.

Table 5-17

UNCERTAINTY FACTORS (MARGINS OF SAFETY) USED IN THE DERIVATION OF REFERENCE DOSES

Uncertainty Factor	Condition of Use
10	A 10-fold uncertainty factor is used with valid experimental results on appropriate durations of exposures of humans.
100	A 100-fold uncertainty factor is used when human data are not available and extrapolation is made from valid results of long-term animal studies.
1,000	A 1,000-fold uncertainty factor is used when human data are not available and extrapolation is made from animal studies of less than chronic exposure.
1-10	An additional uncertainty factor from 1 to 10 when using a lowest observed adverse effect level (LOAEL) instead of a no observed adverse effect level (NOAEL).
Intermediate uncertainty factor	Other uncertainty factors used, according to scientific judgment, when justified.

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Source: EPA 1986a.

may be of potential concern. Conversely, if the estimated exposure is less than the estimated acceptable intake, no adverse affects would be expected and the exposure level is considered acceptable. Noncarcinogenic risks are usually assessed by calculating a hazard index which is the ratio of the estimated exposure to the RfD as follows:

$$HI = \frac{ADI}{RfD}$$

where

HI = Hazard Index

ADI = Average Daily Intake (exposure)

RfD = Reference Dose (acceptable daily intake).

A hazard index greater than 1 indicates that adverse effects may be possible while a value less than 1 means that adverse effects would not be expected. The greater the hazard index is above 1, the more likely that adverse effects would occur at that given exposure level.

Chronic RfDs are presented in Table 5-18.

5.6.1.3 Assessment of Carcinogens

In contrast to noncarcinogenic effects, for which thresholds are thought to exist, scientists differ as to whether such a threshold exists for carcinogenics. This has led to the assumption by federal regulatory agencies (e.g., EPA, Food and Drug Administration (FDA), and Occupational Safety and Health Administration (OSHA)) that any exposure to a carcinogen theoretically presents some finite risk of cancer. However, depending on the potency of a specific carcinogen, and the level of exposure, such a risk could be vanishingly small.

Scientists have developed several mathematical models to estimate low-dose carcinogenic risks from observed high-dose risks. Consistent with current theories of carcinogenesis, EPA has selected the linearized multistage model based on prudent public health policy (EPA 1986b). In addition to using the linearized multistage model, EPA uses the upper 95% confidence limit for doses or concentrations in animal or human

Table 5-18

CHRONIC TOXICITY VALUES: POTENTIAL NONCARCINOGENIC EFFECTS – ORAL EXPOSURE

Chemical	Chronic RfD (mg/kg-day)	Confidence Level	Critical Effect	RfD Basis/ RfD Source	Uncertainty (UF) and Modifying (MF) Factors
ORAL ROUTE					
Arsenic	1×10^{-3}(g)	NS	Cancer, keratosis	HEAST	UF = 1
Barium	5.0×10^{-2}	Medium	Increased blood pressure	Water/IRIS	UF = 100, MF = 1
Benzene	NA				
Cadmium-Food	1×10^{-3}	High	Human studies – chronic	Food/IRIS	UF = 10, MF = 1
-Water	5×10^{-4}	High	Significant proteinurea	Water/IRIS	UF = 10, MF = 1
Chromium III	1.0	Low	Hepatoxicity	Diet/IRIS	UF = 100, MF = 10
Chlorobenzene	2.0×10^{-2}	NS	Liver and kidney effects	Oral/IRIS	UF = 1,000, MF = 1
2-Chlorophenol	5.0×10^{-3}	NS	Reproductive effects	Water/IRIS	UF = 1,000, MF = 1
Copper	3.7×10^{-2}	NS	Local irritation	NA	NA
Cyanide (free)	2×10^{-2}	Medium	Myelin, weight loss, thyroid effects	Diet/IRIS	UF = 100, MF = 5
1,2-Dichlorobenzene	9.0×10^{-2}	NS	Liver effects	Oral/IRIS	UF = 1,000, MF = 1
1,3-Dichlorobenzene	NA	--	--	--	--
1,4-Dichlorobenzene	NA	--	--	--	--
1,2-Dichloroethane	NA	--	--	--	--
1,2-Dichloroethene (trans)	2×10^{-2}	Low	Increased serum alkaline phosphatose	Water/IRIS	UF = 1,000, MF = 1
Lead	NA	--	--	--	--
Manganese	NA	--	--	--	--
Tetrachloroethene	1×10^{-2}	Medium	Hepatotoxicity – mice Weight gain – rats	Oral/IRIS	UF = 1,000, MF = 1
1,2,4-Trichlorobenzene	5.0×10^{-3}	NS	Liver weight and enzyme induction	Diet/HEAST	UF = 1,000
Trichloroethene	9×10^{-1}	Medium	Growth retardation	Inhalation/IRIS	UF = 1,000, MF = 1
Vinyl chloride	NA	--	--	--	--
Zinc	2.0×10^{-1}	NS	Anemia	Oral/HEAST	UF = 10

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Key:
RfD = Reference dose
NA = Not available
NS = Not specified
g = Under review by RfD work group
IRIS = Integrated Risk Information System
HEAST = Health Effects Assessment Summary Tables

Compiled by Ecology and Environment, Inc., 1990.

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studies to estimate low-dose, slope factors (SFs). By using these procedures, the regulatory agencies are unlikely to underestimate the actual slope factors for humans.

Using SFs, lifetime excess cancer risks can be estimated by:

$$\text{Risk} = \Sigma\ \text{LADD}_j \times \text{SF}_j$$

where

LADD_j = exposure route-specific lifetime average daily dose

SF_j = route-specific slope factor.

Using the multistage model, the carcinogenic risks for the oral, dermal, and inhalation routes of exposure are calculated as follows:

$$\text{Risk} = \text{LADD}_o\text{SF}_o + \text{LADD}_d\text{SF}_o + \text{LADD}_i\text{SF}_i$$

where subscript "o" indicates the oral route, subscript "d" the dermal route and subscript "i" the inhalation route. SFs for the chemicals of concern for the oral and inhalation exposure routes are presented in Table 5-19. EPA's weight-of-evidence classification for the chemical and the type of cancer which may be associated with exposure to the chemical are included.

Once substances have been absorbed by the oral or dermal routes, their distribution, metabolism, and elimination patterns (pharmacokinetics) are usually similar. For this reason, and because dermal route RfDs and SFs are usually not available, oral route RfDs and SFs are commonly used to evaluate exposures to substances by both the oral and dermal routes. This approach is not appropriate and is not used if the adverse effect occurs at the point of exposure. Examples would be skin irritation or skin cancer resulting from dermal exposure. Although inhalation route pharmacokinetics differ more from oral route kinetics than do the dermal route kinetics, oral RfDs and SFs may also be used to evaluate inhalation exposures (except in the case of exposure point effects) if inhalation route RfDs and SFs are not available.

Exposure to some chemicals may result in both carcinogenic and noncarcinogenic effects. In these cases, both the carcinogenic and

Table 5-19

TOXICITY VALUES: POTENTIAL CARCINOGENIC EFFECTS– CONFIDENCE IN CLASSIFICATION AND IN SLOPE FACTOR (SF)

Chemical	Slope Factor (SF) $(mg/kg\text{-}day)^{-1}$	Weight-of-Evidence Classification	Type of Cancer	SF Basis/SF Source
ORAL ROUTE				
Arsenic	NA (pending)	A	Skin cancer	--
Benzene	2.9×10^{-2}	A	Leukemia	Occupational Air/IRIS
1,4-Dichlorobenzene	2.4×10^{-2}	B2	--	--
1,2-Dichloroethane	9.1×10^{-2}	B2	--	Gavage/IRIS
Tetrachloroethene	5.1×10^{-2}	B2	--	Inhalation/IRIS
Trichloroethene	1.1×10^{-2}	B2	--	Oral, Inhalation/IRIS
Vinyl chloride	2.3	A	Liver	Oral/inhalation/IRIS

02[IL]JD1900:D3048/1162/21

NS = Not specified
IRIS = Integrated Risk Information System

Compiled by: Ecology and Environment, Inc., 1990.

noncarcinogenic effects were evaluated and considered in the risk assessment process.

Since scientists differ as to whether a "safe" exposure exists for carcinogens, the task becomes one of determining what level of risk will be deemed acceptable. In general, regulatory agencies in the United States (EPA, FDA, and OSHA) have not established a uniform cancer risk level for distinguishing between risks which are deemed acceptable and those which may be of concern. The agencies have generally considered risks in the range of one in ten thousand (1×10^{-4}) to one in ten million (1×10^{-7}) to be acceptable. EPA has recently adopted the policy that acceptable exposures are generally those that represent an excess upper bound lifetime cancer risk to an individual of between 10^{-4} and 10^{-6}. In addition, EPA will use the 10^{-6} risk level as the point of departure for determining remediation goals for National Priorities List (NPL) sites (EPA 1990).

5.7 RISK CHARACTERIZATION

This section combines the information developed in the exposure and toxicity assessment sections to obtain estimates of the risks posed by Front Street site contaminants to human health. The process by which this is done is as follows:

Risks due to carcinogenic and noncarcinogenic contaminants are assessed differently, as discussed previously. Briefly, carcinogenic risks are assessed by multiplying the estimated chronic daily intake (CDI) of a carcinogen by its estimated slope factor (SF) to obtain the estimated risk, expressed as the probability of that exposure resulting in an excess incidence of cancer.

The potential for adverse effects resulting from exposure to noncarcinogens is assessed by comparing the CDI of a substance to its chronic RfD. This comparison is performed by calculating the ratio of the estimated CDI to the corresponding RfD, which is called a hazard quotient or hazard index. If the hazard index is less than 1, no adverse effects would be expected; however, if it is greater than 1, adverse effects could be possible.

The excess cancer risk or the hazard quotient for exposure to each chemical by each route of exposure, exposure pathway, and category of receptor (adult or child) are initially estimated separately.

The separate cancer risk estimates are then summed across chemicals and across all exposure routes and pathways applicable to the same population to obtain the total excess cancer risk for that population. Hazard quotients for noncarcinogens are summed across chemicals that produce the same type of adverse effect (such as liver damage) but are kept separate if their effects are different. Hazard quotients are summed across all chemicals, exposure routes, and pathways applicable to the same population to obtain hazard indices for that population.

5.7.1 Magnitude of Risks Posed by Site Groundwater Contaminants

The estimated risks of excess cancers and noncarcinogenic adverse effects are summarized in Tables 5-20 and 5-21 respectively. Detailed exposure and risk estimates for each exposure pathway are presented in Tables 5-22 through 5-29 at the end of section 5.7. Because of uncertainties as to the origin and mobility of some of the groundwater contaminants at the site, the risks for three chemical groups are given separately, in addition to the total risks for all of the groundwater contaminants of concern. To summarize the discussion in Section 5.2, group 1 chemicals include the monocyclic aromatic hydrocarbons and the chlorinated pesticides. Members of this group were present in both soil and groundwater and are believed to have originated at the site. Group 2 chemicals are the chlorinated aliphatic hydrocarbons, which were found in the groundwater, but not in the soil and may well have migrated onto the site with the groundwater. Group 3 chemicals are the metals and cyanide, which are likely associated with soil sediment particles entrained in the groundwater samples. Consequently, these chemicals are not significantly mobile in the subsurface and would probably not migrate off site with the groundwater.

In accordance with this interpretation, the risks associated with these groups would be viewed as follows. The risks due to the site-derived organics are risks likely to be experienced by receptor populations and attributable to contamination originating from the site. The risks due to non-site-derived organics are risks likely to be experienced by receptors, but which are probably attributable to contaminants originating upgradient from the site rather than the site itself. The

Table 5-20

HUMAN HEALTH RISK ASSESSMENT
SUMMARY OF ESTIMATED CANCER RISKS
3163 BUFFALO AVENUE SITE

		Cancer Risk			
		Group 1	Group 2	Group 3	
Exposure Pathway	Description	Site-Derived Organics	Non-Site-Derived Organics	Metals and Cyanide	Total for all Groundwater Contaminants
Pathway 1:	Worker exposure in the Niagara River gorge: inhalation of vapors	9.0×10^{-9}	3.2×10^{-7}	0	3.2×10^{-7}
	Pathway 1 Total	9.0×10^{-9}	3.2×10^{-7}	0	3.2 x 10-7
Pathway 2:	Recreational Fishermen in the Niagara River gorge:				
2A:	Dermal contact with surface water	8.4×10^{-12}	2.1×10^{-10}	1.5×10^{-15}	2.2×10^{-10}
2B:	Ingestion of surface water	4.6×10^{-12}	1.1×10^{-10}	7.9×10^{-14}	1.2×10^{-10}
2C:	Inhalation of vapors	1.5×10^{-8}	5.6×10^{-7}	0	5.7×10^{-7}
2D:	Ingestion of Contaminated Fish	6.2×10^{-10}	9.1×10^{-10}	2.4×10^{-11}	1.6×10^{-9}
	Pathway 2 Total	1.6×10^{-8}	5.6×10^{-7}	2.4×10^{-11}	5.7×10^{-7}
Pathway 3:	Residential Water Usage:				
3A:	Ingestion as Drinking Water	2.6×10^{-9}	6.3×10^{-8}	4.8×10^{-11}	6.6×10^{-8}
3B:	Inhalation of vapors during showering	1.7×10^{-9}	6.9×10^{-8}	0	7.1×10^{-8}
3C:	Dermal Contact with water during showering	4.7×10^{-12}	1.1×10^{-10}	8.6×10^{-16}	1.2×10^{-10}
	Pathway 3 Total	4.4×10^{-9}	6.4×10^{-8}	4.8×10^{-11}	1.3×10^{-7}
TOTAL GROUNDWATER RISKS SUMMED OVER ALL EXPOSURE PATHWAYS		2.9×10^{-8}	9.4×10^{-7}	7.2×10^{-11}	1.0×10^{-6}

Table 5-21

HUMAN HEALTH RISK ASSESSMENT
SUMMARY OF ESTIMATED NONCARCINOGENIC RISKS
3163 BUFFALO AVENUE SITE

		Noncarcinogenic Chronic Hazard Index			
		Group 1	Group 2	Group 3	
Exposure Pathway	Description	Site-Derived Organics	Non-Site-Derived Organics	Metals and Cyanide	Total Hazard Index for all Contaminants
Pathway 1:	Worker exposure in the Niagara River gorge:				
	Inhalation of vapors:	8.1×10^{-10}	1.2×10^{-9}	0	2.0×10^{-9}
	Pathway 1 Total	8.1×10^{-10}	1.2×10^{-9}	0	2.0×10^{-9}
Pathway 2:	Recreational Fishermen in the Niagara River gorge				
2A:	Dermal contact with surface water	1.5×10^{-9}	3.0×10^{-9}	7.3×10^{-8}	7.8×10^{-8}
2B:	Ingestion of surface water	1.1×10^{-9}	1.6×10^{-9}	9.8×10^{-9}	1.3×10^{-8}
2C:	Inhalation of vapors	3.2×10^{-4}	5.8×10^{-12}	0	3.2×10^{-4}
2D:	Ingestion of Contaminated Fish	2.0×10^{-6}	2.2×10^{-8}	3.2×10^{-6}	5.2×10^{-6}
	Pathway 2 Total	3.2×10^{-4}	2.7×10^{-8}	3.3×10^{-6}	3.3×10^{-4}
Pathway 3:	Residential Water Usage:				
3A:	Ingestion as Drinking Water	6.6×10^{-7}	9.1×10^{-7}	6.0×10^{-6}	7.6×10^{-6}
3B:	Inhalation of vapors during showering	7.2×10^{-7}	6.2×10^{-7}	0	1.3×10^{-6}
3C:	Dermal contact with water during showering	9.0×10^{-10}	1.6×10^{-9}	4.3×10^{-8}	4.6×10^{-8}
	Pathway 3 Total	1.4×10^{-6}	1.5×10^{-6}	6.0×10^{-6}	8.9×10^{-6}
TOTAL GROUNDWATER HAZARD INDEX SUMMED OVER ALL EXPOSURE PATHWAYS		3.2×10^{-4}	1.5×10^{-6}	9.3×10^{-6}	3.4×10^{-4}

risks from the metals and cyanide are attributable to the site, but would probably not be experienced by potential receptors. These contaminants are unlikely to migrate off site and reach the potential receptor populations. Thus, according to this interpretation, only the site-derived organics pose risks that are likely to be experienced and attributable to site contamination.

The risks have been summed for each exposure pathway separately and for all of the pathways together. Because of the nature of the potential exposure pathways, an individual might be exposed by one, two or all three of the pathways. Therefore, it is appropriated to sum the estimated risks across all of the pathways.

EPA has recently adopted the policy that acceptable exposures to known or suspected carcinogens are generally those that represent an excess upper-bound lifetime cancer risk to an individual of between 10^{-4} and 10^{-6}. In addition, EPA will use the 10^{-6} risk level as the point of departure for determining remediation goals for NPL sites. For systemic toxicants (noncarcinogens) EPA defines acceptable exposure levels as those to which the human population, including sensitive subgroups, may be exposed without adverse effects during a lifetime or part of a lifetime, incorporating an adequate margin of safety (EPA 1990). This acceptable exposure level corresponds to a hazard index of 1. If the hazard index is less than 1, no adverse effects would be expected. If the hazard index is greater than 1, adverse effects could be possible. Many state regulatory agencies have also adopted these EPA threshold levels for assessing risks posed by contaminants at hazardous waste sites.

As shown in Table 5-20 the total estimated cancer risk due to site-derived organics (i.e., the estimated risk attributable to site contamination that is likely to be realized) is about 30 times less than 1×10^{-6}, the lowest level considered to pose a significant risk by the regulatory agencies.

The site-derived and non-site-derived organics together account for all of the risks likely to be realized and virtually all of total estimated risks posed by all site groundwater contaminants. The total groundwater risks for all contaminants and all of the potential exposure pathways just reach the 1×10^{-6} risk level.

A number of physical, chemical and ecological processes could result in the removal of many of the contaminants of concern from the groundwater before it reaches the Niagara River. These were not included in estimating contaminant loadings to the river. In view of the conservative nature of the risk assessment process, which is discussed in the next section, this estimate of the potential risks posed by site groundwater contaminants probably substantially overestimated the actual risks potential receptors are likely to experience.

As shown in Table 5-21, the total estimated hazard index for all site groundwater contaminants summed over all of the potential exposure pathways is about 3,000 times less than the threshold value of 1, above which the potential for such effects begins to become of concern. Therefore, site groundwater contaminants do not appear to pose a significant threat of adverse noncarcinogenic health effects.

The concentrations of many of the chemicals found in the groundwater at the site exceed NYSDEC standards or guidelines for Class GA groundwater and state and/or federal drinking water standards. Thus, while the groundwater at the site is classified as a potential drinking water source, it is presently unsuitable for that use because of the contaminants present.

5.8 UNCERTAINTIES IN THE RISK ASSESSMENT PROCESS

5.8.1 Uncertainty in the Exposure Assessment

There are a number of factors that will cause the exposure levels estimated in this section to differ from the exposures potential receptor populations may actually experience. This section will identify these factors, discuss the potential effects of the factors on the exposure estimates, and, where possible and appropriate, estimate the degree of confidence that should be placed in the various assumptions and parameter estimates that have gone into the exposure estimates.

5.8.1.1 Environmental Sampling

The available data characterizing groundwater contamination at the site is somewhat limited. Thirteen wells were installed during the RI, but only one round of groundwater sampling was conducted. The ground-

Table 5-22

EXPOSURE PATHWAY 1: WORKER EXPOSURE VIA INHALATION OF VAPORS FROM THE NIAGARA RIVER IN THE GORGE BELOW THE FALLS RISK ESTIMATES

		Cancer Risk Assessment		Noncarcinogenic Risk Assessment	
Chemical	Exposure Concentration in Air, CA (mg/m^3)	Average Absorbed Dose* (mg/kg/day)	Inhal-ation Cancer Risk	Average Absorbed Dose* (mg/kg/day)	Chronic Hazard Index (Inhal.)
Group 1: Site-Derived Organics					
Benzene	4.6×10^{-6}	3.09×10^{-7}	9×10^{-9}	--	--
Chlorobenzene	5.7×10^{-8}	--	--	8.09×10^{-9}	1.78×10^{-10}
1,2-Dichlorobenzene	1.9×10^{-8}	--	--	3×10^{-9}	6×10^{-10}
1,3-Dichlorobenzene	7.8×10^{-12}	--	--	--	--
1,4-Dichlorobenzene	1.7×10^{-11}	1.14×10^{-12}	3×10^{-14}	2.7×10^{-12}	1.89×10^{-12}
1,2,4-Trichlorobenzene	7×10^{-8}	--	--	1.1×10^{-8}	3.3×10^{-11}
α-HCCH	8.14×10^{-11}	5.46×10^{-12}	3.4×10^{-11}	--	--
β-HCCH	2×10^{-12}	1.34×10^{-13}	2.4×10^{-13}	--	--
γ-HCCH	1.17×10^{-10}	7.9×10^{-12}	1.03×10^{-11}	1.8×10^{-11}	5.4×10^{-15}
δ-HCCH	2.3×10^{-13}	--	--	--	--
Group 1 Subtotals			9.04×10^{-9}		8.1×10^{-10}
Group 2: Non-Site-Derived Organics					
1,2-Dichloroethane	0	0	0	--	--
1,2-Dichloroethene (total)	3.8×10^{-7}	--	--	6×10^{-8}	1.2×10^{-9}
Tetracloroethene	1.3×10^{-9}	8.7×10^{-11}	1.5×10^{-13}	2×10^{-10}	2×10^{-12}
Trichloroethene	9.6×10^{-10}	6.4×10^{-11}	8.3×10^{-13}	--	--
Vinyl Chloride	1.6×10^{-6}	1.07×10^{-7}	3.15×10^{-7}	--	--
Group 2 Subtotals			3.15×10^{-7}		1.2×10^{-9}
Total Pathway Risks/ Hazard Indices			3.24×10^{-7}		2.01×10^{-9}

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*The average absorbed doses differ for the carcinogenic and noncarcinogenic effects assessments because different averaging times are used.

Source: Ecology and Environment, Inc. 1990.

Table 5-23

EXPOSURE PATHWAY 2A: RECREATIONAL FISHERMEN USING THE NIAGARA RIVER – DERMAL CONTACT WITH CHEMICALS IN SURFACE WATER WHILE FISHING RISK ESTIMATES

Chemical	Exposure Media Concentration in Water (mg/L)	Cancer Risk Assessment: Absorbed Dose (mg/kg/day)	Cancer Risk Assessment: Cancer Risk (Oral)	Noncarcinogenic Risk Assessment: Absorbed Dose (mg/kg/day)	Noncarcinogenic Risk Assessment: Chronic Hazard Index (Oral)
Group 1: Site-Derived Organics					
Benzene	6.9×10^{-06}	2.58×10^{-10}	7.48×10^{-12}	--	--
Chlorobenzene	1.02×10^{-07}	--	--	8.90 x 10-12	4.45×10^{-10}
1,2-Dichlorobenzene	3.7×10^{-08}	--	--	3.23 x 10-12	3.59×10^{-11}
1,3-Dichlorobenzene	1.5×10^{-08}	--	--	--	--
1,4-Dichlorobenzene	3.2×10^{-08}	1.20×10^{-12}	2.88×10^{-14}	2.79 x 10-12	3.99×10^{-12}
1,2,4-Trichlorobenzene	1.4×10^{-07}	--	--	1.22 x 10-12	6.10×10^{-11}
α-HCCH	3.0×10^{-09}	1.12×10^{-13}	7.06×10^{-13}	--	--
β-HCCH	9.3×10^{-10}	3.48×10^{-14}	6.26×10^{-14}	--	--
γ-HCCH	3.4×10^{-09}	1.27×10^{-13}	1.65×10^{-13}	2.97 x 10-12	9.90×10^{-10}
δ-HCCH	2.22×10^{-10}	--	--	--	--
Group 1 Subtotals			8.44×10^{-12}		1.54×10^{-09}
Group 2: Non-Site-Derived Organics					
1,2-Dichloroethane	0	0	0	--	--
1,2-Dichloroethene (total)	6.7×10^{-07}	--	--	5.85E-11	2.93×10^{-09}
Tetracloroethene	2.5×10^{-09}	9.35×10^{-14}	4.77×10^{-15}	2.18E-13	2.18×10^{-11}
Trichloroethene	1.8×10^{-09}	6.73×10^{-14}	7.40×10^{-16}	--	--
Vinyl Chloride	2.4×10^{-06}	8.98×10^{-11}	2.07×10^{-10}	--	--
Group 2 Subtotals			2.07×10^{-10}		2.95×10^{-09}

Table 5-23 (Cont.)

Chemical	Exposure Media Concentration in Water (mg/L)	Cancer Risk Assessment: Absorbed Dose (mg/kg/day)	Cancer Risk Assessment: Cancer Risk (Oral)	Noncarcinogenic Risk Assessment: Absorbed Dose (mg/kg/day)	Noncarcinogenic Risk Assessment: Chronic Hazard Index (Oral)
Group 3: Metals and Cyanide					
Arsenic	2.24×10^{-09}	8.38×10^{-16}	1.47×10^{-15}	1.96E-15	1.96×10^{-12}
Barium	4.48×10^{-09}	--	--	3.91E-15	7.82×10^{-14}
Cadmium	2.54×10^{-09}	--	--	2.22E-15	4.44×10^{-12}
Chromium	5.68×10^{-06}	--	--	4.96E-12	4.96×10^{-12}
Lead	5.57×10^{-08}	--	--	4.86E-14	1.62×10^{-10}
Manganese	2.78×10^{-07}	--	--	—	—
Zinc	4.46×10^{-07}	--	--	3.89E-13	1.95×10^{-12}
Cyanides	1.67×10^{-07}	--	--	1.46E-09	7.30×10^{-08}
Group 3 Subtotals			1.47×10^{-15}		7.32×10^{-08}
Total Exposure Pathway Risk/Hazard Indices			2.15×10^{-10}		7.77×10^{-08}

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Table 5-24

EXPOSURE PATHWAY 2B: RECREATIONAL FISHERMEN USING THE NIAGARA RIVER - INGESTION OF CHEMICALS IN SURFACE WATER WHILE FISHING RISK ESTIMATES

Chemical	Exposure Concentration in Water (mg/L)	Cancer Risk Assessment Absorbed Dose (mg/kg/day)	Cancer Risk Assessment Cancer Risk (Oral)	Noncarcinogenic Risk Assessment Absorbed Dose (mg/kg/day)	Noncarcinogenic Risk Assessment Chronic Hazard Index (Oral)
Group 1: Site-Derived Organics					
Benzene	6.9×10^{-06}	1.39×10^{-10}	4.03×10^{-12}		
Chlorobenzene	1.02×10^{-07}	--	--	4.79×10^{-12}	2.40×10^{-10}
1,2-Dichlorobenzene	3.7×10^{-08}	--	--	1.74×10^{-12}	1.93×10^{-11}
1,3-Dichlorobenzene	1.5×10^{-08}				
1,4-Dichlorobenzene	3.2×10^{-08}	6.43×10^{-13}	1.54×10^{-14}	1.50×10^{-12}	2.14×10^{-12}
1,2,4-Trichlorobenzene	1.4×10^{-07}			6.58×10^{-12}	3.29×10^{-10}
α-HCCH	3.0×10^{-09}	6.03×10^{-14}	3.80×10^{-13}	--	--
β-HCCH	9.3×10^{-10}	1.87×10^{-14}	3.37×10^{-14}		
γ-HCCH	3.4×10^{-09}	6.83×10^{-14}	8.88×10^{-14}	1.60×10^{-13}	5.33×10^{-10}
δ-HCCH	2.22×10^{-10}	--	--	--	--
Group 1 Totals			4.55×10^{-12}		1.12×10^{-09}
Group 2: Non-Site-Derived Organics					
1,2-Dichloroethane	0	0	0		
1,2-Dichloroethene (total)	6.7×10^{-07}			3.15×10^{-11}	1.58×10^{-09}
Tetracloroethene	2.5×10^{-09}	5.03×10^{-14}	2.57×10^{-15}	1.18×10^{-13}	1.18×10^{-11}
Trichloroethene	1.8×10^{-09}	3.62×10^{-14}	3.98×10^{-16}	--	--
Vinyl Chloride	2.4×10^{-06}	4.82×10^{-11}	1.11×10^{-10}	--	--
Group 2 Totals			1.11×10^{-10}		1.59×10^{-09}
Group 3: Metals and Cyanide					
Arsenic	2.24×10^{-09}	4.50×10^{-14}	7.88×10^{-14}	1.05×10^{-13}	1.05×10^{-10}
Barium	4.48×10^{-09}	--	--	2.11×10^{-13}	4.22×10^{-12}
Cadmium	2.54×10^{-09}	--	--	1.19×10^{-13}	2.38×10^{-10}
Chromium	5.68×10^{-06}	--	--	2.67×10^{-10}	2.67×10^{-10}
Lead	5.57×10^{-08}	--	--	2.62×10^{-12}	8.73×10^{-09}
Manganese	2.78×10^{-07}	--	--	--	
Zinc	4.46×10^{-07}	--	--	2.10×10^{-11}	1.05×10^{-10}
Cyanides	1.67×10^{-07}	--	--	7.85×10^{-12}	3.93×10^{-10}
Group 3 Totals			7.88×10^{-14}		9.84×10^{-09}
Total Exposure Pathway Risk/Hazard Indices			1.16×10^{-10}		1.26×10^{-08}

Table 5-25

EXPOSURE PATHWAY 2C: RECREATIONAL FISHERMEN USING THE NIAGARA RIVER INHALATION OF VAPORS FROM THE RIVER WHILE FISHING RISK ESTIMATES

		Cancer Risk Assessment		Noncarcinogenic Risk Assessment	
Chemical	Exposure Concentration in Air, CA (mg/m^3)	Average Absorbed Dose* (mg/kg/day)	Inhalation Cancer Risk	Average Absorbed Dose* (mg/kg/day)	Chronic Hazard Index (Inhal.)
Group 1: Site-Derived Organics					
Benzene	2.6×10^{-5}	5.02×10^{-7}	1.46×10^{-8}	--	--
Chlorobenzene	3.5×10^{-7}	--	--	1.6×10^{-8}	3.2×10^{-4}
1,2-Dichlorobenzene	1.1×10^{-8}	--	--	5×10^{-10}	1×10^{-10}
1,3-Dichlorobenzene	4.6×10^{-8}	--	--	--	--
1,4-Dichlorobenzene	1.06×10^{-7}	2.05×10^{-9}	4.92×10^{-11}	4.8×10^{-9}	3.4×10^{-9}
1,2,4-Trichlorobenzene	4.4×10^{-7}	--	--	2×10^{-8}	6×10^{-11}
α-HCCH	3.1×10^{-10}	5.9×10^{-12}	3.7×10^{-11}	--	--
β-HCCH	7.7×10^{-12}	1.5×10^{-13}	2.7×10^{-13}	--	--
γ-HCCH	4.6×10^{-10}	8.9×10^{-12}	1.2×10^{-11}	2.07×10^{-11}	6.2×10^{-15}
δ-HCCH	8.4×10^{-13}	--	--	--	--
Group 1 Totals			1.47×10^{-8}		3.2×10^{-4}
Group 2: Non-Site-Derived Organics					
1,2-Dichloroethane	0	0	0	--	--
1,2-Dichloroethene (total)	2.44×10^{-9}	--	--	1.1×10^{-10}	2.2×10^{-12}
Tetracloroethene	8.1×10^{-9}	1.56×10^{-10}	2.65×10^{-13}	3.6×10^{-10}	3.6×10^{-12}
Trichloroethene	6.1×10^{-9}	1.2×10^{-10}	1.6×10^{-12}	--	--
Vinyl Chloride	9.8×10^{-6}	1.9×10^{-7}	5.6×10^{-7}	--	--
Group 2 Totals			5.6×10^{-7}		5.8×10^{-12}
Total Exposure Pathway Risk/Hazard Indices			5.7×10^{-7}		3.2×10^{-4}

02[IL]JD1900:D3048/1623/20

*The average absorbed doses differ for the carcinogenic and noncarcinogenic effects assessments because different averaging times are used.

Source: Ecology and Environment, Inc. 1990.

Table 5-26

EXPOSURE PATHWAY 2D: RECREATIONAL FISHERMEN USING THE NIAGARA RIVER - INGESTION OF CONTAMINATED FISH RISK ESTIMATES

Chemical	Exposure Concentration in Water (mg/L)	Cancer Risk Assessment Absorbed Dose (mg/kg/day)	Cancer Risk Assessment Cancer Risk (Oral)	Noncarcinogenic Risk Assessment Absorbed Dose (mg/kg/day)	Noncarcinogenic Risk Assessment Chronic Hazard Index (Oral)
Group 1: Site-Derived Organics					
Benzene	6.9×10^{-06}	5.06×10^{-09}	1.47×10^{-10}	--	--
Chlorobenzene	1.02×10^{-07}	--	--	3.36×10^{-10}	1.68 x 10-08
1,2-Dichlorobenzene	3.7×10^{-08}	--	--	1.40 x 10-09	1.56 x 10-08
1,3-Dichlorobenzene	1.5×10^{-08}	--	--	--	--
1,4-Dichlorobenzene	3.2×10^{-08}	5.28×10^{-10}	1.27×10^{-11}	1.23×10^{-09}	1.76 x 10-09
1,2,4-Trichlorobenzene	1.4×10^{-07}	--	--	2.90×10^{-08}	1.45 x 10-06
α-HCCH	3.0×10^{-09}	5.50×10^{-11}	3.47×10^{-10}	--	--
β-HCCH	9.3×10^{-10}	1.70×10^{-11}	3.06×10^{-11}	--	--
γ-HCCH	3.4×10^{-09}	6.23×10^{-11}	8.10×10^{-11}	1.45×10^{-10}	4.83 x 10-07
δ-HCCH	2.22×10^{-10}	--	--	--	--
Group 1 Totals			6.18×10^{-10}		1.97 x 10-06
Group 2: Non-Site-Derived Organics					
1,2-Dichloroethane	0	0	0	--	--
1,2-Dichloroethene (total)	6.7×10^{-07}	--	--	3.31×10^{-10}	1.66 x 10-08
Tetracloroethene	2.5×10^{-09}	2.33×10^{-11}	1.19×10^{-12}	5.43×10^{-11}	5.43 x 10-09
Trichloroethene	1.8×10^{-09}	2.69×10^{-14}	2.96×10^{-14}	--	--
Vinyl Chloride	2.4×10^{-06}	3.96×10^{-10}	9.11×10^{-10}	--	--
Group 2 Totals			9.12×10^{-10}		2.20 x 10-08
Group 3: Metals and Cyanide					
Arsenic	2.24×10^{-09}	1.39×10^{-11}	2.43×10^{-11}	3.24×10^{-11}	3.24 x 10-08
Barium	4.48×10^{-09}	--	--	--	--
Cadmium	2.54×10^{-09}	--	--	6.77×10^{-11}	1.35 x 10-07
Chromium	5.68×10^{-06}	--	--	2.99×10^{-08}	2.99 x 10-08
Lead	5.57×10^{-08}	--	--	8.98×10^{-10}	2.99 x 10-06
Manganese	2.78×10^{-07}	--	--	--	--
Zinc	4.46×10^{-07}	--	--	6.90×10^{-09}	3.45 x 10-08
Cyanides	1.67×10^{-07}	--	--	0	0
Group 3 Totals			2.43×10^{-11}		3.22 x 10-06
Total Exposure Pathway Risk/Hazard Indices			1.55×10^{-09}		5.21 x 10-06

Table 5-27

EXPOSURE PATHWAY 3A: RESIDENTIAL DRINKING WATER INGESTION RISK ESTIMATES

		Cancer Risk Assessment		Noncarcinogenic Risk Assessment	
Chemical	Exposure Concentration (as Modeled) in Water at Mouth of Niagara River (mg/L)	Average Absorbed Dose* (mg/kg/day)	Cancer Risk (Ingestion)	Average Absorbed Dose* (mg/kg/day)	Chronic Hazard Index (Ingestion)
Group 1: Site-Derived Organics					
Benzene	6.5×10^{-6}	7.96×10^{-8}	2.31×10^{-9}	--	--
Chlorobenzene	9.7×10^{-8}	--	--	3.24×10^{-7}	1.39×10^{-7}
1,2-Dichlorobenzene	3.5×10^{-8}	--	--	2.59×10^{-8}	1.11×10^{-8}
1,3-Dichlorobenzene	1.4×10^{-8}	--	--	--	--
1,4-Dichlorobenzene	3.05×10^{-8}	3.73×10^{-10}	8.69×10^{-12}	2.89×10^{-9}	1.24×10^{-9}
1,2,4-Trichlorobenzene	1.3×10^{-7}	--	--	4.34×10^{-7}	1.86×10^{-7}
α-HCCH	3×10^{-9}	3.67×10^{-11}	2.31×10^{-10}	--	--
β-HCCH	9.3×10^{-10}	1.14×10^{-11}	2.05×10^{-11}	--	--
γ-HCCH	3.4×10^{-9}	4.16×10^{-11}	5.41×10^{-11}	7.56×10^{-7}	3.24×10^{-7}
δ-HCCH	2.2×10^{-10}	--	--	--	--
Group 1 Totals			2.62×10^{-9}		6.6×10^{-7}
Group 2: Non-Site-Derived Organics					
1,2-Dichloroethane	--	--	--	--	--
1,2-Dichloroethene (total)	6.3×10^{-7}	--	--	2.1×10^{-6}	9×10^{-7}
Tetracloroethene	2.38×10^{-9}	2.91×10^{-11}	1.49×10^{-12}	1.4×10^{-8}	6×10^{-9}
Trichloroethene	1.7×10^{-9}	2.08×10^{-11}	2.29×10^{-13}	--	--
Vinyl Chloride	2.25×10^{-3}	2.76×10^{-8}	6.34×10^{-8}	--	--
Group 2 Totals			6.34×10^{-8}		9.07×10^{-7}

Table 5-27 (Cont.)

Chemical	Exposure Concentration (as Modeled) in Water at mouth of Niagara River (mg/L)	Cancer Risk Assessment		Noncarcinogenic Risk Assessment	
		Average Absorbed Dose* (mg/kg/day)	Cancer Risk (Ingestion)	Average Absorbed Dose* (mg/kg/day)	Chronic Hazard Index (Ingestion)
Group 3: Metals and Cyanides					
Arsenic	2.24×10^{-9}	2.74×10^{-11}	4.8×10^{-11}	1.49×10^{-7}	6.4×10^{-8}
Barium	4.48×10^{-9}	--	--	5.97×10^{-9}	2.56×10^{-9}
Cadmium	2.54×10^{-9}	--	--	3.83×10^{-7}	1.45×10^{-7}
Chromium	5.68×10^{-6}	--	--	3.78×10^{-7}	1.62×10^{-7}
Lead	5.57×10^{-8}	--	--	1.24×10^{-5}	5.3×10^{-6}
Manganese	2.78×10^{-7}	--	--	--	--
Zinc	4.46×10^{-7}	--	--	1.49×10^{-7}	6.37×10^{-8}
Cyanide	1.67×10^{-7}	--	--	5.58×10^{-7}	2.39×10^{-7}
Group 3 Totals			4.8×10^{-11}		5.98×10^{-6}
Total Pathway Risks/ Hazard Indices			6.61×10^{-8}		7.55×10^{-6}

02[IL]JD1900:D3048/1245/13

*The average absorbed doses differ for the carcinogenic and noncarcinogenic effects assessments because different averaging times are used.

Source: Ecology and Environment, Inc. 1990.

Table 5-28

EXPOSURE PATHWAY 3B: RESIDENTIAL INHALATION FROM AIR IN SHOWER RISK ESTIMATES

Chemical	Exposure Concentration in Air in Shower (mg/m3)	Cancer Risk Assessment: Average Absorbed Dose* (mg/kg/day)	Cancer Risk Assessment: Cancer Risk (Inhalation)	Noncarcinogenic Risk Assessment: Average Absorbed Dose* (mg/kg/day)	Noncarcinogenic Risk Assessment: Chronic Hazard Index (Inhalation)
Group 1: Site-Derived Organics					
Benzene	8.07×10^{-5}	5.93×10^{-8}	1.72×10^{-9}	--	--
Chlorobenzene	1×10^{-6}	--	--	2.01×10^{-7}	8.6×10^{-8}
1,2-Dichlorobenzene	2.89×10^{-7}	--	--	5.76×10^{-9}	2.47×10^{-9}
1,3-Dichlorobenzene	1.22×10^{-7}	--	--	--	--
1,4-Dichlorobenzene	2.77×10^{-7}	--	--	1.58×10^{-9}	6.78×10^{-10}
1,2,4-Trichlorobenzene	1.1×10^{-6}	--	--	1.47×10^{-6}	6.28×10^{-7}
α-HCCH	5.34×10^{-10}	3.92×10^{-13}	2.47×10^{-12}	--	--
β-HCCH	1.29×10^{-11}	9.48×10^{-15}	1.71×10^{-14}	--	--
γ-HCCH	8.02×10^{-10}	5.9×10^{-13}	7.66×10^{-13}	1.07×10^{-8}	4.59×10^{-9}
δ-HCCH	1.41×10^{-12}	--	--	--	--
Group 1 Totals			1.72×10^{-9}		7.22×10^{-7}
Group 2: Non-Site-Derived Organics					
1,2-Dichloroethane	--	--	--	--	--
1,2-Dichloroethene (total)	7.19×10^{-6}	--	--	1.44×10^{-6}	6.16×10^{-7}
Tetracloroethene	2.17×10^{-8}	1.6×10^{-11}	2.71×10^{-14}	8.7×10^{-9}	3.73×10^{-9}
Trichloroethene	1.71×10^{-8}	1.26×10^{-11}	1.63×10^{-13}	--	--
Vinyl Chloride	3.17×10^{-5}	2.33×10^{-8}	6.88×10^{-8}	--	--
Group 2 Totals			6.88×10^{-8}		1.34×10^{-6}
Total Pathway Risks/ Hazard Indices			7.05×10^{-8}		1.34×10^{-6}

02[IL]JD1900:D3048/1249/13

*The average absorbed doses differ for the carcinogenic and noncarcinogenic effects assessments because different averaging times are used.

Source: Ecology and Environment, Inc. 1990.

Table 5-29

EXPOSURE PATHWAY 3C: RESIDENTIAL DERMAL CONTACT WITH SHOWER WATER RISK ESTIMATES

Chemical	Exposure Concentration (as Modeled) in Water at Mouth of Niagara River (mg/L)	Cancer Risk Assessment: Average Absorbed Dose* (mg/kg/day)	Cancer Risk Assessment: Cancer Risk (Dermal Contact)	Noncarcinogenic Risk Assessment: Average Absorbed Dose* (mg/kg/day)	Noncarcinogenic Risk Assessment: Chronic Hazard Index (Dermal)
Group 1: Site-Derived Organics					
Benzene	6.5×10^{-6}	1.43×10^{-10}	4.15×10^{-12}	--	--
Chlorobenzene	9.7×10^{-8}	--	--	5.81×10^{-10}	2.49×10^{-10}
1,2-Dichlorobenzene	3.5×10^{-8}	--	--	4.67×10^{-11}	2×10^{-11}
1,3-Dichlorobenzene	1.4×10^{-8}	--	--	--	--
1,4-Dichlorobenzene	3.05×10^{-8}	6.72×10^{-13}	1.61×10^{-14}	5.23×10^{-12}	2.24×10^{-12}
1,2,4-Trichlorobenzene	1.3×10^{-7}	--	--	7.79×10^{-11}	3.34×10^{-11}
α-HCCH	3×10^{-9}	6.61×10^{-14}	4.17×10^{-13}	--	--
β-HCCH	9.3×10^{-10}	2.05×10^{-14}	3.69×10^{-14}	--	--
γ-HCCH	3.4×10^{-9}	7.49×10^{-14}	9.74×10^{-13}	1.36×10^{-9}	5.83×10^{-10}
δ-HCCH	2.2×10^{-10}	--	--	--	--
Group 1 Totals			4.72×10^{-12}		8.88×10^{-10}
Group 2: Non-Site-Derived Organics					
1,2-Dichloroethane	--	--	--	--	--
1,2-Dichloroethene (total)	6.3×10^{-7}	--	--	3.78×10^{-9}	1.62×10^{-9}
Tetracloroethene	2.38×10^{-9}	5.25×10^{-14}	2.68×10^{-15}	2.85×10^{-11}	1.22×10^{-11}
Trichloroethene	1.7×10^{-9}	3.75×10^{-14}	4.12 x 10	--	--
Vinyl Chloride	2.25×10^{-3}	4.96×10^{-11}	1.14×10^{-10}	--	--
Group 2 Totals			1.14×10^{-10}		1.63×10^{-9}

Table 5-29 (Cont.)

Chemical	Exposure Concentration (as Modeled) in Water at Mouth of Niagara River (mg/L)	Cancer Risk Assessment: Average Absorbed Dose* (mg/kg/day)	Cancer Risk Assessment: Cancer Risk (Dermal Contact)	Noncarcinogenic Risk Assessment: Average Absorbed Dose* (mg/kg/day)	Noncarcinogenic Risk Assessment: Chronic Hazard Index (Dermal)
Group 3: Metals and Cyanides					
Arsenic	2.24×10^{-9}	4.94×10^{-16}	8.46×10^{-16}	2.68×10^{-12}	1.15×10^{-12}
Barium	4.48×10^{-9}	--	--	1.08×10^{-13}	4.61×10^{-14}
Cadmium	2.54×10^{-9}	--	--	6.09×10^{-12}	2.61×10^{-12}
Chromium	5.68×10^{-6}	--	--	6.81×10^{-12}	2.92×10^{-12}
Lead	5.57×10^{-8}	--	--	2.23×10^{-10}	9.55×10^{-11}
Manganese	2.78×10^{-7}	--	--		
Zinc	4.46×10^{-7}	--	--	2.68×10^{-12}	1.15×10^{-12}
Cyanide	1.67×10^{-7}	--	--	1×10^{-7}	4.29×10^{-8}
Group 3 Subtotals			8.46×10^{-16}		4.3×10^{-8}
Total Pathway Risks/ Hazard Indices			1.19×10^{-10}		4.55×10^{-8}

02[IL]JD1900:D3048/1250/13

*The average absorbed doses differ for the carcinogenic and noncarcinogenic effects assessments because different averaging times are used.

Source: Ecology and Environment, Inc. 1990.

water elevation at the site does not fluctuate greatly with the seasons, probably because of the relatively constant upgradient head provided by the Niagara River and the downgradient sink formed by the NYPA tunnel and the Falls Street Tunnel. This would tend to limit seasonal fluctuations in groundwater contaminant concentrations.

Some of the groundwater contaminant concentrations were above the normal calibration range for the analytical methods employed. This caused these concentrations to be estimated (E flag). Such estimates do not have the same degree of precision and accuracy as concentrations measured within the normal calibration range. This marginally decreases the degree of confidence in these values and the exposure and risk estimates dependent on these values.

5.8.1.2 Contaminant Migration Modeling

Uncertainties about the reliability of modeling predictions arise in two areas. The first is the validity and accuracy of the model itself and the second is the selection of appropriate parameter values for use in the calculations. Three general types of models were used in developing exposure estimates.

A simple hydrogeological model based on Darcy's Law, one of the basic laws of hydrogeology, was used to estimate groundwater flow velocities and fluxes through certain subsurface areas. It is widely used and has stood the test of time. This model was used to estimate the concentration of contaminants in the Niagara River below the falls that could result from the discharge of site-derived contaminants from the Falls Street Tunnel to the river. The parameter values required are the porosity and permeability of the bedrock and the differences in the hydraulic head from location to location. All of these values were obtained from geological and hydrogeological investigations at or near the site. The contaminant migration calculations used to estimate the contaminant loading to the river via the groundwater neglect possible decreases in contaminant concentrations in the groundwater due to adsorption, precipitation, volatilization, and chemical and biological degradation of contaminants as the groundwater moves from the site to the river. This is an extremely conservative approach since it is quite likely that the contaminant concentrations in the groundwater will be substantially reduced by one or more of these processes before the groundwater discharges to the river. Neglecting these processes will result in over estimation of the contaminant loading to the river and the potential exposures experienced by receptors by pathways involving initial contaminant migration via the groundwater. The overall results of the hydrogeological modeling should be moderately reliable.

A second type of model was used to estimate the contaminant concentrations in the river resulting from contaminants discharged to the river from the Falls Street Tunnel, and the subsequent evaporation of volatile contaminants from the river. The river dispersion and volatilization model described by Thomann and Mueller (1987), which is a standard stream mixing model, was used for this purpose. It has been thoroughly validated and is recommended in EPA's Water Quality Assessment Manual (EPA 1985). The key parameters are the rate of discharge of contaminants to the river, obtained from the groundwater migration estimates discussed above; the width, depth and velocity of the river, which were obtained from published river flow data and the Henry's Law constants for the volatile contaminants.

The third type of modeling is air dispersion modeling, which was used to estimate ambient air concentrations downwind from the vapor emission source areas of the river below the falls. A simple box model was used to estimate near field concentrations experienced by fisherman and workers in the gorge.

The box model is a basic dilutional model that estimates air concentrations by diluting the contaminant flux emerging from the ground by the volume of air passing over the source area. The key parameters, the size of the source area and the windspeed, are selected on a site-specific basis. The results of this model should be moderately reliable.

5.8.1.3 Exposure Estimation Calculations

The primary uncertainty regarding the exposure estimation calculations is that associated with the selection of appropriate parameter values. The values used and a brief rationale for their selection are given in the tables in Section 5.5, which describe the exposure calculations for the various pathways. Individual parameter values were selected so that the overall pathway exposure estimates would approximate a reasonable maximum exposure. It is important to note that in attempting to estimate the RME, it is generally not appropriate to use a worst-case or an upper-95th percentile estimate for every parameter in the calculation (EPA 1989a) because this will result in a pathway estimate that is so conservative it is in fact an extreme worst-case estimate (perhaps a 99.9999th percentile estimate), rather than a reasonable worst-case estimate in the 99th to 99.9th percentile range.

5.8.1.4 Steady State Assumption

The exposure calculations used in this risk assessment all assume that the contaminant concentrations in the source media are in a steady state and remain constant for the duration of the exposure periods which in most cases was 30 years. The steady state assumption was used because there is no chronological data available for contamination concentrations in the soil or groundwater on site or for the presumed groundwater plume that could serve as a basis for estimating the rate of source decay. The volatile aromatic compounds (BTEXs) found in the soil and groundwater at the site are all readily degradable under suitable conditions. Therefore, assuming that the source concentrations remain constant over those periods and that no degradation occurs in the groundwater, the true exposures will probably be substantially overestimated.

Carbon disulfide was found in the soil and in a dense NAPL sample from well 2B in relatively high concentrations but was not found in the groundwater. Since it was not present in the groundwater it was not considered in the quantitative assessment of the risks posed by groundwater contaminants at the site. Carbon disulfide is relatively mobile in the subsurface therefore its presence in high concentrations in the near surface soils and absence from deeper soils and groundwater may indicate that its presence on site is the result of a relatively recent release. If this is true, carbon disulfide might not have had time to reach a steady state distribution at the site and might appear in groundwater in the future.

5.8.1.5 Exposure Assessment Uncertainty Summary

Overall the exposure estimates obtained are probably moderately reliable. A number of the factors adding uncertainty to the estimates tend to result in overestimation of the exposure. These include:

- The use of conservative parameter values and assumptions in the various contaminant transport models;
- The use of the steady state assumption for source concentration estimates; and
- The assumption, in the absence of data to the contrary, that contaminant degradation is not occurring in the groundwater.

Factors that could result in under estimation of potential exposures are:

- The use of the steady state assumption for carbon disulfide.

The cumulative effect of all of the exposure uncertainties most likely is to overestimate rather than underestimate the true potential exposures.

5.8.2 UNCERTAINTIES RELATED TO THE TOXICITY ASSESSMENT

5.8.2.1 Introduction

In order to evaluate the meaning of any risk assessment, one must consider the uncertainties in the assumptions made, the impact of changing the magnitude of those assumptions on the risk estimates, and the relevance of the findings to real world exposures and risks. Due to the number of assumptions, data points, and calculations, a degree of uncertainty is necessarily associated with the numerical toxicity values in any risk assessment.

This section begins with a discussion of the assumptions used to estimate carcinogenic risks, continues with a discussion of noncarcinogenic risk estimates, and concludes with a discussion of the other major assumptions used in developing the exposure scenarios.

5.8.2.2 Evaluation of Carcinogenic Toxicity Assessment Assumptions

The chemicals of concern have been evaluated by EPA using its weight-of-evidence carcinogenicity evaluation criteria and have been placed in Group A, human carcinogens, or Group B, probable human carcinogens, based on sufficient data in humans and animals and insufficient data in humans, respectively (EPA 1986b).

Rodent bioassay and epidemiological studies, such as those performed for the chemicals of concern, would require tens of thousands of animals or humans in order to determine whether or not a chemical may be carcinogenic at low doses. As the relationship between tumor location, time to appearance, and the proportion of animals with cancer determines the estimated carcinogenic SF, animal bioassay or human epidemiological data are not routinely sufficient for directly estimating SF at low doses. Therefore, by necessity, agencies such as EPA use carcinogenic extrapolation models for estimating low doses SFs. Based upon policy grounds, these agencies assume that there is no threshold dose below which carcinogenic risks will not occur. This is equivalent to the assumption that every dose above zero, no matter how low, carries with it a small but finite risk of cancer. They also assume that the dose-response relationship is linear at low doses. This is contrary to approaches used for other toxic effects, because thresholds are assumed to exist.

The current model favored by EPA and certain other federal regulatory agencies is the linearized multistage model. The agency then uses the statistically derived upper 95% confidence bounds, rather than a maximum likelihood value for SF. The agency has concluded, based on theoretical grounds consistent with human epidemiological and animal data, that cancer follows a series of discrete stages (i.e., initiation, promotion, and progression) which ultimately can result in the uncontrolled cell proliferation known as cancer. Consistent with this conclusion, the use of the linearized multistage model permits an estimation of SF which is not likely to be exceeded if the real slope could be measured. However, compelling scientific arguments can be made for several other extrapolative models which, if used, could result in significantly reduced values for SFs, some tens of millions of times lower than those estimated using the linearized multistage model. The one hit model used to estimate risks due to exposures above the linear range of the multistage model is one such model. Thus, the current EPA SFs calculated in this fashion represent upper-bound values based on animal data which should not be interpreted as necessarily equivalent to actual human cancer potencies. It is this conservative value, nevertheless, that is used in this risk assessment on policy grounds for the protection of public health.

5.8.2.3 Evaluation of Noncarcinogenic Toxicity Assessment Assumptions

Key assumptions used in assessing the likelihood of noncarcinogenic effects are that threshold doses exist below which various noncarcinogenic effects do not occur and that the occurrence or absence of noncarcinogenic effects can be extrapolated between species and occasionally between routes of exposure and over varying exposure durations. The threshold assumption appears to be sound for most noncarcinogens based on reasonably good fits of experimental data to the usual dose response curves. One possible exception to this is lead, which may not have a threshold base for its noncarcinogenic effects (EPA 1988b).

The other assumption generally appears to be true to varying degrees. The effects observed in one species or by one route of exposure may not occur in another species or by another route, or they may occur at a higher or lower dose due to differences in the pharmacokinetics (uptake, distribution, metabolism, and elimination) of a compound in different species or by different routes of exposure. The uncertainty in these assumptions is taken into account in the development of RfDs through the use of safety or uncertainty factors. The uncertainty factors used by EPA are conservative (health protective) in nature in that they tend to overestimate the uncertainties so that the RfDs obtained are unlikely to be too high. Use of the resulting RfDs tends to overestimate the potential for noncarcinogenic effects occurring at a given exposure level. Table 5-17 presents uncertainty factors used to derive the RfDs for chemicals of concern at the Buffalo Avenue site.

Uncertainty factors used to derive RfDs are presented in Table 5-18 for each chemical of concern. For example, an uncertainty factor of 100 was used to derive the RfD for xylenes: 10 for species-to-species extrapolation and 10 to protect sensitive individuals. In addition to uncertainty factors, a modifying factor is applied to reflect a qualitative professional assessment of additional uncertainties in the critical study and in the entire data base for the chemical not explicitly addressed by the preceding uncertainty factors. The modifying factor ranges from >0 to 10 with a default value of 1 (EPA 1989a).

5.8.2.4 Discussion of Confidence in the SF

The degree of confidence presented in Table 5-19 reflects the EPA Carcinogen Risk Assessment Verification Endeavor (CRAVE) work groups' judgments about the ability of the risk measures derived from dose-response assessment to estimate the risks of that chemical to humans. Verification involves consideration of factors that increase or decrease confidence in the numerical risk estimate. The following criteria are considered:

- Appropriateness of data to estimation of human carcinogenic risks;
- Quality of study design;
- Strength of study results;
- Appropriateness of model application to the data; and
- Support of risk estimate from collateral studies.

For example, confidence in the oral RfD for xylenes as defined in IRIS is medium based on a well-designed study in which adequately sized groups of two species were tested over a substantial portion of their lifespan. Comprehensive histology was performed and a NOAEL was defined; clinical chemistries, blood enzymes, and urinalysis were not performed. The data base was given a medium confidence level because, although supporting data exist for mice, and teratogenicity and fetotoxicity data are available with positive results at high oral doses, a LOEL for chronic oral exposure has not been defined. Medium confidence in the RfD follows. Confidence levels for verified RfDs are included in Table 5-18.

5.8.2.5 Unavailable Toxicity Values

Toxicity values suitable for risk assessment purposes are not available for lead.

Lead has been classified as a category B2 carcinogen. However, because of uncertainties in the underlying data and complex pharmacokinetics that may be unique to lead, EPA's carcinogen assessment group recommends that a numerical estimate of lead's carcinogenic potency not be used at this time (IRIS 1990). Lead also exhibits a number of noncarcinogenic adverse effects. However, there is some question about whether a threshold dose (i.e., a no-effect exposure level) exists for these effects. Consequently, the EPA's RfD work group considers it inappropriate to develop an RfD for lead (IRIS 1990).

Despite that quantitative toxicity indices are not available, EPA's Office of Drinking Water proposed a revised maximum contaminant level goal (MCLG) and a revised maximum contaminant level (MCL) for lead in drinking water of 0 and 5 µg/L respectively in August 1988 (EPA 1988b). The proposed MCL was based on health effects considerations, data on existing lead concentrations present in public water supplies, and the technical feasibility of achieving various treatment standards. A purely health-based reference dose was not derived in the proposed rule-making, nor can one be inferred from the discussion of the rule-making process. However, the unit dose to an individual using drinking water containing the MCL concentration of lead can be calculated. Since a young child is probably the most sensitive receptor for lead intoxication a 16 kg child who consumes 1 liter of water per day was assumed to be the receptor for the purpose of calculating the MCL equivalent dose (MCLED). The MCLED is given by the following equations.

$$\text{MCLED} = \frac{(\text{MCL Conc.}) \times (\text{Water Consumer Per Day})}{(\text{Body Weight})}$$

$$= \frac{(0.005\ \mu\text{g/L}) \times (1\ \text{L})}{(16\ \text{kg})}$$

$$= 3.125 \times 10^{-4}\ \text{mg/kg-day}$$

The MCLED can be used in the same way as an RfD in evaluating the potential risks posed by various concentrations of lead, but with a different interpretation. Normally, if the estimated chronic daily intake (CDI) of a substance does not exceed its RfD, this would indicate that no adverse effects would be expected. In contrast, if the estimated CDI of lead does not exceed the MCLED, this would indicate that the risk of adverse health effects would be no greater than the risks associated with drinking water with an MCL concentration of lead, a risk that has been judged tolerable by the EPA Office of Drinking Water.

The lack of verified toxicity indices for lead and the PAHs introduces a significant measure of uncertainty into the toxicity assessment. However, this uncertainty is mitigated to some extent by the use of the MCL equivalent dose.

5.8.2.6 Summary of Toxicity Assessment Uncertainties

The basic uncertainties underlying the assessment of the toxicity of a chemical include:

- Uncertainties arising from the design, execution or relevance of the scientific studies that form the basis of the assessment; and
- Uncertainties involved in extrapolating from the underlying scientific studies to the exposure situation being evaluated, including variable responses to chemical exposures within human and animal populations, between species, and between routes of exposure.

These basic uncertainties could result in a toxicity estimate, based directly on the underlying studies, that either under- or overestimates the true toxicity of a chemical in the circumstances of interest.

The toxicity assessment process compensates for these basic uncertainties through the use of safety factors (uncertainty factors) and modifying factors, when assessing noncarcinogens, and the use of the upper 95% confidence limit from the linearized multistage model for the SF when assessing carcinogens. The use of the safety factors and the upper 95% confidence limit in deriving the RfDs and SFs ensures that the toxicity values used in the risk estimation process are very unlikely to underestimate, and thus, almost always overestimate, the true toxicity of a chemical.

The absence of verified toxicity values for lead introduces a significant measure of uncertainty into the toxicity assessment. However, the use of an alternate toxicity value reduces the uncertainty.

5.8.3 Additional Uncertainty Factors

There are several additional factors that need to be considered when discussing uncertainties associated with the overall risk characterization. These are the cumulative effect of using conservative assumptions throughout the process and the likelihood of the exposures postulated and estimated in the exposure assessment actually occurring.

The cumulative effect of using conservative assumptions throughout the risk estimation process is that the resulting estimate will substantially overestimate the true risks. The Risk Assessment Guidance for Superfund manual (EPA 1989a) recommends that individual parameter values be selected so that the overall estimate of exposure, for example, represents a "reasonable maximum exposure." In many cases, the statistical distribution of a parameter is unknown and the risk assessor is left to select a value, using best professional judgment, that is sufficiently conservative to avoid underestimating the true risk, yet not so conservative that the resulting risk estimate turns out to be unreasonably high. When in doubt, the risk assessor will usually elect to err in favor of protecting human health and select a value that results in overestimating the true risk.

Conservative estimates are typically used at every stage of the risk assessment process, including:

- Selection or derivation of source media concentrations;
- Selection of the parameters used in estimating contaminant migration and receptor exposure;
- Selection or derivation of the average exposure point concentrations over the exposure duration postulated (steady state assumption is often used); and

- o Derivation of quantitative indices of toxicity (safety factors are used in deriving RfDs and the upper 95% confidence limit on the multistage model estimate is used as the carcinogenic potency SF).

In the risk estimation process, these estimates and the parameters contributing to the estimates are usually combined by multiplying them together. If two values, each an upper 95th percentile estimate, are multiplied together, the resulting value would be an upper 99.75th percentile estimate for the product. If three 95th percentile values are multiplied together, the result is an upper 99.9825th percentile estimate and four 95th percentile estimates yield a 99.9994th percentile product, which means the estimate has less than 1 chance in 100,000 of underestimating the actual value. A risk estimate derived in this way would obviously be extremely conservative and would substantially overestimate the true risks. There are many instances in the risk assessment process in which four or more parameters are multiplied together to obtain a risk estimate.

In summary, the nature of the risk estimation process itself, in which many factors are multiplied together, inherent uncertainties exist about parameter values, and conscious decisions are made by risk assessors and the regulatory agencies to err on the side of protecting human health, virtually ensures that the true risks will be overestimated, sometimes by large margins.

The last uncertainty factor to consider is the likelihood of the postulated exposures actually occurring. The exposure pathways identified as complete under existing conditions are all plausible and exposure is either presently occurring, or could potentially occur, by these pathways. The postulated frequencies of occurrence may overestimate routine occurrence but could certainly reflect the reasonable maximum occurrence.

5.9 RISK COMPARISON

5.9.1 Introduction

Section 5.7 presents quantitative estimates of risks for exposed individuals assuming lifetime exposures to chemical contaminants according to the conditions specified in the exposure scenarios.

Judging the potential health significance of these risk estimates is difficult, particularly in light of the fact that the estimates are based on extrapolation from animal data and that other plausible arguments can be made that actual risks may be as low as zero. In order to place the data in perspective, this section will compare the estimated risks with the following three data sets:

1. Current regulatory practices which define acceptable (de minimus) levels of risk;
2. Existing background cancer rates for the general population; and
3. Normal risks in everyday living.

In making these comparisons, distinctions must be made among the types of uncertainty in the data used to generate risk estimates, guidelines, and standards. Carcinogenic risk estimates for the indicator chemicals identified in this report, as well as most risk estimates in general, use as their basis current regulatory practices defining acceptable level risks, which are primarily based on laboratory animal data. The risks so generated represent estimated upper-bound potential human risks, assuming that laboratory animal data are directly applicable to humans. As these data sets were generated using comparable risk assessment methodologies, the risks can be directly compared with one another. However, use of these data is not meant to imply that they represent actual or real risks to humans, but rather are estimates generated on the basis of prudent public health policy which assumes they have value in protecting human health.

On the other hand, existing background cancer rates (number 2, above) and risks of everyday living (number 3, above) are based on actual human data. Thus, though they can be compared, contaminant risks are hypothetical, whereas the comparison data are real.

5.9.2 Current Regulatory Policy or Practice Regarding Acceptable Risks

In general, regulatory agencies in the United States have not established a uniform cancer risk level for distinguishing between risks which are significant and those which are insignificant (de minimis).

EPA has recently adopted the policy that acceptable exposures are generally those that represent an excess upper bound lifetime cancer risk to an individual of between 10^{-4} and 10^{-6}. In addition, EPA will use the 10^{-6} risk level as the point of departure for determining remediation goals for National Priorities List (NPL) sites (EPA 1990).

Travis et al. (1987) have recently reviewed 132 federal regulatory decisions involving potential carcinogens for which risks have been evaluated using the 95% upper confidence level of the linearized multistage model. Both individual risk and population risk factors have been influenced by individual decisions not to regulate specific chemical releases. Table 5-30 summarizes recent decisions by EPA for air quality, radiation programs, and toxic substances; as summarized in the table, while individual risks ranged from 10^{-3} to 10^{-5}, annual total population cancer risks were too low (0.001 to 6) to regulate. The evaluation of potential carcinogenic risks associated with recent EPA-promulgated nationwide drinking water maximum contaminant levels (MCLs) reinforces these conclusions (see Table 5-31). As presented in the table, E & E has estimated that these EPA drinking water MCL regulations for four probable or known human carcinogens were set at levels corresponding to estimated lifetime cancer risks based on EPA SFs (EPA 1986) ranging from 1.3×10^{-4} to 1.4×10^{-6}.

5.9.3 Risks Associated with Everyday Living

Most human activities present some degree of risk. Thus, to place the estimated risks into perspective, background risks should be analyzed. Table 5-32 presents annual average and lifetime risks of death, as well as the time necessary to accumulate a one-in-a-million risk of death during the natural course of a lifetime. For example, the risk of death from such possible occurrences as motor vehicle accidents (1.7×10^{-2}), fires (1.7×10^{-3}), lightning (3.5×10^{-5}), and animal bites or stings (1.7×10^{-5}) typically exceed the risks discussed in this document.

5.9.4 Cancer Rates for the General Population

Using the same risk assessment methodologies, one can estimate everyday cancer risks due to dietary intake and inhalation of probable

Table 5-30

EPA DECISIONS CITING INSIGNIFICANT POPULATION RISK AS THE REASON NOT TO REGULATE

Chemical	Agency Office	Individual Risk	Population Risk (Cancer/year)	Agency Comments	Source
Arsenic					
Zinc oxide production	Air	3×10^{-3}	0.02	Total cancer incidence, on a national basis is likely to be small compared to the incidences associated with smoking and diet.	50 Federal Register 3112
Secondary lead smelters	Air	3×10^{-3}	6		
Primary lead smelters	Air	1×10^{-4}	0.1		
Epichlorohydrin	Air	1×10^{-5}	0.001	Relatively low aggregate risk.	50 Federal Register 24575
Elemental phosphorus	Radiation programs	1×10^{-3}	0.06	The 1×10^{-3} individual risk must be weighed against low aggregate risk and other factors.	50 Federal Register 32632
Vinylidene chloride	Air	8×10^{-4}	0.07	Magnitude of public health risk is too small.	
Radionuclides					
Department of Energy Facilities (DOE)	Radiation programs	7×10^{-4}	0.07	Population impact insufficient to warrant regulation.	50 Federal Register 5190
Nuclear Regulatory Commission non-DOE Facilities	Radiation programs	2×10^{-5}	0.001	Population impact insufficient to warrant regulation	50 Federal Register 5190
Formaldehyde					
Teachers	Toxic substances	7×10^{-5}	0.1	Population risk below one cancer per year which was judged as insignificant risk of widespread harm.	--
Students	Toxic substances	3×10^{-6}	0.001 - 0.1		

02[IL]JD1900:D3048/1169/3

Source: Travis et al. 1987.

Table 5-31

EPA DRINKING WATER STANDARDS FOR FOUR ORGANICS CLASSIFIED AS PROBABLE OR HUMAN CARCINOGENS

Chemical	Standard (μg/L)*	Estimated Lifetime Cancer Risk**
Benzene	5	1.4×10^{-6}
1,2-Dichloroethane	5	1.3×10^{-5}
Trichloroethylene	5	2.0×10^{-6}
Vinyl chloride	2	1.3×10^{-4}

02[IL]JD1900:D3048/1170/32

*Source: 52 FR 25690-25717, July 1987.

**Estimated by E & E using current USEPA SFS for each of the specific chemicals (EPA 1986).

Table 5-32

ESTIMATED LIFETIME ONE-IN-A-MILLION RISKS OF DEATH FOR UNITED STATES RESIDENTS

Incidents	Average Lifetime Risks*
Motor vehicle accident	1.7×10^{-2}
Falls	4.3×10^{-3}
Drowning	2.5×10^{-3}
Fires	1.7×10^{-3}
Firearms	7.0×10^{-4}
Electrocution	3.9×10^{-4}
Floods	4.2×10^{-5}
Lightning	2.5×10^{-5}
Animal bite or sting	1.7×10^{-5}

02[IL]JD1900:D3048/1171/24

*Assumes 70-year lifetime.

Source: Crouch and Wilson 1982.

or known human carcinogens (see Table 5-33). Recent statistical information generated by the National Cancer Institute indicates that the lifetime probability of developing cancer (incidence rate) is about 1 in 4 (0.25) for males and slightly higher for females.

Other cancer risks include those related to smoking (8.4×10^{-2} or 84,000 per million), sharing a room with a smoker (7.0×10^{-4}), eating one tablespoon of peanut butter per day (1.4×10^{-4}), and living at sea level (1.4×10^{-4}), the latter risk attributable to natural background radiation (Crouch and Wilson 1982).

Thus, the carcinogenic risks estimated in this report are considerably lower than these attributable to smoking-related risks and a variety of cancer risks attributable to natural living conditions.

Table 5-33

DERIVED EVERYDAY CANCER RISKS

Activity	Average Lifetime Risk
All cancers	0.25
One transcontinental round trip by air per year*	7×10^{-5}
Natural background radiation at sea level	1.4×10^{-4}
Average diagnostic X-ray	1.4×10^{-4}
Eating one tablespoon of peanut butter per day**	1.4×10^{-4}
Eating one pint of milk per day**	1.4×10^{-4}
Eating one-half pound charcoal broiled steak per week***	2.1×10^{-5}
Smoker	8.4×10^{-2}
Person sharing a room with a smoker	7.0×10^{-4}

02[IL]JD1900:D3048/1172/25

*Estimated based on exposure to cosmic rays.
**Estimated based on exposure to aflatoxin, a natural carcinogen contained in a variety of foods.
***Estimated based on exposure to polycyclic aromatic hydrocarbons formed during charcoal broiling of steaks.

Source: Crouch and Wilson 1982.

6. CONCLUSIONS

6.1 RI CONCLUSIONS

Major contaminants of concern are: benzene, chlorobenzene, 1,2-dichlorobenzene, 1,3-dichlorobenzene, 1,4-dichlorobenzene, and 1,2,4-trichlorobenzene. Arsenic, barium, cadmium, chromium, lead, manganese, zinc, and cyanide were also detected in high concentrations. 1,2-dichloroethane (total), tetrachloroethene, trichloroethene, and vinyl chloride were also found in groundwater on site.

The overburden is a mixture of fill, silt, and clay and ranges in thickness from 6 to 12 feet. The fill is a combination of industrial debris and dolomitic cobbles in a silty sandy matrix. Its average thickness is 1.5 to 3.5 feet. The glacial sediment beneath the fill varies in thickness from 2 to 9 feet and is composed of silt and clay.

Soil sampling from the RI has shown that two areas of high soil contamination (greater than 100,000 microgram/kilogram [µg/kg]) exist at the site. These areas are primarily in the southeast and southwest site quadrants. Benzene, chlorobenzene, and the various derivatives of dichlorobenzene and trichlorobenzene were the most significant chemicals found in these two areas. Elevated concentrations of metals, such as arsenic, barium, cadmium, chromium, manganese, zinc, and cyanide, were also found in these areas.

The bedrock underlying the site is the Lockport Dolomite. All bedrock wells on site terminate in the Oak Orchard member of this unit. Fracture trends generally follow horizontal bedding planes, but vertical and high-angle fractures were also observed in drill cores. For monitoring purposes, the Lockport has been divided into four fracture zones (B, C, CD, and F) which are present on site between 10 and 100 feet below ground surface.

Sixteen monitoring wells were installed in the saturated A-Zone and in the B- through F-Zones of the Lockport Dolomite primarily to characterize groundwater quality on the site. These wells substantiated that the southwest area of the site was a source area for contaminants. This is best evidenced by the fact that very high levels (greater than 100,000 µg/L) of benzene and benzene-related compounds were found in monitoring wells in the southwest area. Groundwater from monitoring wells immediately downgradient of the southeast area yielded concentrations one to two orders of magnitude lower than groundwater from monitoring wells in the southwest. Therefore, while the southeast area must contribute contaminants to the groundwater, its impact is not as significant as the southwest area.

Groundwater in the A- and B-Zones flows across the site toward the north, northwest and west. The CD-Zone also flows to the northwest, and the C- and F-Zones flow toward the northeast. It should be noted, however, that groundwater flow directions in C-, CD-, and F-Zones were determined using only two or three wells per zone and thus should be regarded as approximate. Contaminants in the groundwater (chiefly benzenes and related benzene compounds) flow in the direction of groundwater transport, which is principally toward north, northwest, and west. Moderately high levels (greater than 1,000 µg/L) were detected in monitoring well nest MW-5 and monitoring well nest MW-4, which are located at the north and northwest site boundaries, respectively.

The saturated overburden is an aquitard that stores but does not transmit significant quantities of groundwater. The average hydraulic conductivity of this unit is 4.8 gpd/ft^2, but this value is heavily weighted by the presence of the fill material, which is much more conductive than native soils. Wells which terminate in the glacial sediment are an order of magnitude less conducive than well MW-7A, which terminates in fill. The maximum horizontal groundwater gradient observed in A-Zone is 0.021 ft/ft. Groundwater velocity ranges from 0.12 ft/day to 13.5 ft/day in the A-Zone, based on effective porosity values of 0.1 to 0.001. Velocities of groundwater are not reflective of contaminant velocity, because the compounds are slowed due to retardation. The direction of flow in this zone is to the north and west.

The B- through F-Zones in the Lockport Dolomite are named for water-bearing fracture zones. These zones show an increase in groundwater velocity and hydraulic conductivity with depth. A downward vertical hydraulic gradient is seen from the B- through CD-Zones, while the F-Zone maintains a potentiometric surface equal to that of the B-Zone. Fracture interconnectedness, as well as vertical and high-angle fracture frequency in these zones, dictates groundwater and therefore contaminant migration. Flow direction in these zones is to the northeast in the C- and F-Zones and north and northwest in the B- and CD-Zones.

Levels of compound concentrations and their pattern of distribution on site indicate a source area in the southwest quadrant of the site. Soil samples SB2A2 and SB1B2 from the monitoring well nest 1 and 2 show high levels of contaminants, which are also seen in the groundwater samples from these wells. Subsurface composite soil samples from the southeast corner exhibit high levels of chlorobenzene, the three dichlorobenzene isomers, and 1,2,4-trichlorobenzene; however, the nearest downgradient wells, MW-6A and MW-6B, have significantly lower groundwater concentrations than the soils.

TCE and its degradation products are not found in soil samples to any significant degree on site, but are detected in groundwater samples. The most consistent occurrences are seen in the lower fracture zones, C and F, and are thus considered to have an off-site origin.

Loading rates show a general increase with depth. A maximum loading rate of 13.92 lb/day is seen in the B-Zone, which is heavily weighted by well MW-1B and its correspondingly high hydraulic conductivity. Northwestern boundary wells MW-4A and MW-4B exhibit comparatively heavier loadings than other boundary wells, due to the general northwesterly trend of groundwater flow.

6.2 RISK ASSESSMENT CONCLUSIONS

The RI and risk assessment focused on the possible threat posed by site contamination to the groundwater, and the risks contaminated groundwater could pose to human health. Contamination was found in subsurface soils and groundwater, and, in most cases, the same contaminants were present in both media. Therefore, the potential contaminants of concern used in the quantitative risk assessment were selected from among the groundwater contaminants.

For the purposes of the risk assessment, all site groundwater contaminants were assumed to be intercepted by the Falls Street Tunnel, which discharges to the Niagara River near the Rainbow Bridge. The most likely scenario is that some site groundwater and contaminants would flow to the gorge without being intercepted by the Falls Street Tunnel. Under this model, contaminant loading to the Niagara River would actually be reduced because of attenuation through dispersion, degradation, and volatilization of chemicals that remain in the groundwater. These physical and chemical processes would be enhanced in groundwater because contaminants would have a larger residence time prior to intercepting the gorge compared to a shorter residence time in the Falls Street Tunnel. The groundwater and Falls Street Tunnel reception model has not been used because no off-site groundwater data exists to measure attentuation of contaminants leaving the site. Use of the Falls Street Tunnel as the primary transport receptor is regarded as being more conservative than the groundwater/ Falls Street Tunnel receptor because less time is available for attenuation in the Falls Street Tunnel. Thus, the highest contaminant loading will be via the tunnel.

Three potential exposure pathways were evaluated:

- Inhalation of site-derived contaminants by workers in the gorge near the Rainbow Bridge;
- Exposure of recreational fishermen using the Niagara River near the Power Authority generating station; and
- Use of Niagara River water at the mouth of the river for residential drinking water purposes.

It appeared that some of the groundwater contaminants of potential concern were either coming onto the site with the groundwater from upgradient sources (i.e., volatile organics) or were associated with suspended particulates in the groundwater, which would render them essentially immobile. Therefore, the risks associated with these contaminants were summed separately from those associated with the site-derived contaminants that are likely to be mobile.

The estimated cancer risks due to the mobile, site-derived contaminants were about 30 times less than 1 x 10^{-6}, the minimum risk level considered to be of concern by the EPA and many state regulatory agencies. The total estimated risk due to all of the site groundwater contaminants just reached the 1 x 10^{-6} level. However, because of the conservative approach used in estimating contaminant loadings to the Niagara River, in particular, and in the risk assessment process as a whole, this estimate probably substantially overestimates the actual risks posed by site groundwater contaminants via the exposure pathways evaluated. The estimated noncarcinogenic risks for all of the site groundwater contaminants were about 3,000 times less than the threshold level above which the potential for adverse noncarcinogenic effects begins to be of concern. To summarize, it is unlikely that all of the site groundwater contaminants taken together would pose a significant threat to human health by the exposure pathways evaluated, and even less likely that the mobile, site-derived contaminants would pose such a risk.

The groundwater at the site is classified GA by NYSDEC indicating that its best potential use is as a drinking water supply source. The groundwater at and in the vicinity of the site is not presently used for that purpose, nor is such use likely in the future, because the area is served by a public water supply system. For this reason use of site groundwater as a drinking water source was not considered in the risk assessment. The concentrations of many of the chemicals found in the groundwater at the site exceeds NYSDEC standards or guidelines for Class GA groundwater and state and/or federal drinking water standards. Thus while the groundwater at the site is classified as a potential drinking water source, it is presently unsuitable for that use because of the contaminants present.

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