FINAL PHASE II
REMEDIAL INVESTIGATION REPORT

OLIN CHEMICALS
ROCHESTER PLANT SITE
ROCHESTER, NEW YORK

VOLUME I

Submitted to:

Division of Hazardous Waste Remediation
New York State Department of Environmental Conservation
50 Wolf Road
Albany, New York 12233-4011

Prepared by:

ABB Environmental Services, Inc.
511 Congress Street
Portland, Maine 04101

OCTOBER 1997

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ASEA BROWN BOVERI

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EXECUTIVE SUMMARY

This report presents the findings of the Phase II Remedial Investigation (RI) conducted at the Olin Corporation Chemicals Division (Olin) manufacturing plant in Rochester, New York, between August and December 1995. The Phase II RI was performed to fulfill part of the requirements of the Consent Agreement between the New York State Department of Environmental Conservation (NYSDEC) and Olin (Index No. B8-0343-90-08). The Phase II RI report includes discussions of the purpose of the RI; the technical program; physical characteristics of the site; nature and distribution of contamination; fate and transport; and the supplemental risk assessment.

INTRODUCTION

The purpose of the Phase II RI was to 1) further characterize the nature and distribution of site-related chemicals in the Lab Sample Area, 2) further track groundwater downgradient of the Phase I limit of data, and 3) evaluate the performance of the existing overburden and bedrock pumping systems in capturing groundwater flowing beneath the Olin Plant site.

The Olin Plant is located on McKee Road, a private road in southwestern Rochester. The plant property is approximately 15.3 acres. The surrounding area is industrialized and the nearest residential areas are 1,500 to 4,000 feet from the site. The present Olin Plant operations consist of organic and inorganic chemical manufacturing facilities. The predominant products are specialty organic chemicals, including chloropyridines.

As described in the Phase I RI, chemical releases on-site have resulted from past operating procedures and waste management practices. The operational sources include leakage from underground sewers and infiltration of building washdown water. Several on-site waste management operations have also been identified and specifically investigated in Phase I as possible sources, including an acid neutralizing pond, a Lab Sample, Tank Farm, sodamide, and a building washdown area (Well B-17 Area). Site-related chemical constituents in groundwater are related to residual soil concentrations resulting from the past releases to the environment.

REMEDIAL INVESTIGATION PROGRAM

The technical approach for this Phase II RI was designed to address data gaps identified in the Phase I RI. Although some investigations were conducted on the Olin Plant property (on-site), the focus was on areas outside the plant property (off-site). These investigations included:

- surface-soil, subsurface soil, and groundwater sampling
- monitoring well installations
- borehole geophysical surveys
- surface water and groundwater sampling off-site
- hydraulic conductivity testing
- water level measurements
- surveying
- off-site laboratory sample analysis

SITE PHYSICAL CHARACTERISTICS

Results of investigations undertaken during the Phase II RI at the Olin Study Area have corroborated previous conclusions regarding the geology and hydrogeology of the site, and the direction of groundwater flow. Investigations to date have identified the following physical characteristics of the Olin Plant and surrounding area:

- study area geology consists of 10 to 20 feet of overburden, consisting of soil fill, stratified silt, sand, and gravel, overlying Lockport Dolomite bedrock;

- primary groundwater flow occurs in the saturated parts of the overburden and the uppermost 11 to 40 feet of bedrock, which is generally more fractured and weathered than the deeper bedrock;

- a deeper water-bearing zone was identified within the more competent deep rock, between 60 and 80 feet below ground surface (bgs);

- groundwater beneath the Olin Plant flows primarily to the south, southwest, and west, with a smaller component toward the southeast; and

- hydraulic conductivity estimates range from 1.9×10^{-5} to 7.7×10^{-3} centimeters per second (cm/sec) in the overburden and from 4.0×10^{-5} to 1.7×10^{-3} cm/sec in the shallow bedrock; deeper bedrock hydraulic conductivities were estimated to be approximately 10^{-6} cm/sec in the competent rock and 2.4×10^{-4} cm/sec in the water-bearing zone between 73 and 75 feet bgs.

RECOVERY WELL SYSTEM (AQUIFER TESTING)

Preliminary testing of six overburden wells, proposed for aquifer testing, found most were capable of producing low yields (0.1 gpm or less). Efforts to improve yields through well rehabilitation were unsuccessful. These low yields observed in overburden wells are likely due to either natural properties of the overburden or well/formation clogging.

The pumping test performed in overburden extraction well W-1, a higher yielding overburden well, indicated that the aquifer transmissivity is likely between 1.5 and 340 ft^2/d. Regardless of where the actual overburden transmissivities lies within this range, it appears unlikely that the existing overburden extraction well system is achieving complete capture of overburden groundwater migrating off-site.

Pumping tests performed in bedrock extraction wells BR-6A and BR-7A indicated that shallow bedrock aquifer transmissivity ranges from 250 to 350 ft^2/d, and that these wells are each capable of yields of more than 50 (BR-6A) and 20 (BR-7A) gpm. These results indicate that pumping from these wells should be capable of capturing shallow bedrock groundwater migrating off-site to the south and southwest. In addition, pumping from these wells may also be capable of effecting either partial or complete capture in the overburden by creating bedrock drawdowns that cause either an increase in vertical flow from the overburden or dewatering of the overburden. Additional analysis will be required to evaluate this possibility.

DISTRIBUTION, FATE, AND TRANSPORT OF CHEMICALS

The Phase II RI provided additional understanding of the distribution, fate, and transport of site related chemicals, as summarized below.

Lab Sample Area:

Subsurface Soil. Analytical results from subsurface soil samples at three additional locations at the Lab Sample Area show relatively low concentrations of site-related chemicals. Results from these and borings from the Phase I RI suggest this area is not a high-concentration source of site-related chemicals in groundwater.

Surface Soil. Mercury analysis results for two additional surface soil samples (SS-116 and SS-117) were one or more orders of magnitude less than the previously reported high concentration sample (SS-103), and show that the higher mercury concentration is isolated.

Groundwater. Carbon tetrachloride and chloroform were detected at higher concentrations at one location (SB-3). Lower concentrations of these and other VOCs were detected at SB-1 and SB-2. Chloropyridines and SVOCs were also detected at SB-3 and SB-2. The total inorganics concentration at SB-2 exceeded mean values for overburden wells sampled during Phase II. However, higher inorganics concentrations are likely the result of constituents adsorbing to suspended solids which in turn results in higher reported concentrations.

Comparison of the organics concentrations of chemicals detected in the soil samples to organics in groundwater at the Lab Sample Area does not suggest that the soil at the Lab Sample Area is a significant source of chemicals in groundwater. Groundwater detections are interpreted to be part of an area-wide plume. Results show there is not a specific active or soil-borne chemical source at the Lab Sample Area.

Downgradient Investigations:

Downgradient well installations and sampling, and surface water sampling provided additional information about the off-site extent of site-related chemicals in groundwater and surface water.

Overburden Groundwater. The areal distribution of site-related chemicals in overburden groundwater has been delineated. Site-related chemicals have not migrated beyond the new overburden well to the southeast (MW-114). To the west of the Olin Plant site, the overburden becomes unsaturated. Here the limit of saturation marks the western extent of chemicals in overburden groundwater. The Phase II analytical results support the findings of the Phase I RI, that concentrations of pyridines are distributed more widely than any other group of site-related chemicals. Based on the analytical results, no additional overburden groundwater investigations are recommended.

Bedrock Groundwater. Analytical results show site-related chemicals, specifically pyridines and selected VOCs, are present south and southwest of the Olin Plant site. Pyridines are distributed as far west as the Dolomite Products Quarry in the Town of Gates, where they were detected in groundwater seep samples. Pyridines have not been detected in the water that is pumped from the quarry to the Erie Barge Canal.

Neither the southern nor the western extent of site-related chemicals in bedrock groundwater has been fully delineated. Additional bedrock groundwater sampling in each of the these directions would be required to characterize the areal extent of these constituents.

Surface Water. Surface water analytical results from the Erie Barge Canal show detections of pyridines in two out of five quarterly sampling events. Each detection is at an estimated concentration (less than the detection limit of 10 μg/L). The estimated concentrations were detected at SW-1, located upstream and upgradient of the Olin Rochester Plant site, as well as at the two downstream sampling locations.

The site conceptual model was updated to incorporate results from the Phase II RI. It illustrates that chemicals leach from soil by infiltrating precipitation, and have percolated through the unsaturated overburden to the groundwater. Once in the groundwater, site-related chemicals migrate in the dissolved phase in the saturated overburden and bedrock. Some groundwater may discharge from bedrock to the Erie Barge Canal, but most appears to flow beneath the canal in fractures, based on detections of chloropyridines in groundwater seep samples collected from the Dolomite Products Quarry, located west of the canal. Oxidation/reduction processes, dissolution, degradation, volatilization, and adsorption processes act to reduce concentrations of chemicals in groundwater during migration.

BASELINE RISK ASSESSMENT

Human Health Risk. Health risks were evaluated for potential exposures to off-site media at the Olin Plant, including groundwater (and associated inhalation exposures), surface water, and groundwater seeps. Chemicals of potential concern (CPCs) were selected on a medium- and location-specific basis. Generally, the CPCs identified were VOCs (particularly chlorinated compounds), SVOCs (primarily chloropyridines), and inorganics.

The exposure scenarios quantitatively evaluated include:

- current and potential future recreational exposures to surface water in the Erie Barge Canal,
- current and potential future industrial worker exposures to groundwater seeps at the Dolomite Products Quarry, and
- potential future industrial/commercial worker exposures to groundwater;

The USEPA acceptable risk level for noncarcinogenic risk of an HI of 1 was used to characterize potential noncancer risks. The USEPA risk range of 1×10^{-4} to 1×10^{-6} for carcinogenic risk was used to characterize potential cancer risks. Potential health risks characterized as exceeding the USEPA-acceptable risk level or range are considered significant, whereas those exceeding a level of 1×10^{-6} are considered by the NYSDOH to require additional evaluation (i.e., require remedial goals to be developed). The results of the risk assessment are as follows:

- No significant human health risks were identified for potential exposures to surface water in the Erie Barge Canal or to groundwater seeps in the Dolomite Products Quarry under current or potential future land use conditions; cancer risks did not exceed an excess lifetime cancer risk of 1 x 10^{-6}, and non-cancer risks did not exceed a hazard index of 1.

- Evaluation of potential future worker exposure to off-site groundwater used as industrial process water identified cancer risks above 1 x 10^{-6} but below 1 x 10^{-4} (i.e., within the USEPA acceptable cancer risk range) for mean groundwater concentrations. For maximum concentrations, however, carcinogenic risks were calculated to be 9 x 10^{-4}, above the upper end of the USEPA acceptable cancer risk range. Calculated non-cancer risks were elevated, but the majority of that risk appears to be attributable to iron and zinc associated with turbidity in groundwater samples. In addition, the iron and zinc concentrations detected in off-site wells were higher than any detections on-site. Therefore, risks are unlikely to be related to Olin operations. Because risks for potential future exposures to off-site groundwater used as industrial process water exceeded an excess lifetime cancer risk of 1 x 10^{-6} and a non-cancer hazard index of 1, remedial goals will be calculated for the Feasibility Study.

- Modelling a hypothetical future release of VOCs from groundwater used as process water to industrial facility air did not result in any exceedances of workplace air standards.

- Groundwater concentrations exceeded MCLs and New York Standards for several CPCs. No domestic use of the groundwater is anticipated. For aesthetic reasons, groundwater in the vicinity of the Olin Study Area is not used as a drinking water source. Naturally-occurring sulfide and explosive gases preclude use of bedrock groundwater for drinking water.

Ecological Risk. A supplemental Ecological Risk Assessment (ERA) was conducted to address certain information gaps identified during the Phase I RI, in particular, the potential ecological risks associated with off-site surface water exposures in the vicinity of the Olin Plant. This assessment focused on aquatic receptor and semi-aquatic wildlife exposures to selected pyridine compounds in the Erie Barge Canal. Measured, rather than modeled, surface water analytical data were used to assess the likelihood of adverse impacts to ecological receptor populations that exist in this habitat. Aquatic toxicity benchmarks were developed for all surface water analytes and were compared to the detected estimated surface water concentrations. Food chain-related exposures by semi-aquatic receptors were evaluated using bioconcentration factors to estimate fish tissue concentrations. Finally, potential risk associated with on-site surface soil exposure was reexamined based on a consideration of regional background conditions. The conclusions are listed below.

- Estimated concentrations of the three surface water analytes detected in the Erie Barge Canal were lower than all toxicity benchmarks for aquatic receptors. Consequently, no adverse impacts to these receptors would be anticipated.

- Due to the low-magnitude, low frequency detections of estimated concentrations of pyridines, and the low uptake potential of the surface water analytes, bioconcentration hazards to semi-aquatic wildlife are considered insignificant.

- Based on concentrations of pyridines detected in Phase II wells adjacent to the Erie Barge Canal, no adverse effects to ecological receptors were identified in the ERA should undiluted groundwater discharge into the canal.

- The on-site surface soil constituents chromium and zinc exceed regional background concentration ranges and available toxicological benchmarks. However, habitat constraints most likely limit plant and soil invertebrate populations at the Olin Plant site. The site cover is sparsely vegetated and provides poor habitat for these populations.

RECOMMENDATIONS FOR FUTURE WORK

Based on the information collected during the Phase I and II RIs and previous investigations, the following tasks are planned as future work:

- Continue to sample and analyze surface water from the Erie Barge Canal for pyridines, and continue assessment of impacts to the Erie Barge Canal.

- Periodically monitor groundwater seeping into the quarry and surface water discharged from the quarry into the Erie Barge Canal, for the presence of pyridines at the Dolomite Products Quarry.

- Monitor groundwater for pyridines from selected monitoring and/or industrial wells located west of the Erie Barge Canal and south of the Ness site to further assess the off-site distribution of pyridines in bedrock groundwater.

- Based on off-site results from the above mentioned off-site wells, consider the installation of new bedrock monitoring wells to further define the distribution of chemicals in groundwater.

- Further evaluate performance of the on-site groundwater recovery system and consider options for adjusting pumping rates to optimize capture.

1.0 INTRODUCTION

This report presents results of a Phase II Remedial Investigation (RI) conducted at the Olin Corporation Chemicals Division (Olin) manufacturing plant in Rochester, New York, between August and December 1995, and selected subsequent surface water and seep sampling. Figure 1-1 shows the location of the Olin plant site. Also presented herein are interpretations and recommendations based on these results and previous investigations at the plant. This report is issued as an addendum to the Final Phase I RI report (ABB-ES, 1995). The Olin RI was performed under a consent agreement between the New York Sate Department of Environmental Conservation (NYSDEC) and Olin (Index No. B8-0343-90-08). The Phase II RI was conducted in accordance with a Work Plan approved by NYSDEC on July 13, 1995.

1.1 REPORT ORGANIZATION

This report is organized into six sections. Section 1 discusses the purpose and scope of the Phase II Investigation, and briefly presents the organization of the report. Section 2 presents the technical approach and data findings of the field program. Section 3 discusses and evaluates the findings presented in section 2, incorporating the new information into the revised site conceptual model. Section 4 updates and revises the Phase I Baseline Risk Assessment based on the new data and new exposure scenarios from the Phase II Investigation. Section 5 provides a summary and conclusions of the Phase II RI, and outlines Olin's approach to further characterization and remediation of the Rochester site.

1.2 PURPOSE AND SCOPE OF THE PHASE II INVESTIGATION

Summary of Phase I RI results: The Final Phase I RI report identified site-related chemicals in subsurface soil and groundwater beneath the site that were determined to have been released during historical plant operations. No new source areas were identified during the investigation and, in general, the limits of site-related chemicals in on-site soil were identified.

Pyridines were the most frequently-detected organic chemicals in both overburden and bedrock groundwater, and the distribution of pyridines is believed to represent the greatest extent of site-related chemicals in groundwater. Total pyridine concentrations were lower in deep bedrock than in the adjacent shallow bedrock wells. In overburden groundwater, total pyridine concentrations were delineated to 10 µg/L in all directions except the southeast, where they were delineated to 4,600 µg/L. In shallow bedrock groundwater, the distribution of total pyridine concentrations was delineated in all directions except south and southwest of the Olin Plant, where concentrations up to 3,000 and 23,000 µg/L respectively, were detected at the limit of explorations. Pyridines and VOCs were detected in the single deep bedrock well installed during the Phase I RI. The lateral distribution of site-related chemicals in the deep bedrock groundwater was not delineated in the Phase I RI.

A conceptual model was developed as part of the Phase I RI which illustrates that chemicals leach from soil by infiltrating precipitation, or formerly percolated as a separate phase liquid through the unsaturated overburden to the groundwater. Once in the groundwater, chemicals migrate in the dissolved phase in the saturated overburden and shallow bedrock. Groundwater may move vertically into deeper bedrock, discharge from the bedrock to the Erie Barge Canal, or flow beneath the canal in fractures. Oxidation/reduction processes, dissolution, degradation, volatilization, and adsorption processes act to reduce concentrations of chemicals in groundwater during migration.

Based on the information collected during the Phase I RI and previous investigations, additional investigations were recommended at the Olin Plant site and in surrounding areas. Subsequent to completion of the Phase I RI, information became available indicating that the Lab Sample and Off-Specification Material Disposal Area (Lab Sample Area), one of the five identified on-site source areas, may have been more extensive than initially estimated. Therefore, the area of investigation at the Lab Sample Area was broadened to complete its characterization. The Phase II RI was developed to further characterize this area of the Plant site, to further track groundwater downgradient of the existing data, and to evaluate the performance of the existing overburden and bedrock pumping systems in capturing groundwater flowing beneath the Olin Plant site.

Major Phase II work tasks included:

1. Additional soil sampling in the vicinity of the Lab Sample Area to further characterize potential impact at locations where laboratory chemicals may have been released to the ground. Although Phase I did <u>not</u> identify a significant area of site-related chemicals in soil associated with the disposal of waste laboratory samples, anecdotal information collected subsequent to Phase I more precisely defined an expanded area of the potential sample disposal sites. This area was investigated during Phase II.

2. Downgradient investigations to further characterize bedrock groundwater flow and assess the degree of impact to groundwater and surface water to the south and west of the site. These investigations included: installation and sampling of monitoring wells and related activities (i.e. borehole geophysics, hydraulic conductivity testing, water level measurement, etc.), Erie Barge Canal surface water sampling and analysis, and quarry seep sampling and analysis at the Dolomite Products Quarry approximately 3/4 mile downgradient from the Olin Plant. The quarry seep is a potential discharge point for area groundwater.

3. Further characterization of overburden and bedrock groundwater flow beneath the site through pumping tests conducted in site extraction wells. Data from the pumping tests were used to derive hydraulic properties of the overburden and bedrock flow systems. The results would be used to determine the effectiveness of the existing site groundwater collection system and to assist in planning for its long term operation and/or enhancement.

4. Public health and ecological risk assessments were expanded to characterize potential risks related to surface water exposures, based upon the additional Phase II data.

2.0 PHASE II REMEDIAL INVESTIGATIONS

2.1 LAB SAMPLE AREA

The Lab Sample Area was a pit north of the laboratory where laboratory waste quality control samples were reportedly buried from the 1950's until 1970 (Olin, 1990). These wastes were exhumed during subsequent construction activities for the boiler house and disposed of at a commercial landfill (Olin, 1990). Phase I soil sampling found no evidence of remaining samples or indications that a significant soil source area exists in this part of the site. After Phase I, interviews with Olin facility personnel identified three potential additional locations in the vicinity where laboratory waste quality control sample disposal might have taken place. Three soil borings were completed during the Phase II investigation, one at each of these locations. Figure 2-1 shows the location of each boring.

Phase II investigations also included collection and analysis of two surface soil samples within the previously investigated Lab Sample Area to further characterize the distribution of mercury in surface soil (See Figure 2-1). These samples were collected near SS-103, where the maximum concentration (214 mg/kg) of mercury was detected during the Phase I investigation.

Section 2.1.1 describes the scope of work completed at the Lab Sample Disposal Area. Analytical findings are presented in Section 2.1.2.

2.1.1 Scope of Lab Sample Area Work

2.1.1.1 Soil Borings and Groundwater Sampling. Three soil borings were completed on August 17, 1995. The borings, (SB-1, SB-2, SB-3), were completed using a Geoprobe drilling technique, which is a direct-push hydraulic drilling technique. Both soil and groundwater samples were collected from each boring.

Soil. Soils from approximately 2 to 6 feet below ground surface (bgs) were collected using 4-foot long disposable samplers. Sample depths were initiated at 2' because of the presence of asphalt and coarse fill material. Sampling was extended only to the water table. This was done to avoid sampling in saturated soil that may be affected by chemicals in groundwater migrating through this area from other sources. The water table was measured at approximately 5 to 6 feet bgs in each borehole. Field photoionization detector (PID) and flame ionization detector (FID) measurements for each soil core were all less than 2 ppm. Samples from the following depths were collected and composited for laboratory analysis:

Boring	Sample Depth
SB-1	2'-6'
SB-2	2'-5'
SB-3	2'-6'

Laboratory analyses included Target Compound List (TCL) Volatile Organic Compounds (VOCs) (USEPA Method 8240/8260), TCL Semivolatile Organic Compounds (SVOCs) and selected pyridines (USEPA Method 8270), and Target Analyte List (TAL) inorganics (USEPA Method 6010/7000 series) for each soil sample.

Groundwater. Upon completion of soil sampling, each soil boring was advanced approximately 5 feet into the saturated zone so that groundwater samples could be collected. After reaching the desired completion depth, a temporary 3/4-inch inside diameter (ID) polyvinyl chloride (PVC) screen and riser pipe was placed in each boring. Groundwater samples were later collected from each temporary well using a small diameter bailer. Groundwater samples from these borings were to be analyzed for TCL VOCs, TCL SVOCs and selected pyridines, and TAL inorganics (6010/7000s). The full suite of analyses was run on sample SB-2. Because of very slow recharge rates, however, sample volume from SB-1 was sufficient only for VOC analysis, and sample volume from SB-3 was sufficient for VOC and SVOC analyses.

2.1.1.2 Surface Soil Sampling. Surface soil samples SS-116 and SS-117 were collected 10 feet and 20 feet east of SS-103, respectively (see Figure 2-1). Sample locations were selected to the east because buildings or pavement surface cover are present immediately north, south, and west of SS-103. Soil samples were collected beneath a 2 to 3 inch layer of crushed stone that covers this unpaved area. Each sample was analyzed for mercury (USEPA Method 7471).

2.1.2 Analytical Findings

The following subsections present the data quality evaluation and results for samples collected from the Lab Sample Area, including the surface soil samples analyzed for mercury. The findings are presented for both soil and groundwater. The groundwater results are also incorporated into the groundwater chemical distribution discussion in Section 3.0.

2.1.2.1 Data Quality Evaluation. Chemical analysis was performed in accordance with data quality objectives and procedures described in the Phase I Work Plan and Quality Assurance Project Plan (QAPP), as modified in the Phase II Work Plan and approved by NYSDEC. Chemical data generated during the Phase II RI from the Lab Sample Area were evaluated to determine if multiple analyses (i.e., dilutions and reanalyses) were conducted on any of the samples. If a dilution was performed because a compound concentration exceeded the calibration, then the result from the diluted analysis replaced the original result. Chemical data were then tabulated by parameter (e.g., VOCs, SVOCs) and are provided in Appendix B.

During reporting of organic analytical results, the laboratory flagged all results quantified above the laboratory method detection limit, but below the reporting limit as estimated (J). This qualifier was retained as results were tabulated. Interpretation of parameter-specific comments provided by the analytical laboratory are summarized below.

VOCs. Ortho-xylene and meta and para-xylene eluted separately during sample analysis; however, they were reported by the laboratory as total xylenes. The concentration was calculated by adding the areas of ortho-xylene and meta and para-xylene and using the response factor from ortho-xylene to calculate the amount. No data quality issues were noted for VOC analysis.

SVOCs/Pyridines. No data quality issues were noted for SVOC analysis. The pyridine analysis did however, indicate that the matrix spike blank (MSB) and its duplicate (MSBD) and the matrix spike/matrix spike duplicate (MS/MSD) analyses exhibit low recovery for one or more of the selected pyridine spiking compounds. The contract laboratory default control limits for the pyridines were set at 75 to 100 percent recovery. However, because these compounds are highly polar and soluble in water, the contract laboratory was requested to perform a method study in March 1996 to generate accuracy and precision control limits for the Olin suite of selected pyridines.

QC limits for these compounds were established using Olin's sample-specific MS/MSD data generated during 1995. Recovery ranges were calculated using the mean +/- three standard deviations, omitting data points falling below one half of the mean or above twice the mean. The compound-specific limits generated by the laboratory are listed below, and are believed to more accurately represent the recovery efficiency for these compounds in water.

Compound	Relative Percent Difference	Percent Recovery
2-chloropyridine	30	10 - 128
3-chloropyridine	30	10 - 103
2,6-dichloropyridine	30	10 - 120
p-fluoroaniline	30	10 - 95

NYSDEC Analytical Services Protocol (ASP) semivolatile compound matrix spike recoveries typically have wide control limits (e.g., 9 - 103 percent, 12 - 110 percent, 27 - 123 percent), and it is expected that the chloropyridines and p-fluoroaniline would have similar spiking recoveries. Furthermore, there were small relative percent differences between the recoveries of these compounds for the MSB/MSBD and MS/MSD samples, indicating the laboratory was able to reproduce compound recoveries.

Inorganics. No data quality issues were noted for inorganic (including mercury) analyses.

2.1.2.2 Analytical Results. Soil. Soil analytical results from each boring show concentrations of SVOCs and 2,6-Dichloropyridine, with the highest total SVOC concentration detected in SB-2. Trace concentrations of VOCs, including carbon tetrachloride, toluene, and tetrachloroethene (PCE) were measured at less than 5 µg/kg. Figure 2-2 shows the VOCs and SVOCs detected in the soil boring samples. For the inorganics results, SB-2 soil showed the highest concentrations. A summary of the non-nutrient inorganic soil results is presented in Table 2-1

Groundwater. Figure 2-3 is a detailed summary of organics detected in groundwater at the Lab Sample Area. Selected VOC and SVOC results from the groundwater samples are incorporated into the discussion and presentation of groundwater chemical distribution in Section 3.0. Groundwater analytical results (Figure 2-3) show maximum concentrations of carbon tetrachloride (260,000 µg/L) and chloroform (80,000 µg/L) at SB-3. Lower concentrations of these and other VOCs were detected at SB-1 and SB-2. Chloropyridines and SVOCs were also detected at SB-3 and SB-2.

Inorganics results from SB-2 show concentrations of analytes that exceed mean values for overburden wells sampled during Phase II. The high concentration of analytes is likely the result of inorganics adsorbing to the suspended solids, resulting in higher reported results. Among the inorganics detected at higher concentrations than the mean for on-site overburden wells during the Phase I RI are:

Aluminum
Barium
Beryllium
Calcium
Chromium
Cobalt
Copper
Iron
Lead
Magnesium
Manganese
Nickel
Potassium
Vanadium

The complete analytical results for both soil and groundwater samples from the Lab Sample Area are presented in Appendix B.

Organics and inorganics were detected in groundwater beneath the Lab Sample Area. However, comparison of concentrations of chemicals detected in the soil samples to groundwater concentrations at the Lab Sample Area did not suggest that the soil at the Lab Sample Area was a significant source of site-related chemical in groundwater. Therefore, it is concluded that the groundwater detections are part of an area-wide plume and that there is no specific active or soil-borne chemical source at the Lab Sample Area.

<u>Surface Soil.</u> Mercury concentrations measure 0.15 mg/kg for SS-116 and 7.2 mg/kg for SS-117. These concentrations are comparatively lower than the 214 mg/kg measured at nearby SS-103, which was collected during the Phase I RI. The concentratoin for SS-117, however, is above the background level of <2 μg/kg.

<u>Background Comparison</u>. A comparison of the inorganics concentrations measured in surface and subsurface soil samples to background levels was completed for all surface soil samples (SS-101 through SS-117) and the three soil boring samples (SB-1, SB-2, and SB-3), collected during Phase I and Phase II RIs. This comparison is shown on Table 2-2. Background values for inorganics were taken from New York State Department of Environmental Conservation (NYSDEC) Technical and Administrative Guidance Memorandum (TAGM) on Determination of Soil Cleanup Objectives and Cleanup Levels (NYSDEC, 1994). Three inorganic analytes were detected above background levels at a majority of locations: magnesium, mercury, and zinc. Eight other inorganics were detected above background levels at one or more locations: arsenic, cadmium, calcium, chromium, copper, iron, lead, and nickel. Nine inorganics were either not detected or measured below background levels. No background concentrations were available for four other inorganics. These analytes were compared to the USEPA Region III risk-based concentration for industrial soil (USEPA, 1994) in lieu of a background value. The range of concentrations for these four analytes is below the risk-based concentration. The location of the maximum number of background exceedances is SS-104, within the Tank Farm. Figure 2-4 shows the location of inorganics background exceedances. Additional

soil background comparisons are made for the ecological risk assessment at selected habitat locations (See Section 4.0).

2.2 DOWNGRADIENT INVESTIGATIONS

Phase II tasks were completed downgradient of the Olin Plant site (off-site) to provide further understanding of the nature and distribution of site-related chemicals. off-site. An additional objective of the tasks was to characterize the bedrock flow system that is the primary pathway for transport of dissolved phase compounds away from the site. Figure 2-4 shows the downgradient monitoring wells, and surface water and seep sample locations. Downgradient investigations included the installation and sampling of paired monitoring wells south and west of the site, and associated tasks such as borehole geophysics, hydraulic conductivity testing, and water level measurements. Groundwater seeps within the Dolomite Products quarry (approximately 3/4 mile southwest of the Olin Plant) and surface water in the Erie Barge Canal (approximately 1/4 mile west and southwest of the Olin Plant) were sampled to assess water quality at areas of potential groundwater discharge to surface receptors.

Section 2.2.1 describes the scope of downgradient work completed during Phase II investigative tasks. Analytical and hydrogeologic findings are presented in Section 2.2.2.

2.2.1 Scope of Downgradient Work

Phase II investigative tasks downgradient of the site were centered on (1) activities associated with the installation and sampling of monitoring wells, and (2) surface water and seep sampling. These tasks were detailed in the Phase II Remedial Investigation Final Work Plan (ABB-ES, 1995b).

2.2.1.1 Monitoring Well Installation and Sampling. A total of eight monitoring wells were installed during the Phase II RI. Paired shallow and deep bedrock wells were constructed at three locations to the west and southwest of the Olin Plant site, along the east bank of the Erie Barge Canal (see Figure 2-5). Paired overburden and shallow bedrock wells were also installed at one location southeast of Olin Plant site. Well installation and development was accomplished between August 21, 1995 and October 13, 1995.

The wells were installed in accordance with procedures described in the Phase I Work Plan and QAPP, as modified in the Phase II Work Plan and approved by NYSDEC. The shallow bedrock wells (BR-111, BR-112A, BR-113, BR-114) were completed as open

3.8-inch diameter coreholes beneath 6-inch steel casing that is grouted into the top of the bedrock. The deep bedrock wells (BR-111D, BR-112D, BR-113D) were completed as 2.0-inch ID PVC wells constructed in 3.8-inch diameter coreholes. To prevent cross contamination or communication between shallow and deeper bedrock groundwater systems, a 4.0-inch diameter casing was seated through the shallow bedrock to a depth of 5 feet below the adjacent shallow bedrock well. The overburden well (MW-114) is a 2-inch PVC well constructed from the top of bedrock to within 6 feet of the ground surface.

Geophysical logging was conducted to aid in the selection of the well screen interval for the three deep bedrock monitoring wells. The objective was to identify hydraulically-active fractured intervals in the bedrock. Logging suites conducted were: (1) fluid temperature and resistivity, (2) caliper, (3) single point resistance and spontaneous potential, and (4) natural gamma. The logging results were reviewed in conjunction with core descriptions and notes on water loss during drilling to determine potential water-productive zones. Based on this evaluation, the deep wells were constructed at similar elevations, with open intervals approximately 50 to 75 feet bgs. Borehole geophysical results are included in Appendix A. Rock core and monitoring well construction logs are included in Appendix A. The construction details for the newly installed monitoring wells are as follows:

Well No.	Total Depth (feet bgs)	Open Interval (feet bgs)	Elevation - Ground (MSL)	Elevation - Open Interval (MSL)
Overburden				
MW-114	15.9	4.0 - 15.9	539.7	535.7 - 523.8
Shallow Bedrock Wells				
BR-111	45.0	14.5 - 45.0	537.2	522.7 - 492.2
BR-112A	40.0	12.5 - 40.0	545.0	532.5 - 505.0
BR-113	45.0	11.0 - 45.0	540.4	529.4 - 495.4
BR-114	39.6	19.5 - 39.6	539.8	520.3 - 500.2
Deep Bedrock Wells				
BR-111D	100.0	52.0 - 74.6	537.8	485.8 - 463.2
BR-112D	100.0	47.0 - 70.0	545.7	498.7 - 475.7
BR-113D	100.0	54.0 - 76.6	540.5	486.5 - 463.9

During a check of well BR-112 and subsequent to construction of BR-112D, grout was discovered within the open corehole of BR-112. Bentonite, introduced to seat the 4-inch casing into bedrock at BR-112D, had migrated through a significant fracture to BR-112. Because the hydraulic connection to the surrounding aquifer had been altered, Olin elected to abandon the well and drill replacement well BR-112A. BR-112 was abandoned by grouting in place and cutting off the protective casing below ground level. BR-112A was drilled approximately 30 feet to the southeast of BR-112D, or approximately 40 feet from BR-112, to avoid the grout that had apparently migrated from BR-112D.

Bedrock wells were developed by pumping to remove fine-grained materials and mitigate the effects of well construction, including the introduction of water during drilling. The wells were developed using submersible electric pumps discharging to a tanker truck. Pumped water was discharged to the Olin on-site treatment system. Temperature, pH, specific conductance, and turbidity were measured during development of each well at 15-minute to 1-hour intervals. Well development logs are provided in Appendix A.

Hydraulic Conductivity Testing. Hydraulic conductivity tests were performed on the seven new bedrock monitoring wells. The tests were completed using solid slugs to displace water within the wells. Both falling and rising head tests were run. Tests were not run on MW-114 because of insufficient water column height. Hydraulic head was measured using pressure transducers connected to a Hermit 2000 datalogger. Hydraulic conductivity values were estimated using the Bouwer-Rice (1976) method in AQTESOLV (Geraghty & Miller Modeling Group, 1989). Hydraulic conductivity test data plots and calculations are presented in Appendix A and discussed in subsection 2.2.2.1, Hydraulic Conductivity. Additional aquifer testing conducted on two on-site bedrock wells and six on-site overburden wells is described in Section 2.3.

Water Level Measurements. Water level measurements were collected from all accessible site and offsite wells and piezometers from November 6 to 8, 1995. In all, 83 wells and piezometers were measured. The objective was to obtain a comprehensive database of water table and piezometric head elevations in the vicinity of the site under non-pumping conditions. At the time of this measurement task the groundwater extraction system had been shut-down for 47 days and aquifer conditions are presumed to have rebounded to static levels. Water level data were used to develop piezometric flow maps for the overburden and shallow bedrock groundwater systems. The piezometric maps, are shown and discussed in Section 3.0. A tabulation of the piezometric data is provided in Appendix A.

Semiannual Groundwater Sampling Event. To support the downgradient investigations of the Phase II RI, information from the September 1995 semiannual groundwater sampling event is being incorporated into this report. Twenty-seven wells that are located both on and off the Olin Plant site are included in the current semiannual groundwater monitoring network. Each well is analyzed for VOCs (8240) and selected pyridines (8270).

Phase II RI Groundwater Sampling Event. One complete round of groundwater sampling was conducted for the Phase II RI on two separate dates (November 20, and December 7, 1995). The sampling 1995 semiannual groundwater sampling sampling was done on two dates, because some locations inadvertently were not sampled for the pyridines during the November 20 sampling event. On November 20, samples were collected for TCL VOCs, TCL SVOCs, and TAL Inorganics. Ten wells, including the eight new monitoring wells and two former bedrock industrial wells (NESS-E and NESS-W) at Ness Precision Products were sampled as part of this episode. Samples were analyzed for TCL VOCs (8240), TCL SVOCs (8270), and TAL inorganics (6010/7000s). On December 7, samples were collected from each of the ten wells and submitted for selected pyridine analysis by USEPA Method 8270.

2.2.1.2 Surface Water And Seep Sampling. Quarterly Sampling Events. Surface water sampling occurs on a quarterly basis at locations in the Erie Barge Canal. Data from the following previous sampling events at three locations (SW-1, SW-2 and SW-3) are incorporated into this report:

November 1994
March 1995
May 1995
September 1995

All samples were analyzed for selected pyridines by NYSDEC ASP91 Category B protocols.

Phase II Surface Water Sampling. Surface water samples were collected from three quarterly monitoring locations in the Erie Barge Canal (i.e., SW-1, SW-2, and SW-3) shown on Figure 2-5. The most recent Phase II samples were collected on November 20, 1995. In addition, results form surface water sampling conducted subsequent to the Phase II RI are included for purposes of making the risk assessment more comprehensive. The samples have been reported in quarterly reports to NYSDEC.

Quarry Seep and Surface Water Sampling. Seep and surface water samples were collected within the Dolomite Products Company Quarry, located 4000 feet southwest of the Olin Plant site. A total of seven water samples were collected during two events, September 6 and October 25, 1995, and analyzed for selected pyridines.

Four seep samples (QS-1 through QS-4) were collected on September 6, 1995. These samples were collected from water discharging from a prominent horizontal fracture zone on the eastern wall of the quarry. The seep samples were obtained by filling sample bottles with water flowing or cascading down the quarry wall from the horizontal fracture plane. The water from the seeps collects in a drainage pond at the bottom of the quarry and is pumped to the Erie Barge Canal as part of the de-watering system for the quarry. Samples from the quarry drainage pond (QP-1) and the outflow pipe at the Erie Barge Canal (QO-1) were collected on October 25, 1995. In addition, seep sample location QS-4 was resampled at this time to confirm pyridine detections in the September sampling.

Field Survey. A field survey was conducted to measure vertical and horizontal positions of the eight newly installed wells as well as the seep sample and fracture locations in the Dolomite Products Quarry. All horizontal positions were measured to the nearest 1 foot and were referenced to New York State Plane Coordinate System. Vertical measurements were referenced to mean sea level (MSL). Well casing elevations were measured to the nearest 0.01 feet and the fracture elevation measurements were made to the nearest 0.1 feet. Three locations on the water-bearing fracture plane in the quarry were surveyed; two along the east face and one along the south face. This triangular configuration of points was measured so that the orientation of the fracture plane could be calculated. Survey coordinates for each measurement point are presented in Appendix A.

Quarry Outfall Sampling. Following completion of the Phase II RI investigation program, Olin discovered the location of a ditch and outfall through which water from the quarry is discharged to the canal. Results of sampling from the outfall have been provided to NYSDEC as part of the qaurterly reports, and are included in the risk assessment in Section 4.0.

2.2.2 Analytical and Hydrogeologic Findings

This subsection presents the findings of the downgradient investigations. Discussions of these findings, including interpreted groundwater flow and chemical distribution plots are presented in Section 3.0.

2.2.2.1 Hydraulic Conductivity Test Results. All slug test data were analyzed using the method of Bouwer and Rice (1976), as applied by the AQTESOLV computer program (Geraghty & Miller Modeling Group, 1989). Tests conducted in shallow bedrock wells (BR-111, BR-112A, BR-113, and BR-114) resulted in calculated hydraulic conductivity values ranging from 8.0 x 10^{-2} to 1.4 x 10^{-3} centimeters per second (cm/sec). Deep bedrock wells (BR-111D, BR-112D, and BR-113D) yielded results ranging from 7.3 x 10^{-3} to 3.0 x 10^{-3} cm/sec. Table 2-2 is a summary of the hydraulic conductivity results; the test plots and raw data are provided in Appendix A.

Shallow Bedrock Wells. Values from three of the four shallow bedrock wells were in the 10^{-2} cm/sec range, which is at the high end of the range of conductivities measured for Phase I shallow bedrock wells (4.0×10^{-5} to 1.7×10^{-2} cm/sec). These values are also higher than the 10^{-4} cm/sec range reported for wells installed at Olin in the early 1980's (Olin, 1982). Test plots show that head oscillations occurred in BR-111, BR-113, and BR-114 during testing. These oscillations are likely the result of very rapid recovery to static levels after immersion or removal of the testing slug.

Deep Bedrock Wells. The range of values for the three deep bedrock wells (7.3×10^{-3} to 3.0×10^{-3} cm/sec) tested during Phase II are higher than the 10^{-4} cm/sec range measured in BR-105D during the Phase I RI. The difference may be due to variability in the type and extent of water-bearing fractures at different locations and depths. The three new deep bedrock locations are installed within 50 to 100 feet of the eastern wall of the Erie Barge Canal, and may be installed in rock that may have been disturbed by excavation of the canal.

2.2.2.2 Analytical Results. This subsection presents analytical results for downgradient groundwater and surface water samples collected during the Phase II RI field program. Presentations include data summaries and tabulations. Section 3.0 discusses the nature and distribution of site-related chemicals based on these results. Complete tabulations of the laboratory analytical results are in Appendix B.

Groundwater. Groundwater results are presented in this subsection by groups of chemicals that occur most frequently and at higher concentration within the study area. The groupings, listed below, are based on chemical detections during the groundwater investigation (Olin, 1990) and Phase I RI (ABB-ES, 1995a) and chemical similarity. In particular, the selected VOCs were selected for field laboratory analysis during the Phase I RI.

Pyridines
2,6-dichloropyridine
2-chloropyridine
3-chloropyridine
4-chloropyridine
p-fluoroaniline
pyridine

Selected VOCs
1,1,1-trichloroethane
1,1-dichloroethane
1,2-dichloroethene (total)
carbon tetrachloride
chloroform
methylene chloride
tetrachloroethene
trichloroethene

Chlorinated Ethenes

BTEX compounds

1,2-Dichloroethene (Total)	benzene
tetrachloroethene	toluene
trichloroethene	ethylbenzene
vinyl chloride	total Xylenes

These chemical groupings are also presented in Section 3.0 for showing chemical distribution and discussing possible migration pathways.

Pyridines. A total of four chloropyridine compounds and p-fluoroaniline were detected in groundwater samples. 2-chloropyridine (2-CPL), 2,6-dichloropyridine (2,6-CPL), p-fluoroaniline, and 3-chloropyridine (3-CPL) were detected more frequently than other compounds in this grouping. 4-chloropyridine was detected in only one sample out of the ten collected from the eight new offsite wells and two Ness wells, whereas pyridine was not detected in any of the these samples. Tables 2-4 and 2-5 show concentrations of 2-CPE, 2,6-DCPE, 3-CPE, and p-fluoroaniline in overburden and bedrock wells, respectively.

Selected VOCs. This grouping includes halogenated VOCs that occur at higher frequency or concentration than other VOCs on-site. Selected VOCs were detected in the new off-site wells. Tables 2-6 and 2-7 show concentrations of these constituents in overburden and bedrock groundwater, respectively.

Chlorinated Ethenes. Because 1,2-dichloroethene (1,2-DCE) (total) and vinyl chloride (VC) commonly occur as degradation products of PCE and/or trichloroethene (TCE), the chlorinated ethenes are grouped so that transport pathways can be better understood. Each of the selected chlorinated ethenes was detected in onsite and offsite groundwater. Tables 2-8 and 2-9 show concentrations of these chemicals in overburden and bedrock groundwater, respectively.

Benzene, Toluene, Ethylbenzene, and Total Xylenes (BTEX). BTEX compounds are grouped together because they are significant volatile constituents of lighter fuels. Each of the BTEX constituents was detected in on-site and off-site groundwater. Tables 2-10 and 2-11 show concentrations of these chemicals in overburden and bedrock groundwater, respectively.

Other Constituents. Other constituents detected in groundwater include inorganics and TCL SVOCs. Results for these parameter groups from the eight new offsite wells and the two Ness wells are presented here. Most of the highest inorganic concentrations were measured at the NESS-E well, where values for some analytes are up to an order of magnitude higher than at any of the other wells. Table 2-12 is a summary of the inorganics detected in groundwater. A median concentration has been calculated for this table to show representative concentrations for each inorganic analyte. TCL SVOCs were detected in the ten offsite wells with less frequency and at lower concentrations than other organics (i.e. VOCs and selected pyridines). SVOCs detected include 2-methylnaphthalene (up to 14 µg/L), 4-methylphenol (up to 8 J µg/L), naphthalene (up to 16 µg/L), and bis(2-ethylhexyl)phthalate (up to 16 µg/L).

Surface Water And Seeps. Chloropyridines were detected in surface water from the quarry pond, from all canal sampling locations, and in samples from three of four seep samples. The highest chloropyridine concentrations were measured in QS-4, a seep sample that was collected from the water bearing fracture at the Dolomite Products Quarry. Sporadic lower concentrations of chloropyridines were measured in several surface water samples from the Erie Barge Canal. These concentrations, however, were estimated ("J" qualifier) because they were measured below the practical quantitation limit of 10 µg/L. Figure 2-6 presents pyridines results for seep samples collected in 1995 and five rounds of surface water samples from November 1994 to November 1995.

2.2.2.3 Analytical Data Quality Evaluation. Chemical analyses were performed in accordance with data quality objectives and procedures described in the Phase I Work Plan and QAPP, as modified in the Phase II Work Plan and approved by NYSDEC. Chemical data generated during the Phase II RI groundwater investigation were evaluated to determine if multiple analyses (i.e., dilutions and reanalyses) were conducted on any of the samples. If a dilution was performed because a compound concentration exceeded the calibration, then the result from the diluted analysis replaced the original result. Data generated during the quarterly canal sampling events were evaluated in accordance with NYSDEC ASP91 Category B protocols. Chemical data were then tabulated by parameter (e.g., VOCs, SVOCs) and are provided in Appendix B.

During reporting of organic analytical results, the laboratory flagged all results quantified above the laboratory method detection limit, but below the reporting limit as estimated (J). This qualifier was retained as results were tabulated. Interpretation of parameter-specific comments provided by the analytical laboratory are summarized below.

VOCs. Ortho-xylene and meta- and para-xylene eluted separately during sample analysis; however, they were reported by the laboratory as total xylenes. The concentration was calculated by adding the areas of ortho-xylene and meta- and para-xylene and using the response factor from ortho-xylene to calculate the amount. Additionally, the MS/MSD analyses for samples BR-114 and NESS-W exhibited a relative percent difference between the two spike recoveries for benzene as above laboratory quality control (QC) limits. The benzene recoveries, however, were within QC limits; therefore, it is not expected that this occurrence has an adverse impact on the benzene results for samples BR-114 and NESS-W. No other data quality issues were noted for VOC analysis.

SVOCs/Pyridines. SVOC analysis indicated a few samples (BR-112D, BR-114, MW-114) exhibited surrogate standard recoveries outside QC limits. All samples, however, were re-extracted outside the allowable holding time by several days. The original results, therefore, were retained and tabulated. The sample and its field duplicate collected from NESS-E exhibited the recovery of one of its internal standards as below QC limits indicating a potential matrix interference.

The pyridine analyses indicated that the MSB and MSBD and the batch MS and MSD exhibited spike recoveries of one or more pyridine spiking compounds below laboratory control limits of 75-125 percent. Sample results were not flagged as estimated, however, because it is believed that these control limits do not accurately represent the recovery efficiency for these compounds. NYSDEC ASP semivolatile compound matrix spike recoveries typically have wide control limits (e.g., 9 - 103 percent, 12 - 110 percent, 27 - 123 percent), and it is expected that the chloropyridines and p-fluoroaniline would have similar spiking recoveries. Furthermore, there were small relative percent differences between the recoveries of these compounds for the MSB/MSBD and MS/MSD samples, indicating the laboratory was able to reproduce compound recoveries.

Inorganics. No data quality issues were noted for inorganic (including mercury) analyses.

2.3 GROUNDWATER EXTRACTION SYSTEM EVALUATION

Olin operates a groundwater extraction system to prevent offsite migration of impacted groundwater. The system Olin operates, active since 1983, consists of overburden extraction wells concentrated around the site perimeter and selectively positioned bedrock extraction wells. Wells discharge to common subsurface headers that gravity flow to inground concrete sumps. Groundwater is pumped from the sumps to the onsite pretreatment plant, treated by steam distillation and/or activated carbon, and discharged to the municipal publicly-owned treatment works (POTW). The Phase II Investigation included aquifer testing of overburden and bedrock extraction wells to determine hydraulic properties of the soil and bedrock flow systems. This information will be used to determine if any adjustment is needed for the extraction system to effect efficient long-term capture of impacted groundwater.

2.3.1 Technical Approach

Pumping tests were planned for six of the 18 overburden extraction wells and two of the five active bedrock extraction wells. The wells planned for testing included overburden wells distributed along the western and southern boundaries of the site and bedrock wells BR-6 and BR-7A, also in the southwest part of Olin's property. Because groundwater flows predominantly towards the southwest, beneath the site (as determined during Phase I), it is critical to characterize hydrogeologic conditions in this part of the site.

The entire groundwater extraction system was shut-down in mid-September 1995, allowing piezometric levels to rebound from depressed long-term pumping levels and providing static conditions for the start of the tests. The tests were planned as separate constant-rate discharge events that would stress the surrounding aquifer by pumping to produce measurable drawdown in nearby monitoring wells or piezometers. Pumping rates were to be determined based on each well's estimated yield. Also, during the shutdown Olin replaced BR-6 with a larger diameter well to accommodate a more efficient submersible pump. Therefore, the new well BR-6A, and not BR-6, was used for one of the pumping tests.

The following sections discuss the actual overburden and bedrock tasks performed.

2.3.1.1 Overburden Testing. Pumping tests were planned at overburden wells W-1, W-2, W-4, W-5, S-1, and S-3. Initial tasks included the installation of small-diameter piezometers to provide observation points close to four of the wells and preliminary yield testing to determine the methodology for longer-term pumping tests. The yield testing unexpectedly revealed that most of the wells could sustain only very low pumping rates. After well rehabilitation failed to increase discharge, a longer-term pumping test was attempted only at W-1, the well with the highest yield. These efforts are described below:

Piezometer Installation. Small-diameter piezometers were installed near overburden wells S-3, W-2, W-4, and W-5. Marcor of New York, Inc. installed the piezometers (PZ-201, PZ-202, PZ-203, PZ-204) on August 17, 1995 using Geoprobe drilling technique. One-inch ID PVC piezometers installed at W-2, W-4, and W-5 and a 1.25-inch galvanized steel piezometer placed at S-3. The piezometers were placed within 10 feet of the extraction wells and screened across the water table at bottom depths ranging from 10.5 to 14 feet bgs. They are temporary installations with no surface seals or protective casings and will be removed after this investigation.

Piezometer	Nearest Well	Distance (ft)	Open Interval (ft bgs)
PZ-201	S-3	9	6.5 - 11
PZ-202	W-5	8	6.5 - 10.5
PZ-203	W-4	8	10 - 14
PZ-204	W-2	8	9 - 13

Yield Testing. Preliminary pumping tests were performed at each of the six overburden wells to establish well yields and provide information needed to design the longer-term pumping tests. The tests, performed between September 28 and October 2, 1995, were completed using a surface peristaltic pump fitted with 1/2-inch tubing. The intake end of the tubing was inserted to the bottom of each well. With the exception of W-1, water stored within each well was quickly evacuated and, as pumping continued, water was removed at the rate it infiltrated into each well. Well yield was established by periodically measuring discharge rate into calibrated containers. Each test continued for 30 to 60 minutes until successive measurements indicated relatively consistent flow rates. At W-1, higher rates of inflow allowed pumping to continue at a constant rate until drawdown was observed in the closest piezometer.

These initial yield tests produced the following results:

Well	Discharge Rate	Piezometer Response (distance)
W-1	0.75 gpm	0.1 ft at B-1 (8 ft)
W-2	0.1	none at PZ-204 (8 ft)
W-4	0.03	none at PZ-203 (8 ft)
W-5	0.12	none at PZ-202 (8 ft)
S-1	0.003	none at B-14 (17 ft)
S-3	0.03	none at PZ-201 (9 ft)

With the exception of W-1, the yields of the wells were too low to produce a drawdown response in the nearest observation wells during these short-duration pumping tests. Since some encrustation and oily sludge was observed on most well riser pipes, Olin tried to improve the yields by aggressively cleaning the wells. Parratt-Wolff Inc. of Syracuse, New York performed the well rehabilitation on October 23-25, 1995.

The rehabilitation attempted to restore higher yields by a combination of physical and chemical methods including: addition and surging of trisodium phosphate, surging and bailing, addition of chlorine, soaking for at least two hours, and final surging and pumping. The wells were pumped immediately after cleaning to determine yield. Flow rates did not increase significantly as a result of the cleaning. Olin believes that the wells are not as productive as earlier in their pumping history. The soil matrix around the wellscreens may have decreased transmissivity due to biological or mineral deposits that have accumulated since the system began pumping. This type of fouling is generally impossible to remove except within a few inches from the wellscreen. No records exist of individual well yields when installed. Plant personnel report that combined production of the 18 overburden extraction wells has historically been about 4 gallons per minute (gpm).

Longer term pumping tests would be impractical at the current low flow rates due to the time needed to produce perceptible drawdown in observation wells and the difficulty in accounting for natural water table fluctuations when compared to those small drawdowns. Therefore, a constant-rate pumping test was attempted only at W-1, where higher flow rates would produce a larger drawdown during a short pumping interval.

W-1 Pumping Test. W-1 is located at the northwestern corner of the site and is screened from 6 to 16 ft bgs. This 2-inch PVC well is housed within a concrete vault that extends from four feet below to one foot above grade. Static water level and well bottom were approximately 10 feet (ft) and 17 ft below the top surface of the vault, respectively. B-1, a 2-inch PVC observation piezometer, is located 8 feet north of W-1 and screened from 5.5 to 15.5 ft bgs.

The W-1 pumping test was accomplished using a single stage submersible purge pump, fitted with 1/2-inch flexible tubing and powered by a 12-volt car battery. The pump was placed one foot above well bottom. Pumped water was discharged to a 55-gallon drum staged adjacent to the vault. A surface diaphragm pump pumped water from the drum to the extraction system's subsurface header pipe. Flow rate was measured by timing flow into a calibrated 5-gallon bucket and was controlled by varying the elevation of the outflow into the drum. Pressure transducers were installed in W-1 and B-1. A barometric probe was placed within the vault to record barometric pressure changes.

Background trend of water table elevation head was determined by collecting measurements of water levels in W-1 and B-1 during the 24 hours prior to pumping. Weather during this period was dry and measurements collected on November 6th and 7th differed by less than 0.02 feet (within instrument measurement error) indicating that there was no significant trend at the time of the test start.

The W-1 pumping test began at 1300 hours on November 7, 1995, and ended at 0700 hours on November 8th (total duration 1080 minutes). The pump stopped at about 1080 minutes, most likely due to clogging by sediment. Discharge was steady at 1.1 gpm from pump start to 450 minutes, at 0.92 gpm from 450 to 500 minutes, and at 1.1 gpm from 500 to 1080 minutes. The rate changes were caused by switchover between different batteries and could not be wholly compensated for by altering the outflow head elevation.

Light rain began several hours prior to the start of pumping and this caused ponding in a drainage depression within 20 feet (north) of W-1, a condition that was present throughout the test. Runoff also created puddles in low areas along the unpaved shoulder of McKee Road southwest of W-1. The rain changed to snow during the morning of November 8th and temperatures stayed at or below the freezing level during the recovery phase. After the pump failure, electronic data collection continued until 0700 hours on November 9th. At that time, the test was stopped, equipment removed and decontaminated, and the datalogger downloaded to a field computer. Results of the W-1 pumping test are presented and discussed in Section 2.3.2.

2.3.1.2 Bedrock Aquifer Testing. Constant-rate pumping tests were performed on BR-6A and BR-7A. The objective of each test was to stress the bedrock aquifer by pumping to effect measurable drawdown in a number of existing monitoring wells. Each test included set-up, pumping, and recovery periods. Drawdown data, collected from the pumping wells and selected observation wells, were used to create graphs of drawdown response versus time. These graphs were analyzed by published methods to compute aquifer properties such as transmissivity (T) and storage coefficient (S). Data collection included continuous electronic measurement of background barometric pressure and water levels in selected wells, and periodic hand measurement of water levels in surrounding observation wells or piezometers.

The following sections discuss the scope of work performed for each bedrock test.

2.3.1.2.1 BR-7A Pumping Test

Pre-Test Measurements and Set-up. BR-7A is a 6-inch ID well, cased through the overburden to a depth of 16 ft and open below the casing as a cored bedrock hole to about 51 ft bgs. The surface casing extends three feet above ground. All water level measurements are referenced to the rim of the surface casing. Static water level was approximately 26 ft from this rim. BR-7A has been pumped extensively as part of the active bedrock groundwater extraction system. The dedicated submersible pump and associated downhole equipment was removed in August 1995 for maintenance prior to the overall pumping system shut-down in mid-September.

The BR-7A pumping test was accomplished using a Grundfos® Redi-Flo-2 two-inch diameter submersible pump fitted with 3/4-inch ID plastic tubing. The pump was hung 45 feet below the well rim, approximately 19 feet below static water level. Pumped water was discharged to a 55-gallon drum staged adjacent to the wellhead. A surface diaphragm pump discharged water from the drum to the subsurface extraction system header pipe. A gasoline generator provided the power for the pump. Discharge rate was controlled by varying the output amperage of the pump controller unit.

The equipment was assembled at the site on October 15, 1995 and a brief test (25 minutes) was conducted to assure that the pump and discharge equipment were fully operational. Pressure transducers were installed in BR-7A, PZ-104, and PZ-105 and connected to a Hermit 2000 datalogger. They were programmed to record drawdown at logarithmic time intervals to a maximum interval of 60 minutes. A barometric probe, hung at ground level within the test well casing, provided a continuous record of barometric pressure changes.

Background trend of bedrock piezometric head was determined by collecting daily measurements of water levels for four days preceding the test. Measurements were collected using a Slope water level meter in wells PZ-103, PZ-104, PZ-105, PZ-107, and BR-7A.

Constant-Rate Testing. The pumping phase of the BR-7A constant-rate test began at 0800 hours on October 16, 1995, and ended at 1530 hours on October 17th (total duration of 1890 minutes). Pumping rate varied between 5.2 and 6.2 gpm during the first several hours of pumping and then settled at 6.1 gpm for the remainder of the test (about 300 to 1890 minutes). Discharge rate measurements were collected periodically throughout the test by timing flow into a calibrated 5-gallon bucket. Attempts to adjust flow rate during the early part of the test were only partially successful and were not attempted after about 230 minutes. The early pumping rate variability was apparently caused by fouling of the pump intake screen by biological slime (observed upon removal of the pump at the end of the test).

During aquifer recovery, drawdown measurements were collected until 0800 hours on October 19, 1995, for a total recorded duration of 4320 minutes. Following recovery, data from the test were downloaded to a field computer and equipment was removed from the wells and decontaminated. Water generated during the decontamination process was discharged to the groundwater extraction system.

2.3.1.2.2 BR-6A Pumping Test.

Pre-Test Measurements and Set-up. BR-6A is a new well, installed as a replacement for BR-6 on November 13, 1995. It is cased with 6-inch iron pipe through fill and native overburden to a depth of 19 feet below current grade and open below the casing as a 6-inch diameter corehole to 57.5 feet. Water level measurements were taken from the top of the casing, approximately three feet above grade. The elevation of this surface casing will likely be altered upon pump installation and hookup of the well to the extraction system. Static water level prior to pumping was 18 ft below top of casing.

The BR-6A pumping test was accomplished using a Franklin 4-inch diameter submersible electric pump fitted with 250 feet of one-inch ID rubber water hose. The pump was hung 45 feet below the top of casing, approximately 27 feet below static water level. Pumped water was discharged directly into the Area C sump, a subsurface concrete vault 200 feet east of BR-6A that is the collection point for groundwater extracted from wells in the southern part of the site.

The pump was powered by a gasoline generator positioned 20 feet from the well. A gate valve was placed at the outflow end of the discharge line to control flow rate. As it transpired, the test was completed with the valve open and no adjustments to flow rate were necessary. The equipment was assembled at the site on November 16, 1995 and a brief test (5 minutes) was conducted to assure that the pump and discharge equipment were fully operational. Discharge rate measurements were collected throughout the test by timing flow to a measured 5-gallon mark on a plastic bucket.

Pressure transducers were installed in BR-6A, PZ-105, and PZ-107, and these, along with a barometric probe were attached to a Hermit 2000 datalogger. Transducers were also placed in PZ-106 and BR-2A and connected to a Hermit 1000c datalogger. The dataloggers were programmed to record drawdown at logarithmic time intervals with a maximum interval of 60 minutes.

Water level measurements were collected on November 15th and 16th to determine background trend prior to test start. A Slope water level meter was used to take hand measurements of water levels during pre-pumping, pumping, and recovery periods to augment the electronic data record.

Constant-Rate Testing. The pumping phase of the BR-6A constant-rate test began at 1537 hours on November 16, 1995 and ended at 0942 hours on November 17th (total duration of 1085 minutes). Pumping rate was consistent at 20.1 gpm throughout the test.

Weather was below freezing with occasional snow flurries throughout the test. A snowstorm on November 15th dropped approximately 14 inches of snow across the site.

During the pumping phase, review of data from BR-2A showed that the transducer in this well was not functioning properly and hand measurements were collected to monitor drawdown. The dataloggers were stepped just prior to pump shut-off to resume short-interval logarithmic measurements and automated recovery data were collected until 0924 hours on November 18, 1995. At that time data from the two dataloggers were downloaded into a field computer and equipment removed from the wells and decontaminated.

2.3.2 Findings

Data from the three pumping tests were evaluated to determine aquifer response to pumping and derive soil (W-1) and bedrock (BR-7A and BR-6A) values of transmissivity and storativity. The evaluation included interpretation of graphical plots of well drawdowns versus time using standard analytical models for transient groundwater flow. Drawdown data and graphical plots of well response are presented by test in Appendix C. Test results and calculated aquifer values are summarized below. Conclusions based on the test results are presented in Section 5.0

2.3.2.1 W-1 Test (overburden). Plots of drawdown versus time pumping were created from the automated data collected at W-1 and B-1. The test's discharge rate of 1.1 gpm produced maximum drawdowns of approximately 2.2 ft in W-1 and 0.2 ft in B-1. Drawdown in both wells was overcome after the start of pumping by rising water levels, apparently due to recharge from surface water that was ponding in the vicinity of the wells. Because water levels in both wells were stable (constant) when measured on November 6th and prior to pumping on November 7th, it appears that recharge did not influence drawdown until sometime after the start of pumping. This effect is clearly shown on the time-drawdown plots. The combination of the effects of this recharge, variations of discharge rate, and barometric changes altered drawdown responses and limited data interpretation beyond about the first hour of pumping.

Early-time data were interpreted using curve matching techniques based on the method of Theis (1935). The techniques consist of overlaying a type curve on each data plot and identifying a match point that can be used to identify the values required by the method's solution. At B-1, the drawdown data fit a theoretical response from one to about 30 minutes (time pumping). The Theis solution, shown on the data plot, yields a transmissivity of 340 ft^2/day and a storativity of 0.0001.

Data from W-1 were analyzed using the method of Boulton (1963), a Theis-based method that appears to fit the observed data curve better than Theis. Boulton developed a solution for unconfined aquifers to account for delayed drainage. This method is used when the rate of drawdown slows due to gravity drainage from pores in the vicinity of the well, and then resumes as gravity drainage is depleted. Data from W-1 conform with the theoretical Boulton response until about an hour after the start of pumping, but they show no resumption of drawdown following depletion of delayed drainage, probably due to the effects of the recharge event. This limits the usefulness of this solution. The Boulton method yields a value for transmissivity of approximately 1.5 ft^2/day. This is quite different than the calculated value from B-1 (340 ft^2/day).

A third calculation for transmissivity was derived by analyzing recovery data from the initial yield test of W-1 that was performed on September 29, 1995. (Recovery data from the November pumping test were not collected at close-spaced time intervals due to the unanticipated pump shut-off and were not therefore available). The September test pumped W-1 at several different discharge rates over 141 minutes of total pumping time (average time-weighted pumping rate of 0.54 gpm). Recovery data, which are less affected by variations in discharge rate than pumping phase data, were analyzed using the method of Harrill (1970) and produced a transmissivity of 8 ft^2/day.

In summary, the W-1 test produced estimates of the overburden transmissivity that range over more than two orders of magnitude (1.5 to 340 ft^2/day) and appear to include within this range the likely true value for the aquifer. The wide range of transmissivity values produced by the test may have resulted from the influences of delayed drainage in the unconfined aquifer and the observed recharge event on water levels during pumping. Although pumping tests in unconfined aquifers, such as the overburden, are typically run for a long enough period of time to allow observation of drawdown responses after delayed drainage from dewatered pores has been exhausted, this was not done for the W-1 test. The test was initially cut short by an unplanned pump stoppage, but it was not rerun because it was believed that the large influence from the recharge event would make the data from a longer test difficult or impossible to analyze.

2.3.2.2 BR-7A Test. Data plots of well drawdowns versus time were created and evaluated to derive aquifer characteristics. The test's average discharge rate of 6.1 gpm produced a measured head change (apparent drawdown) in the pumping well of approximately 7.2 feet. Apparent drawdown in observation wells ranged from none to about 0.4 feet.

Data adjustments. Before drawdown data were analyzed, the results were examined to assess whether adjustments for external influences such as barometric effects or background water level trends were required. Barometric change was slight throughout the test interval and a comparison of BR-7A water level and barometric trend during the pumping and recovery phases shows no apparent barometric effect. Therefore, no adjustment for barometric pressure changes was applied to well data.

Pre-test water-level measurements indicated slight rising piezometric trends ranging from about 0.04 to 0.14 feet per day in the observation piezometers. Drawdown data from PZ-103, PZ-105, PZ-106, and PZ-107 were adjusted based on the background levels projected by these trends. PZ-104 remained nearly static throughout the pumping and recovery intervals, although it exhibited a rising trend similar to the other piezometers in the days before the pumping test. Because the magnitude of adjusted drawdown would be wholly dependent on the value assigned to background trend, data from this well were not analyzed.

Pre-test measurements at BR-7A seemed to indicate a similar rising trend, however, hand-held meter readings were difficult to reproduce with consistency throughout the testing period. Oily scum and/or biological matter floating on the water surface are believed to have interfered with these measurements. The trend adjustment for BR-7A data was determined by examination of post-recovery electronic data. Data beyond 3780 minutes (2x the pumping interval) shows an upward trend of about 0.3 ft/day, about twice the magnitude of observation well trends.

Data Results. Data plots of drawdown versus time were created using the adjusted drawdowns shown on the data tables. Log-log plots of drawdown versus time were produced for observation wells PZ-103, PZ-105, PZ-106, and PZ-107. The data were all hand measured water levels because the only electronic transducer (PZ-105) malfunctioned and did not yield usable data. Aquifer properties were calculated from these plots using the method of Theis (1935). Results are presented in Table 2-12 and calculations are shown on the data plots. Figure 2-8 shows the interpreted maximum drawdown produced by the pumping of BR-7A.

In general, the data fit the early part of the Theis curve moderately well. Drawdown measurements collected by hand-held water level meter, which are generally accurate and reproducible to about 0.04 feet, may have introduced some inconsistency. Also, on most of the plots, the last one or two data points suggest a rise in water level. The cause of this is unknown, but it may reflect a change in the rate of background water level trend.

Semi-log plots of drawdown versus time were also generated from adjusted drawdown data from the four observation wells. These plots were analyzed by the method of Cooper and Jacob, (1946), a Theis-based method that requires the data to plot along a straight line when plotted on a semi-log scale. This technique is applicable for late time drawdown data, generally requiring that the value of u be less than about 0.01 to 0.03. While this condition was not met for all wells, (it would require several days of pumping), the Cooper and Jacob analyses are included as they show general agreement with the Theis results and consistency of values between observation piezometers.

At BR-7A, the variation in pumping rate during the early part of the test affected the rate of drawdown and rendered the pumping phase data unusable. Fortunately, water level recovery data are relatively unaffected by such variations during pumping and could be used to interpret transmissivity at the test well using the method of Harrill (1970).

As discussed previously, PZ-104 did not exhibit significant head change during the pumping or recovery phases of the test and therefore is not interpreted to have been affected by pumping. Two other bedrock wells, BR-3A and BR-8 were also not effected by pumping. The water level within BR-3A rose at a consistent rate of 0.00014 ft/minute throughout the pumping and recovery phases. BR-8, measured during the recovery phase only, showed no apparent change in water level.

Calculated results show excellent consistency and suggest that the shallow bedrock system in the vicinity of this well has a transmissivity of between 300 and 600 ft^2/day. Storage coefficient values exhibited more variability, but suggest semi-confined conditions in the shallow bedrock aquifer. The absence of drawdown response in PZ-104, BR-3, and BR-8, (which are closer to BR-7A than PZ-106 and PZ-107) may indicate that the fractures intercepted in BR-7A are most transmissive in an east-west direction. Not enough is known, however, about the condition of the non-responding wells to make a judgment about directional response.

2.3.2.3 BR-6A Test. Data plots of well drawdowns versus time were created and evaluated to derive aquifer characteristics from the BR-6A test. Drawdown data from the pumping well and observation wells PZ-105, PZ-106, PZ-107, and BR-2A were plotted and evaluated. The test's average discharge rate of 20.1 gpm produced an apparent drawdown of 8.6 feet in the test well after 1000 minutes of pumping, and drawdowns of up to 1.5 ft in observation wells.

Data adjustments. Before drawdown data were analyzed, the results were examined to assess whether adjustments for external influences such as barometric effects or background water level trends were required. A plot of drawdown and recovery in the pumping well and barometric trend versus time is included in Appendix C. Barometric pressure change was slight throughout the test interval and had no discernible effect on well responses, therefore adjustments to drawdown data for barometric pressure were not necessary.

Linear scale plots of pre-pumping, pumping, and recovery water levels versus time indicate a slight downward background piezometric trend at the start of the pumping phase. This trend was calculated to be 0.43 ft/day downward at wells BR-6A, PZ-105, PZ-106, and PZ-107 and 0.20 ft/day downward at BR-2A. Drawdown data from BR-2A and PZ-106 were adjusted based on these trends. The trend does not affect the interpretation of data from BR-6A due to the relatively large drawdown in the pumping well, and the plots from this well have not been trend-adjusted. Also, PZ-105 recovered fully to pre-pumping static levels so it is not clear if the falling background trend persisted during pumping. Therefore, data from PZ-105 were interpreted as collected, without trend adjustment. Finally, although various trend adjustments were applied to the data from PZ-107, the resultant drawdown plots did not appear to fit any of the commonly used models for analyzing aquifer responses to pumping and could not, therefore, be interpreted to derive aquifer values.

Data Results. Log-log plots of drawdown versus time were produced for wells BR-6A, PZ-105, PZ-106, and BR-2A. Semi-log plots were produced for wells BR-6A and PZ-105. Aquifer properties were calculated from the log-log plots using the Theis method (1935) and from the semi-log plots after the method of Cooper and Jacob (1946). Results are presented in Table 2-13 and calculations are shown on the data plots. Figure 2-9 shows the interpreted maximum drawdown produced by the pumping of BR-7A.

The log-log plots show that the five wells responded to pumping consistent with the Theis model. Good type curve matches were generally achieved with the electronic data from wells BR-6A, PZ-105, and PZ-106. A good match was also obtained from the limited number of hand measurements collected at BR-2A.

The semi-log plots yielded straight line responses as required for the Cooper and Jacob method at wells PZ-105, PZ-106, and BR-2A. As noted earlier, this technique also requires that the value u be less than about 0.01 to 0.03. This condition was met at PZ-105 and PZ-106, but would require much longer pumping at BR-2A.

BR-7A showed a large response to pumping. Hand measurements indicate a head drop during pumping of greater than 4 feet and full restoration to pre-pumping static levels during recovery. There were not enough measurements collected from this well to use curve-matching techniques.

Two wells did not apparently respond to pumping. A few hand measurements taken during the pumping phase suggest that BR-3 and PZ-103 were did not respond to pumping of BR-6A. Water levels in both wells did not decline during the test.

The results agree well with the findings of the BR-7A pumping test. The large drawdown response in BR-7A indicates that the two test wells (BR-6A and BR-7A) are in close hydraulic connection, apparently along a highly transmissive fracture. The other nearby wells (PZ-105, PZ-106 and PZ-107) show similar drawdown response and calculated transmissivities. This test confirms the BR-7A test in general response of the shallow aquifer to pumping in the southern part of the site. It again confirmed that BR-3 is hydraulically isolated from the fracture system to the south.

Based on the results of the two tests, the bulk transmissivity of the bedrock along the southern boundary of the site is between 250 and 350 ft^2/day. This range of T may be higher than a general value for the bedrock beneath the site since it is calculated from wells generally along an east-west orientation. The absent or low drawdowns in wells located more northerly or southerly during the two tests (PZ-103, PZ-104, BR-2A, BR-3) may indicate that the test results represent conditions in a more transmissive east-west trending fracture zone. In that case, the results are biased by representing only those wells that responded to pumping during these short-duration pumping events. Drawdowns would be greater under long-term steady-state pumping conditions because less water would be available from the system as a whole than quickly available along this fracture zone. A discussion of the significance of these test results in configuring long-term capture of groundwater beneath the site is presented in Section 3.2.

3.0 DISCUSSION OF FINDINGS

This section discusses the findings of the Phase II RI, presented in Section 2.0. The discussion includes the physical characteristics of the study area and chemical distribution, transport, and fate. The findings are integrated into a site conceptual model based on the site conceptual model developed from the Phase I RI, and augmented by Phase II findings.

3.1 HYDROGEOLOGY

This subsection provides a description of the hydrogeology of the study area as it is understood from both the Phase I and Phase II RI. A summary of the overburden and bedrock geology, bedrock structural features, and groundwater flow conditions is presented. Because the Phase II RI field work was focused on additional off-site bedrock well installations and groundwater sampling, special emphasis is placed on further characterization of the bedrock groundwater system.

3.1.1 Surficial Geology

The surficial geology of the study area is characterized by Late Pleistocene glacially-deposited sands and silty sands, interpreted to be till or stratified till. Based on observations, localized glaciolacustrine sediments may also be present toward the eastern regions of the study area. In general, sediments in the upper part of the till are better sorted than the lower part. Upper till sediments show signs of stratification. The till is covered locally by fill interpreted to be a recompacted till. Collectively, the till and fill are referred to as overburden in this report. Overburden thickness in the McKee Road area ranges from approximately 10 to 20 feet. Borings completed along the east side of the Erie Barge Canal during the Phase II RI show a slight reduction in overburden thickness, when compared to borings nearer the Olin Plant site to the east. Overburden thicknesses range from 9 to 12 feet at each of the three paired monitoring well locations (BR-111/111D, BR-112A/112D, and BR-113/113D).

3.1.2 Bedrock Lithology

Bedrock underlying the overburden has been identified as the Lockport Dolomite (Olin, 1990). Regionally, this formation consists of very gently dipping medium-to-thick bedded fine-grained dolomite with interbedded shales (Williams, 1990). In the study area, the formation is characterized by light gray color, medium bedding, and fine-grained texture with interbedded shale lenses and stringers. The three members that comprise the Lockport Dolomite in the Rochester area are:

Oak Orchard -	Granular (sucrosic), bituminous dolomite.
Penfield -	Coarse-grained, argillaceous dolomite containing mineralized vugs.
Decew -	Fine-grained, argillaceous dolomite with contorted bedding.

In the Rochester area, the Lockport Dolomite is reported to be up to 180 feet thick, whereas the thickness within the study area is reported to be up to 100 feet (Haley and Aldrich, 1990).

The bedrock surface is interpreted to have little to moderate relief, with elevations ranging from approximately 520 to 530 feet above MSL. Based on rock core logs at the Olin Plant site and logs from the Dolomite Products Quarry in the Town of Gates, the interpreted ranges of thickness of each member of the Lockport Dolomite are as follows:

Oak Orchard -	0 to 25 feet
Penfield -	50 to 60 feet
Decew -	10 to 15 feet

The contact between each member is gradational. A gradational contact also exists between the Decew Member and the underlying Gates Member of the Rochester Shale. This contact is estimated at 450 to 460 feet above MSL (~ 75 to 85 feet bgs) in the area of the Olin Plant site. Two interpreted geologic cross sections have been developed, showing bedrock lithologic features. Figure 3-1 illustrates the location of each cross section and Figures 3-2 and 3-3 show the north-south (A-A') and east-west (B-B') geologic cross sections, respectively. Data that was used to develop the cross sections includes boring log information from the Olin Plant, Cumberland Farms Petroleum Terminal, and Dolomite Products Quarry Sites.

3.1.3 Bedrock Structural Features

Bedrock structural features may influence the occurrence of groundwater and the pattern and rates of groundwater flow. Groundwater flows primarily through open fractures and joints that are present in the Lockport Dolomite. The degree of connectivity between these features in part determines the rate at which groundwater will flow in bedrock. Bedrock joints and fractures are depicted in Figure 3-1. Joint patterns, mapped from nearby Monroe County Pure Waters' Combined Sewer Overflow Abatement Program (CSOAP) tunnels, trend from 60° to 80° east of north (Haley and Aldrich, 1990). Although there is no dip information reported for these measurements, the majority of open fractures and joints in rock cores from the Olin Plant, Cumberland Farms Petroleum Terminal and Dolomite Products Quarry Sites, are oriented parallel or subparallel to bedding planes or at very high angles and perpendicular to bedding planes.

One key bedrock feature observed in the study area is an open water-bearing fracture in the Dolomite Products Quarry. The water-bearing portion of the fracture zone measures approximately 750 feet horizontally, along the east and south faces of the quarry (See Figure 3-1). This fracture is interpreted to be concordant with bedding planes and strikes N 61° E with an approximate 1° dip to the south.

The quarry is reportedly dewatered at a rate of approximately 700,000 gallons per day from the drainage pond (Haley and Aldrich, 1987). Assuming that all water in the drainage pond is from seepage along the quarry face, the daily pumping due to groundwater seepage would range up to approximately 500 gpm. The quarry, however, also receives runoff from precipitation in the surrounding area. Therefore, groundwater seepage likely is less than 500 gpm.

In addition to joints and fractures, the presence of vuggy zones may have an influence on the groundwater flow system. Vugs are cavities ranging from less than $^1/_8$ -inch to several inches in diameter created by dissolution of carbonate or sulfate minerals. Essentially, the minerals partially dissolve over long periods of time, leaving voids in the rock. Because dissolution is accomplished by a relatively large amount of water passing through the bedrock, there may be inherent zones of preferential groundwater flow associated with vuggy zones. Several drilling logs from the Phase II RI and previous investigations show water loss while drilling at depths of vuggy zones. Vugs were observed at various depths within the Penfield Member of the Lockport Dolomite. One vuggy zone was observed at approximately 45 to 50 feet below ground surface in nearly all bedrock wells that penetrated that depth in the Olin Plant Area. Generally, from 1 to 3 small vugs were seen in cores here, although in some cores, numerous small pits characterized this vuggy zone. This zone was traced westward onto the Cumberland Farms Petroleum Terminal Site where vugs where reported in rock cores from four locations in equivalent stratigraphic position. As shown on Figures 3-2 and 3-3, this vuggy zone is interpreted to be concordant with bedding and may be at a similar elevation as the water-bearing fracture in the Dolomite Products quarry.

An inferred zone of structural deformation has been reported in the study area (Haley and Aldrich, 1990). Available geologic reports show a zone of deformation, consisting of low amplitude folds that trend northwest to southeast, from an area southwest of the Olin Plant site toward commercial buildings to the southeast. This deformation zone, along with several others in the Rochester area, reportedly trend from 50° to 70° west of north (Haley and Aldrich, 1990).

3.1.4 Groundwater Flow Conditions

Groundwater beneath the Olin Plant site is present in both overburden and bedrock. The saturated thickness of the overburden ranges up to 10 feet beneath the Olin Plant site. However, the overburden becomes unsaturated between the Olin Plant and the downgradient Erie Barge Canal to the west (see Figure 3-4). In bedrock, groundwater flow occurs in the upper, more fractured zones of the bedrock and in deeper discrete fracture zones such as observed in the Dolomite Products Quarry. Groundwater flow is interpreted to be somewhat restricted in more competent rock between the upper and deeper zones. Thus, the focus of bedrock groundwater flow discussion and interpretation is placed on a shallow bedrock groundwater system, monitored by wells extending 15 to 40 feet into rock, and a deep bedrock groundwater system, monitored by wells extending up to 70 feet below the top of rock. No barrier to flow between the overburden and the shallow bedrock groundwater systems has been identified.

Flow conditions are characterized using water level data from November 6 , 7 ,and 8, 1995 as referenced to MSL. At the time of measurement, both overburden and bedrock pumping wells had been shut off since September 20, 1995, to conduct various aquifer pumping tests. In addition, the water level in the Erie Barge Canal was at its "summer level", or high level. The Erie Barge Canal, in this area of Rochester, is maintained at its summer level (~ 510' to 512' MSL near the Buffalo Road Bridge) between early May and late November. The lower "winter level" (~ 501' to 503' MSL) is maintained from late November until early May.

Overburden Groundwater Flow Conditions. The groundwater table in the overburden is generally less than 10 feet bgs throughout the study area. Overburden groundwater flow appears to be controlled to some degree by the underlying bedrock surface topography, and the nature and distribution of water-bearing fractures and flow directions in bedrock. Figure 3-4 presents interpreted piezometric contours for overburden groundwater that were developed using the November 1995 non-pumping water level measurements.

The piezometric contours indicate that overburden groundwater flows mainly to the west and south from the Olin property, toward the Erie Barge Canal and Buffalo Road. In addition, new monitoring well MW-114, southeast of Olin, further supports the interpretation of a southeastward flow component from the Olin Property. A groundwater divide exists along the eastern side of the Olin Property, extending southward from a piezometric high point (~ 533' MSL) in the northeast corner of the property. With the exception of localized effects of the pumping wells, the non-pumping flow conditions appear similar to those when pumps are running (see Figure 3-4). Water level measurements for November 1995 and in all seasons over the past two years show the overburden becomes unsaturated between the Olin Plant site and the Erie Barge Canal (see Figure 3-4). Vertical hydraulic gradient measurements between paired overburden and bedrock wells MW-103/BR-103, MW-104/BR-104, MW-106/BR-106, MW-107/BR-107, MW-114/BR-114, suggest overburden groundwater moves downward into the shallow bedrock groundwater system. This downward trend is consistent with past quarterly measurements when overburden and bedrock pumping wells were in service.

Horizontal hydraulic gradients calculated from the November 1995 water level measurements average about 0.012 feet per foot (ft/ft) in flow directions west, southwest, and south of Olin's Main Plant building. Based on hydraulic conductivities ranging from 1.3×10^{-4} cm/sec to 7.3×10^{-3} cm/sec (calculated from the results of Phase I RI slug tests) and an effective porosity of 30 percent, this gradient results in a linear flow velocity range of approximately 5 to 300 feet per year.

Bedrock Groundwater Flow Conditions. Figure 3-5 presents interpreted piezometric contours and flow directions for bedrock groundwater systems based on November 1995 water level measurements. The shallow bedrock contours were constructed assuming a constant head value of 510.89' MSL along the length of the Barge Canal. The water elevation was measured at the Buffalo Road Bridge. Beneath most of the study area, the shallow bedrock is overlain by, and is in hydraulic communication with, saturated overburden.

Based on the piezometric contours, bedrock groundwater is interpreted to flow primarily west and south from the Olin property toward the Erie Barge Canal and Buffalo Road. Measurements at BR-103 and new monitoring well BR-114 show that southward and eastward groundwater flow components also occur under non-pumping conditions. A groundwater piezometric high appears to exist at BR-5A and BR-4 along the eastern boundary of the Olin property. Figure 3-5 shows that most of the bedrock groundwater under non-pumping conditions is expected to flow toward the west and south from these high points.

Using measurements at BR-105D, BR-111D, BR-112D and BR-113D, deep bedrock groundwater flows generally east to west, toward the canal. The two deep bedrock wells at the Olin Plant site, BR-2D and BR-3D, show markedly lower water levels (< 461' MSL) than any of the other deep wells, and are interpreted to be connected to no significant water-bearing fractures. These wells were reported to sustain low yields (on the order of 0.1 gpm) during sampling (Olin, 1990).

With the exception of localized changes at the pumping wells, the November 1995 shallow bedrock groundwater contours are similar to those previously interpreted under pumping conditions. One shallow bedrock well that does not appear to fit into the interpreted flow pattern is BR-3. This well is a pumping well that is located in the south central portion of the site, and its water level was consistently measured approximately 8 to 9 feet lower than expected (521.91' MSL) based on other shallow bedrock piezometric evaluations/measurements. Pumping tests run on BR-6A and BR-7A, showed little response in BR-3 as compared to other wells at equal or greater distance. The pumping test data suggest that BR-3 is not hydraulically connected to other shallow bedrock wells on the Olin Property (see Section 2.3). The response in BR-3 may also indicate local dewatering or clogging of fractures.

Horizontal hydraulic gradients for shallow bedrock wells range from 0.01 to 0.05 ft/ft in areas nearer the Olin Plant site, while to the west of McKee Road, gradients range from 0.001 to 0.002 ft/ft. As shown on Figure 3-5, the changes in gradient magnitude suggest the presence of three zones of different transmissivity within the shallow bedrock. The lowest transmissivity zone, correlating with the area of highest gradient, appears to extend northeast to southwest across the northwest corner of the Olin Plant site and just to the west and southwest of the southern part of the site. Higher transmissivities are indicated by the lower gradients beneath the rest of the Olin Plant site. The very low gradients east of the Erie Barge Canal suggest that this zone has the highest transmissivity in the study area.

Water levels from the deep and shallow bedrock wells near the Erie Barge Canal are within several tenths of a foot of the canal water elevation, suggesting a high degree of hydraulic connection. Each of these wells is within 80 feet of the eastern wall of the canal. As shown on Figure 3-2, the shallow bedrock wells intercept the elevation of the canal water zone, whereas the top of screen elevation for the deep bedrock wells ranges from 5 to 17 feet below the bottom of the canal.

The type of influence the Erie Barge Canal has on bedrock groundwater probably changes seasonally in response to natural groundwater level changes and to the seasonal changes in the canal water level. The seasonal lowering of the canal in winter and raising of it in summer appears to cause the biggest change in the relationship between groundwater and the canal. The approximate 10-foot drop in the canal water elevation that is maintained during winter has a significant impact on local hydraulic gradients that must either increase the rate of groundwater discharge to the canal and/or reduce the rate of discharge from the canal. In fact, it may cause both responses in different stretches of the canal. The resumption of high water in the canal each spring then reverses this influence.

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Of the four shallow and deep bedrock well pairs to the west of the Olin Plant site, two well pairs (BR-105/105D and BR-113/113D) show a downward vertical hydraulic gradient. Gradients for these well pairs range from 0.07 to 0.21 ft/ft. Slight upward gradients of 0.002 to 0.01 ft/ft are calculated for well pairs BR-111/111D and BR-113/113D, respectively. These gradients may be influenced by groundwater discharging to the canal in an area that is in close proximity to the canal.

Typical linear flow velocities for shallow bedrock groundwater calculated from the Phase I RI ranged from 1.8 to 9.2 feet per day (ABB-ES, 1995a). Using the average hydraulic conductivity value of 3.5 x 10^{-2} cm/sec for shallow bedrock wells at the Erie Barge Canal (BR-111, BR-112A, and BR-113) with an assumed porosity of 0.1, linear flow velocities in the shallow bedrock system nearer the canal are estimated to range from 1.0 to 2.0 feet per day, at the lower end of the range calculated in Phase I. These lower flow velocities are due primarily to the lower estimated horizontal hydraulic gradients between McKee Road and the canal (0.001 ft/ft to 0.002 ft/ft).

West of the Ness Site and the Erie Barge Canal, shallow and deep bedrock groundwater has been interpreted to flow generally to the west southwest, towards the Dolomite Products Quarry in the Town of Gates (Haley and Aldrich, 1987). Groundwater seeps primarily along the east face of the quarry along a distinct water-bearing fracture (described in Section 3.1.3). Groundwater from this seepage face is collected in a drainage pond at the southeast corner of the quarry. Groundwater and surface water from this discharge pond are pumped into the Erie Barge Canal to the east. The elevation of the water bearing fracture zone in the quarry ranges from approximately 486 to 490 feet MSL. The discharge pond elevation is estimated to be 400 feet MSL. The projected plane of the quarry fracture zone is intercepted by deep bedrock wells along the Erie Barge Canal.

3.2 GROUNDWATER CAPTURE EVALUATION

Overburden and bedrock well pumping tests conducted as part of the Phase II RI provided the basis for evaluating the ability of the existing groundwater extraction system to capture chemicals migrating from the Olin Plant site in groundwater. These tests indicated that most of the existing overburden extraction wells are only capable of producing very low yields (0.1 gpm or less), but that the two tested bedrock wells were capable of higher yields (20 to 50 gpm each) than expected. These results suggested that recent operation of the existing system has provided incomplete capture of groundwater in both aquifers, but that capture might be accomplished through modifications to the system.

Complete capture of overburden groundwater by the existing overburden system appears unlikely both because of the low yields of existing wells and because the overburden aquifer has a small saturated thickness relative to the aquifer's hydraulic gradient. The low yields from existing wells may be the result of well clogging due to long-term operation (previous overburden system total pumping rates of more than 3 gpm were reported by Olin [1990]).

While the existing overburden extraction system appears unlikely to achieve complete capture of overburden groundwater, the higher than expected yields from the two tested shallow bedrock wells (BR-6A and BR-7A) suggests that pumping from these wells may be capable of providing better than expected capture. However, available data from operation of the bedrock extraction system indicate that each well in the bedrock system has been pumped typically at rates up to about 10 gpm, including BR-5A located on the eastern site boundary. Using an average transmissivity of 300 ft^2/d (from the BR-6A and BR-7A pumping tests) and an average gradient of 0.018 ft/ft, pumping at least 20 gpm has been estimated to capture this portion of the groundwater moving off-site. Single-well capacities of more than 20 gpm at BR-6A and BR-7A suggest that this rate, and thus capture, may be achieved by increasing the rate of pumping from existing wells. More recently, however, the results of groundwater modeling indicate that pumping wells BR-6A and BR-7A at 10 gpm each would be sufficient to capture both bedrock and overburden groundwater flowing from the southern half of the property (ABB-ES, 1996b).

In addition to capturing shallow bedrock groundwater, the existing bedrock extraction wells may also be capable of effecting either partial or complete capture in the overburden by creating bedrock drawdowns that cause either an increase in vertical flow from the overburden or dewatering of the overburden. To evaluate this possibility and provide a more thorough assessment of bedrock groundwater capture, a numerical model may be required.

3.3 CHEMICAL DISTRIBUTION, TRANSPORT, AND FATE

3.3.1 Groundwater Chemical Distribution

This section discusses the nature and distribution, transport, and fate of chemicals in groundwater associated with the Olin Study Area. Also presented is the site conceptual model that has been further refined based on findings of the Phase II field investigation.

The following is a discussion of the distribution of selected chemical groups in overburden and bedrock groundwater in the study area. The rationale for these groupings is provided in Section 2.0. Concentration isopleth maps constructed for groupings of pyridines and VOCs represent the current understanding of how these chemicals are distributed both on and off the Olin Plant site and are discussed in the following subsections. The isopleth maps are based on analytical results from September to December 1995 for 10 overburden, 18 shallow bedrock, and 8 deep bedrock wells. Data used to construct the isopleth maps are tabulated in Section 2.0. It should be noted that analytical results from a larger number of wells were available for the isopleth maps constructed for the Phase I RI and resulted in some isopleths having different shapes than these presented here. Appendix B provides a summary of historical groundwater sampling results for all off-site wells.

3.3.1.1 Semivolatile Organics and Pyridines. Chloropyridines have been the most frequently detected organic chemicals in both overburden and bedrock wells in the Phase I and Phase II RIs. Several TCL SVOCs, including bis(2-chloroethyl)ether and naphthalene, were also detected in several wells, but at lower concentrations than the pyridines. Olin is the only known potential source of pyridines in the area, and pyridines appear to be more water soluble and, therefore, more mobile than other site-derived organic compounds. Because of this mobility, the distribution of pyridines in overburden and bedrock groundwater systems is believed to represent the widest areal distribution of site-related chemicals in groundwater.

Overburden. Figure 3-6 presents interpreted concentration isopleths for total pyridines in overburden groundwater. On-site, the highest pyridine concentrations (greater than 100,000 mg/L) are at well B-17, matching the area of highest concentration for the Phase I RI. Off-site, a lobe of higher pyridines concentrations appears to extend westward to well MW-106, where the sum of pyridines measures 100,000 μg/L. The new monitoring well, MW-114, which is situated downgradient of the southeast concentration lobe, showed no pyridines. This pattern is consistent with the overburden groundwater flow contours, which suggest a more southerly flow direction from the Olin Plant site.

The Phase I report (1993/1994) showed two main concentration lobes originating from the Olin Plant site. One lobe extended toward the west and northwest to the limit of saturated overburden. The other was located to the southeast where pyridines were detected as far south as Buffalo Road. Because overburden groundwater is not present in areas west of the Olin Plant site, further delineation was not necessary in that direction. The one data gap that did exist was the distribution of pyridines to the southeast. The absence of pyridines at MW-114, serves to delineate this southeastern boundary of pyridines in overburden groundwater.

Bedrock. Figure 3-7 presents interpreted concentration isopleths for total pyridines in shallow and deep bedrock wells. On-site, the location of the highest pyridine concentrations (BR-6 and BR-3) in shallow bedrock is in the general area of the highest overburden groundwater concentrations in the south-central area of the Olin Plant site. A relatively wide lobe of pyridines, elongated in a southwesterly direction from the Olin Plant site, describes the distribution in shallow bedrock. Nearer the Erie Barge Canal, shallow bedrock wells show total concentrations ranging from non-detect to 4 μg/L for pyridines. Hydraulically, the distribution of pyridines would be expected to extend more westward to coincide with the westward groundwater flow direction. The more southerly-trending distribution of pyridines may have been the result of the operation of the two deep bedrock industrial wells which were in use until the mid-1980's at the NESS site (i.e. NESS-E and Ness-W).

At the Dolomite Products Quarry, located 4000 feet west-southwest of the Olin Plant, pyridines were detected in quarry seep samples. The concentration of pyridines was shown to increase from north to south along the east wall of the quarry. The highest concentrations, at QS-4, are within the same order of magnitude as in the Ness wells which are nearer the Olin Plant site (see Figure 2-5, Section 2.0). Based on the distribution of pyridines on the east side of the Erie Barge Canal and the location of the highest pyridine concentrations in the quarry, it is expected that these chemicals would be present along a south-southwest trending flow path between the Ness site and the quarry. The elevation of the interpreted deep bedrock fracture zone toward Olin and the Ness site is within the same range as the water-bearing feature observed at the quarry. The distribution of pyridines in bedrock groundwater to the west of the Erie Barge Canal will be further characterized through additional groundwater sampling between the canal and the Dolomite Products Quarry. The recent sampling results in the deeper Ness wells and at the quarry seep do suggest, however, that the southern extent of the pyridines in bedrock groundwater, especially deep bedrock groundwater, has not been fully delineated.

3.3.1.2 Volatile Organics. VOCs detected in both overburden and bedrock groundwater at significant concentrations include carbon tetrachloride, chloroform, methylene chloride, PCE, TCE, 1,2-DCE (total), 1,1,1-trichloroethane (1,1,1-TCA), and 1,1-dichloroethane (1,1-DCA). The distribution of these chemicals is illustrated by the sum of selected VOC concentration isopleth maps shown on Figures 3-8 and 3-9. Other groups of VOCs detected and discussed in this section include chlorinated ethenes (PCE, TCE, 1,2-DCE, and vinyl chloride) and BTEX compounds. The distributions of chlorinated ethenes are shown on Figures 3-10 and 3-11 while BTEX distributions are shown on Figures 3-12 and 3-13. Other VOCs detected in a high percentage of groundwater samples included chlorobenzene (up to 13,000 μg/L onsite and 1,400 μg/L offsite) and carbon disulfide (up to 3,100 μg/L onsite and 36 μg/L offsite). The distribution of VOCs in overburden and bedrock groundwater systems is discussed in the following paragraphs.

Selected VOCs in Overburden Groundwater. The total selected VOCs concentration isopleths (Figure 3-8) show that the highest concentrations of these constituents extend beneath the Well B-17 (140,000 μg/L) and Lab Sample Areas (340,000 μg/L). Carbon tetrachloride and chloroform comprise over 90% of the concentration totals in each of these areas. The highest concentrations of selected VOCs offsite are at MW-106, where chloroform was detected at 89 μg/L. Neither carbon tetrachloride nor methylene chloride were detected in any off-site overburden wells. Detections of PCE and TCE at MW-114 are interpreted to be unrelated to the Olin Plant site because of the absence of these compounds in Phase I sampling between the Olin Plant site and this well and the absence of pyridines in this well. The distribution of selected VOCs in overburden groundwater has been delineated to less than 10 µg/L in all directions from the Olin Plant site. Only at MW-106, where the overburden becomes unsaturated, does the total sum of selected VOCs exceed 10 µg/L.

Selected VOCs in Bedrock Groundwater. The selected VOCs distribution in both shallow and deep bedrock groundwater is characterized by higher carbon tetrachloride, chloroform, and methylene chloride concentrations on-site and higher 1,2-DCE and 1,1-DCA concentrations off-site. In shallow bedrock groundwater, the selected VOCs have been delineated to less than 10 µg/L in all directions except to the northwest where 1,2-DCE was detected in BR-107. In addition, concentrations over 10 µg/L for 1,2-DCE and 1,1-DCA constitute much of the plume of VOCs in deep bedrock wells extending to the south and southwest. This plume of VOCs has not been fully delineated in these directions. Additional sampling is required from wells south and southwest to characterize the distribution of site-related VOCs in these directions.

Chlorinated Ethenes in Overburden Groundwater. The distribution of total chlorinated ethenes shown on Figure 3-10, is similar to that of the total selected VOCs. This grouping adds vinyl chloride to the three chlorinated ethenes included in the selected VOCs group. Vinyl chloride, however, was only detected at MW-106 (8 J µg/L). As with the selected VOCs, the distribution of chlorinated ethenes in overburden groundwater has been delineated to less than 10 µg/L in all directions from the Olin Plant site.

Chlorinated Ethenes in Bedrock Groundwater. The distribution of total chlorinated ethenes shown on Figure 3-11, is similar to the offsite distribution of selected VOCs, due mostly to the presence of 1,2-DCE in many of the offsite wells. Higher concentrations of vinyl chloride and 1,2-DCE in BR-106 and BR-107, offsite, account for the significant lobe of these constituents that is detected toward the west northwest. In deep bedrock groundwater, chlorinated ethenes have not been delineated to the south and southwest. The pattern of higher concentrations in shallow bedrock on-site suggests chemicals in the deep bedrock groundwater found at the Ness site and along the canal have migrated from the south central area of the Olin Plant site. This pattern appears similar to that observed for the pyridines, which shows a more southerly distribution pattern (See Figure 3-7).

BTEX in Overburden Groundwater. In the 1993/1994 Phase I RI, the highest BTEX concentrations in overburden groundwater on site were centered off the south and eastern sides of the Olin main plant building. Total BTEX concentrations in these areas were above 1000 µg/L. Figure 3-12 shows the distribution interpreted for the Phase II RI. MW-106, to the west of the Olin Plant site, shows that total BTEX concentrations above 1000 µg/L are now detected off-site. BTEX concentrations at MW-114 (2 µg/L) to the southeast of the Olin Plant site serve to delineate the southeastern extent of BTEX: The 10 µg/L concentration isopleth was not previously closed off in this direction. The distribution of BTEX in overburden groundwater has been delineated to less than 10 µg/L in all directions from the Olin Plant site, except to the west where the overburden becomes unsaturated.

BTEX in Bedrock Groundwater. The highest BTEX concentrations in shallow bedrock groundwater are at BR-101 in the central part of the Olin Plant site (See Figure 3-13). In general, concentrations of BTEX in the 100 µg/L range are present throughout the southern half of the Olin property. In offsite shallow bedrock groundwater, higher concentrations were detected to the west at BR-106. A higher total BTEX concentration at BR-113 is not consistent with the distribution pattern of other chemicals. No other site-related VOCs were detected in this well, and concentrations of total chloropyridines were estimated ("J" qualifier) at only 2 µg/L. The source of BTEX in BR-113 is not known, and could be indicative of an off-site source..

To the northwest of the Olin Plant site, BTEX was detected at 120 µg/L in shallow bedrock well BR-107. The source or sources of this BTEX is not clear. Discussion in the Phase I RI suggested a possible alternate source of this BTEX. The northwest is the only direction for which BTEX has not been delineated below 10 µg/L in shallow bedrock groundwater.

In deep bedrock groundwater offsite, the BTEX distribution shows a somewhat different trend. The highest concentrations, both west and south of well BR-105D are measured in BR-111D, a well in which neither pyridines nor chlorinated ethenes were detected. These data imply a source other than the Olin Plant site. Lower BTEX concentrations to the south and southwest at wells NESS-E and NESS-W have not been fully delineated.

3.3.1.3 Inorganics. TAL inorganics concentrations detected in the 8 new monitoring wells and the Ness wells (NESS-E and NESS-W) do not exhibit distributions that can be clearly linked to a site or source. One notable finding was the relatively high concentrations of inorganics in the NESS-E well. As presented in Section 2.0, some inorganic concentrations in this well measured up to an order of magnitude higher than any of the other wells sampled during the Phase I or Phase II RIs.

3.3.2 Chemical Transport

This subsection evaluates the migration potential of site-related chemicals for the Olin Plant site. Discussions of transport are based upon findings of both the Phase I and

Phase II RIs with particular emphasis on new findings from Phase II (1995) investigations. Transport of site-related chemicals identified during the Phase I Investigation are discussed in Section 5.0 of the Final Phase I RI (ABB-ES, 1995a). Any changes in the understanding of chemical transport mechanisms, as a result of Phase II Investigation data, are discussed in this subsection. As the dominant transport mechanisms are identified, the chemical distribution can be interpreted in terms of past events, and the future chemical distribution can be better predicted.

Chemicals found on-site include pyridines, VOCs, SVOCs, and inorganics. The observed distributions of these constituents are the result of their physio-chemical properties and site-conditions. As discussed in the Phase I RI, site conditions governing transport (i.e., migration) of chemicals include location of likely on-site sources, topography, meteorological conditions, and hydrogeology. Site conditions and the physio-chemical properties of site-related chemicals determine which transport mechanisms will predominate. Applicable physio-chemical properties for organic chemicals include specific gravity, solubility, and the organic carbon partition coefficient (K_{oc}). Applicable physio-chemical properties for inorganic constituents include oxidation state, pH, and specific solute species.

Physio-chemical properties of site-related chemicals are listed in Table 3-1. Specific gravity is the ratio of the mass of a given volume of a liquid substance to the mass of an equal volume of water. Liquids with specific gravities greater than 1 are termed "heavier" than water. Solubility values and the K_{oc} represent measures of the tendency of a material to move from one phase to another. Solubility measures the partitioning between the pure liquid or solid form of a chemical and the aqueous phase, or the tendency of a material to dissolve in water. Substances with relatively low solubilities are more likely to remain in a separate phase when in contact with water; substances with high solubilities will dissolve and move with water. K_{oc} measures the extent that an organic chemical partitions between a solid phase (organic carbon) and a liquid phase, and is used to predict whether a chemical could be adsorbed to soil organic carbon (Ney, 1990). Chemicals with a K_{oc} of greater than 10,000 will adsorb strongly to soil organic carbon. Chemicals with a K_{oc} in the range of 1,000 to 10,000 can be partially adsorbed or retarded during transport with water. Chemicals with a K_{oc} of less than 1,000 will generally not adsorb strongly to soil organic carbon and are relatively mobile.

Note that specific gravities, solubilities, and K_{oc} values are not provided for inorganics listed in Table 3-1. This is because inorganic analyses measure the total amount of a particular constituent in the sample; not the actual chemical form (such as a metal hydroxide complex or metal-ligand complex) or metal oxidation state. The distribution of specific solute species, pH, and oxidation state are important in determining the total solubility or mobility of a given inorganic.

3.3.2.1 Atmospheric Migration. Atmospheric migration of chemicals occurs primarily by: (1) volatilization of the chemical into air; and (2) release of fugitive dust with chemicals adsorbed to soil or other particulates. The scope of soil sampling during the Phase II investigation was limited to the Lab Sample Area. The first mechanism, volatilization, was addressed during the Phase I RI and was identified as a major pathway for VOCs from surface soils and filled materials to on- or off-site receptors. Analysis of surface soil samples during the Phase II investigation was limited to mercury. Because the presence of mercury in surface soil is limited to a small area, significant mercury transport via a volatilization mechanism is not anticipated. The second method, fugitive dust release, predominates for organic compounds with high adsorption characteristics (i.e., high K_{oc} values) such as SVOCs and PCBs, and for inorganics. Atmospheric migration of mercury via wind-blown particulate matter is a possible chemical migration pathway. The extent to which these mechanisms operate is governed, in part, by meteorological conditions and the amount of exposed contaminated surface materials. Very little exposed soil is present at the Olin Plant site, therefore, there is little likelihood of migration in dust.

Atmospheric migration via volatilization from the subsurface into basements or floor slabs through cracks, openings, or sumps represents a possible migration pathway for VOCs and, to a lesser extent, the pyridines in subsurface soil from the Lab Sample Area. This pathway was examined in the Phase I RI and was shown to be insignificant.

3.3.2.2 Surface water Migration. Surface water can transport chemicals either as a dissolved phase or adsorbed onto entrained particulate matter. Dissolved and adsorbed phase chemicals move to surface water via either runoff from contaminated surface soils or discharge from contaminated groundwater. As discussed during the Phase I RI, runoff from contaminated surface soil is not expected to be a significant migration pathway because on-site runoff is collected in storm sewers and combined with process water for disposal at the POTW. Surface water discharge from the shallow bedrock groundwater system or the deep bedrock fracture into the Erie Barge Canal downgradient of the Olin Plant site is a possible migration pathway suggested by the similarity of water levels of bedrock groundwater and the canal.

3.3.2.3 Groundwater Migration Chemicals may enter groundwater at the study area through the downward seepage of chemicals, either as pure liquids or dissolved in water by the action of rain, surface water, or shallow groundwater percolating (leaching) through the subsurface soil. In all cases, the concentrations in groundwater depend on the solubility of the chemical in water and the concentrations present in the source areas. Chemicals entering groundwater as a dissolved phase move with groundwater flow.

If chemicals enter groundwater as a non-aqueous phase liquid, they will migrate in a direction dependent on the specific gravity of the chemical phase, groundwater flow, entry pressures, and the surface topography of any confining layers. Over time, and depending on the characteristics of the bedrock fractures, some fraction of DNAPL will diffuse into the pore water of the rock matrix where it will become relatively immobile, but will continue to be a source of groundwater contamination. Groundwater data from the Phase I RI and prior sampling events show concentrations of organic chemicals for several VOCs exceeding one percent of solubility limits (ABB-ES, 1995a). A separate phase liquid has been observed in the past in two bedrock wells (BR-3 and BR-5) (Olin, 1990). However, no separate phase liquid was observed during either the Phase I or Phase II RI.

Vertical Migration. Hydrogeologic data were used to assess potential vertical migration of dissolved chemicals in groundwater from the Lab Sample Area and in areas downgradient from the site. Monitoring well and piezometer cluster analytical data at the Olin Plant show vertical hydraulic gradients that indicate groundwater moves downward from the overburden groundwater to the shallow bedrock groundwater system. Shallow bedrock groundwater is interpreted to migrate to deep bedrock through vertical fractures. The vertical seepage rate is interpreted to be significantly less than the horizontal seepage rate.

Horizontal Distribution. As identified previously in the Phase I RI, the horizontal distribution of site-related chemicals suggests a more widespread pattern in the shallow bedrock groundwater than in the overburden groundwater system. As a result of the Phase II investigation showing pyridines in the groundwater at the Dolomite Products Quarry, there is indication of a further downgradient distribution pattern of site-related chemicals than known during the Phase I RI. VOCs detected at high concentrations on-site (e.g., carbon tetrachloride, chloroform, and methylene chloride) appear to have migrated only a short distance off-site. In contrast, the chloropyridines which are more miscible with water, and the chlorinated volatile organics that are believed to be degradation products of PCE and/or TCE, have migrated off-site to the south and southwest. Chloropyridines are found as far as 4000 feet from the Olin Plant site in the Dolomite Products Quarry.

3.3.2.4 Migration in Soil. Soil sampling during the Phase II investigation occurred at the Lab Sample Area only. Migration of chemicals in soil can occur primarily by: (1) volatilization of the chemical in the surface or near surface soils or (2) leaching of the chemical constituent (either dissolved phase or entrained on particulate matter) via shallow groundwater percolating through the subsurface. Once in the air space of the unsaturated soil zone, the chemical will either be emitted to the atmosphere or be resolubilized and carried back down to the groundwater.

Chemical partitioning between the soil and groundwater retards the migration of site-related chemicals with respect to groundwater velocity. This may allow other attenuative processes, such as degradation, to be more effective.

3.3.3 Chemical Fate

The primary constituents potentially migrating from the site are pyridines, and VOCs. These are migrating from past releases at the Olin Plant to groundwater. There is potential for their discharge into the Erie Barge Canal and/or migration beyond the Erie Barge Canal to the Dolomite Products Quarry.

3.3.3.1 Pyridines. Processes that control the fate of pyridines at, and downgradient of, the study area include biodegradation and volatilization. Over time, pyridine and chloropyridines are expected to leach from pyridine-contaminated soils on-site into the groundwater. Once in the water, pyridine is expected to migrate in the groundwater regime and eventually undergo biodegradation, photo-oxidation (after discharging to surface water), and volatilization.

Biological degradation and reductive mechanisms constitute the major dissipation processes of selected pyridines (Sims and O'Loughlin, 1989). Reduction of pyridine in the environment, however, is proposed to be by both aerobic and anaerobic microorganisms. Pyridine is readily degraded by microorganisms, but the biodegradation rate of pyridine derivations appears to be affected dramatically by the nature and position of ring substitutes present in the chloropyridines.

In general, the chloropyridines are more persistent than pyridine and increasing the number of halogen substituents increases the persistence of the pyridine ring (Sims and O'Loughlin, 1989).

Based on detections in samples from the off-site monitoring wells , the chloropyridines have migrated farther from the site in groundwater than other site-related chemicals. Chloropyridines have consistently been detected in the deep bedrock groundwater discharging from the eastern wall of the quarry. Concentrations detected in the quarry pond where this discharge accumulates are substantially lower than those at the discharge point. This concentration reduction appears to largely be the result of dilution by unaffected groundwater that also discharges to the quarry, but some reductions through photo-oxidation and volatilization may also be occurring. No chloropyridines have been detected in the pond water at the point where it is discharged by pumping into the Erie Barge Canal.

3.3.3.2 Volatile Organic Compounds. Site-related VOCs are classified as halogenated hydrocarbons (e.g., carbon tetrachloride, chloroform, and methylene chloride) which contain one or more halogens, and aromatic hydrocarbons (i.e., BTEX) which possess one benzene ring as the basic structural unit. Variables that will control the fate of VOCs at the study area include volatilization, degradation, and dissolution.

Dissolution of VOCs from past release sources to groundwater and degradation are believed to be the most significant fate processes for VOCs at, and downgradient of, the study area. Factors affecting dissolution and degradation of VOCs include: (1) water table elevation in contaminated soil, (2) flow rate (residence time) of the groundwater in the contaminated material, and (3) oxygen content.

Biodegradation reactions can reduce the total mass of VOCs in groundwater. Studies have identified naturally occurring soil and aquatic microorganisms capable of degrading aromatic hydrocarbons (Jamison, et al., 1975; and Bailey, et al., 1973). These microorganisms require oxygen for aerobic biodegradation activity.

Halogenated VOCs are degraded by different mechanisms than are aromatic hydrocarbons. The primary halogenated VOCs at the study area are carbon tetrachloride, chloroform, methylene chloride, and to a lesser degree, PCE and TCE. 1,2-Dichlorobenzene and chlorobenzene were also observed in the groundwater systems at the Olin Plant. Under aerobic conditions, halogenated VOCs are quite stable and persistent in the environment. Under anaerobic conditions, however, halogenated VOCs are believed to undergo biologic transformation as the dominant fate process. The anaerobic biologic transformation for PCE is well-documented (Vogel and McCarty, 1985; Vogel and McCarty, 1987) and shown as follows:

$$(1) \quad \text{PCE} \rightarrow \text{TCE} \rightarrow \text{1,2-DCE} \rightarrow \text{vinyl chloride}$$

$$(2) \quad \text{PCE} \rightarrow \text{TCE} \rightarrow \text{1,2-DCE} \rightarrow \text{1,2-DCA} \rightarrow \text{chloroethane}$$

The physio-chemical properties of the chlorinated aliphatic methanes which include carbon tetrachloride, chloroform, methylene chloride, chloromethane, and methane are such that water solubility and vapor pressure increases with decreasing chlorinated substitution. These compounds are expected to leach into groundwater where they may reside for long periods of time, (Howard, 1990). In groundwater under reducing conditions, these compounds, given time, would most likely undergo reductive dehalogenation (i.e., the removal of one Cl atom and the addition of one H atom) (Dragun, 1988). The reaction products that may be formed (in order of reductive dehalogenation) are: carbon tetrachloride degrading to chloroform degrading to methylene chloride degrading to chloromethane degrading to methane. Each of these compounds, with the exception of methane, is an analyte in the VOC analysis.

Biodegradation of chlorinated aliphatic methanes (e.g., carbon tetrachloride) is also possible, but would occur slowly and only in the presence of soil microorganisms capable of degrading the chemical.

Because compounds indicative of biodegradation were detected at the study area, and in areas downgradient, anaerobic degradation (reductive dechlorination) of PCE is interpreted to be a fate for these compounds. The net result of this fate mechanism is that the degradation products (mono and dichlorinated ethenes) are persistent and mobile. These compounds can be volatilized to the atmosphere or can be further degraded. Complete reductive transformation of PCE, TCE, 1,2-DCE isomers, and vinyl chloride to ethene and ethane has been documented (Bruin et. al., 1992).

Although VOCs have been detected in groundwater as far downgradient of the Olin Plant site as the Erie Barge Canal and the Ness wells, which are the farthest downgradient samples analyzed for VOCs, the VOC concentrations at these locations show a large decrease from those detected on-site. This suggests that VOCs have not migrated as far downgradient from the Olin Plant site as the pyridines, and may not have reached the Dolomite Products Quarry. A large portion of any VOCs that do reach the quarry would be expected to volatilize upon discharge.

3.3.3.3 Non-Pyridine SVOCS. Processes that control the fate of non-pyridine SVOCs (primarily PAHs and phthalates) at the study area, and areas downgradient, include adsorption, biodegradation, and dissolution. The TCL SVOCs detected at the study area are expected to be relatively immobile because of adsorption to the organic carbon fraction of the soil predicted through organic carbon-water partition coefficients and low solubilities (Tinsley, 1979; Kenaga and Goring, 1978). However, leaching of some PAHs to groundwater is observed to have occurred at the study area.

In aerobic environments, natural biodegradation processes can decrease the concentrations of PAHs (Kenaga and Goring, 1978; Weil, Dune, and Quentin, 1973). Overall, adsorption to soil and sediment with subsequent degradation is the expected fate of PAHs and pesticides at the Olin Plant site.

3.3.3.4 Inorganics. Several inorganics were detected in groundwater at the study area, and in areas downgradient. With the exception of one well (NESS-E), concentrations were generally consistent, suggesting the inorganics may be ubiquitous naturally occurring elements that are unrelated to the Olin Plant; however, some areas of higher concentrations of metals were noted. All groundwater analyses, however, were conducted on unfiltered samples, some of which had observable suspended solids content, suggesting particulate matter may have affected results. As such, the following is limited to a general discussion regarding fate of inorganic analytes.

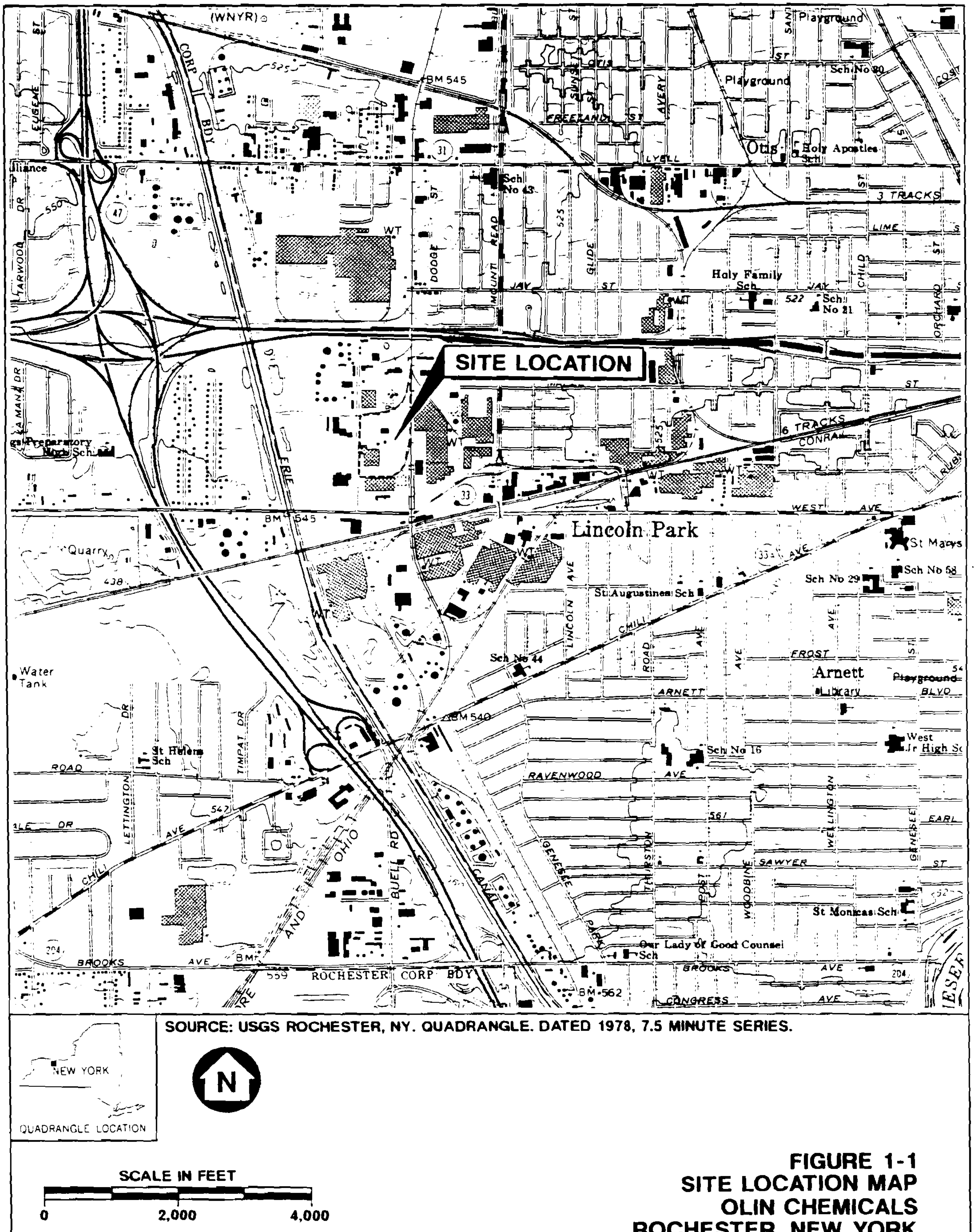
The discussion in this subsection remains qualitative because of the complex nature of inorganic chemistry. Mobility of inorganics in soil-groundwater systems is strongly affected by compound solubility, pH, soil action exchange capacity, soil type, oxidation-reduction potential, adsorption processes, major ion concentrations, and salinity. At the Olin Plant, geologic materials contain natural inorganics that could be available for transport to groundwater.

Several analytes readily form complexes with organic matter, carbonates, sulfates, or hydroxides. High concentrations of metals in groundwater can be observed where a relatively low oxidation potential exists because the metals can be reduced to more mobile species (Hem, 1989). If groundwater comes in contact with air, some analytes become oxidized, and may subsequently precipitate as hydroxides. Bacteria are also known to cause precipitation (oxidizing bacteria) or dissolution (reducing bacterial processes).

In natural waters, some analytes readily precipitate with carbonates, hydroxides, and sulfides to form relatively insoluble compounds. However, others may be quite stable in aqueous solutions, and have the potential to migrate over long distances (IRP, 1990).

3.3.4 SITE CONCEPTUAL MODEL

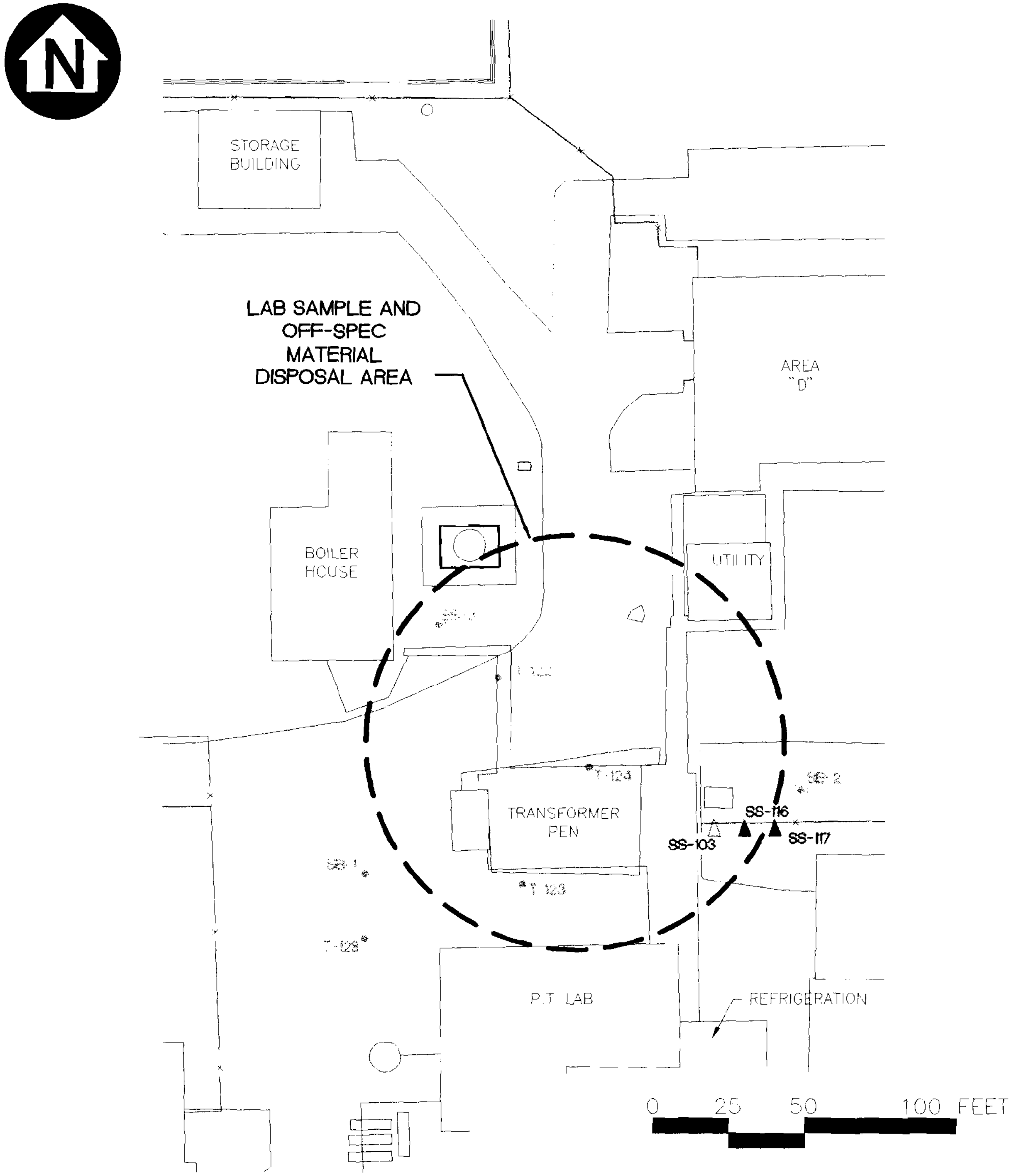
This subsection presents the site conceptual model as developed from both the Phase I and Phase II RI. Based on the discussion of chemical distribution, transport, and fate presented in the preceding subsection, a conceptual model of the site was developed to illustrate chemical migration pathways and physio-chemical processes resulting in the known distribution of site-related chemicals. A schematic cross section illustrating the conceptual model is presented in Figure 3-14. Figure 3-15 illustrates the conceptual model flow diagram for the study area. As did the model from the Phase I RI, this model shows the primary migration pathway at the Olin Plant site as leaching of chemicals from materials at the site by infiltrating precipitation in the unsaturated zone. DNAPLs are also a secondary source of dissolved-phase contamination, as groudnwater passes through the zone(s) of DNAPL. The conceptual model also shows contaminated groundwater then traveling in the overburden and shallow bedrock groundwater systems, where some moves vertically to the deeper bedrock and the remainder travels beneath or discharges to the canal. As shown on Figure 3-14, the migration of site-related chemicals in deep bedrock fractures beneath the canal, has been extended to the Dolomite Products Quarry, where chloropyridines are found in groundwater seeps. Along the chemical migration pathways, oxidation/reduction processes, dissolution, degradation, volatilization, dispersion and adsorption processes act to reduce the overall concentrations of the chemicals detected.



SOURCE: USGS ROCHESTER, NY. QUADRANGLE. DATED 1978, 7.5 MINUTE SERIES.

FIGURE 1-1
SITE LOCATION MAP
OLIN CHEMICALS
ROCHESTER, NEW YORK

ABB Environmental Services, Inc.



LEGEND

△ SURFACE SOIL SAMPLE (PHASE I)

* SOIL BORING (PHASE I)

▲ SURFACE SOIL SAMPLE (PHASE II)

* SOIL BORING (PHASE II)

FIGURE 2-1

LAB SAMPLE AREA SAMPLE LOCATIONS

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.

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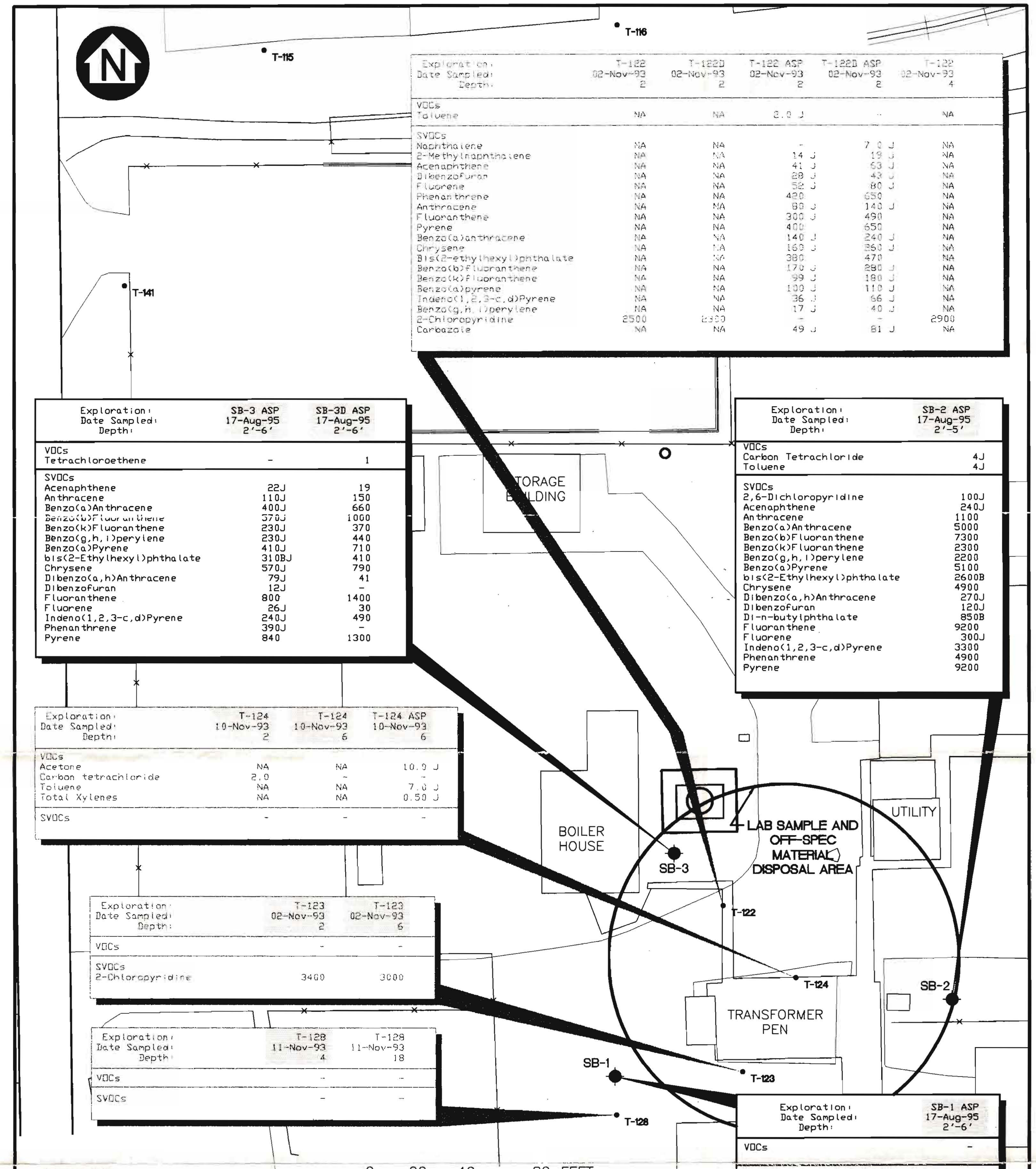


Exploration: Date Sampled: Depth:	T-122 02-Nov-93 2	T-122D 02-Nov-93 2	T-122 ASP 02-Nov-93 2	T-122D ASP 02-Nov-93 2	T-122 02-Nov-93 4
VOCs					
Toluene	NA	NA	2.0 J	-	NA
SVOCs					
Naphthalene	NA	NA	-	7.0 J	NA
2-Methylnaphthalene	NA	NA	14 J	19 J	NA
Acenaphthene	NA	NA	41 J	63 J	NA
Dibenzofuran	NA	NA	28 J	43 J	NA
Fluorene	NA	NA	52 J	80 J	NA
Phenanthrene	NA	NA	420	650	NA
Anthracene	NA	NA	80 J	140 J	NA
Fluoranthene	NA	NA	300 J	490	NA
Pyrene	NA	NA	400	650	NA
Benzo(a)anthracene	NA	NA	140 J	240 J	NA
Chrysene	NA	NA	160 J	260 J	NA
Bis(2-ethylhexyl)phthalate	NA	NA	380	470	NA
Benzo(b)fluoranthene	NA	NA	170 J	280 J	NA
Benzo(k)fluoranthene	NA	NA	99 J	180 J	NA
Benzo(a)pyrene	NA	NA	100 J	110 J	NA
Indeno(1,2,3-c,d)Pyrene	NA	NA	36 J	66 J	NA
Benzo(g,h,i)perylene	NA	NA	17 J	40 J	NA
2-Chloropyridine	2500	2300	-	-	2900
Carbazole	NA	NA	49 J	81 J	NA

Exploration: Date Sampled: Depth:	SB-3 ASP 17-Aug-95 2'-6'	SB-3D ASP 17-Aug-95 2'-6'
VOCs		
Tetrachloroethene	-	1
SVOCs		
Acenaphthene	22J	19
Anthracene	110J	150
Benzo(a)Anthracene	400J	660
Benzo(b)Fluoranthene	570J	1000
Benzo(k)Fluoranthene	230J	370
Benzo(g,h,i)perylene	230J	440
Benzo(a)Pyrene	410J	710
bis(2-Ethylhexyl)phthalate	310BJ	410
Chrysene	570J	790
Dibenzo(a,h)Anthracene	79J	41
Dibenzofuran	12J	-
Fluoranthene	800	1400
Fluorene	26J	30
Indeno(1,2,3-c,d)Pyrene	240J	490
Phenanthrene	390J	-
Pyrene	840	1300

Exploration: Date Sampled: Depth:	SB-2 ASP 17-Aug-95 2'-5'
VOCs	
Carbon Tetrachloride	4J
Toluene	4J
SVOCs	
2,6-Dichloropyridine	100J
Acenaphthene	240J
Anthracene	1100
Benzo(a)Anthracene	5000
Benzo(b)Fluoranthene	7300
Benzo(k)Fluoranthene	2300
Benzo(g,h,i)perylene	2200
Benzo(a)Pyrene	5100
bis(2-Ethylhexyl)phthalate	2600B
Chrysene	4900
Dibenzo(a,h)Anthracene	270J
Dibenzofuran	120J
Di-n-butylphthalate	850B
Fluoranthene	9200
Fluorene	300J
Indeno(1,2,3-c,d)Pyrene	3300
Phenanthrene	4900
Pyrene	9200

Exploration: Date Sampled: Depth:	T-124 10-Nov-93 2	T-124 10-Nov-93 6	T-124 ASP 10-Nov-93 6
VOCs			
Acetone	NA	NA	10.0 J
Carbon tetrachloride	2.0	-	-
Toluene	NA	NA	7.0 J
Total Xylenes	NA	NA	0.50 J
SVOCs	-	-	-

Exploration: Date Sampled: Depth:	T-123 02-Nov-93 2	T-123 02-Nov-93 6
VOCs	-	-
SVOCs		
2-Chloropyridine	3400	3000

Exploration: Date Sampled: Depth:	T-128 11-Nov-93 4	T-128 11-Nov-93 18
VOCs	-	-
SVOCs	-	-

Exploration: Date Sampled: Depth:	SB-1 ASP 17-Aug-95 2'-6'
VOCs	-
SVOCs	
2,6-Dichloropyridine	450
Benzo(a)Anthracene	50J
Benzo(b)Fluoranthene	84J
Benzo(k)Fluoranthene	20J
Benzo(g,h,i)perylene	8J
Benzo(a)Pyrene	44J
bis(2-Ethylhexyl)phthalate	140BJ
Chrysene	65J
Di-n-butylphthalate	48BJ
Fluoranthene	80J
Indeno(1,2,3-c,d)Pyrene	20J
Phenanthrene	33J
Pyrene	110J

0 20 40 80 FEET

SCALE: 1"=40'

LEGEND:

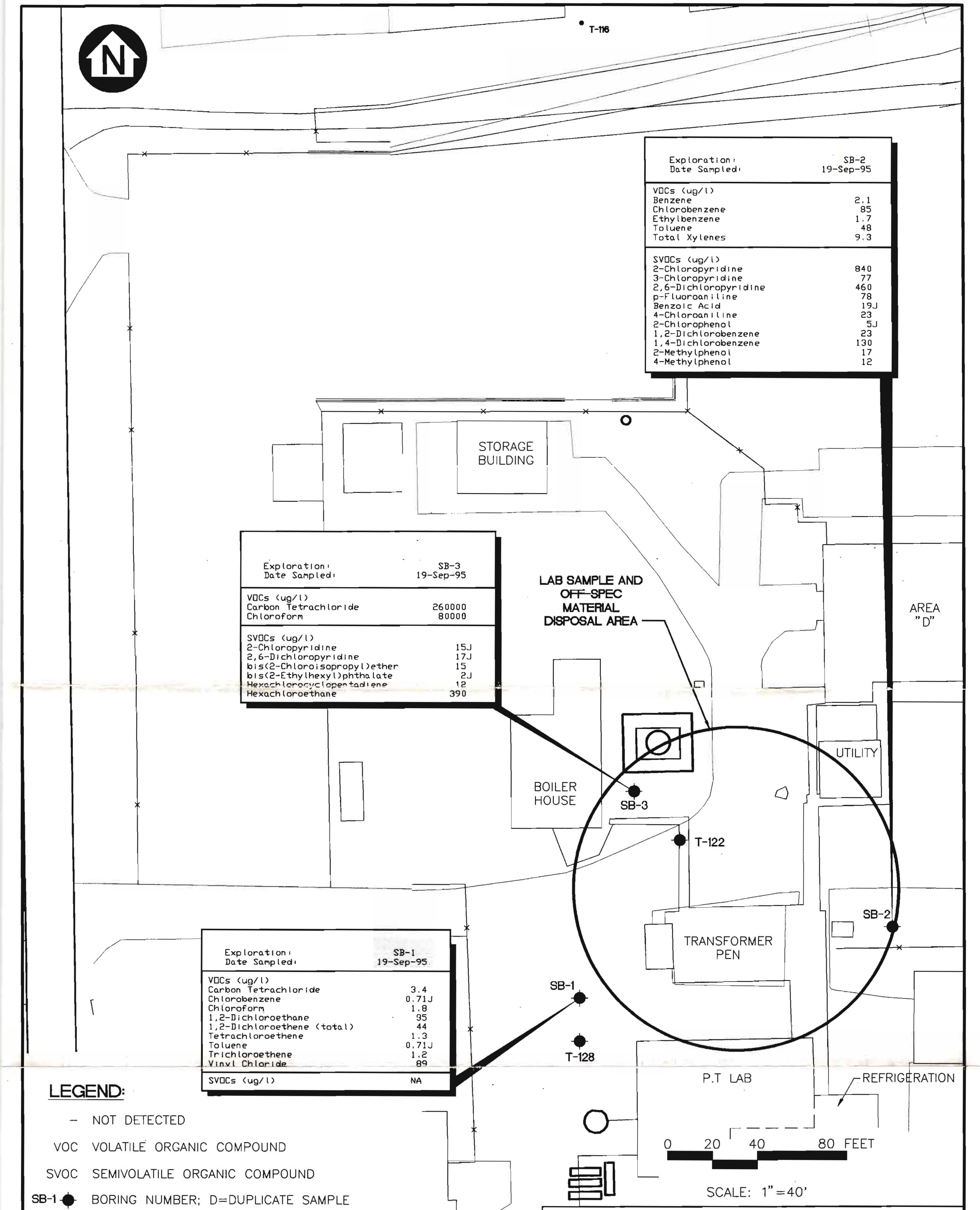
-	NOT DETECTED
VOC	VOLATILE ORGANIC COMPOUND
SVOC	SEMIVOLATILE ORGANIC COMPOUND
T-126D	BORING NUMBER; D=DUPLICATE SAMPLE
ASP	SAMPLE ANALYZED BY NYSDEC OR ASP METHODOLOGY. ALL OTHERS ANALYZED BY ON-SITE GC's.
NA	NOT ANALYZED
J	ESTIMATED; SEE APPENDIX B FOR EXPLANATION OF DATA QUALIFIERS

ALL CONCENTRATIONS REPORTED IN UNITS OF MICROGRAMS PER KILOGRAM. RESULTS IN LIGHTER TONE BOXES FROM PHASE I RI; DARKER TONE BOXES SHOW RESULTS FROM 1995 SAMPLING. UNSHADED SAMPLE ID's SHOW RESULTS FOR SATURATED SOIL.

FIGURE 2-2

VOCs AND SVOCs IN SUBSURFACE SOIL LAB SAMPLE AREA

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.



Exploration:	SB-2
Date Sampled:	19-Sep-95
VOCs (ug/l)	
Benzene	2.1
Chlorobenzene	85
Ethylbenzene	1.7
Toluene	48
Total Xylenes	9.3
SVOCs (ug/l)	
2-Chloropyridine	840
3-Chloropyridine	77
2,6-Dichloropyridine	460
p-Fluoroaniline	78
Benzoic Acid	19J
4-Chloroaniline	23
2-Chlorophenol	5J
1,2-Dichlorobenzene	23
1,4-Dichlorobenzene	130
2-Methylphenol	17
4-Methylphenol	12

Exploration:	SB-3
Date Sampled:	19-Sep-95
VOCs (ug/l)	
Carbon Tetrachloride	260000
Chloroform	80000
SVOCs (ug/l)	
2-Chloropyridine	15J
2,6-Dichloropyridine	17J
bis(2-Chloroisopropyl)ether	15
bis(2-Ethylhexyl)phthalate	2J
Hexachlorocyclopentadiene	12
Hexachloroethane	390

Exploration:	SB-1
Date Sampled:	19-Sep-95
VOCs (ug/l)	
Carbon Tetrachloride	3.4
Chlorobenzene	0.71J
Chloroform	1.8
1,2-Dichloroethane	35
1,2-Dichloroethene (total)	44
Tetrachloroethene	1.3
Toluene	0.71J
Trichloroethene	1.2
Vinyl Chloride	89
SVOCs (ug/l)	NA

LEGEND:

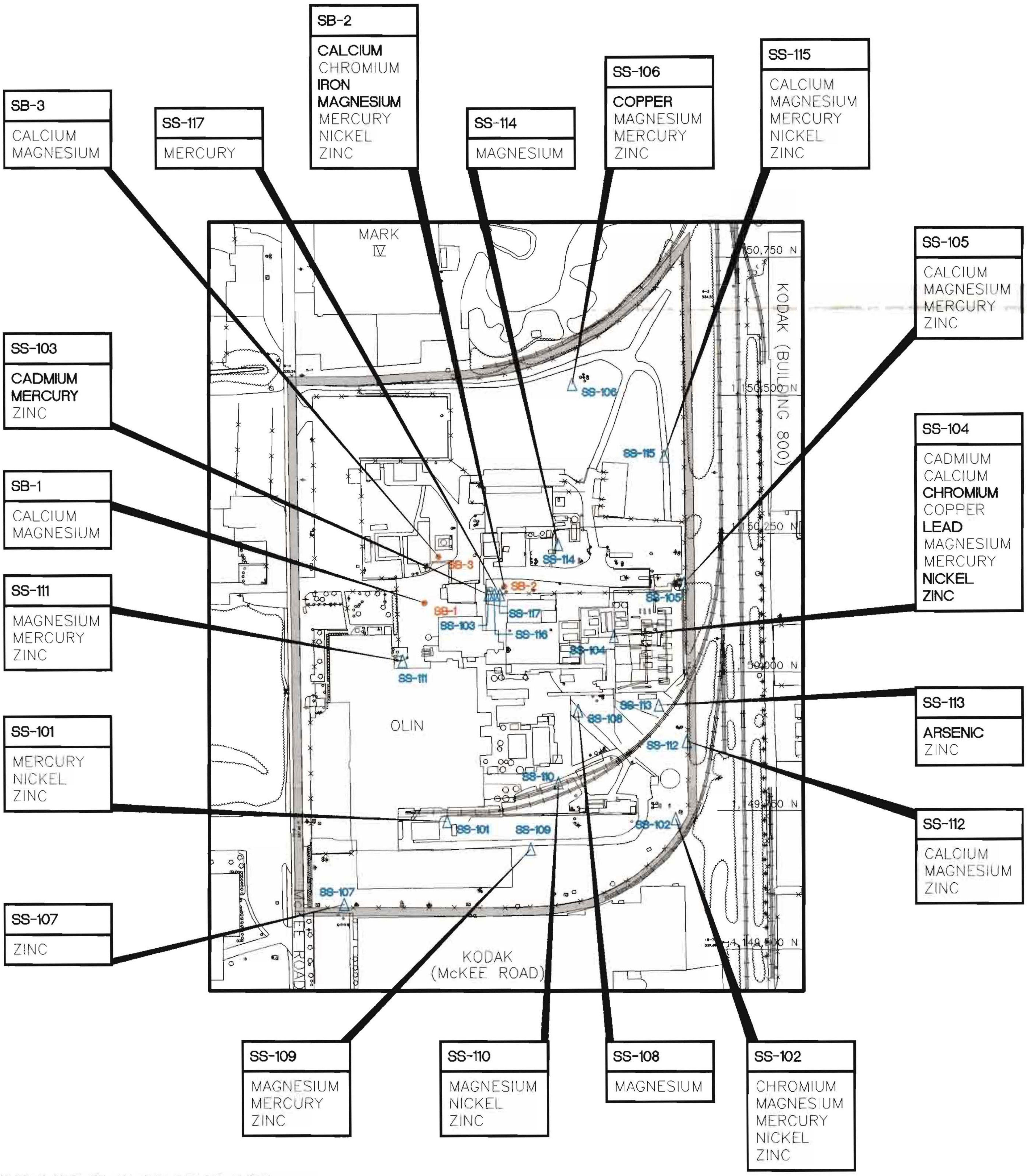
-	NOT DETECTED
VOC	VOLATILE ORGANIC COMPOUND
SVOC	SEMIVOLATILE ORGANIC COMPOUND
SB-1 ●	BORING NUMBER; D=DUPLICATE SAMPLE
NA	NOT ANALYZED
J	ESTIMATED; SEE APPENDIX B FOR EXPLANATION OF DATA QUALIFIERS

ALL CONCENTRATIONS REPORTED IN UNITS OF MICROGRAMS PER LITER

FIGURE 2-3

VOCs AND SVOCs IN
GROUNDWATER
LAB SAMPLE AREA

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.



NOTE:

INORGANICS LABELED IN BOLDFACE TYPE
INDICATE LOCATION OF MAXIMUM CONCENTRATION.

LEGEND

△ SURFACE SOIL SAMPLE LOCATION

● SOIL BORING LOCATION

▨ OUTLINE OF OLIN PROPERTY BOUNDARY

NOTE:

SEE TABLE 2-2 FOR LISTING OF
SPECIFIC BACKGROUND CONCENTRATIONS.

0 100 200 400 FEET

SCALE: 1"=200'

FIGURE 2-4

SOIL INORGANICS IN EXCEEDANCE OF BACKGROUND SURFACE SOIL CONCENTRATIONS

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.

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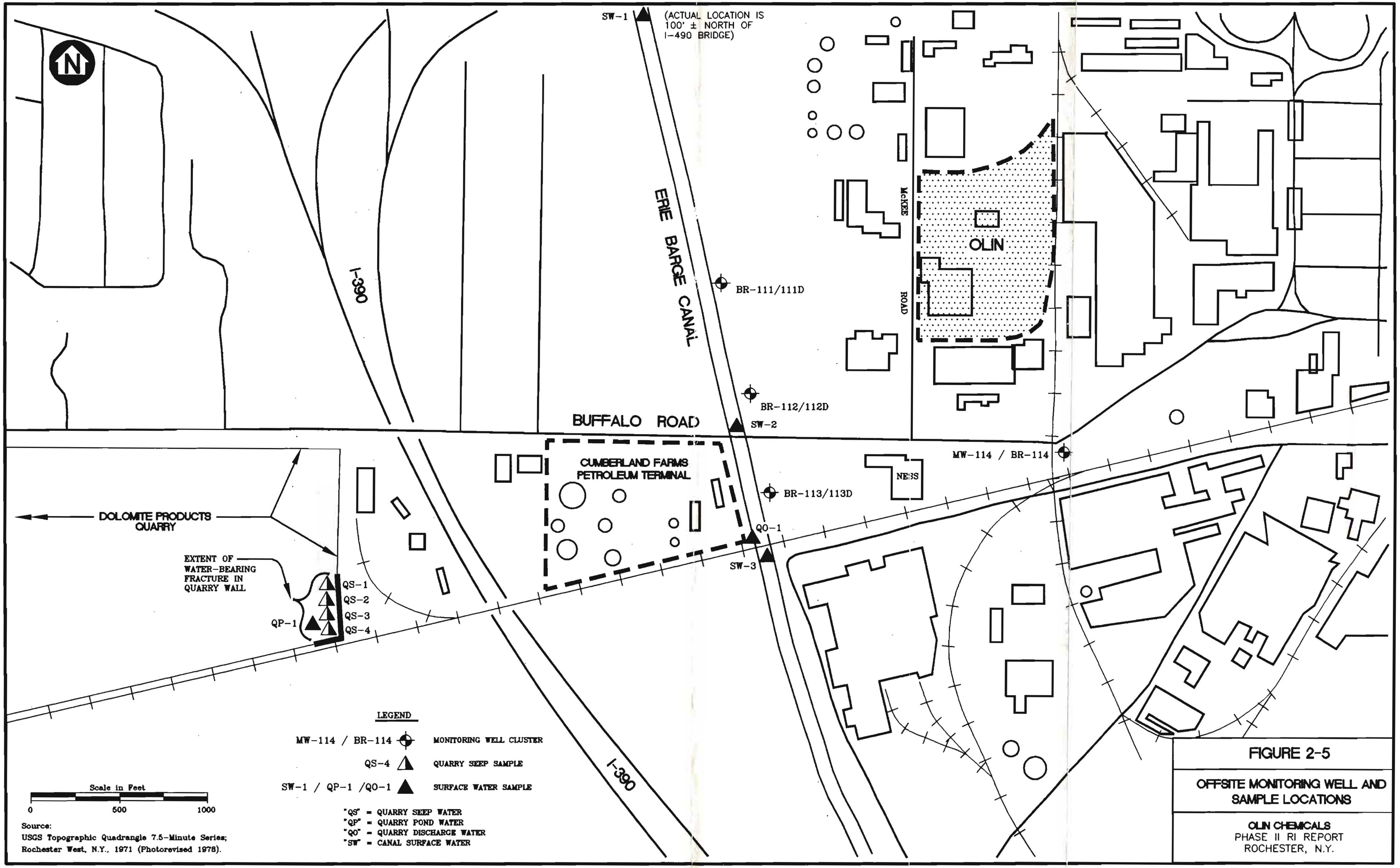


Source:
USGS Topographic Quadrangle 7.5-Minute Series;
Rochester West, N.Y., 1971 (Photorevised 1978).

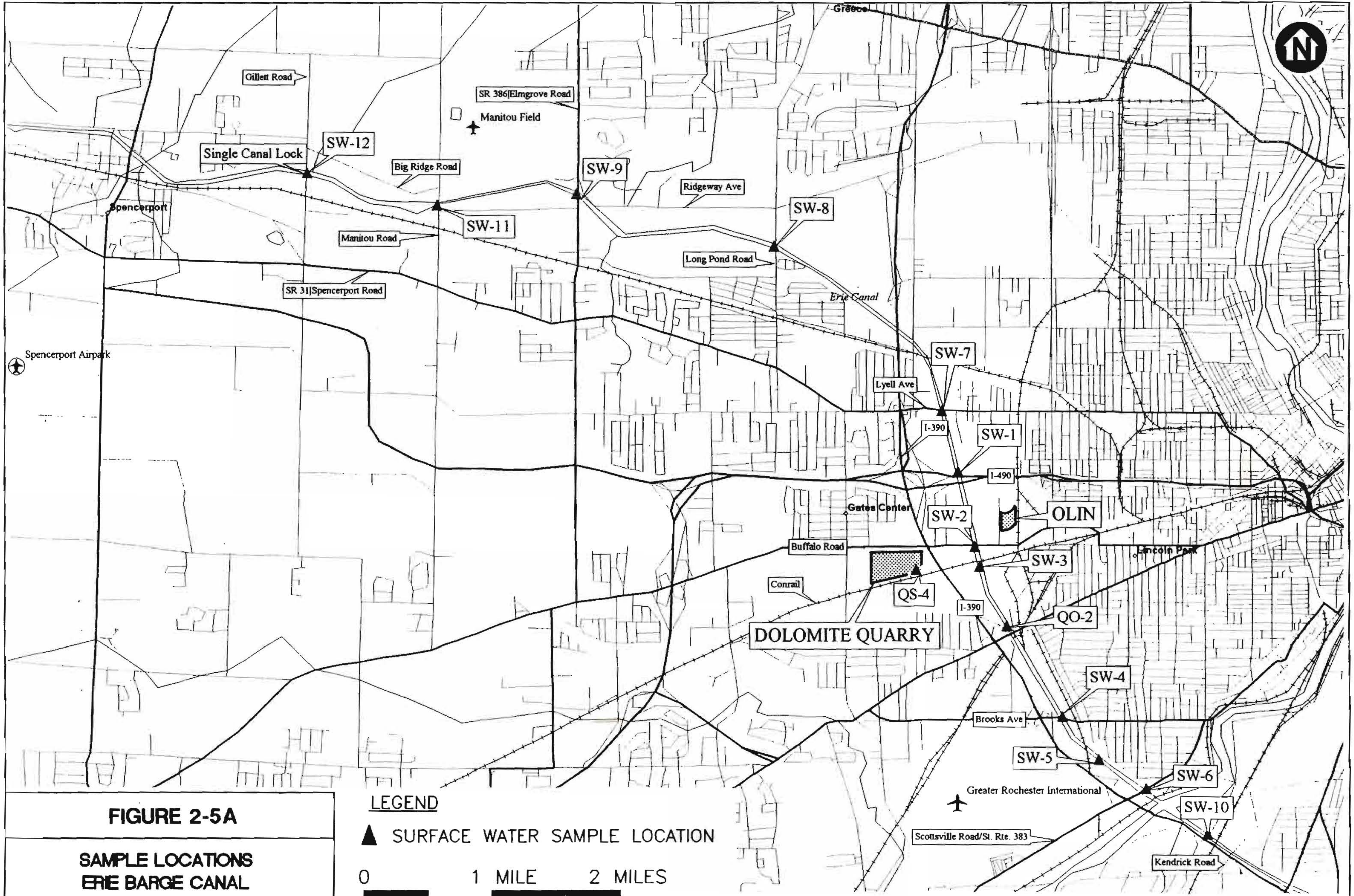
FIGURE 2-5

OFFSITE MONITORING WELL AND SAMPLE LOCATIONS

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.

T:\7311-34\023D-1.DWG 3-26-96

T:\7311-34\7311F501. C&P 4/1/97



LEGEND

▲ SURFACE WATER SAMPLE LOCATION

0 1 MILE 2 MILES

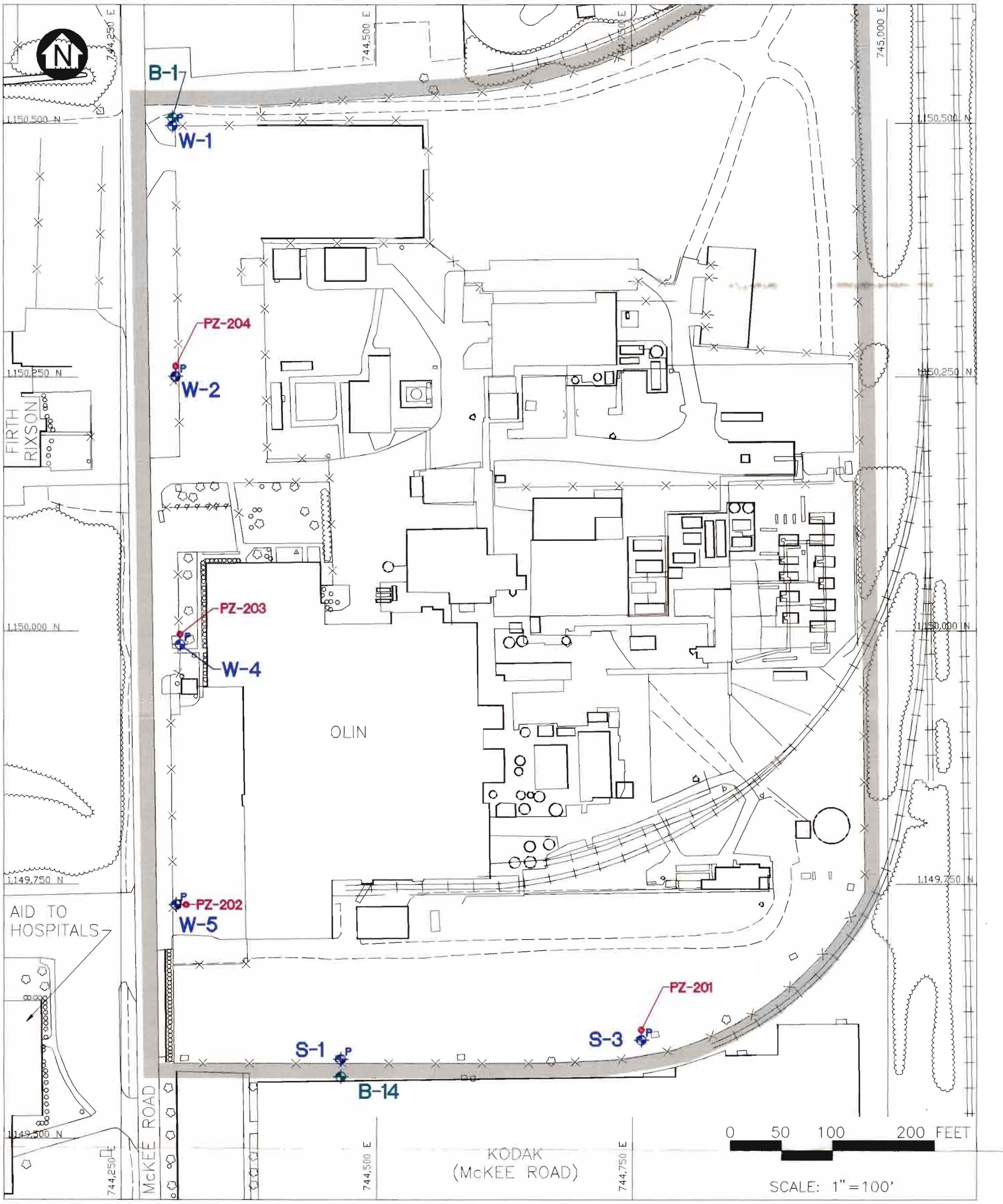
SCALE: 1"=1 MILE

SOURCE: DeLorme Mapping 1995

FIGURE 2-5A

SAMPLE LOCATIONS
ERIE BARGE CANAL

OLIN CHEMICALS
ROCHESTER, N.Y.



LEGEND

● TEMPORARY OVERBURDEN PIEZOMETER

◆ OVERBURDEN PIEZOMETER

◆P OVERBURDEN PUMPING WELL

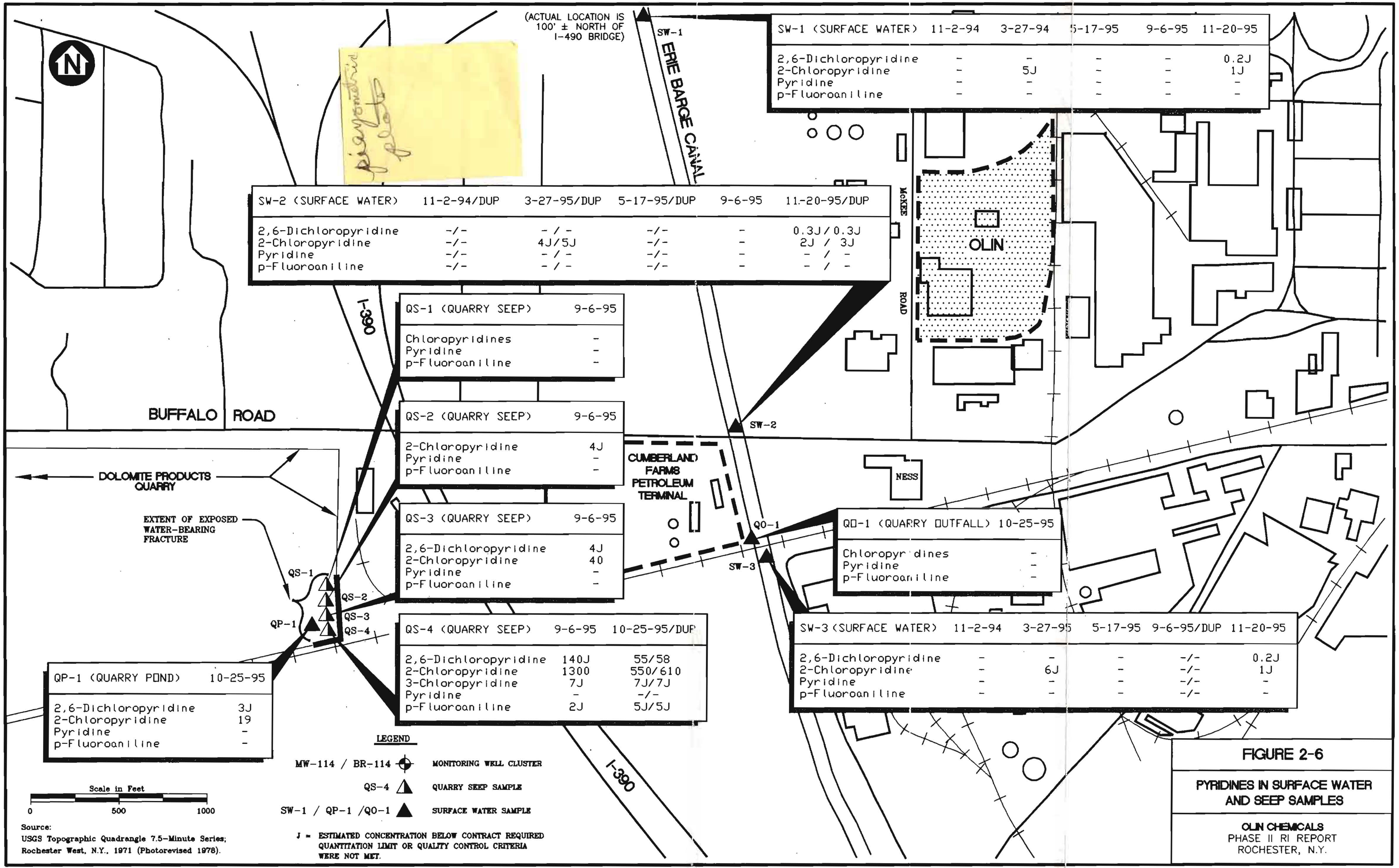
▨ OUTLINE OF OLIN PROPERTY BOUNDARY

FIGURE 2-7

OVERBURDEN WELL LOCATIONS FOR AQUIFER TESTING

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.

T:\7311-34\7311f104.DWG 3-26-96



piezometric plot

SW-1 (SURFACE WATER)	11-2-94	3-27-94	5-17-95	9-6-95	11-20-95
2,6-Dichloropyridine	-	-	-	-	0.2J
2-Chloropyridine	-	5J	-	-	1J
Pyridine	-	-	-	-	-
p-Fluoroaniline	-	-	-	-	-

SW-2 (SURFACE WATER)	11-2-94/DUP	3-27-95/DUP	5-17-95/DUP	9-6-95	11-20-95/DUP
2,6-Dichloropyridine	-/-	- / -	-/-	-	0.3J/0.3J
2-Chloropyridine	-/-	4J/5J	-/-	-	2J / 3J
Pyridine	-/-	- / -	-/-	-	- / -
p-Fluoroaniline	-/-	- / -	-/-	-	- / -

QS-1 (QUARRY SEEP)	9-6-95
Chloropyridines	-
Pyridine	-
p-Fluoroaniline	-

QS-2 (QUARRY SEEP)	9-6-95
2-Chloropyridine	4J
Pyridine	-
p-Fluoroaniline	-

QS-3 (QUARRY SEEP)	9-6-95
2,6-Dichloropyridine	4J
2-Chloropyridine	40
Pyridine	-
p-Fluoroaniline	-

QO-1 (QUARRY OUTFALL)	10-25-95
Chloropyridines	-
Pyridine	-
p-Fluoroaniline	-

QS-4 (QUARRY SEEP)	9-6-95	10-25-95/DUP
2,6-Dichloropyridine	140J	55/58
2-Chloropyridine	1300	550/610
3-Chloropyridine	7J	7J/7J
Pyridine	-	-/-
p-Fluoroaniline	2J	5J/5J

QP-1 (QUARRY POND)	10-25-95
2,6-Dichloropyridine	3J
2-Chloropyridine	19
Pyridine	-
p-Fluoroaniline	-

SW-3 (SURFACE WATER)	11-2-94	3-27-95	5-17-95	9-6-95/DUP	11-20-95
2,6-Dichloropyridine	-	-	-	-/-	0.2J
2-Chloropyridine	-	6J	-	-/-	1J
Pyridine	-	-	-	-/-	-
p-Fluoroaniline	-	-	-	-/-	-

LEGEND

MW-114 / BR-114 ⊕	MONITORING WELL CLUSTER
QS-4 △	QUARRY SEEP SAMPLE
SW-1 / QP-1 /QO-1 ▲	SURFACE WATER SAMPLE

J = ESTIMATED CONCENTRATION BELOW CONTRACT REQUIRED QUANTITATION LIMIT OR QUALITY CONTROL CRITERIA WERE NOT MET.

Source:
USGS Topographic Quadrangle 7.5-Minute Series;
Rochester West, N.Y., 1971 (Photorevised 1978).

FIGURE 2-6

PYRIDINES IN SURFACE WATER AND SEEP SAMPLES

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.

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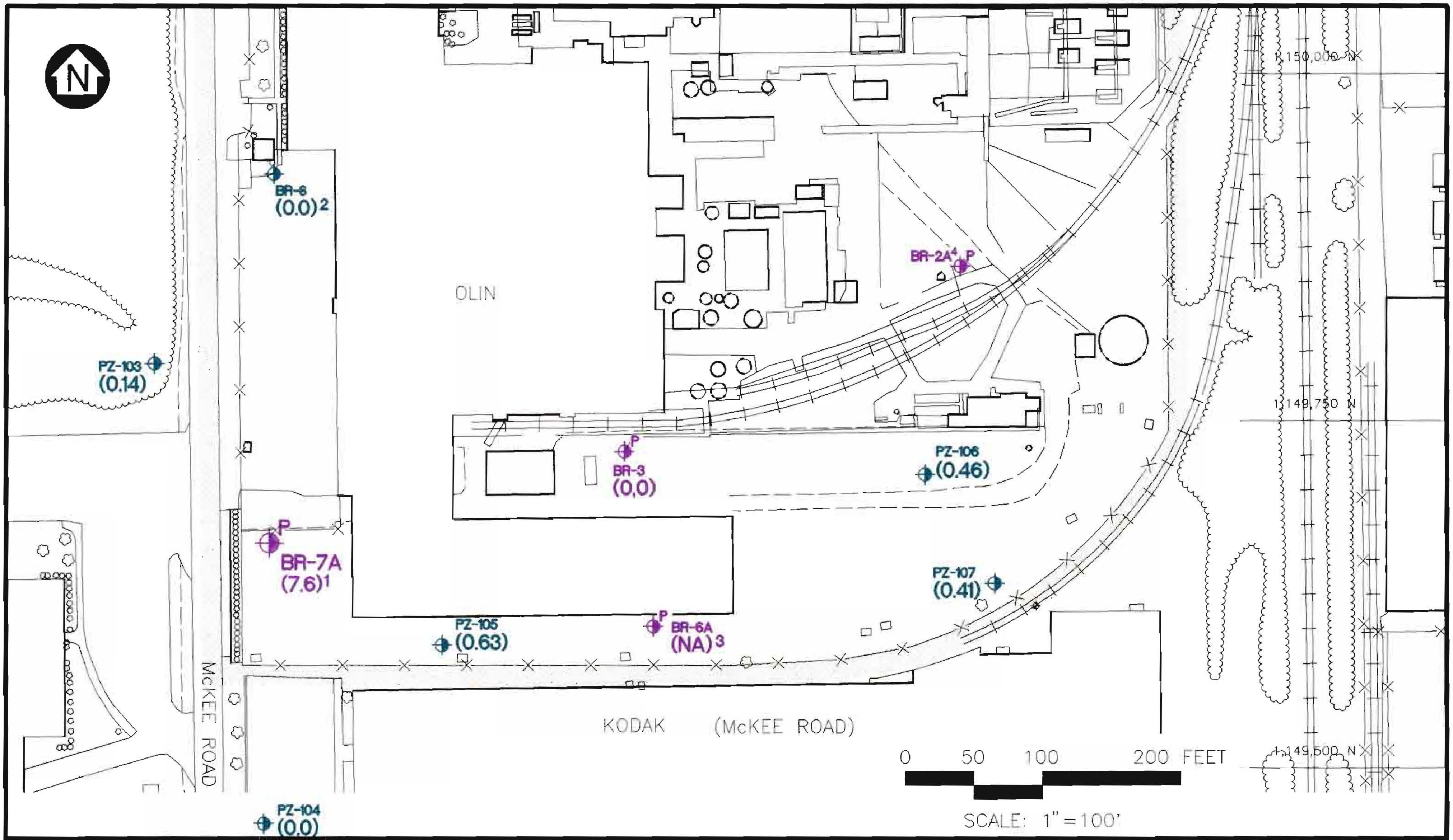
T:\7311-34\7311F401 3-27-96



LEGEND

⊕	BEDROCK PIEZOMETER/MONITORING WELL
⊕P	BEDROCK PUMPING WELL
(7.6)	MAXIMUM INTERPRETED DRAWDOWN
—x—x— ▨	OUTLINE OF OLIN PROPERTY BOUNDARY

NOTES

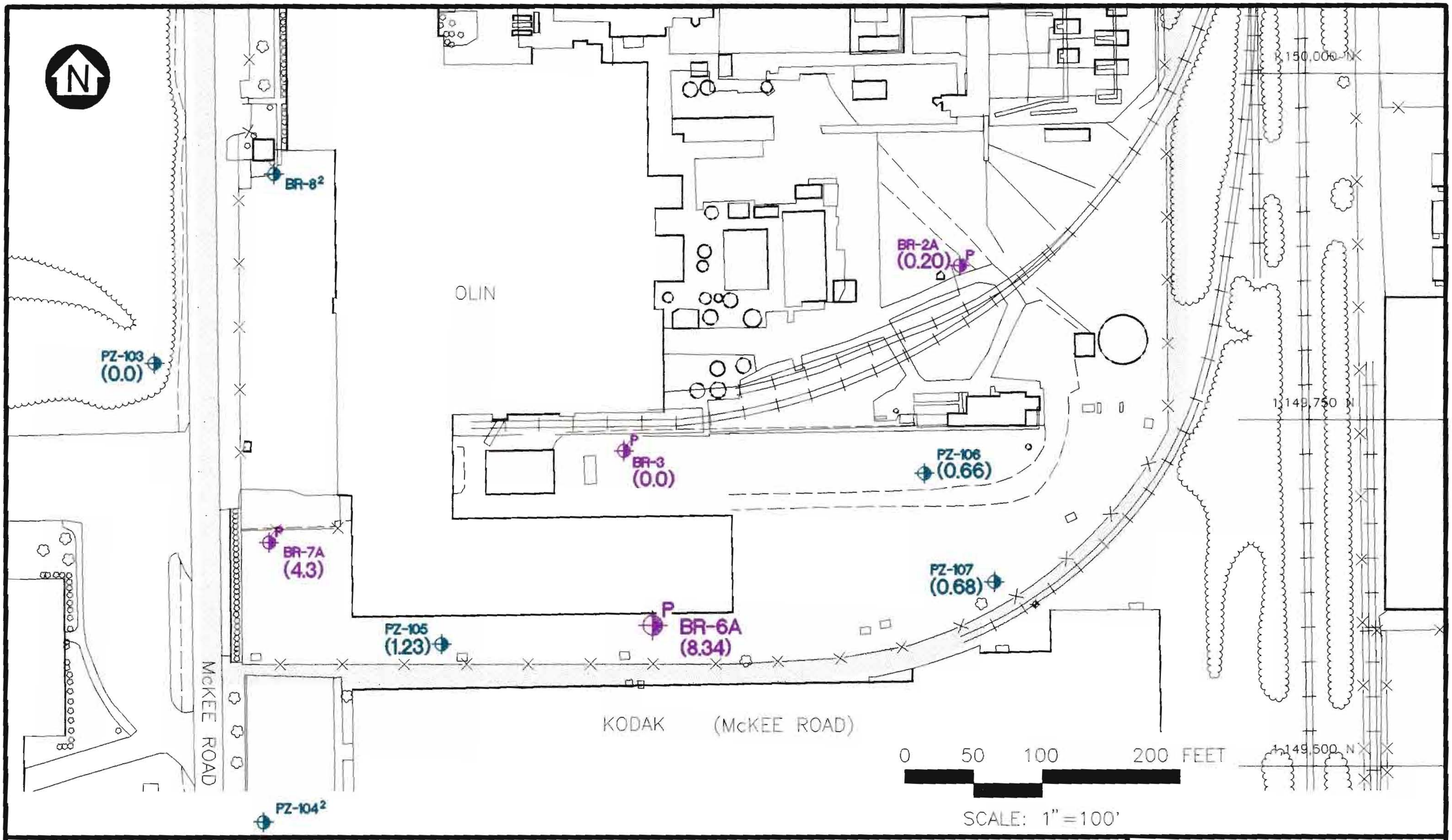
1. BR–7A PUMPED AT 6.1 gpm FOR 1090 MINUTES BEGINNING 11/16/95.
2. BR–8 NO HEAD CHANGE DURING RECOVERY PHASE (NOT MEASURED DURING PUMPING)
3. BR–6A NOT INSTALLED AT TIME OF TEST.
4. NOT MEASURED.

FIGURE 2-8

BR–7A PUMPING TEST INTERPRETED MAXIMUM DRAWDOWNS

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.

T:\7311-34\7311F400 3-27-96



LEGEND

⊕	BEDROCK PIEZOMETER/MONITORING WELL
⊕ᴾ	BEDROCK PUMPING WELL
(4.3)	MAXIMUM INTERPRETED DRAWDOWN
—×—×—	OUTLINE OF OLIN PROPERTY BOUNDARY

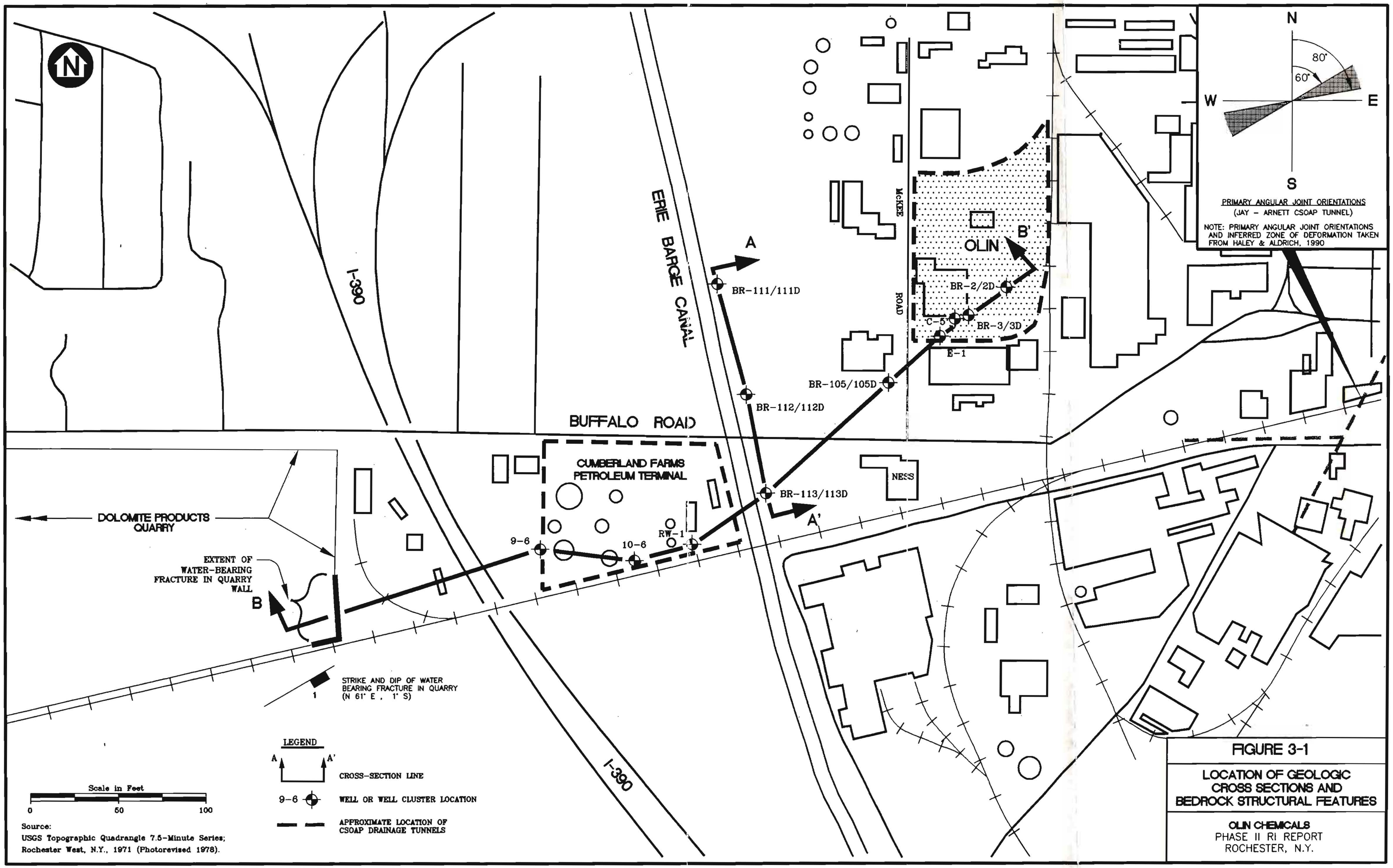
NOTES

1. BR-6A PUMPED AT 20.1 gpm FOR 1085 MINUTES BEGINNING 11/16/95.
2. NOT MEASURED.

FIGURE 2-9

BR-6A PUMPING TEST INTERPRETED MAXIMUM DRAWDOWNS

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.



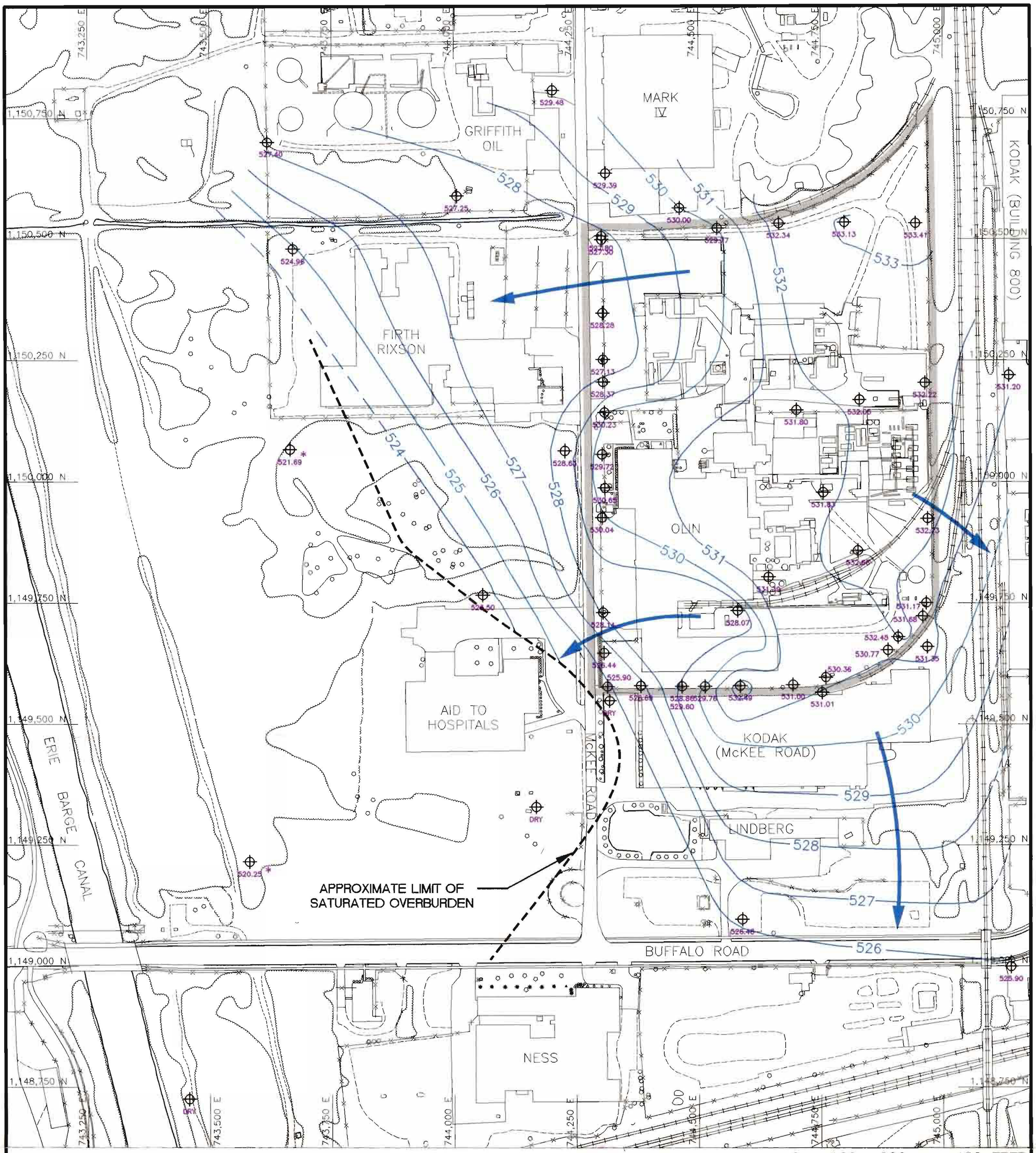
Source:
USGS Topographic Quadrangle 7.5-Minute Series;
Rochester West, N.Y., 1971 (Photorevised 1978).

FIGURE 3-1

LOCATION OF GEOLOGIC CROSS SECTIONS AND BEDROCK STRUCTURAL FEATURES

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.

T:\7311-34\023D.DWG 2-30-96



LEGEND

▨	OUTLINE OF OLIN PROPERTY BOUNDARY
—525—	OVERBURDEN GROUNDWATER ELEVATION CONTOUR (MSL) 5' INTERVAL
—526—	OVERBURDEN GROUNDWATER ELEVATION CONTOUR (MSL) 1' INTERVAL
⊕ 525.76	PIEZOMETRIC ELEVATION AT WELL OR PIEZOMETER (MSL)
⊕ 520.25*	PIEZOMETRIC ELEVATION AT WELL SCREENED IN OVERBURDEN AND SHALLOW BEDROCK; WATER LEVEL IS BELOW BEDROCK SURFACE.
←	INTERPRETED GROUNDWATER FLOW DIRECTION

NOTE:

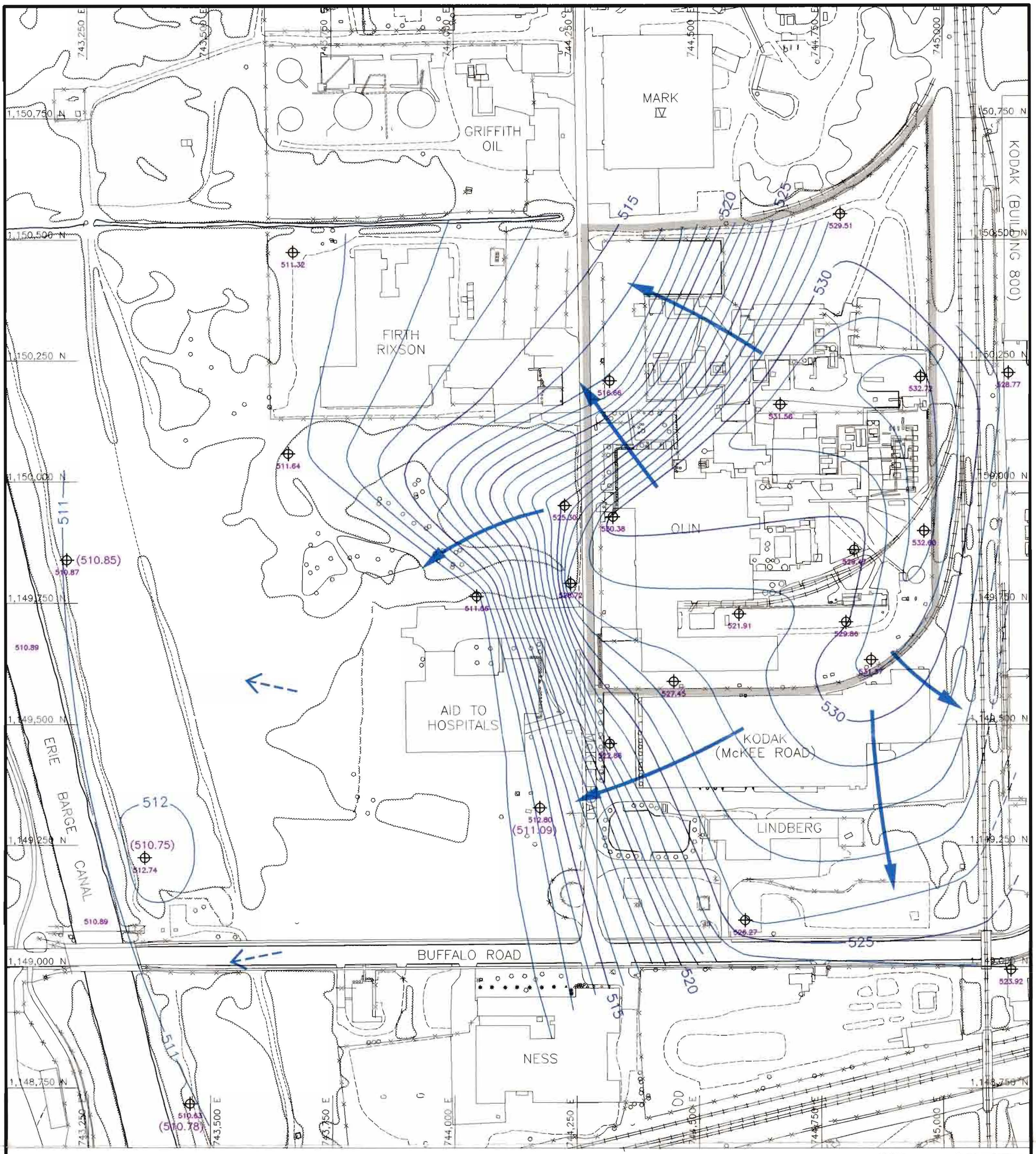
WATER LEVELS MEASURED ON NOVEMBER 6-8,1995; PUMPING WELLS SHUT DOWN ON SEPTEMBER 20, 1995

FIGURE 3-4

NOVEMBER 1995 OVERBURDEN GROUNDWATER INTERPRETED PIEZOMETRIC CONTOURS - NON-PUMPING CONDITION

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.

T:\7311-34\7311F101 3-22-96



LEGEND

▨ OUTLINE OF OLIN PROPERTY BOUNDARY

—525— BEDROCK GROUNDWATER ELEVATION CONTOUR (MSL) 5' INTERVAL

——— BEDROCK GROUNDWATER ELEVATION CONTOUR (MSL) 1' INTERVAL

⊕ 525.76 PIEZOMETRIC ELEVATION AT WELL OR PIEZOMETER (MSL)

◀—— INTERPRETED GROUNDWATER FLOW DIRECTION (SHALLOW BEDROCK SYSTEM)

(510.75) PIEZOMETRIC ELEVATION AT DEEP BEDROCK WELL

←--- INFERRED GROUNDWATER FLOW DIRECTION (DEEP BEDROCK SYSTEM)

NOTE:

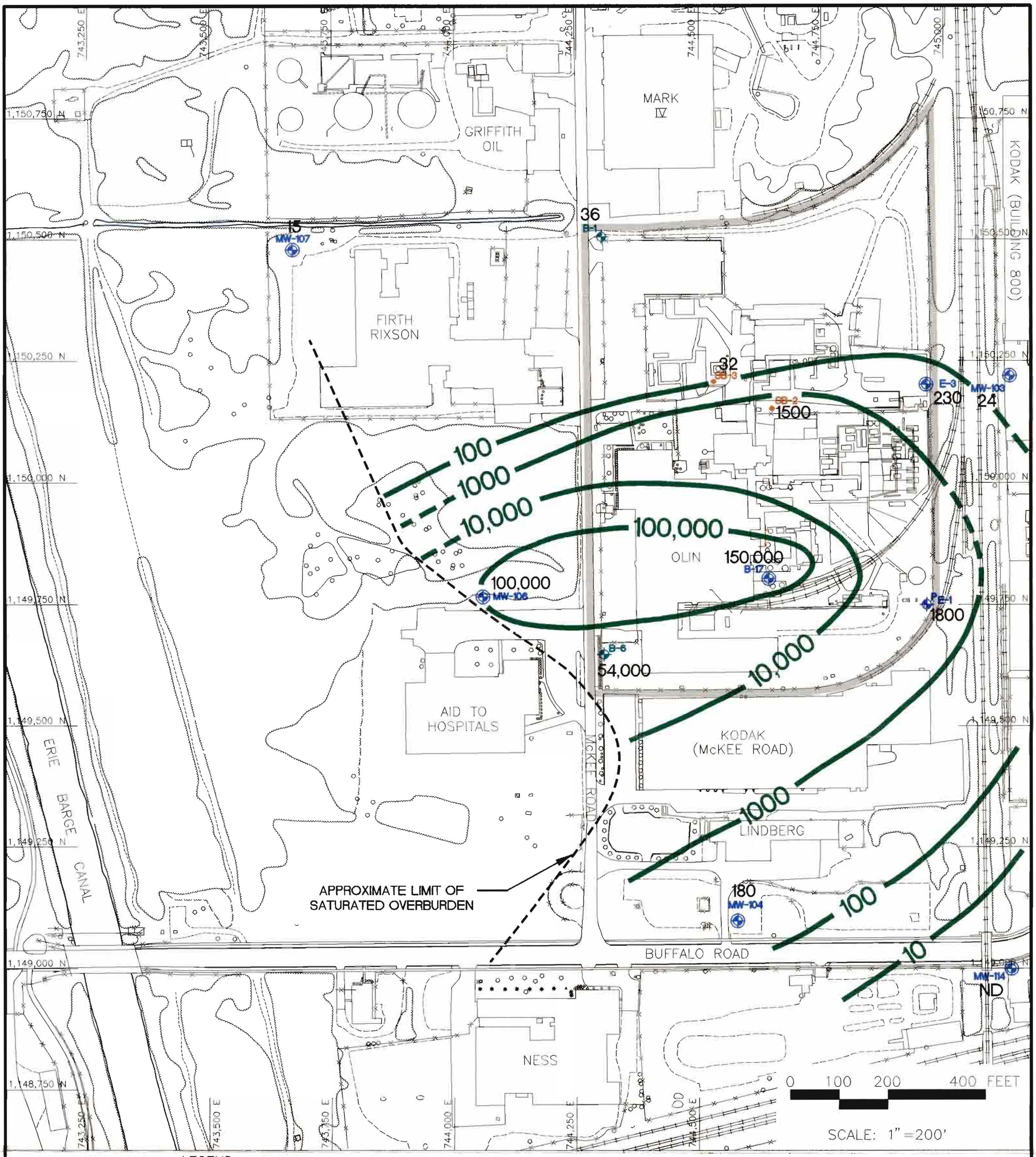
WATER LEVELS MEASURED ON NOVEMBER 6-8,1995; PUMPING WELLS SHUT DOWN ON SEPTEMBER 20, 1995

FIGURE 3-5

NOVEMBER 1995 BEDROCK GROUNDWATER INTERPRETED PIEZOMETRIC CONTOURS - NON-PUMPING CONDITION

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.

T:\7311-34\7311F100 3-22-96



LEGEND

⊕	OVERBURDEN PIEZOMETER
⊕	OVERBURDEN MONITORING WELL
⊕P	OVERBURDEN PUMPING WELL
•	TERRAPROBE BORING
▬×▬	OUTLINE OF OLIN PROPERTY BOUNDARY
—10—	OVERBURDEN GROUNDWATER ISOPLETH
180	DEEP BEDROCK WELL CONCENTRATION TOTAL

NOTES: RESULTS REPORTED AS SUM OF CONCENTRATIONS IN MICROGRAMS PER LITER FOR THE FOLLOWING CHEMICALS:

2,6-DICHLOROPYRIDINE
2-CHLOROPYRIDINE
3-CHLOROPYRIDINE
P-FLOUROANILINE

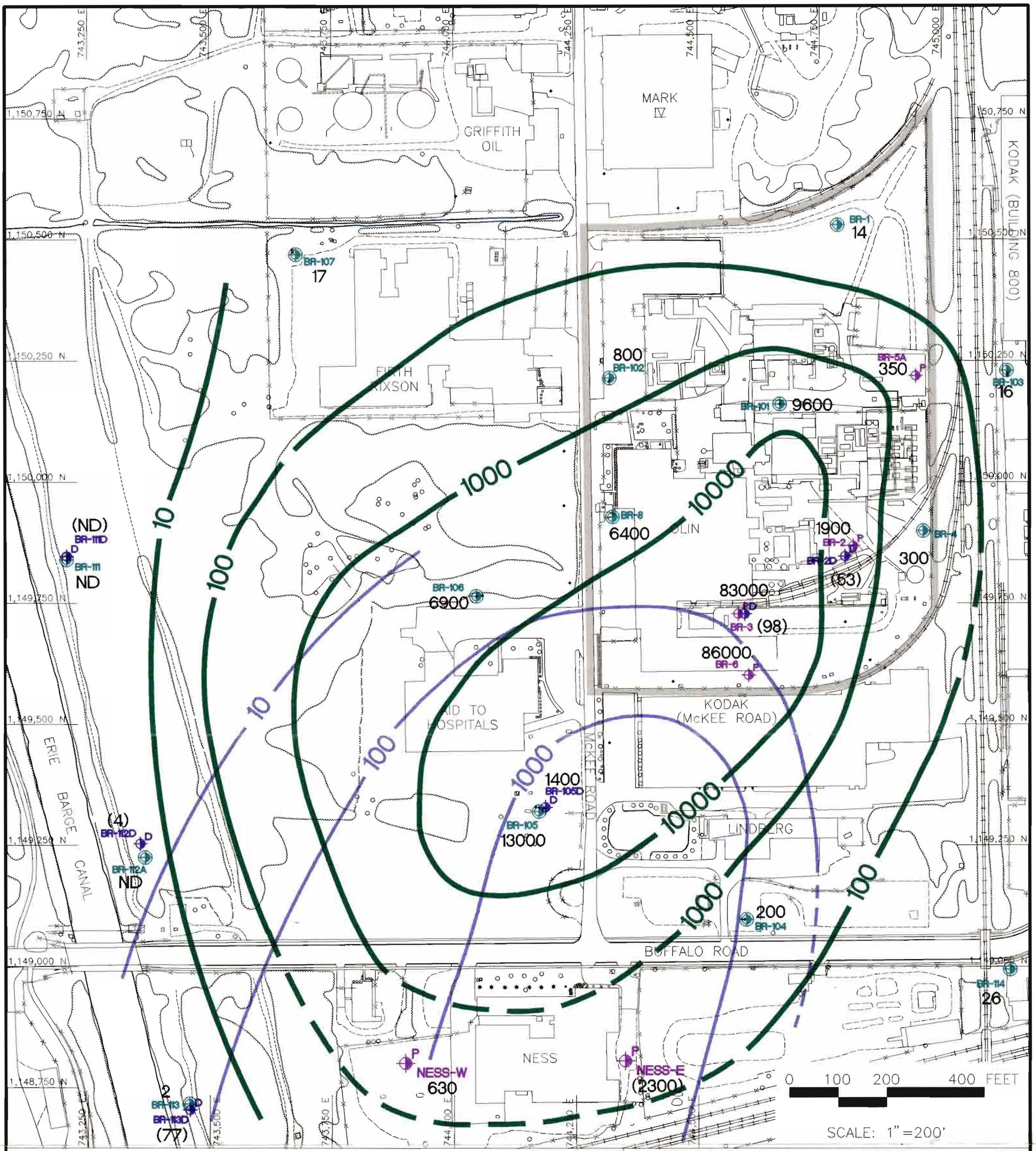
NOTE: DATA FROM SAMPLES COLLECTED FROM 9-95 TO 11-95 USED FOR THIS INTERPRETION.

FIGURE 3-6

TOTAL PYRIDINES INTERPRETED CONCENTRATION ISOPLETHS FOR OVERBURDEN GROUNDWATER

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.

T:\7311-34\7311F301 2-29-96



LEGEND

⊕	BEDROCK PIEZOMETER
⊕	BEDROCK MONITORING WELL
⊕D	DEEP BEDROCK MONITORING WELL
⊕P	BEDROCK PUMPING WELL
▨	OUTLINE OF OLIN PROPERTY BOUNDARY
—10—	SHALLOW BEDROCK ISOPLETH
—10—	DEEP BEDROCK ISOPLETH
(77)	DEEP BEDROCK WELL CONCENTRATION TOTAL

NOTES: RESULTS REPORTED AS SUM OF CONCENTRATIONS IN MICROGRAMS PER LITER FOR THE FOLLOWING CHEMICALS:

2,6-DICHLOROPYRIDINE	P-FLUOROANILINE
2-CHLOROPYRIDINE	
3-CHLOROPYRIDINE	

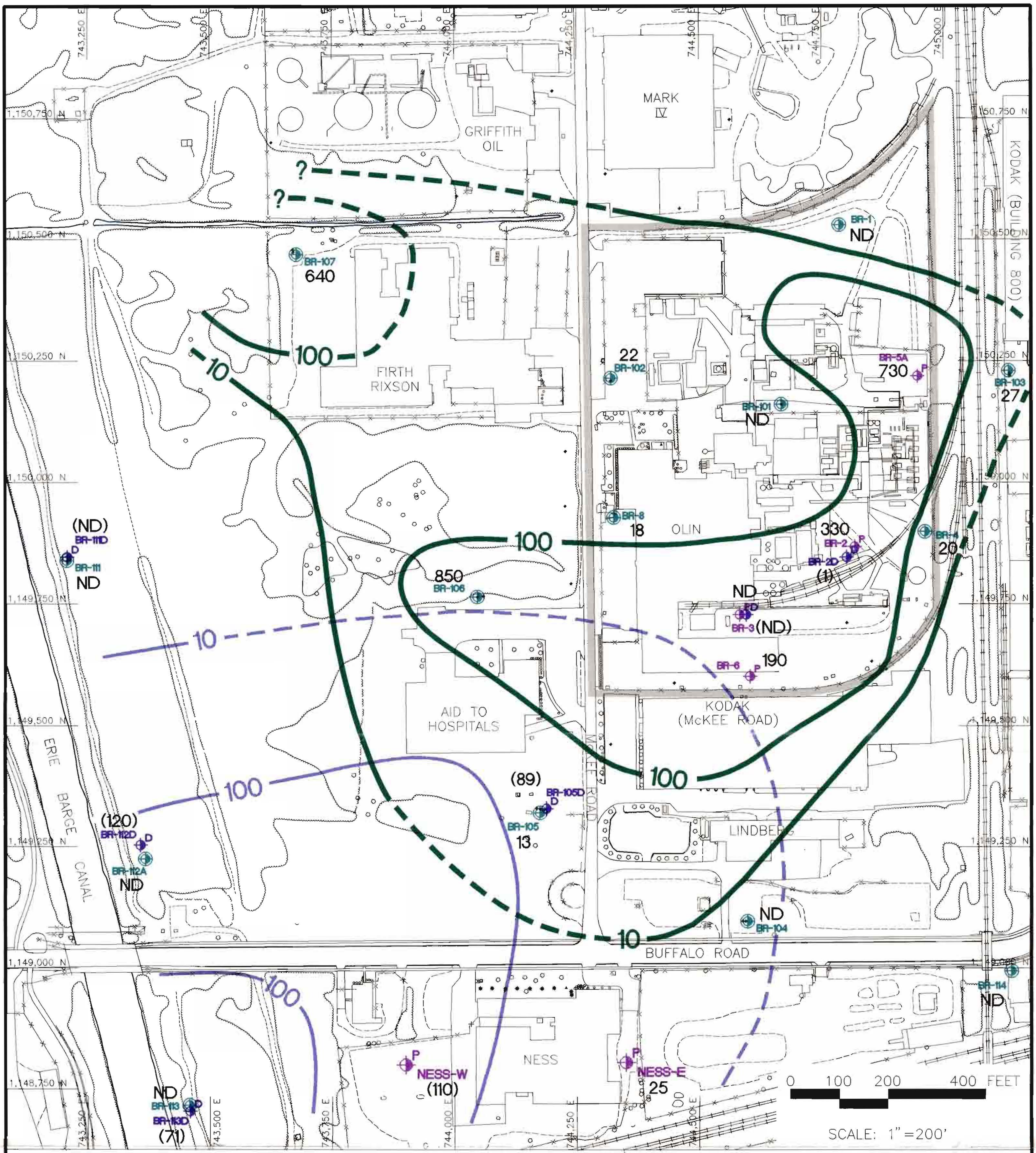
NOTE: DATA FOR BEDROCK WELLS AND PIEZOMETERS SAMPLED FROM 9/95 TO 11/95 USED FOR THIS INTERPRETATION.

FIGURE 3-7

TOTAL PYRIDINES INTERPRETED CONCENTRATION ISOPLETHS FOR DEEP AND SHALLOW BEDROCK GROUNDWATER

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.

T:\7311-34\7311F300 4-19-96



LEGEND

⊕	BEDROCK PIEZOMETER
⊕	BEDROCK MONITORING WELL
⊕D	DEEP BEDROCK MONITORING WELL
⊕P	BEDROCK PUMPING WELL
▬×▬	OUTLINE OF OLIN PROPERTY BOUNDARY
—10—	SHALLOW BEDROCK ISOPLETH
—10—	DEEP BEDROCK ISOPLETH
(26)	DEEP BEDROCK WELL CONCENTRATION TOTAL

NOTES: RESULTS REPORTED AS SUM OF CONCENTRATIONS IN MICROGRAMS PER LITER FOR THE FOLLOWING CHEMICALS:

1,2-DICHLOROETHENE (TOTAL)
TETRACHLOROETHENE
TRICHLOROETHENE
VINYL CHLORIDE

NOTE: DATA FOR BEDROCK WELLS AND PIEZOMETERS SAMPLED FROM 9/95 TO 11/95 USED FOR THIS INTERPRETATION.

FIGURE 3-11

TOTAL CHLORINATED ETHENES INTERPRETED CONCENTRATION ISOPLETHS FOR DEEP AND SHALLOW BEDROCK GROUNDWATER

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.

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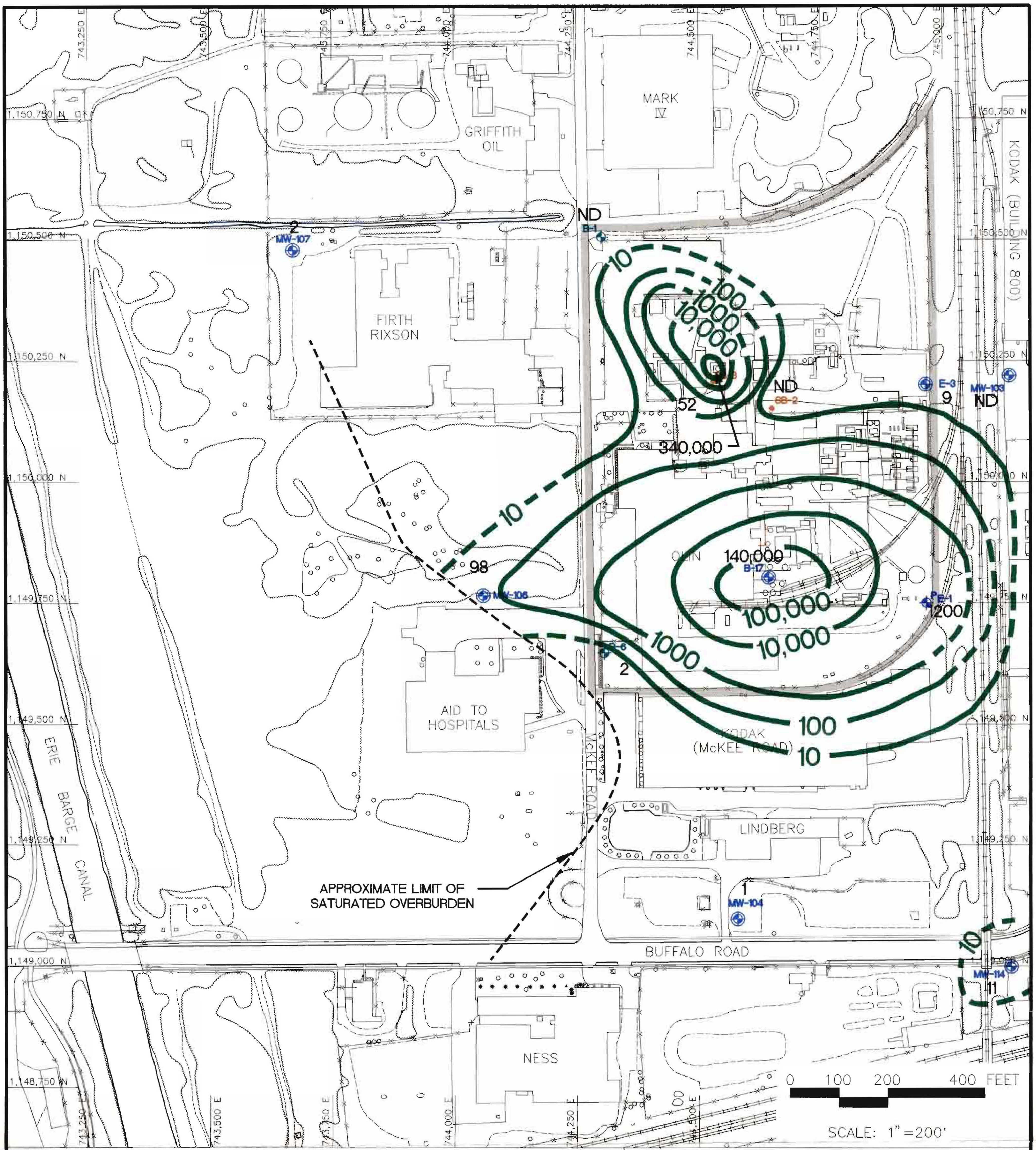

LEGEND

⊕	OVERBURDEN PIEZOMETER
⊕	OVERBURDEN MONITORING WELL
⊕P	OVERBURDEN PUMPING WELL
•	TERRAPROBE BORING
▨	OUTLINE OF OLIN PROPERTY BOUNDARY
—10—	OVERBURDEN GROUNDWATER ISOPLETH
9	CONCENTRATION TOTAL

NOTES: RESULTS REPORTED AS SUM OF CONCENTRATIONS IN MICROGRAMS PER LITER FOR THE FOLLOWING CHEMICALS:

1,2-DICHLOROETHENE (TOTAL)	CHLOROFORM
1,1,1-TRICHLOROETHANE	METHYLENE CHLORIDE
1,1-DICHLOROETHANE	TETRACHLOROETHENE
CARBON TETRACHLORIDE	TRICHLOROETHENE

NOTE: DATA FROM SAMPLES COLLECTED FROM 9-95 TO 11-95 USED FOR THIS INTERPRETION.

FIGURE 3-8

TOTAL SELECTED VOCS INTERPRETED CONCENTRATION ISOPLETHS FOR OVERBURDEN GROUNDWATER

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.

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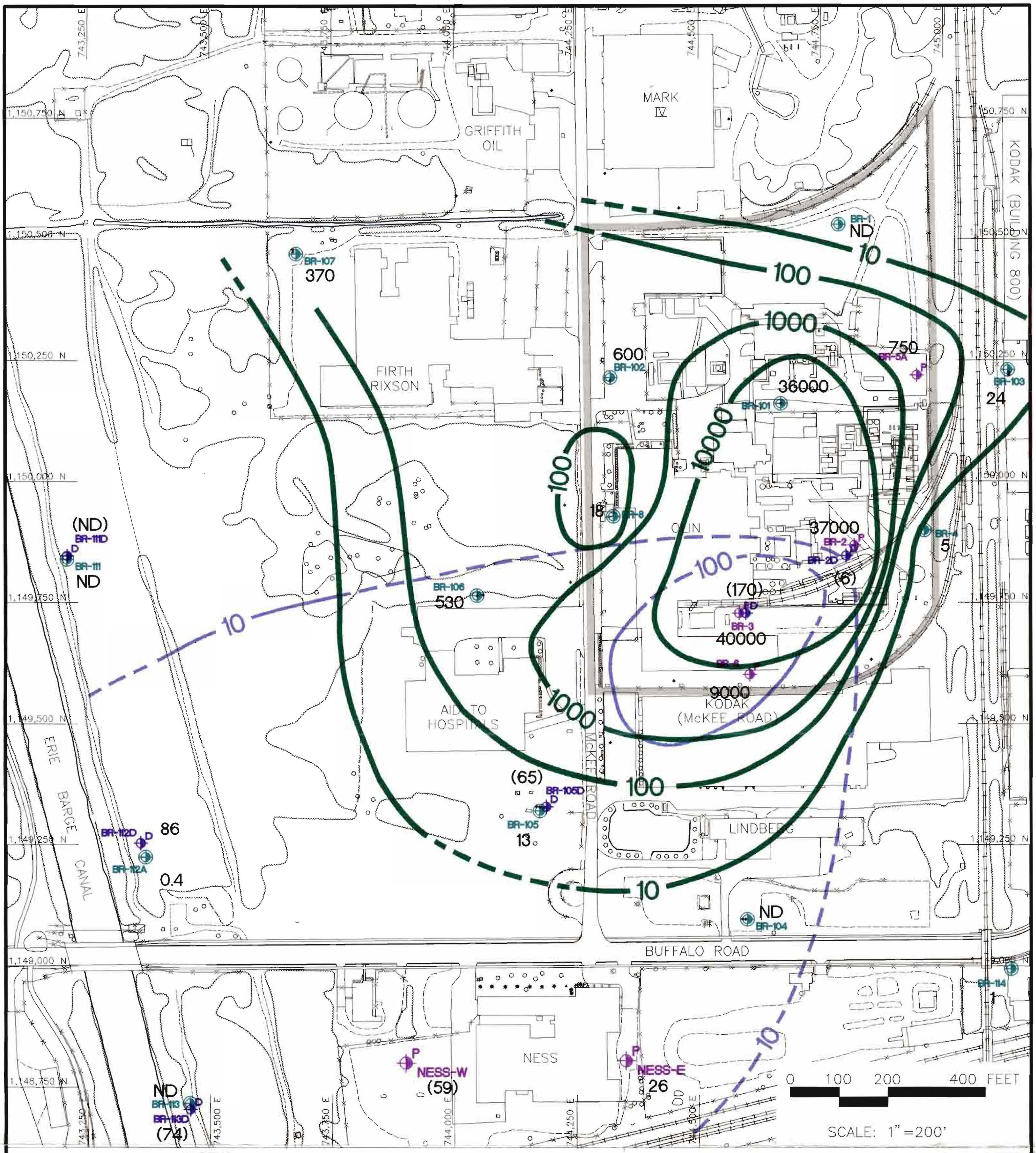

LEGEND

⊕ BEDROCK PIEZOMETER

⊕ BEDROCK MONITORING WELL

⊕D DEEP BEDROCK MONITORING WELL

⊕P BEDROCK PUMPING WELL

▨ OUTLINE OF OLIN PROPERTY BOUNDARY

—10— SHALLOW BEDROCK ISOPLETH

—10— DEEP BEDROCK ISOPLETH

(74) DEEP BEDROCK WELL CONCENTRATION TOTAL

NOTES: RESULTS REPORTED AS SUM OF CONCENTRATIONS IN MICROGRAMS PER LITER FOR THE FOLLOWING CHEMICALS:

1,2-DICHLOROETHENE (TOTAL)	CHLOROFORM
1,1,1-TRICHLOROETHANE	METHYLENE CHLORIDE
1,1-DICHLOROETHANE	TETRACHLOROETHENE
CARBON TETRACHLORIDE	TRICHLOROETHENE

NOTE: DATA FOR BEDROCK WELLS AND PIEZOMETERS SAMPLED FROM 9/95 TO 11/95 USED FOR THIS INTERPRETATION.

FIGURE 3-9

TOTAL SELECTED VOCS INTERPRETED CONCENTRATION ISOPLETHS FOR SHALLOW AND DEEP BEDROCK GROUNDWATER

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.

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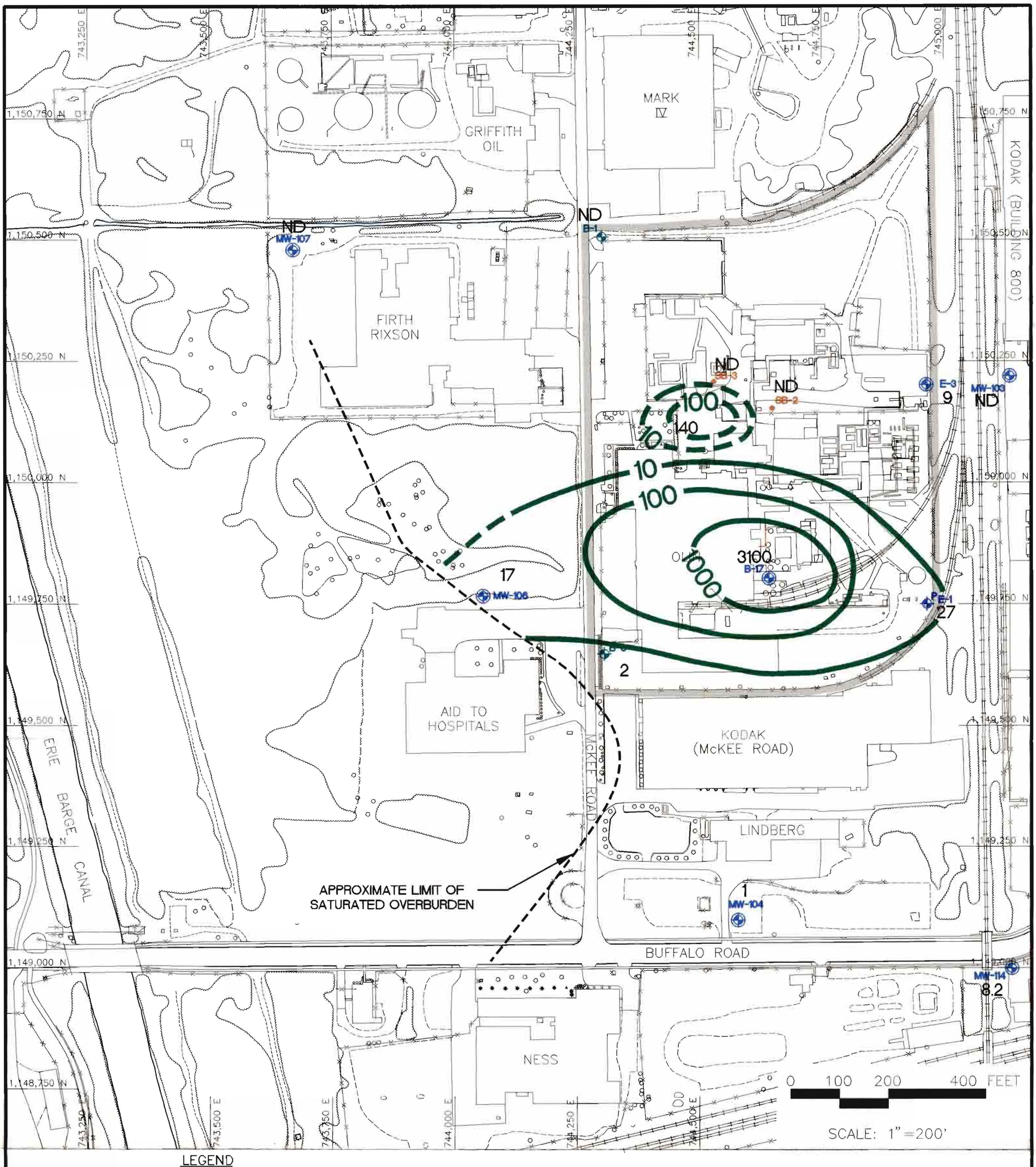

LEGEND

⊕	OVERBURDEN PIEZOMETER
⊕	OVERBURDEN MONITORING WELL
⊕P	OVERBURDEN PUMPING WELL
•	TERRAPROBE BORING
▨	OUTLINE OF OLIN PROPERTY BOUNDARY
—10—	OVERBURDEN GROUNDWATER ISOPLETH
(26)	DEEP BEDROCK WELL CONCENTRATION TOTAL

NOTES: RESULTS REPORTED AS SUM OF CONCENTRATIONS IN MICROGRAMS PER LITER FOR THE FOLLOWING CHEMICALS:

1,2–DICHLOROETHENE (TOTAL)
TETRACHLOROETHENE
TRICHLOROETHENE
VINYL CHLORIDE

NOTE: DATA FROM SAMPLES COLLECTED FROM 9–95 TO 11–95 USED FOR THIS INTERPRETION.

FIGURE 3-10

TOTAL CHLORINATED ETHENES INTERPRETED CONCENTRATION ISOPLETHS FOR OVERBURDEN GROUNDWATER

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.

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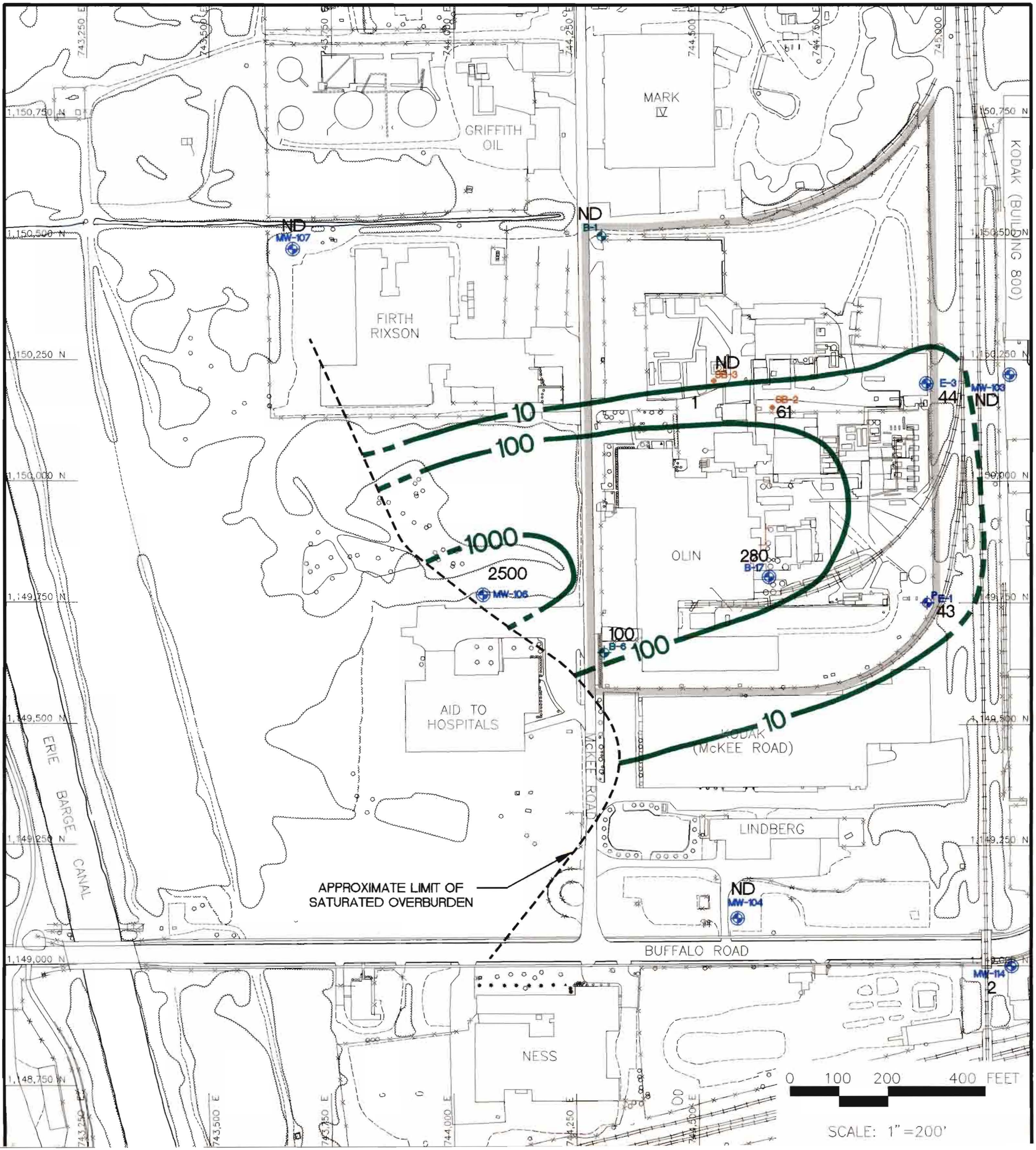
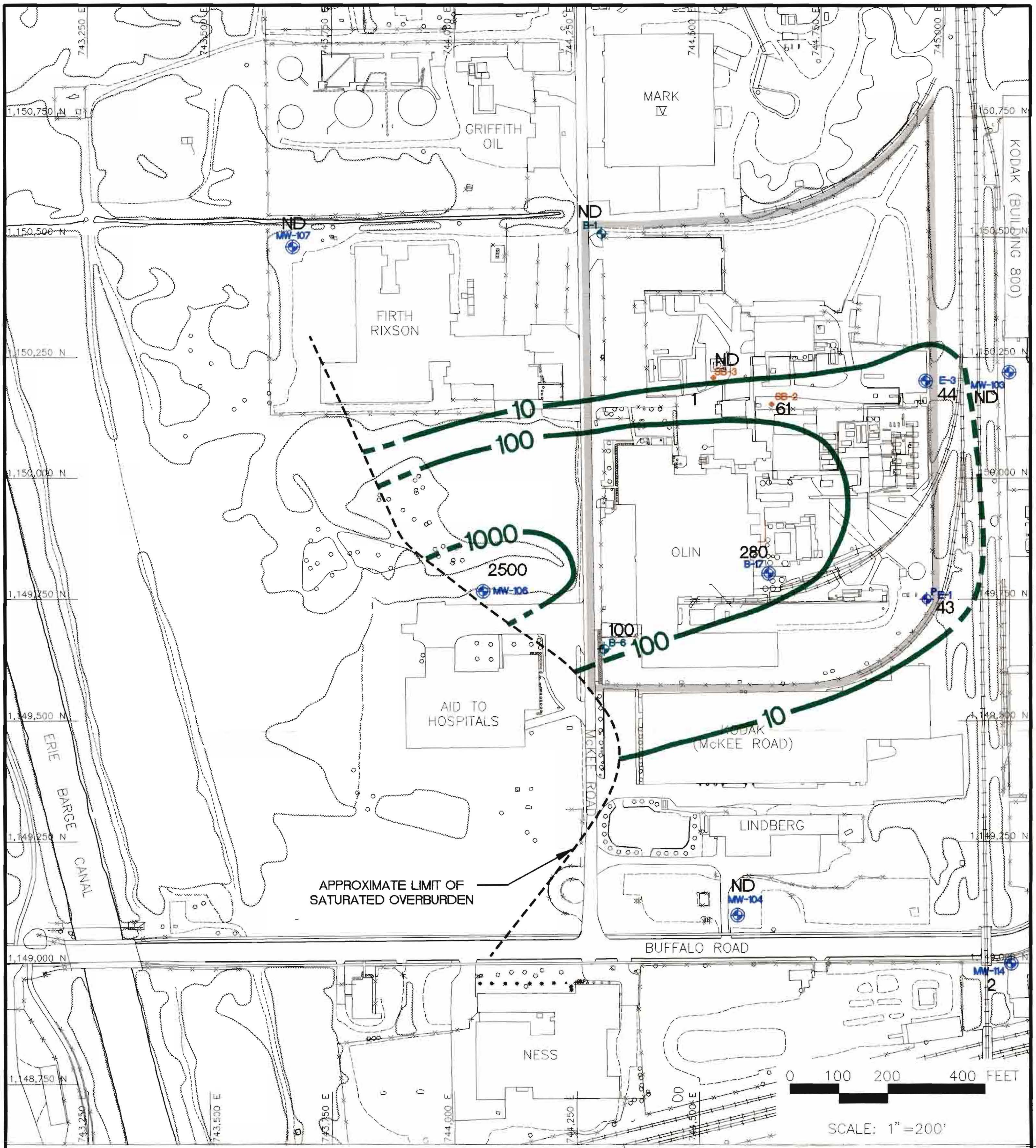

743,250 E
743,500 E
743,750 E
744,000 E
744,250 E
744,500 E
744,750 E
745,000 E
1,150,750 N
1,150,500 N
1,150,250 N
1,150,000 N
1,149,750 N
1,149,500 N
1,149,250 N
1,149,000 N
1,148,750 N
GRIFFITH OIL
MARK IV
KODAK (BUILDING 800)
FIRTH RIXSON
OLIN
AID TO HOSPITALS
KODAK (McKEE ROAD)
LINDBERG
McKEE ROAD
BUFFALO ROAD
NESS
ERIE BARGE CANAL
ND
MW-107
ND
B-1
ND
SB-3
1
SB-2
61
E-3
44
MW-103
ND
10
100
1000
2500
MW-106
280
B-17
PE-1
43
100
B-6
100
10
ND
MW-104
MW-114
2
APPROXIMATE LIMIT OF SATURATED OVERBURDEN
0
100
200
400 FEET
SCALE: 1"=200'



LEGEND

⊕	OVERBURDEN PIEZOMETER
⊕	OVERBURDEN MONITORING WELL
⊕ P	OVERBURDEN PUMPING WELL
•	TERRAPROBE BORING
▬	OUTLINE OF OLIN PROPERTY BOUNDARY
—10—	OVERBURDEN GROUNDWATER ISOPLETH
26	CONCENTRATION TOTAL

NOTES: RESULTS REPORTED AS SUM OF CONCENTRATIONS IN MICROGRAMS PER LITER FOR THE FOLLOWING CHEMICALS:

BENZENE
TOLUENE
ETHYLBENZENE
TOTAL XYLENES

NOTE: DATA FROM SAMPLES COLLECTED FROM 9–95 TO 11–95 USED FOR THIS INTERPRETION.

FIGURE 3-12

TOTAL BTEX INTERPRETED CONCENTRATION ISOPLETHS FOR OVERBURDEN GROUNDWATER

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.

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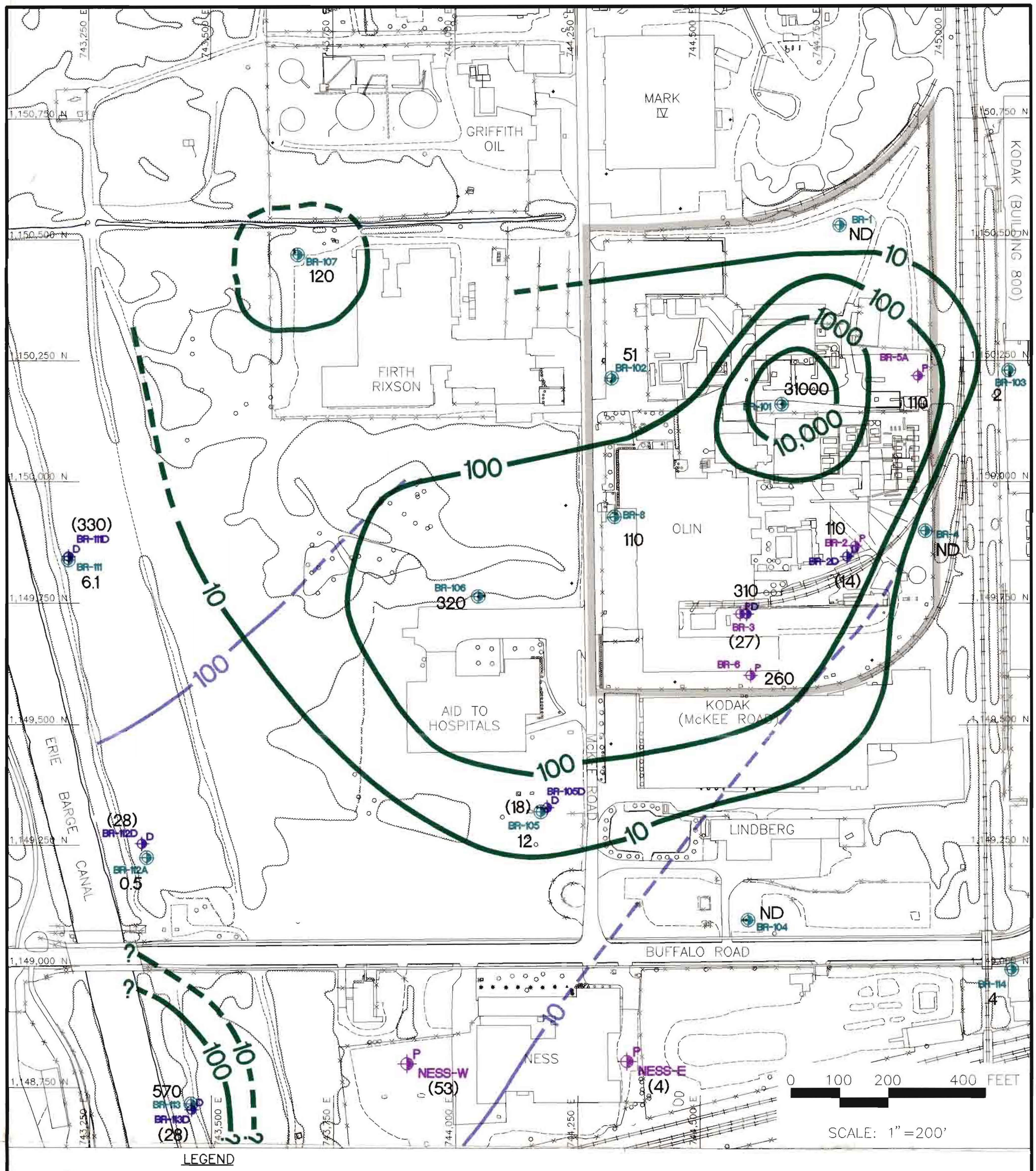

LEGEND

BEDROCK PIEZOMETER

BEDROCK MONITORING WELL

DEEP BEDROCK MONITORING WELL

BEDROCK PUMPING WELL

OUTLINE OF OLIN PROPERTY BOUNDARY

SHALLOW BEDROCK ISOPLETH

DEEP BEDROCK ISOPLETH

(26) DEEP BEDROCK WELL CONCENTRATION TOTAL

NOTES: RESULTS REPORTED AS SUM OF CONCENTRATIONS IN MICROGRAMS PER LITER FOR THE FOLLOWING CHEMICALS:

BENZENE
TOLUENE
ETHYLBENZENE
TOTAL XYLENES

NOTE: DATA FOR BEDROCK WELLS AND PIEZOMETERS SAMPLED FROM 9/95 TO 11/95 USED FOR THIS INTERPRETATION.

FIGURE 3-13

TOTAL BTEX INTERPRETED CONCENTRATION ISOPLETHS FOR DEEP AND SHALLOW BEDROCK GROUNDWATER

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.

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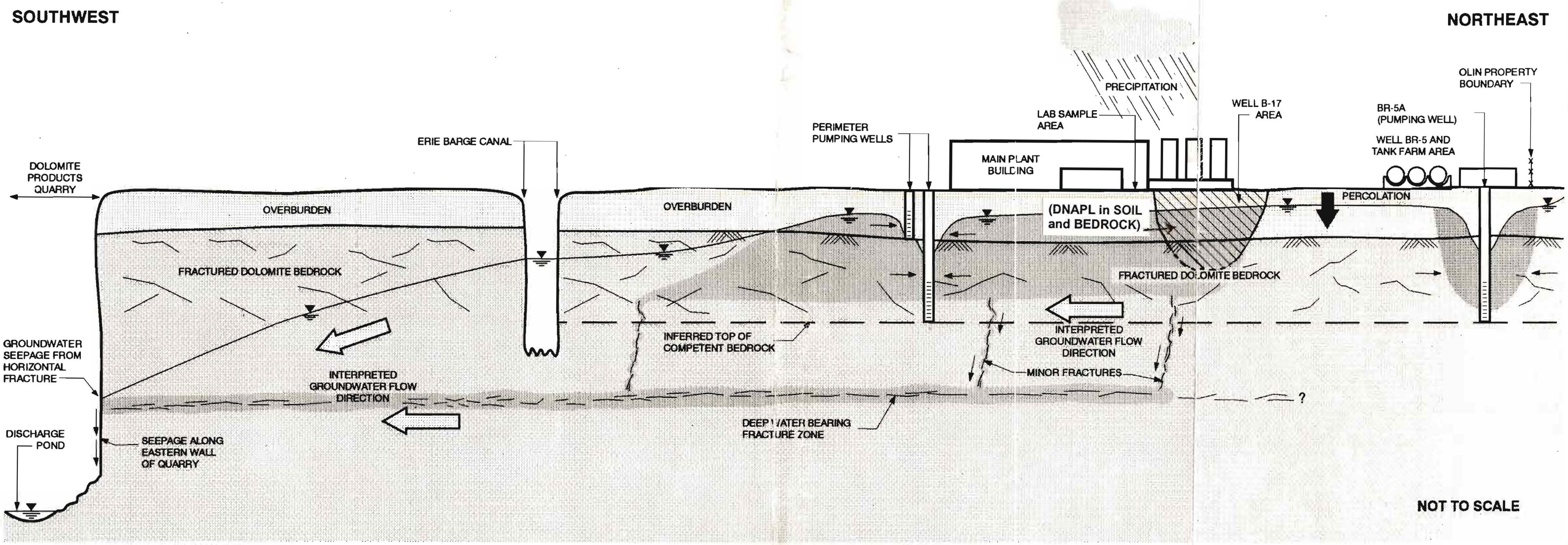


POTENTIAL TRANSPORT MECHANISMS

- LEACHING AND TRANSPORT OF RESIDUAL SOIL CONTAMINANTS AND NAPL TO WATER TABLE.
- GROUNDWATER TRANSPORT OF SUSPENDED AND DISSOLVED CONTAMINANTS IN BEDROCK.
- GROUNDWATER TRANSPORT OF DISSOLVED AND SUSPENDED CONTAMINANTS IN OVERBURDEN.
- GROUNDWATER TRANSPORT OF DISSOLVED CONTAMINANTS FROM BEDROCK GROUNDWATER TO SURFACE WATER.

FIGURE 4

SITE CONCEPTUAL MODEL

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, NEW YORK

W9603025D

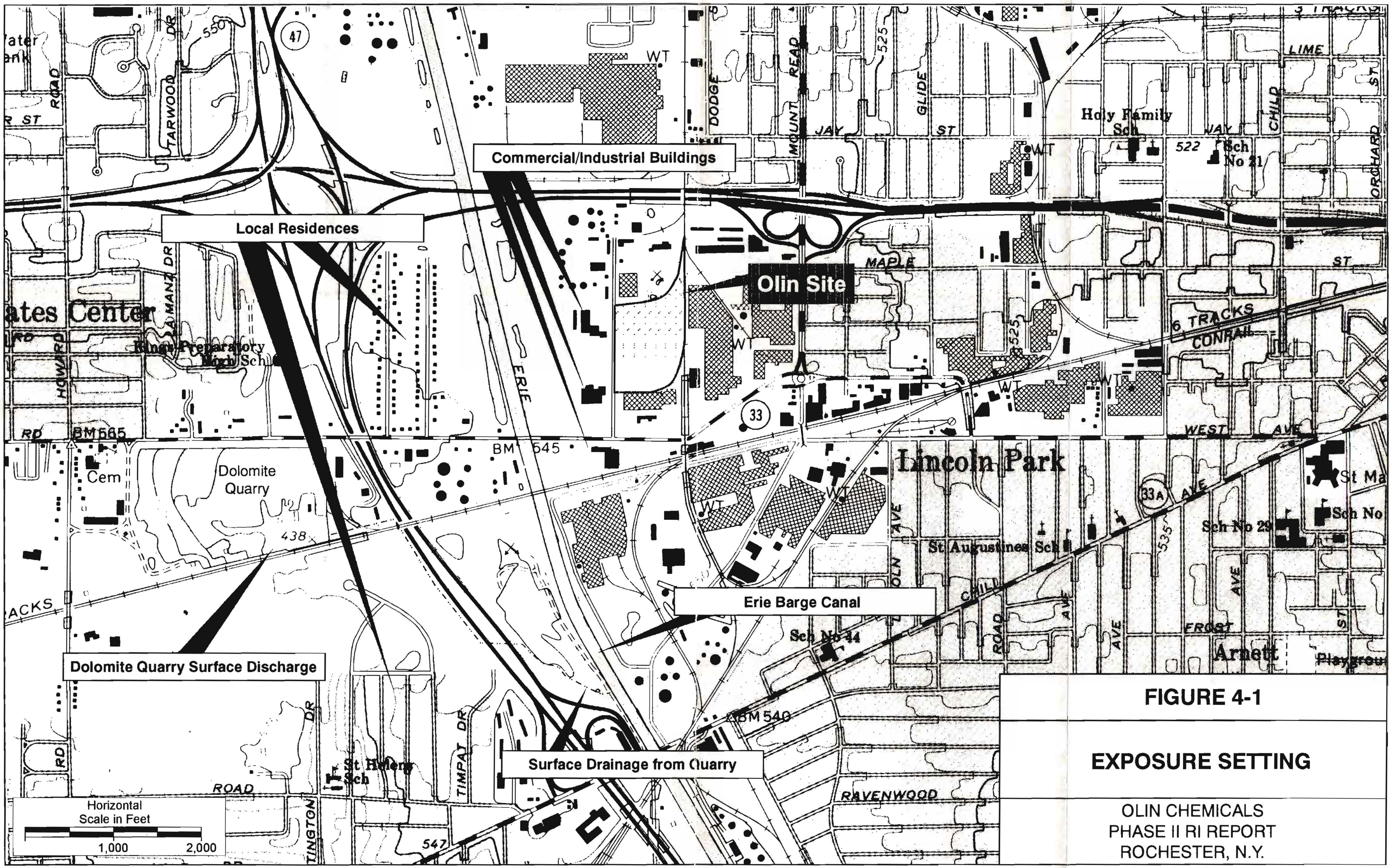
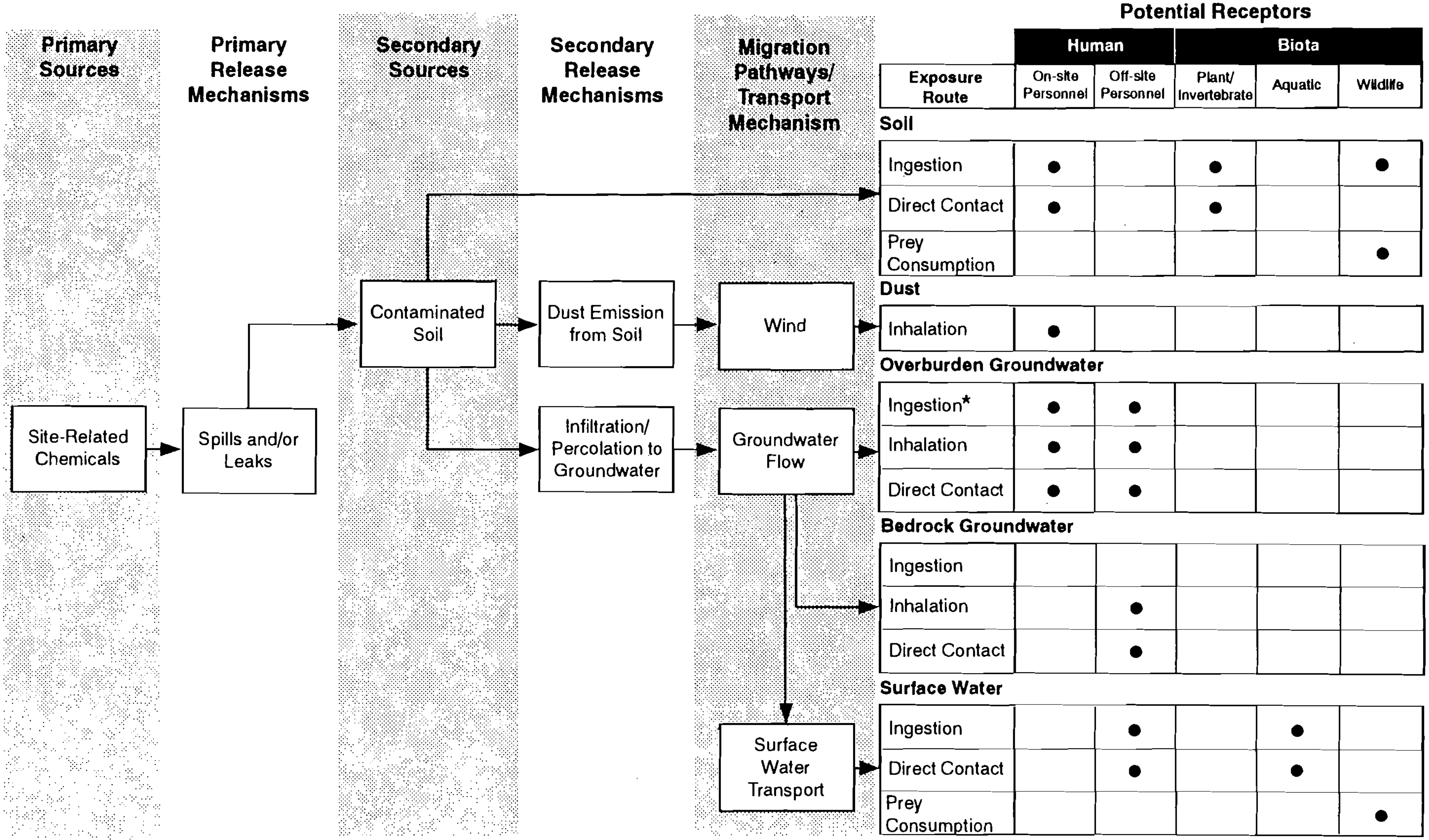


FIGURE 4-1

EXPOSURE SETTING

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, N.Y.

W9705009D(b)



Potential Receptors

Exposure Route	Human		Biota		
	On-site Personnel	Off-site Personnel	Plant/ Invertebrate	Aquatic	Wildlife
Soil					
Ingestion	●		●		●
Direct Contact	●		●		
Prey Consumption					●
Dust					
Inhalation	●				
Overburden Groundwater					
Ingestion*	●	●			
Inhalation	●	●			
Direct Contact	●	●			
Bedrock Groundwater					
Ingestion					
Inhalation		●			
Direct Contact		●			
Surface Water					
Ingestion		●		●	
Direct Contact		●		●	
Prey Consumption					●

*Incidental ingestion during excavations.

FIGURE 3-15

CONCEPTUAL MODEL FLOW DIAGRAM

OLIN CHEMICALS
PHASE II RI REPORT
ROCHESTER, NEW YORK

4.0 BASELINE RISK ASSESSMENT

This section presents the results of the human health and ecological risk assessments conducted as part of the Phase II RI. These assessments supplement the risk assessment conducted as part of the Phase I investigation and focus on the off-site groundwater and surface water. Together with the results of the Phase I risk assessment, these assessments provide a complete picture of the potential risks associated with environmental media in the vicinity of the Olin Plant site that may have been affected by past releases from the plant. A comprehensive summary of the human health risk assessments performed in support of the Phase I and Phase II RIs is presented in the Feasibility Study.

4.1 BASELINE HUMAN HEALTH RISK ASSESSMENT

A human health risk assessment has been conducted to evaluate potential health risks to individuals under current or foreseeable future site conditions associated with the Olin Study Area, particularly focused on the environmental data collected in the Phase II investigation.

For the human health risk assessment, the study area was subdivided into location-specific areas for evaluation. The study area is considered to be all of the areas and media investigated as part of this Phase II RI. Within this general study area there are the on-site areas and the off-site area. The on-site area is considered to be the area within the property boundaries of the Olin Plant. The on-site area is further subdivided into areas associated with the active chemical plant facility (the facility), and areas that do not involve the plant and are open, usually grassy areas on plant property (non-facility). Risks for potential exposures to the on-site area were evaluated in the Phase I RI.

The purpose of this assessment is to evaluate potential health risks from exposure to off-site media, which may not be under Olin management. Media sampled at off-site locations included groundwater, groundwater seeps, and surface water. No surface or subsurface soil samples were collected off-site because no source area associated with the Olin Plant was identified off-site, and because surface soil is not expected to migrate off-site. Media sampled at on-site locations in Phase II were surface soil (0-2 inches bgs), subsurface soil (0-10 feet bgs), and groundwater (overburden and bedrock). These data were collected to supplement data gaps identified in the Phase I RI, and were not evaluated in this risk assessment; on-site exposures, therefore, are not further evaluated. The Olin Plant is expected to remain an active chemical plant under Olin management and exposures to on-site chemicals would involve work place conditions under Occupational Safety and Health Administration regulations.

The risk assessment is consistent with relevant guidance and standards developed by USEPA (USEPA, 1989d,f; 1991a,c; 1992d,e,f) and NYSDEC (NYSDEC, 1994a), reflects comments and guidance received from USEPA Region II, and incorporates data from the scientific literature used in conjunction with professional judgment. NYSDEC, in general, follows USEPA guidance for risk assessment and does not have specific promulgated guidances for risk assessment methodology.

The risk assessment for the study area consists of the following components:

- Identification of Chemicals of Potential Concern (Subsection 4.1.1)
- Exposure Assessment (Subsection 4.1.2)
- Toxicity Assessment (Subsection 4.1.3)
- Risk Characterization (Subsection 4.1.4)
- Uncertainty Evaluation (Subsection 4.1.5)
- Summary and Conclusions (Subsection 4.1.6)

In summary, the risk assessment evaluated exposures to recreational visitors and industrial workers who might contact chemicals of potential concern in surface water, groundwater, or groundwater seeps. The amount of chemical that those receptors might be exposed to was estimated and combined with relevant toxicity information to calculate estimates of cancer and non-cancer risk. The only exposure that was associated with cancer risk estimates above an excess lifetime cancer risk of $1x10^{-6}$ or non-cancer risk estimates above a hazard index of 1 was for future industrial workers using off-site groundwater as industrial process water.

4.1.1 Identification of Chemicals of Potential Concern

The first step in the risk assessment was to collect, summarize, and analyze the study area data to identify those chemicals present in environmental media and related to the Olin Plant site. Study-area-related chemicals that were selected for quantitative evaluation were termed Chemicals of Potential Concern (CPCs) and defined as those chemicals that are present as a result of past activities at the Olin Plant site. The procedures used to summarize available data and to screen data for the selection of CPCs are discussed below.

4.1.1.1 Data Summary Procedures. In selecting CPCs, the analytical data for surface water, groundwater seeps and groundwater samples collected during the field investigation were first grouped and summarized. Tables 4-1 through 4-3 present a summary of data used to perform this risk assessment. Sampling and analysis procedures are described in Subsections 2.1 and 2.2. Off-site laboratory results are used for the evaluation of these media. The following steps, which are in accordance with USEPA (1989d) guidance, were used to summarize the analytical data for this risk assessment:

- Data were summarized by environmental medium (i.e., groundwater, surface water, and seep water). All chemicals detected in at least one sample in each medium were listed. All groundwater data collected for a given location in Phase II were averaged to generate a single concentration representative of that location.

- Frequency of detection was calculated as the number of samples in which the chemical was detected, divided by the total number of samples collected. Duplicate samples were considered as one data point for determining frequency of detection.

- The maximum detected concentration of each chemical was reported. For this determination, duplicate samples were considered individually to ensure that any reported maximum concentration was an actual measured number, and not the average of two samples.

- The arithmetic mean of duplicate samples was calculated and this averaged value was used to represent the concentration for that location for the purpose of calculating the arithmetic mean.

- The arithmetic mean was calculated for each chemical using the detected concentration(s), or using one-half the sample quantitation limit (SQL) for the non-detect sample(s). If the reporting limit for a non-detect sample was two or more times higher than the maximum detected concentration in that medium, the sample was not included in the calculation of the mean for that chemical. Duplicate samples for a given sampling point were also treated in this manner if a chemical was detected in only one sample of a duplicate pair.

- Tentatively identified compounds (TICs), which are chemicals identified during a library search of mass spectra, were not included in the analyte list for a specified analysis but show up as additional peaks in the laboratory analysis.

> Because of uncertainties regarding the identity and concentration of TICs, these data were not used to make quantitative assessments of risk. However, these TICs and their potential impact on total risk estimates is discussed in the uncertainty section of the risk assessment (subsection 4.1.5).

Summary sampling data for the study area are presented by medium in Tables 4-1 through 4-3. Summary data were then used in the data screening procedures to select CPCs.

4.1.1.2 Data Screening Procedures. The selection of CPCs following procedures based on USEPA (1989b) guidance is described below. The results are indicated in Tables 4-1 through 4-3.

- Sampling data were compared to blank (laboratory, field, and trip) concentration data as described in Section 2. For purposes of the risk assessment, if all concentrations of a chemical within a sample grouping were considered to be due to blank contamination, then those analytes may be eliminated from consideration as CPCs. However, no chemicals met this criterion and, therefore, no chemicals were eliminated due to blank contamination.

- Because there are no site-specific background concentrations available for naturally-occurring chemicals, the summary data were not screened to eliminate these chemicals. It should be noted that some organic chemicals may be present due to general urban/industrial anthropogenic activities (e.g., pesticides, PAHs) and not specifically related to activities at the Olin Plant. Ambient conditions, which include both naturally-occurring compounds and anthropogenic compounds, are evaluated qualitatively.

- If the number of organic compounds detected was twenty or more, a concentration/toxicity screening procedure (USEPA, 1989d) was used to limit the number of chemicals in a particular medium to those most likely to contribute the majority of risk. A concentration/toxicity screen was performed for groundwater, and it is included in Appendix D.1 as Table D.1-1.

The toxicity screening was performed by scoring each chemical in a medium according to its concentration and toxicity to obtain a risk factor (R_{ij}). Separate scores were calculated for each medium being evaluated using the following formula:

$$R_{ij}=(C_{ij})(T_{ij})$$

where:

R_{ij} = risk factor for chemical i in medium j;

C_{ij} = concentration of chemical i in medium j; and

T_{ij} = toxicity value for chemical i in medium j (i.e., either the cancer slope factor or 1/risk reference dose [RfD]).

The concentration used in the above equation was the maximum detected concentration for each compound (USEPA, 1989b). In some cases, both the oral and inhalation toxicity factors were available. Normally, in these cases, the most conservative toxicity value (i.e., one yielding the larger risk factor) is used unless an inhalation exposure scenario is unlikely (e.g., sediment).

Chemical risk factors were summed to obtain the total risk factor for all CPCs in a medium. Separate risk factors were calculated for carcinogenic and noncarcinogenic effects. The ratio of the individual risk factor for each chemical to the total risk factor approximates the relative risk for each chemical in a medium. Chemicals with very low ratios (i.e., less than 0.01) were eliminated as CPCs unless they belonged to a class of compounds in which one or more of the compounds exceed the risk ratio of 0.01 (e.g., pyridines). Degradation products of a compound which exceed the risk ratio were retained in the risk assessment.

CPCs retained in the selection process are presented in Tables 4-1 through 4-3 for the various media and are briefly discussed below.

Groundwater. Overburden and bedrock groundwater samples were combined into a single data set. CPCs selected in off-site groundwater samples included VOCs (e.g., 1,2-DCE, PCE, TCE and vinyl chloride), SVOCs (e.g., chloropyridines), and inorganic analytes (Table 4-1).

Surface Water. Because the Phase II surface water sampling was specifically designed to further investigate potential migration of chloropyridines and other site-related chemicals in groundwater, all analytes detected in surface water during the Phase II activities were retained as CPCs. The chloropyridines were also selected as CPCs because they are directly related to the Olin plant site. Olin is the only manufacturer of chlorinated pyridines in the United States. In the Erie Barge Canal, 2-chloropyridine and 2,6-

dichloropyridine were identified as CPCs as shown on Table 4-2. In the quarry seep samples, 2-chloropyridine, 3-chloropyridine, 2,6-dichloropyridine and p-fluoroaniline were identified as CPCs as shown on Table 4-3.

4.1.2 Exposure Assessment

Potential exposures associated with the study area and evaluated in the Phase II risk assessment are off-site exposure scenarios. Persons involved in recreational activities might contact surface water at the Erie Barge Canal, and workers at the Dolomite Products Quarry might contact surface water that has originated from groundwater seeps. Workers at future facilities which may use groundwater for industrial process water may be exposed to the groundwater or chemicals released from groundwater. On-site exposures are considered under Olin management and were previously addressed in the Phase I investigation. Because no source areas from the Olin Plant are identified for off-site soil, no surface soil samples were taken off-site and no soil exposure scenarios are evaluated.

CPCs associated with the Olin Plant may have migrated from Olin property by groundwater transport. The off-site exposures to groundwater were also assessed because of differences in CPCs (on-site versus off-site) and off-site exposures are not necessarily under direct Olin management.

For groundwater, seep water, and surface water off-site, potential exposure pathways were identified. An exposure pathway (i.e., the sequence of events leading to contact with a chemical) generally consists of four elements:

(1) A source and mechanism of chemical release to the environment;

(2) A retention or transport medium for the released chemical;

(3) A point of potential human contact with the contaminated medium (i.e., the exposure point); and

(4) A route of exposure (e.g., ingestion, dermal contact) for a potential receptor.

When all four of these elements are present, an exposure pathway is considered "complete." In some cases, element (2) is not necessary if exposure to the medium to which the chemical was released occurs. In the risk assessment, only complete or potentially complete exposure pathways are evaluated. The exposure assessment is performed to identify complete pathways at the study area. It draws on information regarding the source, fate and transport of chemicals, and information on human populations potentially exposed to chemicals in environmental media.

In evaluating potential human exposure pathways, exposures under both current and potential future site and surrounding land use conditions were evaluated. Current land use conditions were evaluated to take into account actual or possible exposures. Future site land use conditions were considered to address exposures which may occur as a result of any future activities or land use changes.

The Olin Plant Site is located on the east side of the Erie Barge Canal, and the area in the immediate vicinity of the site is heavily industrialized. The Dolomite Products Quarry is located on the west side of the Erie Barge Canal. There are residences on the north and south sides of the quarry, and the ditch leading from the quarry to the Barge Canal passes along the edge of a residential development. Figure 4-1 identifies the locations of these features.

The basic future site and surrounding land use conditions at the study area were assumed to be similar to current conditions. Future residential use of the Olin site and Dolomite Products Quarry is not considered plausible, and therefore, future residential exposure was not evaluated. However, recreational exposures to surface water in the Erie Barge Canal, worker exposure to groundwater seeps at the Dolomite Products Quarry, and industrial exposures to off-site groundwater used as industrial process water, may potentially occur. Possible exposure pathways encompassing both current and future conditions are summarized in Figure 4-2 and Table 4-4, and are discussed below.

4.1.2.1 Potential Exposures Under Current Site Use. Appropriate exposure scenarios for the facility reflect the industrial/commercial use of the property. Residential exposures are not appropriate. However, due to the location of residences with respect to the Erie Barge Canal and the Dolomite Products Quarry (Figure 4-1), recreational activities are possible in the canal and the quarry, and industrial activities occur at the quarry. Groundwater is not used for residential or industrial purposes under current land use. Exposure to groundwater, however, could occur at the quarry seeps.

Surface Water. The presence of a rope swing overhanging the Erie Barge Canal in the vicinity of the study area suggests that children may swim in the canal. Discharge of the quarry pond water to the canal through the quarry outfall pipe, and the detection of chloropyridines in canal surface water indicate that exposure pathways may be complete. Older children (ages 7-17) and adults who swim or boat in the canal may be exposed to canal surface water CPCs through ingestion and dermal contact with the surface water.

People have reportedly been observed fishing at the Erie Barge Canal within 200 meters of the quarry outfall. Fish may bioconcentrate chemicals in the surface water, and people who consume the fish that they catch may then be exposed to those chemicals. An evaluation of exposure to site-related compounds via ingestion of fish from the Erie Barge Canal has been performed as a component of a separate report titled Phase II Remedial Investigation, Supplemental Human Health Risk Evaluation, Erie Barge Canal, November 1996 (ABB-ES, 1996a). This report is included as Appendix D-3 of the Phase II RI Report. A comprehensive summary of the human health risk assessments performed in support of the Phase I and Phase II RIs, including risks associated with fish ingestion, is presented in the Feasibility Study.

Workers in the Dolomite Products Quarry might infrequently come into contact with groundwater seeps via dermal contact. Chloropyridines have been detected in the quarry seeps, indicating that exposure pathways may be complete. The groundwater seeps are evaluated as surface water on the rock walls of the quarry. The area where the seeps are located is remote from the areas of activity at the quarry.

4.1.2.2 Potential Exposures Under Future Site Use. In addition to potential exposures discussed under current conditions, other exposures may occur through future-industrial activities.

Surface Water. Future exposures to surface water in the Erie Barge Canal and the Dolomite Products Quarry groundwater seeps would be similar to those described for the current land use scenario. Should the quarry become inactive, it is possible that recreational users or trespassers could contact groundwater seeps in the quarry. Were this to occur, however, it is unlikely that exposures would exceed those assumed for present-day quarry workers. Because the quarry pond is not an aesthetically inviting place to swim, it is very unlikely that children or adults would swim in the quarry pond.

Groundwater. Exposures to groundwater may also occur through future use of off-site groundwater as industrial process water. Dermal contact with the water and inhalation of VOCs released from the water during its use in an operating facility may occur.

4.1.2.3 Development of Exposure Point Concentrations. To quantitatively estimate the magnitude of exposures, and thus the risks that may be experienced by an individual, the concentration of the CPC in the contact medium must be known or estimated. This concentration is referred to as an exposure point concentration (EPC). To estimate exposures, the EPC is combined with assumptions on the rate and magnitude of chemical contact. EPCs for each pathway were determined using data collected during the RI and are described below.

Quantitative exposure estimates are derived by combining EPCs with information describing the extent, frequency, and duration of exposure for each receptor of concern. An overview of the approaches used to quantify exposures is given below, followed by specific details for potential exposure pathways. The approaches described in the following paragraphs to quantify exposures are consistent with guidance provided by USEPA (1989d, 1991a, 1992e,f).

Based on USEPA risk assessment guidance (USEPA, 1989d, 1991a), exposures were quantified by estimating the reasonable maximum exposure (RME) associated with a pathway of concern. The term RME is defined as the maximum exposure that is reasonably expected to occur at a site (USEPA, 1989d). Used in combination with conservative dose-response values that are protective for sensitive subpopulations (see subsection 4.1.3), the RME is intended to place a conservative upper-bound on the potential risks. Consequently, the risk estimate is unlikely to be underestimated but it may very well be overestimated. The likelihood that this RME scenario may actually occur is small, due to the combination of conservative assumptions incorporated into the scenario. The RME estimate for a given pathway is derived by combining the selected EPC (based on the maximum detected concentration) of each chemical with reasonable maximum values describing the extent, frequency, and duration of exposure (USEPA, 1989d). Many of the exposure parameter values used in this assessment have been defined by USEPA (1989b, 1989g, 1991a) for the RME case.

In order to provide a range of risk estimates to be used for risk management decisions, EPCs were also calculated using the average concentration. This provides a more likely EPC than that calculated using only the maximum detected concentration and maximum exposure values. EPCs for groundwater, Barge Canal surface water, and quarry groundwater seeps are the maximum and arithmetic mean concentrations presented in Tables 4-1 through 4-3.

EPCs for the study area are medium- and location-specific. For surface water, the average CPC concentrations represent the arithmetic mean concentrations of CPCs detected in Barge Canal surface water during sampling events performed in April, June, and September 1996, and April and June, 1997, subsequent to other Phase II sampling activities. These data are the most recent data collected, and reflect the temporal average of concentrations during months when swimming in the Barge Canal may occur. Because sampling data suggest that surface water concentrations measured during 1996 and 1997 sampling events are higher than concentrations measured during 1994 and 1995 sampling events, the temporal average of the most recent data are an appropriate representation of the average exposure scenario EPC. The maximum Barge Canal surface water concentrations are represented by the maximum detected concentrations of CPCs in the quarry outfall water. The EPC based on these data is appropriate for conservatively modeling exposures to swimmers who may swim directly beneath the quarry outfall, a scenario which represents the RME for recreational swimmers.

Groundwater was divided into on-site and off-site areas. Groundwater samples taken at the Olin property are considered on-site, while those taken beyond the property line are considered off-site. Seep groundwater, evaluated as surface water for exposure purposes, was collected from four seeps on the face of the quarry wall during the September, 1995 sampling event. No site-related compounds were detected in sample QS-1 and, therefore, data for this sample were excluded from the EPC calculation. Concentrations in sample QS-4 were highest and, therefore, this seep was resampled in October 1995, and March, June, September, and December, 1996. Data for these sampling events were included in the average and RME EPC calculations.

The general equation for calculating chemical intake is as follows:

$$Intake = \frac{(C \times CR \times RAF \times EF \times ED)}{BW \times AT \times CF}$$

where:

Intake = daily intake averaged over the exposure period
C = concentration of the chemical in the exposure medium
CR = contact rate for the medium of concern
RAF = relative absorption factor
EF = exposure frequency
ED = exposure duration
BW = body weight of the hypothetically exposed individual
AT = averaging time (for carcinogens, AT = 70 years; for noncarcinogens, AT = ED)
CF = units conversion factor (365 days/yr)

Specific equations for each exposure scenario are provided in Appendix D.1 on Table D.1-2. Standard parameters from USEPA guidance were used to the extent that is appropriate in the intake equations. Table D.1-2 delineates the parameters used in each scenario and lists a source for each.

The contact rate reflects the amount of contaminated medium contacted per unit of time or event. The contact rate for dermal exposure to CPCs in water is estimated by combining information on exposed skin surface area, the dermal permeability of the CPC, and the exposure time. Dermal permeability of CPCs in water was evaluated using an approach identified in "Dermal Exposure Assessment: Principles and Application" (USEPA, 1992e). For inorganics, a steady-state approach was used, wherein the permeability coefficient for the inorganic is multiplied by the exposure time, assuming that the contact rate depends only on the amount of chemical crossing the skin barrier. For organic CPCs, a nonsteady-state approach was used which accounts for the total amount of chemical crossing the exposed (outside) skin surface rather than the amount which has traversed the skin and entered the blood during the exposure period (i.e., under a steady-state condition). Therefore, the nonsteady-state approach more accurately reflects normal exposure conditions (under which steady-state often may not occur) and accounts for the dose that may enter the circulatory system after the exposure event due to the storage of chemicals in skin lipids (USEPA, 1992e). In this approach, the permeability coefficient is modified by various factors to account for partitioning properties of the chemical, thickness of the skin, and diffusivity of the chemical within the skin layer. The equations to adjust the permeability coefficient vary according to whether the actual exposure time is more or less than the time it takes for the chemical to reach steady-state. The equations and factors used for each identified CPC in groundwater and surface water are listed in Table D.1-3.

<u>Calculation of theoretical indoor air concentrations - industrial process water scenario</u>. Since there is no means of measuring indoor air concentrations for a potential facility which might use groundwater as industrial process water, a theoretical calculation was conducted to estimate conservatively the concentration in a facility's air in the event that volatile organic compounds were to be released from groundwater which might be used as industrial process water in a manner that is open to the air.

The industrial process water scenario assumes a theoretical building 34.1 meters (approximately 112 feet) long, 34.1 meters (approximately 112 feet) wide, and 3 meters (approximately 10 feet) high. It is assumed that the air exchange rate in the industrial facility is 1 building volume per hour. Residential dwellings in this area of the U.S. typically have air exchange rates between 0.25 and 0.82 building volumes per hour (Murray et al., 1995) with a mean over the four seasons of the year of 0.40 building volumes per hour. Industrial buildings typically have greater air exchange rates than residential dwellings.

It is assumed that a very large volume of groundwater is used as industrial process water in the theoretical facility. One million liters per day (264,200 gallons per day), used at a constant rate throughout the day, is the assumed groundwater usage. It is further assumed that all of the groundwater used in the facility contains all of the compounds which have been detected in the Phase II investigation. The inhalation evaluation was conducted in two ways, using both average and maximum reported groundwater concentrations. In evaluating potential inhalation exposures, it is assumed that the volatile compounds in the groundwater are released from the water to the building interior immediately and are immediately dispersed upon use of the water.

The total mass of each volatile compound released to the building interior per day can be calculated as the concentration of the compound in groundwater multiplied by the volume of groundwater used in the facility per day. Further, the concentration of the compound in the building interior air can be calculated as the total mass released divided by the volume of air passing through the building per day. That volume of air is simply the volume of the building multiplied by the air exchange rate (building volumes per hour) multiplied by 24 hours. Table D.2-3 presents these calculations and the estimated building interior air concentrations based on both the average and maximum reported groundwater concentrations.

4.1.3 Toxicity Assessment

The objective of the dose-response assessment is to define the relationship between the dose of a substance and the likelihood that a toxic effect, either carcinogenic or noncarcinogenic, will result from exposure to that substance. Dose-response values were identified and used to estimate the likelihood of adverse effects as a function of human exposure to an agent. Dose-response summaries are presented in Appendix D.1 on Tables D.1-4 through D.1-6.

There are two types of dose-response values: cancer slope factors (CSFs) and reference doses (RfDs). The derivation of each value for a particular compound depends on the toxicity of that compound and whether it displays carcinogenic or noncarcinogenic effects. USEPA has derived CSFs and RfDs to evaluate carcinogenic and noncarcinogenic (systemic) risks, respectively. The definition of CSFs and RfDs, as stated in USEPA guidance are:

- Cancer Slope Factor - a plausible upper bound estimate of the probability of a response per unit intake of a chemical over a lifetime. The CSF is used to estimate an upper-bound probability of an individual developing cancer as a result of a lifetime exposure to a particular concentration of a potential carcinogen (USEPA Class A or B carcinogens) (USEPA, 1989d).

- Chronic Reference Dose - an estimate of a daily exposure concentration for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effects during a lifetime. Chronic RfDs are specifically developed to be protective from long-term exposure to a compound (e.g., as a Superfund program guideline, seven years to lifetime) (USEPA, 1989d).

- Subchronic Reference Dose - an estimate of a daily exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effects during a portion of a lifetime (e.g., as a Superfund program guideline, two weeks to seven years) (USEPA, 1989d).

In addition, because the toxicity and/or carcinogenicity of a compound can depend on the route of exposure (e.g., oral or inhalation), unique dose-response values (e.g., CSFs and RfDs) have been developed for the oral and inhalation exposure routes.

The primary source for identifying dose-response values is the Integrated Risk Information System (IRIS) (USEPA, 1996). If no information is found in IRIS, the USEPA Health Effects Assessment Summary Tables (HEAST) (USEPA, 1995) are used. If appropriate dose-response values are not available from either of these two sources, other USEPA sources are consulted (e.g., the USEPA National Center for Environmental Assessment [NCEA]). If no data exist to support the derivation of a toxicity value for a given substance, a surrogate assignment may be made or the compound is discussed qualitatively in the uncertainty section.

The methodology used to develop dermal toxicity values is obtained from Risk Assessment Guidance for Superfund, Appendix A (USEPA, 1989d). In general, the oral toxicity value is adjusted from administered dose to absorbed dose, if necessary. The absorption efficiency of a particular compound is used to calculate the RfD based on absorbed dose. For example: if the RfD based on administered dose was 20 mg/kg/day, and the absorption efficiency in the study, which is the basis of the RfD, was 10 percent, then: 20 mg/kg/day x 0.10 = 2 mg/kg/day. Therefore, the adjusted RfD is 2 mg/kg/day. The adjusted RfD is compared to the amount estimated to be absorbed from dermal exposure. This adjusted value is the dermal reference dose (RfD_{derm}). Similarly, the dermal cancer slope factor (SFD) is adjusted from the oral CSF. For example: if the CSF based on administered dose was 1.6 $(mg/kg/day)^{-1}$, and the absorption efficiency in the study, which is the basis of the CSF, is 20 percent, then: 1.6 $(mg/kg/day)^{-1}/0.20 = 8$ $(mg/kg/day)^{-1}$. The adjusted CSF is compared to the amount estimated to be absorbed from dermal exposure. This adjusted value is the SFD.

The oral (or in some cases inhalation) absorption efficiency for individual compounds is obtained from IRIS, HEAST or Agency for Toxic Substances and Disease Registry (ATSDR) toxicity profiles. If the absorption efficiency is not available from these sources, the efficiency is assumed to be similar to structurally similar compounds.

No dose-response health effects criteria were available for some of the CPCs. Therefore, risks associated with these chemicals could not be quantitatively evaluated although they may be retained as CPCs as indicated in the appropriate tables. Chemicals not quantitatively evaluated include lead, nutrients in groundwater, and a number of TICs. Because of the relatively high concentrations of chloropyridines detected, these compounds were quantitatively evaluated using chlorobenzene and/or 1,4-dichlorobenzene as a surrogate compound, although this adds to the uncertainty of the risk evaluation. The following discussion presents the rationale for selection of these compounds as surrogates with respect to potential toxicity of chloropyridines.

4.1.3.1 Surrogate Dose-Response Values. Because there are no published USEPA RfDs or CSFs for the chloropyridine compounds detected at the site, alternative sources of toxicological information were accessed to either develop compound-specific dose-response values or to estimate the toxicity of these compounds based on the toxicity of structurally similar compounds which have published dose-response values. The following paragraphs provide documentation and rationale for the selection of surrogate dose-response values. The toxicological literature was searched to identify appropriate toxicity data. Unfortunately, data suitable for derivation of dose-response values were very limited for chloropyridine compounds; only acute toxicity data (e.g., LD_{50} data) and mutagenicity studies were located. However, data presented in a number of mutagenicity studies provided sufficient information to identify suitable surrogate compounds for chloropyridine compounds, as described below.

Available evidence suggests that some chloropyridine compounds are mutagenic, whereas others are not; the mutagenic potential appears to be related to the position of the chlorine atom(s) relative to the nitrogen atom. In Salmonella reversion assays, 3-chloropyridine and 4-chloropyridine are not mutagenic (Claxton, 1987; Dearfield, 1986, 1993). No information for these compounds in other test systems is available. In contrast, 2-chloropyridine is mutagenic in both the Salmonella reversion assay and mouse lymphoma cells, but only in the presence of metabolic activation. Di-substituted pyridines with one halogen atom in the ortho-position (such as 2,6-dichloropyridine) are also mutagenic in Salmonella in the presence of metabolic activation (Claxton, 1987; Dearfield, 1986, 1993; Chlopkiewicz, 1993). These findings suggest that chloropyridines are more potent mutagens when the chlorine atom is in the ortho position relative to the nitrogen atom. In addition, since mutagenicity appears to occur only in the presence of metabolic activation, a metabolite or reactive chemical intermediate produced during chloropyridine biotransformation is likely responsible for the observed mutagenic effects. This possibility is supported by the observation that reactive species such as peroxides and hydroxide radicals, which are often products of ring-hydroxylation metabolism and are known to react with cellular macromolecules such as DNA, were produced during 2-chloropyridine biotransformation (Chlopkiewicz, 1993).

Although information concerning the biotransformation of other chloropyridine compounds is not available, it is likely that they are biotransformed through a similar pathway. Likewise, the potential carcinogenicity of chloropyridines is unknown, since no bioassay data are available. However, the mutagenic activity demonstrated in the in vitro test systems suggests that chloropyridines are potential carcinogens.

Given the toxicity data presented above, an appropriate surrogate for 2-CPL and 2,6-CPL should be a chemical with a structure that consists of a single aromatic ring, preferably substituted, that is potentially carcinogenic and yields mutagenic intermediates or by-products during metabolism. A chemical with a similar structure, but possessing less carcinogenic or mutagenic potential, should be a suitable surrogate for 3-CPL and 4-chloropyridine (4-CPL). Based on chemical structure, pyridine and chlorobenzenes are possible choices as surrogate chemicals for chloropyridines.

Both pyridine and chlorobenzenes possess chemical attributes similar to chloropyridines. Pyridine contains the nitrogen atom present in chloropyridines, whereas chlorobenzenes contain the chlorine substituents present in chloropyridines. However, the commonality of chlorine substituents on chlorobenzenes and chloropyridines suggest that pharmacokinetics and metabolism may be more similar between these chemicals than between pyridine and chloropyridines. A comparison of available toxicological data for 1,4-dichlorobenzene, chlorobenzene, and pyridine support this hypothesis.

Available data indicate that 1,4-dichlorobenzene may be mutagenic in mammalian cell cultures when tested in the presence of metabolic activation. The results of a cancer bioassay indicate that 1,4-dichlorobenzene is carcinogenic to mice and rats. The metabolic pathway for 1,4-dichlorobenzene is not well characterized, but may involve ring-hydroxylation with subsequent formation of epoxide intermediates and generation of peroxides (ATSDR, 1990a). This evidence suggests 1,4-dichlorobenzene acts as a tumor promoter, rather than a direct-acting carcinogen. Based on this evidence, USEPA has classified 1,4-dichlorobenzene as a Group C "possible human carcinogen".

Chlorobenzene was not mutagenic in mammalian cell cultures or bacterial cultures when tested in the presence or absence of metabolic activation. Although chlorobenzene produced neoplastic nodules in male rats in a cancer bioassay, it did not produce neoplastic lesions in female rats or in either sex of mice. The metabolism of chlorobenzene involves ring-hydroxylation with subsequent formation of epoxide intermediates and peroxides (ATSDR, 1989). Together, this evidence suggests that chlorobenzene is, at best, a weak carcinogen. Nonetheless, USEPA has ranked chlorobenzene as a Group C "possible human carcinogen".

Pyridine was not mutagenic in mammalian or bacterial cell cultures in the presence or absence of metabolic activation, nor was it considered carcinogenic in a cancer bioassay. Most pyridine biotransformation pathways involve metabolism of the nitrogen atom, and not ring-hydroxylation. Therefore, reactive chemical intermediates are not produced (ATSDR, 1990b).

Based on these toxicological considerations, chloropyridines and chlorobenzenes may yield similar mutagenic and potentially carcinogenic chemical intermediates and metabolites during biotransformation. In contrast, the biotransformation of pyridine does not appear to produce mutagenic or potentially carcinogenic products. Based on this information, chlorobenzenes are the preferred surrogates for chloropyridines.

The assigning of surrogates for the various chloropyridine compounds is based on a comparison of the relative potential carcinogenic potency among chloropyridine compounds to the relative carcinogenic potency among chlorobenzene compounds. Based on the limited data available, ortho-substituted chloropyridines appear to be more potent mutagens than other chloropyridine compounds (i.e., non-ortho-substituted), just as 1,4-dichlorobenzene appears to be a more potent mutagen than chlorobenzene. Although no cancer bioassay data are available for chloropyridines, the available mutagenicity data suggest that 2-CPL and 2,6-CPL may be potential carcinogens.

The lack of mutagenic activity of 3-CPL and 4-CPL does not discount them as potential carcinogens, but suggests that potential carcinogenic potency is lower. Likewise, the positive carcinogenicity data for 1,4-dichlorobenzene and the ambiguity of the carcinogenicity data for chlorobenzene suggest that 1,4-dichlorobenzene is a more potent carcinogen than chlorobenzene. Given these data, the more potent chloropyridine compounds, 2-CPL and 2,6-CPL, are assigned the more potent chlorobenzene compound, 1,4-dichlorobenzene as a surrogate. Therefore, the oral CSF for 1,4-dichlorobenzene of

0.024 $(mg/kg/day)^{-1}$ has been assigned to those two compounds. The chloropyridine compounds of lesser potency, 3-CPL and 4-CPL, are assigned chlorobenzene as a surrogate, which appears to be a less potent carcinogen than 1,4-dichlorobenzene. Therefore since a CSF has not been developed for chlorobenzene, the oral RfD of 0.02 mg/kg/day for chlorobenzene has been assigned to those compounds. This assumes that noncancer effects are more significant for chlorobenzene, due to its low potential carcinogenic potency.

4.1.4 Risk Characterization

In this final step of the risk assessment process, the exposure and toxicity information are integrated to develop both quantitative and qualitative evaluations of risk. To quantitatively assess risks associated with CPCs in an environmental medium, the average daily intakes calculated in the Exposure Assessment were combined with the health effects criteria presented in the Toxicity Assessment. The methodology used to quantitatively assess risks is described in detail below.

Methodology. USEPA (1989d, 1992f) has developed guidance for assessing the potential risks to individuals from exposure to carcinogenic and noncarcinogenic chemicals. The USEPA uses separate methodologies for estimating the risks from chemicals causing cancer and from chemicals causing adverse noncarcinogenic effects.

For exposures to a chemical exhibiting carcinogenic effects, an individual upper bound excess lifetime cancer risk was calculated by multiplying the estimated daily intake by the relevant CSF. The resulting risk estimate is an estimate of the probability of contracting cancer as a result of exposure to the potential carcinogen over a 70-year lifetime under the specified exposure conditions. A risk level of 1×10^{-6}, for example, represents an upper bound probability of one in one million that an individual will contract cancer. The upper bound cancer risk estimates provide estimates of the upper limits of risk, and the risk estimates produced are likely to be greater than the 99th percentile of risks faced by actual receptors (USEPA 1992f). To assess the upper bound individual excess lifetime cancer risks associated with simultaneous exposure to all carcinogenic chemicals of concern, the risks derived from the individual chemicals were summed within each exposure pathway. This approach is consistent with the USEPA's guidelines for evaluating the toxic effects of chemical mixtures (USEPA 1989d), but is not realistic if maximum concentrations occurring in different locations were used as exposure point concentrations. The relative significance of risk estimates were evaluated by comparison to a target risk level of 10^{-4} to 10^{-6} established by USEPA (USEPA, 1989b), and to the lower value of this range, which the NYSDOH considers to be a bound between cancer risks that are negligible and those that require further evaluation.

Unlike carcinogenic effects, noncarcinogenic effects are not expressed as incidence probabilities. Rather, potential noncarcinogenic impacts were calculated by means of a hazard quotient (HQ)/hazard index (HI) technique as recommended by USEPA (1989d). To assess impacts associated with noncarcinogenic exposures, the ratio of the daily intake to the reference dose was calculated for each noncarcinogenic chemical to derive an HQ. In general, HQs that are less than one indicate that the associated exposure is not likely to result in any adverse health effects, whereas HQs greater than one indicate that adverse health effects may occur. The effects from simultaneous exposures to all CPCs were computed by summing the individual HQs within each exposure pathway. This sum, known as the HI, serves the same function for exposures to a mixture as the HQ does for exposures to an individual compound. HIs greater than one indicate the potential for the occurrence of adverse health effects. A conclusion should not be categorically drawn, however, that all HIs greater than one are "unacceptable," because of the multiple conservative assumptions built into the exposure estimates and toxicity characterization. For these same reasons, the HIs less than one are generally regarded as being "safe." If an HI calculated in this assessment was greater than one, the CPCs were subdivided into categories based on target organ/critical effect affected by exposure (e.g., liver, skin, etc.) in accordance with USEPA guidance (USEPA, 1989d). HIs were then reexamined for these categories to better identify the potential for noncarcinogenic effects to occur.

Inhalation exposures for workers in operating facilities have been evaluated by comparing estimated indoor air chemical concentrations to workplace indoor air standards (Permissible Exposure Levels or PELs) issued by the American Conference of Governmental and Industrial Hygienists (ACGIH).

Results. Potential human health risks associated with the various environmental media investigated at the Olin Study Area were characterized using USEPA guidance. The media evaluated were groundwater (and associated inhalation exposures), surface water, and groundwater seeps. Cancer risks were characterized by comparison to the USEPA acceptable risk level of 1×10^{-4} to 1×10^{-6}. Noncancer risks were evaluated by comparison to the USEPA HI of 1.0. An HI of 1.0 or less indicates that no adverse health risks are expected from exposures at the study area. NYSDEC has established guidance risk levels for residential exposures, but not industrial exposures (NYSDEC, 1994a). The NYSDOH considers excess lifetime cancer risks below 1×10^{-6} to be negligible, and those above that level to require further evaluation.

The risk characterization tables for the individual media and exposure scenarios are presented in Appendix D-2 spreadsheets. Quantitative potential health risks are summarized by medium in Table 4-5, and by receptor in Table 4-6.

Surface Water. Surface water is quantitatively evaluated in the risk assessment because of the potential for exposures to the surface water in the Erie Barge Canal during recreational use, and groundwater seeps in the Dolomite Products Quarry during industrial use. Risks are calculated for exposures to an older child (ages 7 through 17) and adult who are assumed to swim in the Barge Canal. As discussed previously, the series of groundwater seeps on the face of the quarry walls is evaluated as a "surface water" exposure to a quarry worker. The evaluation of risks to an angler is presented in Appendix D-3.

As shown on Table 4-5, cancer risks for potential exposure to these media are below an excess lifetime cancer risk of 1×10^{-6} for both the average and RME scenarios. Likewise, non-cancer risks are below a hazard index of one for both scenarios. Risk calculations are presented in Tables D.2-4 through D.2-9. Evaluation of risks to site-related chemicals from potential ingestion of fish taken from the Erie Barge Canal are also below a cancer risk of 1×10^{-6} and a hazard index of 1 (Appendix D-3).

Groundwater. Groundwater associated with the study area was characterized as a single data set because the presumed potential future use of the groundwater is industrial process water, which might be withdrawn from any depth within the aquifer. Groundwater is quantitatively evaluated in the risk assessment because of the potential for exposures to workers in industrial facilities that might use groundwater as industrial process water in the future. There are no uses of groundwater under current land use. As shown in Table 4-5, cancer risk for exposure to offsite groundwater (mean concentrations) is 7×10^{-5}, which is within the USEPA acceptable carcinogenic risk range of 1×10^{-6} to 1×10^{-4}. The Cancer risk for the RME to off-site groundwater is 9×10^{-4}, which exceeds the upper end of the USEPA acceptable carcinogenic risk range. Noncancer risks exceed the generally accepted levels for maximum reported concentrations (hazard index of 29) and for mean concentrations (hazard index of 4). Risk calculations are presented in Tables D.2-1 and D.2-2.

The predominant contributors to carcinogenic risk (mean concentrations) are vinyl chloride (48.6% of the risk), 2-CLP (31.6% of the risk) and 2,6-DCLP (12:6% of the risk). The major contributors to noncancer risk are iron (79% of the risk), benzene (13.3% of the risk), and zinc (4.5% of the risk), for the dermal contact exposure scenario. It appears that the high levels of iron and zinc observed in the groundwater may be associated with turbidity in the groundwater samples, and, therefore, the concentrations and risks may not be representative of the exposure scenario which was evaluated. In addition, iron and zinc do not appear necessarily to be site-related. As a consequence, the results do not indicate significant potential health risks associated with dermal exposure to inorganics in off-site groundwater in an industrial process water scenario.

An inhalation-based exposure scenario for the use of groundwater as industrial process water was evaluated using a very conservative screening assessment for potential releases of VOCs from groundwater into the air within an operating industrial facility. As shown in Table D.2-3, it has been assumed that the maximum reported concentrations of VOCs in groundwater are released into the air from uncontained processing equipment. Concentrations in air for a theoretical facility were compared to permissible exposure limits (PELs) published by ACGIH; none of the estimated concentrations of volatiles exceeded any PELs (Table D.2-3). Therefore, estimated concentrations meet workplace air standards.

Summary. Table 4-6 provides a summary of risk estimates for current recreational and potential future recreational and industrial land use conditions. As indicated in Table 4-6, cancer risks for a recreational child and adult swimmer exposed to Erie Barge Canal surface water and Quarry Outfall water are below an excess lifetime cancer risk of 1 x 10^{-6}, and non-cancer risks for these exposures are below a hazard index of 1. In addition, risks for ingestion of site-related chemicals in fish taken from the Erie Barge Canal are below these levels (Appendix D-3). The risks for recreational uses of the Erie Barge Canal and industrial uses of the Dolomite Products Quarry are at a level which USEPA and NYSDOH consider to be negligible.

Cancer risk estimates for a future industrial worker exposed to groundwater used as industrial process water exceed the USEPA acceptable cancer risk range of 1 x 10^{-6} to 1 x 10^{-4} under RME conditions, but are within this range for exposures to average groundwater concentrations. Non-cancer risks for these exposure scenarios are above a hazard index of 1. Estimated air concentrations of chemicals that may volatilize from the groundwater used as industrial process water to indoor air do not exceed permissible occupational exposure limits, indicating that inhalation exposures to volatile chemicals in groundwater are not a concern for workers. Because cancer risks for potential future industrial use direct-contact exposures to groundwater exceed an excess lifetime cancer risk of 1 x 10^{-6}, and non-cancer risks exceed a hazard index of 1, the need for establishing specific remedial goals will be evaluated in the Feasibility Study.

The results of the risk assessment for the Phase II RI are consistent with previous risk assessments. Sirrine Environmental (Olin, 1990) conducted an assessment of potential human health risks associated with surface water in the Erie Barge Canal. The assessment was part of an investigation of the groundwater at the Olin Plant site. The risk assessment modeled the transport of site-related CPCs to the Erie Barge Canal. Exposure to the

CPCs was assumed to occur through swimming in the canal and consumption of fish caught from the canal. The CPCs identified were benzene, dibromochloromethane, bromoform, carbon tetrachloride, chlorobenzene, 1,2-DCE, 1,3-dichlorobenzene, TCE, PCE, chloroform, p-fluoroaniline, methylene chloride, pyridine, monochloropyridines, 2,6-CPL, and vinyl chloride. The risk characterization identified a noncancer HI of only $7\text{x}10^{-4}$, well below the USEPA guidance level of 1.0. The calculated cancer risk, $4.5\text{x}10^{-8}$, was also below the USEPA target risk range of $1\text{x}10^{-4}$ to $1\text{x}10^{-6}$.

A summary of the risk assessments performed in support of the Phase I and Phase II RIs for the Olin Chemicals Facility is provided in the Feasibility Study.

4.1.5 Evaluation of Uncertainty

The interpretation of risk estimates is subject to a number of uncertainties as a result of multiple assumptions inherent in risk assessment. All quantitative estimates of risk are based on numerous assumptions, most intended to be protective of human health (i.e., conservative). As such, risk estimates are not truly probabilistic estimates of risk, but rather conditional estimates given a series of assumptions, usually conservative, about exposure and toxicity.

In general, sources of uncertainty are categorized into general uncertainties inherent in most risk assessments (e.g., toxicity assessment methods), and site-specific factors (e.g., variability in analytical data, modeling results, and exposure parameter assumptions). Major sources of uncertainty and their potential effects (e.g., to over- or underestimate risks) are presented in Table 4-7. Site-specific uncertainties that lend to over- or under-estimation of risks, and therefore have the greatest bearing on interpretation of the risks estimated in this risk assessment, are discussed below.

Tentatively Identified Compounds Tentatively identified compounds (TICs), which are chemicals identified during a library search of mass spectra, were not included in the analyte list for a specified analysis but show up as additional peaks in the laboratory analysis. Because of uncertainties regarding the identity and concentration of TICs, these data were not used to make quantitative assessments of risk. A review of the TIC data (available for groundwater only) indicates that several petroleum hydrocarbon compounds were detected in groundwater at estimated concentrations ranging from 1 ug/L to 260 ug/L. The majority of compounds were classified as substituted benzene derivatives, alkylbenzene derivatives, hexane, or pentane. No dose-response data are available for these compounds, and both the estimated identity and concentrations of these compounds are uncertain. However, substituted benzenes and alkyl benzenes are anticipated to have the same general pharmacokinetic and toxicological properties as specific compounds in this chemical group for which considerable information is available (e.g., ethylbenzene, xylenes). Ethylbenzene, toluene, and xylenes, which were detected at concentrations of up to 2,300 ug/L, were eliminated as groundwater CPCs in this risk assessment following the toxicity screening procedure. Therefore, it is unlikely that the petroleum hydrocarbon TICs that were identified in groundwater at considerably lower concentrations would pose a risk of concern. Excluding these compounds from the risk assessment has not resulted in a substantial underestimation of risk.

Surrogate Dose-Response Values Toxicity information for many chemicals is very limited, leading to varying degrees of uncertainty associated with calculated toxicity values. Sources of uncertainty for calculating toxicity factors include extrapolation from short-term to long-term exposures, amount of data (e.g., number of studies) supporting the toxicity factors, consistency of different studies for the same chemical, and responses of various species to equivalent doses.

The assignment of surrogate toxicity factors for the chloropyridine compounds is a source of uncertainty. By assigning the cancer slope factors for 1,4-dichlorobenzene to 2-CPL and 2,6-CPL, a conservative approach has been taken in evaluating risks for those compounds. Likewise, the assignment of the RfD for chlorobenzene to 3-chloropyridine and 4-CPL is believed to represent a conservative approach to the evaluation of non-carcinogenic health risks. The risks associated with exposure to p-fluoroaniline were evaluated using the RfD for 4-chloroaniline. In the absence of a published RfD for p-fluoroaniline, this is considered a conservative approach to the evaluation of risks.

Dose response values are not available for calcium, magnesium, potassium, and sodium. Therefore, risks for exposure to these inorganics could not be quantified. However, these four inorganics are essential nutrients that are required for maintenance of normal physiological functions. The Food and Drug Administration has established Recommended Dietary Allowances (RDAs) of these nutrients (NRC, 1989). Intakes below or above the RDA may result in toxicity, however, humans can tolerate intakes several times greater than the RDA before adverse effects develop (NRC, 1989). The intakes of essential nutrients calculated in this risk assessment are below RDAs indicating that, even with additional exposure to these substances in environmental media, adverse effects would not be expected.

This in particular is the case for iron, which is also an essential nutrient but for which a provisional dose-response value has been published by the National Center for Environmental Assessment (NCEA). The NCEA provisional RfD for iron is not based on a threshold dose for toxicity, but is instead based on the average intake of iron required as an essential nutrient. As a result, hazard quotients that would be calculated for potential exposures to iron do not represent increased likelihood of adverse health effects. Use of the iron RfD for calculating human health risks for iron exposures only reflects the ratio of iron intake received from environmental media to the dose required for normal physiological functions. It does not reflect the ratio of iron intake received from environmental media to the threshold dose for iron toxicity. A hazard quotient of 1 for iron, for example, indicates that the dose of iron theoretically received from environmental media is equal to the daily dose required as an essential nutrient, and not a dose which is associated with toxicity. Moreover, a hazard quotient of 1 calculated using the NCEA provisional iron RfD would fall below the RDA for a child and within the NOAEL dose range for both children and adults. The hazard quotients for iron for potential future industrial worker dermal exposures to groundwater, therefore, are not considered representative of substantial risks.

Volatile Migration to Building Air Olin has researched groundwater use in the site area and believes that there is little likelihood that there is any resident using groundwater for landscaping or other purposes. If such a case were to be present however, the potential exposure could be thought to be similar to the pathway discussed in Section 4.1.2.2 - dermal exposure to quarry workers. This pathway was examined in the risk assessment and was not associated with any unacceptable risks.

One potential exposure pathway, migration of VOCs from groundwater to indoor air via basement seepage, was not quantitatively evaluated in the Phase II RI risk assessment. However, the risk assessment incorporated a very conservative evaluation of VOC migration from groundwater used as process water to indoor air. This evaluation, which assumed that a hypothetical facility uses a large amount of groundwater (i.e., 1 million liters per day) and that the entire mass of all VOCs detected in groundwater was released from the process water to indoor air, concluded that no indoor air concentrations would exceed workplace air standards. Seepage of groundwater to buildings (e.g., into basements) would not result in indoor air concentrations as high as those estimated for the process water scenario, and groundwater VOCs migrating to indoor air via soil gas would not result in indoor air concentrations as high as those estimated in the process water scenario.

In order for indoor air concentrations of any compound detected in groundwater to exceed workplace air standards, it would be necessary to assume that either: 1) the contribution of VOCs to indoor air from groundwater migration via soil gas was equal to the contribution from process water, and that both fate and transport processes occurred simultaneously (to result in indoor air concentrations twice those estimated for the process water scenario), or 2) the building air exchange rate was reduced from one building volume air exchange per hour, to 0.4 building volume air exchanges per hour (representing the mean annual building air exchange rate for residential dwellings). These assumptions are not realistic for a well-vented commercial/industrial building. Even using these unrealistic assumptions and maximum groundwater concentrations, only benzene and p-fluoroaniline would theoretically occur in indoor air at concentrations up to 7 mg/m^3 and 9.5 mg/m^3, respectively, values slightly above the workplace air standards of 3 mg/ m^3 and 8 mg/ m^3, respectively.

The low likelihood that groundwater VOCs would occur in indoor air at concentrations of concern is further supported by soil gas data collected during the Phase I investigation. Of the soil gas measurements recorded at 87 locations in the vicinity of the Olin Plant, only one compound (trichloroethene) was detected at a single location at a concentration above the OSHA workplace air standard.

Swimmer Exposure Assumptions To help describe the uncertainty associated with the chemical-physical data and the exposure assumptions used in the swimmer exposure scenario, risks were developed for minimum, average, and maximum exposure assumptions. Risk-based concentrations (RBCs) were then developed from these risk estimates. This evaluation is described in detail in Appendix D-3. The RBCs can be compared to surface water concentrations in the Barge Canal or Quarry outfall to gauge the risks to humans potentially exposed to those media (via recreational swimming). As described in Appendix D-3, only the maximum concentrations of 2,6-dichloropyridine and 2-chloropyridine, which are represented by the data for the quarry outfall water, exceed RBCs. The RBCs exceeded are those based on maximum exposure conditions for the $1x10^{-6}$ cancer risk level; RBCs based on average exposure conditions or non-cancer effects are not exceeded. Again, comparison of quarry outfall water concentrations to RBCs that are based on the most stringent exposure conditions represents an extremely conservative evaluation of potential risks.

4.1.6 Human Health Risk Assessment Summary and Conclusions

Health risks associated with potential exposures to media off-site at the Olin Plant were evaluated for groundwater (including associated inhalation exposures) and surface water, including groundwater seeps. CPCs were selected on a medium- and location-specific basis. Generally, the CPCs identified were VOCs (particularly chlorinated compounds), SVOCs (primarily chloropyridines), and inorganics. The exposure scenarios quantitatively evaluated include industrial/commercial worker and recreational exposures. Potential health risks are characterized using USEPA-acceptable risk levels. The potential health risks are summarized below.

- No significant human health risks were identified for potential exposures to surface water in the Erie Barge Canal or to groundwater seeps in the Dolomite Products Quarry under current or potential future land use conditions; cancer risks did not exceed an excess lifetime cancer risk of 1 x 10^{-6}, and non-cancer risks did not exceed a hazard index of 1.

- Evaluation of potential future worker exposure to off-site groundwater used as industrial process water identified cancer risks above 1 x 10^{-6} but below 1 x 10^{-4} (i.e., within the USEPA acceptable cancer risk range) for mean groundwater concentrations. For maximum concentrations, however, carcinogenic risks were calculated to be 9 x 10^{-4}, above the upper end of the USEPA acceptable cancer risk range. Calculated non-cancer risks were elevated, but the majority of that risk appears to be attributable to iron and zinc associated with turbidity in groundwater samples. In addition, the iron and zinc concentrations detected in off-site wells were higher than any detections on-site. Therefore, risks are unlikely to be related to Olin operations. Because risks for potential future exposures to off-site groundwater used as industrial process water exceeded an excess lifetime cancer risk of 1 x 10^{-6} and a non-cancer hazard index of 1, the need for establishing specific remedial goals will be evaluated in the Feasibility Study.

- Modelling a hypothetical future release of VOCs from groundwater used as process water to industrial facility air did not result in any exceedances of workplace air standards.

- Groundwater concentrations exceeded MCLs and New York Standards for several CPCs. No domestic use of the groundwater is anticipated. For aesthetic reasons, groundwater in the vicinity of the Olin Study Area is not used as a drinking water source. Naturally-occurring sulfide and explosive gases preclude use of bedrock groundwater for drinking water.

4.2 HABITAT-BASED ECOLOGICAL ASSESSMENT

This subsection presents the results of a supplemental ecological risk assessment (ERA) for the Olin Study Area conducted as part of the Phase II RI. This assessment was performed in accordance with NYSDEC (1989, 1991) guidance, which provides an approach for the "characterization of the fish and wildlife values and threats at hazardous waste sites being considered for remediation".

This assessment supplements, rather than replaces, the Phase I ERA (ABB-ES, 1994), and focuses on an evaluation of additional site data collected to fill information gaps identified during the Phase I RI. Surface water data that were collected following the completion of the Phase I RI were used to characterize potential risks to aquatic receptors and semi-aquatic wildlife that may occur in the Erie Barge Canal. A computer search of a USEPA aquatic toxicity database (AQUIRE) was conducted and regression models employed to more fully characterize the potential toxicity of the primary groundwater chemicals of concern to ecological receptors. In addition, NYSDEC historically has collected stream and river aquatic macroinvertebrate data as a component to long-term water quality assessment studies. Macroinvertebrate data collected in the 1970's and early 1980's from several locations within the Erie Barge Canal in the general vicinity of the Olin Plant site were also evaluated in order to characterize the nature of the macroinvertebrate communities in this aquatic habitat. Finally, the Phase I ERA conclusions regarding ecological risks associated with surface soil exposure within the Olin Plant site were re-evaluated using regional background data for inorganic analytes. The background data, described in Section 2.1, are from a NYSDEC (1994) document on determination of soil cleanup objectives.

The ERA for the study area includes the following elements:

- Data Evaluation (Subsection 4.2.1)
- Identification of Potential Ecological Receptors (Subsection 4.2.2)
- Ecological Exposure Pathways (Subsection 4.2.3)
- Ecological Effects Assessment (Subsection 4.2.4)
- Ecological Risk Characterization (Subsection 4.2.5)
- Ecological Risk Assessment Uncertainties (Subsection 4.2.6)
- Ecological Risk Assessment Summary and Conclusions (Subsection 4.2.7)

4.2.1 Data Evaluation

The analytical data considered in this ERA include surface water samples collected from the Erie Barge Canal, off-facility groundwater, and two additional surface soil samples collected at the Olin Plant site. Phase II analytical data were collected to address certain data gaps that were identified during the Phase I RI. As a result, no CPC screening was conducted in this ERA.

Surface Water Samples. Surface water samples have been collected quarterly since September 194 at three sampling locations (SW-1, SW-2, and SW-3) within the Erie Barge Canal in the vicinity of the Olin Plant site (Figure 2.5a). Beginning in 1996, additional locations have been added to the surface water sampling program in order to better define the presence of site-related constituents in the canal. Additional sampling locations include SW-7, SW-8, SW-9, SW-11, and SW-12, which are located progressively upstream from SW-1, and SW-4, SW-5, and SW-6, which are located between SW-3 and the confluence with the Genessee River (Figure 2.5a). A surface water sample was also collected at SW-10, located in the canal south of the Genessee River, in December 1996. The potential effected of discharge from the dolomite quarry, located southwest of the Olin Plant, has been evaluated by collecting quarterly samples from the outfall (Quarry Outfall) since June 1996. Surface water samples have also been collected from the Erie Barge Canal at distances of 100 and 200 feet above and below the Quarry Outfall (QO-2N1, -2N2, -2S1, and -2S2) (Figure 2.5a). All surface water samples have been analyzed for pyridine, 2-CPL, 3-CPL, 4-CPL, 2,6-CPL, and p-fluoroaniline and analytical results are presented in Appendix B.

A subset of the available data was used to develop surface water exposure concentrations for aquatic biota in the Erie Barge Canal for the ERA. Surface water data collected prior to 1996 were excluded as these are historical and not representative of current conditions (the maximum concentrations of all detected analytes are included in the 1996/1997 samples). None of the target analytes were detected in the surface water sample collected at SW-10, located in the Erie Barge Canal south of the confluence with the Genessee River, and this location was excluded from the data summaries. The analytical results for the Quarry Outfall (QO-2) were also not evaluated because these samples were collected directly from the outfall pipe and are not representative of exposure conditions within the canal.

Surface water sampling locations were segregated into two sets in order to distinguish the potential contribution of the Quarry Outfall on Erie Barge Canal water quality. Sampling locations upstream of the Quarry Outfall include SW-1, SW-2, SW-3, SW-7, SW-8, SW-9, SW-11, and SW-12. Sampling locations in the vicinity, and downstream, of the quarry outfall include QO-2N1, QO-2N2, QO-2S1, QO-2S2, SW-4, SW-5, and SW-6. Tables 4-8 and 4-9 present summaries of the 1996-1997 analytical results for the sampling locations upstream of and adjacent to/downstream of the Quarry Outfall, respectively. Three of the target analytes, pyridine, p-fluoroaniline, and 4-CPL have never been detected in Erie Barge Canal surface water samples (including pre-1996 data) and are not listed in the summary tables. The tables present detection frequencies, arithmetic average and maximum detected concentrations of 2,6-

CPL, 2-CPL, and 3-CPL. The arithmetic average was calculated using one-half of the reporting limit for non-detect results.

Groundwater Samples. Although no direct ecological exposure to groundwater is likely, future exposures may occur in the scenario of discharge to the Erie Barge Canal. Consequently, groundwater data for the six surface water analytes were evaluated in this ERA. Groundwater data collected from monitoring wells located adjacent to the Erie Barge Canal were compared to canal surface water data to evaluate the relationship between these two media. In addition, concentrations of the six surface water analytes in the overall off-facility groundwater data set were also evaluated to determine whether exposure conditions would likely change in the future.

Surface Soil Samples. With the exception of two locations in the immediate vicinity of the SS-103, no surface soil data were collected as part of Phase II sampling activities. Table 4-10 provides a summary of the surface soil data that were evaluated in the Phase I ERA and regional background inorganic concentration ranges (NYSDEC, 1994). The background data were presented previously in Section 2.2. Six surface soil sampling locations (i.e., SS-102, SS-105, SS-109, SS-112, SS-113, and SS-115) were selected to represent ecological exposures at the Olin Plant site in the Phase I ERA (ABB-ES, 1995a). These locations were generally located along the eastern perimeter of the facility in areas that were not covered by coarse gravel and compacted and where ecological exposures could reasonably be expected. Maximum concentrations of chromium, mercury, nickel, and zinc exceed background concentration ranges, and only the average zinc concentration exceeds the maximum concentration associated with background conditions.

During the Phase I RI, mercury was detected at a concentration of 214 mg/kg in this surface soil sample SS-103 (see Figure 2-4). Mercury was also detected in seven other surface soil samples collected within the Olin Plant site during this same sampling program, at concentrations that ranged from 0.16 to 2.2 mg/kg. The mercury concentration at SS-103 does not appear to be consistent with the other analytical results, and therefore, two additional surface soil samples were collected in the vicinity of this location as part of the Phase II RI field activities. The Phase II RI surface soil analytical results (these samples were only analyzed for mercury) are presented in Appendix B-1. The detected mercury concentrations at locations SS-116 and SS-117 are 0.15 and 7.2 mg/kg, respectively. These results further suggest that the Phase I analytical result at SS-103 is atypical.

4.2.2 Identification of Potential Ecological Receptors

The general types of ecological receptors that are expected to occur in the Erie Barge Canal in the vicinity of the Olin Plant site have been described in the Phase I ERA. Additional information on the invertebrate fauna characteristic of this section of the canal was received after the submission of the Phase I RI report. Although these data are primarily of historical interest (the barge canal immediately upstream of the Genessee River was sampled in 1975 and 1981) and are not necessarily representative of current conditions, they provide an indication of types of organisms that would be expected to occur in this aquatic habitat. Between the years 1972 and 1992, NYSDEC's Stream Biomonitoring Unit collected macroinvertebrate community data from New York State streams and rivers. A sampling location in the Erie Barge Canal located in the vicinity of the Olin Plant site was sampled in 1975 and 1981. During this time period, NYSDEC biologists reported a general improvement in water quality as measured by macroinvertebrate community structure and function (NYSDEC, 1993). In 1975, high organic inputs were noted in the Erie Barge Canal upstream of the Olin Plant site. This organic enrichment, which was also observed in the sampling location within the study area, was correlated with large standing crops of pollution-tolerant organisms. By 1981, standing crops had declined in this area and macroinvertebrate abundances were relatively consistent throughout the sampled portion of the canal. In addition, relatively pollution-sensitive organisms such as the mayfly (*Stenonema femoratum*) and caddisfly (*Cheumatopsyche* sp.) were collected throughout the entire reach sampled (NYSDEC, 1993). The water quality at the sampling location near the Olin Plant site was classified as "non-impacted" in 1981. Although the invertebrate community at this sampling location was still dominated by oligochaete worms (*Nais* sp.) and pollution-tolerant chironomid midge larvae (e.g., *Dicrotendipes* sp.), standing crop decreased and species diversity dramatically increased over the 6 year interval. The researchers suggested that these changes may have been due to improvements in several point source inputs to the Erie Barge Canal upstream of the Olin Plant site (NYSDEC, 1993).

In addition to aquatic receptors (i.e., fish, amphibians, invertebrates, and plants), semi-aquatic wildlife, such as piscivorous birds and mammals could be exposed as a result of feeding on contaminated prey items from the Erie Barge Canal. Although the heavy residential and industrial land use of the area surrounding the potentially affected portion of the Erie Barge Canal probably does not offer suitable habitat for the more reclusive large predatory species, it is likely that piscivores such as raccoons (*Procyon lotor*) and belted kingfisher (*Ceryle alcyon*) would utilize this foraging area.

4.2.3 Ecological Exposure Assessment

The purpose of the ecological exposure assessment is to evaluate the potential for ecological receptor exposures to chemical constituents in the study area.

4.2.3.1 Aquatic Biota. Tables 4-8 and 4-9 present summaries of the 1996-1997 analytical results for the sampling locations upstream of and adjacent to/downstream of the Quarry

Outfall, respectively. Of the three detected target analytes, 2-CPL has been detected most frequently, and at highest concentrations; this analyte was detected in 23 of 36 samples collected upstream of the Quarry Outfall (Figure 4-8). 2-Chloropyridine is also the only analyte detected at sampling locations in the vicinity, and downstream, of the outfall (Table 4-9), with estimated values ranging from 0.2 µg/L to 4 µg/L. With the exception of analytical results collected in March and April 1996, detected 2-CPL concentrations have been less than 10 µg/L. Analytical results for surface water samples collected at SW-1, SW-2, and SW-3 in March and April 1996, and at SW-7 and SW-8 in April 1996, range from 25.5 µg/L to 45 µg/L (Appendix B). 2,6-Chloropyridine was detected at a maximum (estimated) concentration of 5 µg/L at SW-3 in April 1996 and 3-CPL was detected at a maximum (estimated) concentration of 3 µg/L at SW-2 in March 1996.

4.2.3.2 Semi-Aquatic Wildlife. Table 4-11 presents the estimated exposure body dose estimates for two representative piscivorous wildlife receptors, belted kingfisher and raccoon. For both species, fish tissue concentrations were conservatively estimated by applying a bioconcentration factor (BCF) to the maximum surface water concentration detected throughout the sampling program. Ingestion body dose estimates were then derived by multiplying the estimated fish tissue concentration by the daily ingestion rate and dividing by the receptor body weight. It was assumed that the representative receptors consume only barge canal fish and that the fish have bioaccumulated these three surface water analytes as predicted from a long-term exposure to the maximum concentrations detected in surface water.

4.2.4 Ecological Effects Assessment

The purpose of the ecological effects assessment is to describe the toxic or adverse ecological effects associated with the six surface water analytes and evaluate the relationship between these measured concentrations to which an organism is exposed and the potential adverse effects due to such exposures. The primary aspect of the effects assessment is the identification of threshold or reference toxicity values (RTVs) for each of the chemicals of concern. Information provided in the effects assessment is used in conjunction with exposure information to evaluate ecological risks to aquatic receptors and wildlife in the ecological risk characterization.

4.2.4.1 Toxicity to Aquatic Receptors in the Erie Barge Canal. The primary sources of aquatic toxicity information used to develop aquatic RTVs were the USEPA AQUIRE and ecological structure activity relationship (ECOSAR), a computer program which estimates aquatic toxicity of chemicals based on structure activity relationship (SAR) regression models.

AQUIRE database. The AQUIRE database presents information extracted from independently-compiled data files and from published literature that focuses on the toxicity of chemicals to aquatic organisms. A search of the AQUIRE database was conducted for the following six surface water analytes (i.e., pyridine, 2-CPL, 3-CPL, 4-CPL, 2,6-CPL and p-fluoroaniline). One Hundred and Thirty One records were retrieved as a result of the database search, consisting of 123 records for pyridine, one record for each of the monochloropyridines and 2,6-CPL, and four records for p-fluoroaniline.

Each record is assigned a code representing the estimated reliability of the study, as established by the USEPA. Code categories are assigned to indicate whether a specific study meets all established criteria, meets some of the established criteria, does not meet any criteria, or was not reviewed; these categories are designated as "1", "2", "3", and "4", respectively. In addition, studies designated with a reliability code of "5" are from the "Guilford file", which contains acute toxicological data derived from studies which focused on acute toxicity of organic chemicals to fathead minnows. For pyridine, 99 records were assigned reliability codes of either 1 or 2, twenty-two were designated as either a 3 or 4, while the remaining two were assigned a category of "5". All of the chloropyridine records were designated with a reliability code of 2. Finally, for p-fluoroaniline one record was designated with a 1 and the remaining three were designated with 3s. Selected fields from the entire set of retrieved records from the AQUIRE database search are presented in Table 4-12.

Figure 4-2 presents a summary of the acute toxicity data for pyridine obtained from the AQUIRE database. Pyridine was the only study analyte with sufficient data to develop a cumulative effects distribution. This figure presents only those studies which derived an acute LC_{50} (the single dose lethal to 50 percent of the test population). LC_{50} results range from 1,100 μg/L to 9,550,000 μg/L, a range of almost 4 orders of magnitude. Based on the toxicological data included in the AQUIRE database, the most sensitive aquatic organism is the pink salmon (*Oncorhynchus gorbuscha*) with an LC_{50} concentration of 1,100 μg/L. The least sensitive organism included in the database is the clawed toad (*Xenopus laevis*), which had the highest LC_{50} value of 9,550,000 μg/L. However, it is important to note that considerable variation in toxicological response within taxonomic categories is evident in the AQUIRE results. For instance, LC_{50} values for salmonid fish range from 1,100 (pink salmon) to 560,000 μg/L (rainbow trout), which differ by approximately 500 times. LC_{50} values for the clawed toad range from 1,000,000 to 9,550,000 μg/L, or nearly one order of magnitude (Table 4-12). Assuming that the available data are normally distributed and representative of the toxicological response of most aquatic species, less than 1 percent of all LC_{50}s are expected to fall below 115,000 μg/L and 50 percent are expected to be 1,900,000 μg/L or greater (Figure 4-2).

Structure-Activity Relationships (SARs). A computer model ECOSAR, developed by the USEPA (Clements and Nabholz, 1994) was also employed to estimate effect thresholds for the study analytes. As discussed above, with the exception of pyridine, few data were obtained from the AQUIRE database search. The ECOSAR program estimates threshold effect concentrations for different aquatic taxa; separate regression models are available for different types of endpoints (including LC_{50}s, EC_{50}s and chronic values (CVs). Currently, the program contains over 100 SAR regression models for 42 chemical classes. The majority of these regression models were developed using measured aquatic toxicity data and octanol/water partition coefficients (K_{ow}). The majority of which have been developed for acute toxicity to fish (fresh and saltwater), water fleas (daphnids), green algae, however SARs have been also been developed for earthworms and other endpoints such as chronic toxicity and bioconcentration factors.

The inputs required to run the ECOSAR model include chemical name, Chemical Abstract Service (CAS) number, molecular weight, melting point, solubility, physical state, and $\log K_{ow}$. The physio-chemical information that was utilized to estimate effect concentrations for the surface water analytes is provided in Table 4-13. The first step in estimating toxicological thresholds utilizing the ECOSAR program requires selecting a specific chemical class for which SARs have been developed that is appropriate for the chemical in question. The classes chosen for pyridine compounds and p-fluoroanilines were halogenated aromatic hydrocarbons and anilines, respectively. Physio-chemical data are input and the types of organisms, exposure duration, and endpoints of interest are selected from the menu of available regression models. The output generated by the program is a SAR report, which includes all chemical-specific input data and predicted toxicity values for the selected endpoints. The user can also request information about the selected regression models, including the compounds used to develop the SAR, recommendations regarding applications and limitations of the particular model, and the primary literature reference(s). The SAR reports and cover sheets generated for the 6 surface water analytes are presented in Appendix E.

The ECOSAR model results including the LC_{50}, and EC_{50} and CV results are presented in Figures 4-3 and 4-4, respectively. The specific SAR models used in developing these toxicity estimates are presented in Table 4-14. As indicated in Figure 4-3, daphnids appear to be more sensitive to the surface water analytes than are the other modeled taxa. Based on the SAR model estimates, pyridine toxicity appears to be related to increasing chlorination, and p-fluoroaniline appears to be more toxic to aquatic organisms than are pyridines, in general.

Surface water Benchmark Development. The lowest chronic values from the evaluated toxicological data were used to develop RTVs for each of the surface water analytes. These RTVs, which represent a threshold concentration for effects to aquatic organisms, are expressed in µg/L. Although only 2-CPL, 3-CPL, and 2,6-CPL have been detected in barge canal surface water samples, and only at estimated concentrations, RTVs were developed for all 6 analytes in order to evaluate potential risks associated with the future discharge of groundwater.

The AQUIRE database included few chronic toxicity data for the surface water analytes; these data are necessary in order to develop RTVs that are protective of chronic exposures. Where possible, taxon-specific acute to chronic ratios were developed for each pyridine compound based on the estimated ECOSAR results (Appendix E). For those compounds having sufficient acute data to adequately characterize the lethal endpoint in aquatic receptors, the maximum acute/chronic ratio (derived from the ECOSAR model) was then applied to the lowest LC_{50} value to estimate a chronic RTV. For pyridine, the maximum acute/chronic ratio (9.22) was applied to the lowest LC_{50} concentration (1,100 µg/L) reported in the AQUIRE database. This resulted in a surface water RTV of 120 µg/L. In the case of the monochloropyridine compounds and 2,6-CPL, available toxicological data are very limited and the may not be characteristic or protective of organisms for which data do not exist. Consequently, the lowest estimated ECOSAR model results were selected as the RTVs for the chloropyridine compounds. As presented in Appendix C, the selected RTVs are as follow:

2-chloropyridine	14,000 µg/L
3-chloropyridine	12,900 µg/L
4-chloropyridine	15,300 µg/L
2,6-dichloropyridine	4,700 µg/L

For p-fluoroaniline, the acute/chronic ratio value of 219 was applied to the available acute value (16,900 µg/L from a single LC_{50} study for fathead minnow). The estimated chronic value (77.1 µg/L) based on this approach was compared to the lowest chronic value (32 µg/L) estimated using the ECOSAR program and the lower of the two values was selected as the RTV for p-fluoroaniline.

4.2.4.2 Toxicity to Semi-Aquatic Wildlife Receptors in the Erie Barge Canal. Published laboratory-derived toxicological data were evaluated in order to develop ingestion RTVs for the selected representative wildlife receptors, the belted kingfisher (*Ceryle alcyon*) and raccoon (*Procyon lotor*). RTVs were developed for the three detected surface water analytes, 2,6-CPL, 2-CPL, and 3-CPL. Very limited published data are available for these compounds; in fact, a single acute oral LD_{50} was obtained for each. A safety factor of 20 percent was applied to this value to generate an acute lowest observed adverse effects level (LOAEL). From this number a chronic LOAEL was calculated by applying an acute/chronic ratio of 10. The ATSDR (1990b) for pyridine provided additional toxicological information on this group of organic compound. Acute and chronic values presented for pyridine were slightly higher than the derived numbers for the chlorinated pyridines as would be expected. The ATSDR document also provides limited toxicological data for sublethal effects associated with chronic exposure to pyridines. Sublethal effects to mammals associated with chronic pyridine exposure include hepatic effects (e.g., increased liver weight and inflammatory lesions), decreased weight gain, and central nervous system toxicity.

4.2.5 Ecological Risk Characterization

This subsection characterizes the risk to aquatic receptors from exposure to estimated concentrations of surface water analytes detected in the Erie Barge Canal. In addition, the concentrations of the surface water analytes detected in groundwater monitoring wells were evaluated to assess the relative magnitude of future exposures associated with the discharge of contaminated groundwater into the canal. The exposure information combined with the ecological effects information provides the basis for this risk characterization.

4.2.5.1 Risks to Aquatic Receptors in Barge Canal. A groundwater dilution model was employed in the Phase I RI ERA to assess aquatic risks associated with the discharge of contaminated groundwater into the Erie Barge Canal. It was concluded that the estimated surface water concentrations under both high- and low-water conditions were several orders of magnitude lower than screening benchmark values and that no risks to aquatic organisms were indicated. The results of the Phase II surface water sampling program confirm these conclusions. The few estimated concentrations of surface water analytes detected are considerably lower than the established RTVs for aquatic receptors (Tables 4-8 and 4-9).

Aquatic receptors may be exposed to the six surface water analytes in the future if contaminated groundwater were to discharge into the Erie Barge Canal. The concentrations of the surface water analytes detected in groundwater from monitoring wells located adjacent to the barge canal were compared with the surface water data in order to assess whether future exposures would likely result in greater risks than are currently estimated. Analytical data for monitoring wells BR-111, BR-111D, BR-112A, BR-112D, and BR-113, and BR-113D are presented in Appendix B-1. Of the six pyridines selected as surface water analytes, only 2-CPL and 2,6-CPL were detected in these adjacent wells during the October and December 1995 groundwater sampling events. These are also the only two surface water analytes that were detected in Erie Barge Canal surface water samples collected during this sample event (i.e., November 1995). 2,6-CPL was detected in the adjacent monitoring wells at a maximum concentration that is only three to five times greater than the estimated concentrations detected in surface water samples collected during the same time period. On the other hand, 2-CPL was detected in the adjacent monitoring wells at a maximum concentration that is 20 to 75 times greater than the estimated concentrations detected in surface water samples collected during the same time period. The maximum detected concentrations of both of these analytes are less than the established surface water RTVs; and no risks to aquatic receptors would be expected even if they were to be exposed directly to the concentrations detected in groundwater in Phase II wells along the Erie Barge Canal.

The entire off-site groundwater data set was also evaluated because the surface water analytes were detected at highest concentrations in monitoring wells that are located some distance from the Erie Barge Canal (Appendix B-1). Maximum concentrations of the surface water analytes detected in the off-site groundwater data were compared to the established RTVs in order to estimate the likelihood of future aquatic impacts under worst-case exposure assumptions. Of the six surface water analytes, only pyridine was not detected in the Phase II groundwater monitoring wells (Appendix B-1). The maximum concentrations of 2,6-CPL, 2-CPL, and p-fluoroaniline exceed the surface water RTVs. 2,6-CPL was detected in groundwater at a maximum concentration of 15,000 μg/L, which exceeds the surface water RTV (4,700 μg/L) by approximately 3.2 times; the average concentration is below the surface water RTV. The maximum concentration of 2-CPL (84,000 μg/L) exceeds the surface water RTV (14,000 μg/L) by approximately 6 times. p-fluoroaniline was detected at a maximum concentration (320 μg/L) in Phase II groundwater samples, which is 10 times greater than the surface water RTV (32 μg/L). Considering the attenuation and dilution processes that would occur prior to ecological exposure occurring, these relatively minor exceedances of the surface water benchmarks by the maximum detected concentrations of these compounds in groundwater suggests that future risks associated with the groundwater discharge will similarly be minimal.

4.2.5.2 Risks to Semi-aquatic Wildlife Receptors in Barge Canal. Table 4-11 presents a comparison of the total body dose estimates to ingestion toxicity values for each of the three analytes detected in surface water. In all cases, HQs are several orders of magnitude below 1. These results indicate that risks to semi-aquatic wildlife receptors associated with exposure to pyridine compounds and p-fluoroaniline in the Erie Barge Canal are virtually non-existent. The semi-aquatic wildlife risk estimates are based on extremely conservative exposure assumptions.

4.2.5.3 Risks to Terrestrial Plants and Invertebrates. Risks to terrestrial plants and invertebrates were evaluated by comparing detected surface soil concentrations with available RTVs, and are presented in Tables 6-12 and 6-13 in the Phase I RI. Plant screening benchmarks for aluminum, chromium, lead, vanadium, and zinc were exceeded by the maximum concentrations of these inorganics detected in the 6 surface soil samples evaluated. Invertebrate screening benchmarks were exceeded by the maximum detected concentrations of chromium, copper, and zinc. HIs based on a comparison of the maximum detected surface soil CPC concentrations to toxicity screening benchmark values were 980 and 7.5 for plants and soil invertebrates, respectively.

As many of these potential risk drivers are naturally-occurring analytes, these risk estimates should be viewed in the context of background conditions. Although no site background data are available, regional background concentrations were compared to the surface soil inorganic analytical data (Table 4-10). The maximum detected concentration of chromium, mercury, nickel, silver, and zinc in the surface soil dataset evaluated in the Phase I ERA, exceeded the maximum range of background levels. Several analytes, including aluminum, lead, and vanadium, contributed to the plant risk estimates but were detected at maximum concentrations that are well within reported background concentration ranges. The HQ for aluminum represents nearly 90% of the HI for potential phytotoxicological effects; this inorganic was detected at a maximum concentration (8,700 mg/kg) which is considerably below the background concentration for the eastern USA (33,000 mg/kg). Chromium was also a substantial plant risk contributor, with a HQ of 75. The maximum chromium concentration is approximately 3 times greater than the maximum concentration detected in background samples; the average chromium concentration falls within the background range. It is important to note that the screening benchmark phytotoxicological value for chromium (2 mg/kg) is equal to the low end of the range of background concentrations. Consequently, the screening benchmark value for chromium is overly conservative because it is unlikely that plants could be adversely affected in most background situations. The average and maximum detected concentrations of zinc exceeded the maximum background level, however this analyte only contributed approximately 1 percent of the total potential risk to plants.

The Phase I ERA suggested that exposure to the maximum detected concentrations of chromium, copper, and zinc could potentially adversely affect soil invertebrates, although the risks appear to be minimal. Of these three risk contributors, copper was detected at concentrations that appear consistent with regional background concentrations (Table 4-10). The maximum detected concentrations of chromium and zinc exceed background ranges; the average zinc concentration also exceeds maximum background concentration (Table 4-10). Ecological risks associated with soil invertebrate exposures to these inorganic soil constituents are possible, although the toxicological benchmarks employed in the Phase I ERA are intended for use in risk screening only (Will and Suter, 1994). As indicated in the Phase I ERA, on-site habitat conditions appear to be most limiting to these receptor populations.

4.2.6 Ecological Risk Assessment Uncertainties

The general risk assessment uncertainties are discussed in the Phase I ERA. The risk uncertainties that apply to this assessment relate to the limited toxicological data available for all surface water analytes with the exception of pyridine. The RTVs developed for chloropyridines are considerably larger than the aquatic benchmark established for pyridine. This result is contrary to the anticipated increase in toxicological effect associated with increasing chlorination, which is also predicted with the ECOSAR program. It is likely that this effect is principally due to the large amount of toxicological data available for pyridines, and the inclusion of west coast salmonid species in the genus *Oncorhynchus* (e.g, pink salmon, chinook salmon, chum salmon, and coho salmon) that appear to be particularly sensitive to pyridine exposure. Although rainbow trout (*Oncorhynchus mykiss*) is distributed throughout the east coast, it does not occur in the warm water Erie Barge Canal (NYSDEC, 1994). The carp (*Cyprinus carpio*) is the species with the lowest LC_{50} value that would be expected to occur in this habitat, and this LC_{50} is 25 times greater than the sensitive pink salmon.

Although there is considerable uncertainty involved with exposure modeling to semi-aquatic wildlife, the fact that no risks were evident using very conservative exposure assumptions suggests that these uncertainties would not affect the general conclusions of this ERA.

4.2.7 Ecological Risk Assessment Summary and Conclusions

A supplemental ERA was conducted to address certain information gaps identified during the Phase I RI. In particular, the potential ecological risks associated with off-site surface water exposures in the vicinity of the Olin Plant site were evaluated in this ERA. This assessment focused on aquatic receptor and semi-aquatic wildlife exposures to selected pyridine compounds in the Erie Barge Canal. Measured, rather than modeled, surface water analytical data were used to assess the likelihood of adverse impacts to ecological receptor populations that exist in this habitat. Aquatic toxicity benchmarks were developed for all surface water analytes and were compared to the detected estimated surface water concentrations. Food chain-related exposures by semi-aquatic receptors were evaluated using bioconcentration factors to estimate fish tissue concentrations. Finally, potential risk associated with on-site surface soil exposure were reexamined based on a consideration of regional background conditions. The conclusions are listed below.

- Estimated concentrations of the three surface water analytes detected in the Erie Barge Canal were lower than all toxicity benchmarks for aquatic receptors. Consequently, no adverse impacts to these receptors would be anticipated.

- Due to the low-magnitude, low frequency detections of estimated concentrations, and the low uptake potential of the surface water analytes, bioconcentration hazards to semi-aquatic wildlife are considered insignificant.

- Based on concentrations of pyridines detected in Phase II wells adjacent to the Erie Barge Canal, no adverse effects to ecological receptors were identified in the ERA should undiluted groundwater discharge into the canal.

- The on-site surface soil constituents, chromium and zinc, exceed regional background concentration ranges and available toxicological benchmarks. However, the benchmarks are considered to be overly-conservative and habitat constraints most likely limit plant and soil invertebrate populations at the Olin Plant site.

5.0 CONCLUSIONS

This section presents the conclusions from the Phase II RI for: 1) assessment of the recovery well system, 2) the distribution, fate, and transport of chemicals, and 3) the human health and ecological risk assessments.

5.1 RECOVERY WELL SYSTEM (AQUIFER TESTING)

Preliminary testing of six overburden wells, proposed for aquifer testing, found most were capable of producing low yields (0.1 gpm or less). Efforts to improve yields through well rehabilitation were unsuccessful. These low yields observed in overburden wells are likely due to either natural properties of the overburden or well/formation clogging.

The pumping test performed in overburden extraction well W-1, a higher-yielding overburden well, indicated that the aquifer transmissivity is likely between 1.5 and 340 ft^2/d. This wide range of values resulted from limitations in the W-1 test data, caused by the influence of a precipitation event on groundwater levels. Regardless of where the actual overburden transmissivity lies within this range, it appears unlikely that the existing overburden extraction well spacing is achieving complete capture of overburden groundwater migrating off-site. A much closer well spacing would be required to achieve capture because the small saturated thickness in the overburden limits the area of groundwater flow an individual well can capture. An extraction well spacing of approximately 25 feet appears necessary for overburden wells to achieve capture.

Pumping tests performed in bedrock extraction wells BR-6A and BR-7A indicated that shallow bedrock aquifer transmissivity ranges from 250 to 350 ft^2/d, and that these wells are each capable of yields of more than 50 (BR-6A) and 20 (BR-7A) gpm. These results indicate that pumping from these wells should be capable of capturing shallow bedrock groundwater migrating off-site to the south and southwest. In addition, pumping from these wells may also be capable of effecting either partial or complete capture in the overburden by creating bedrock drawdowns that cause either an increase in vertical flow from the overburden or dewatering of the overburden. Additional analysis will be required to evaluate this possibility.

5.2 DISTRIBUTION, FATE, AND TRANSPORT OF CHEMICALS

The Phase II RI provided additional understanding of the distribution fate, and transport of site-related chemicals, as summarized in the following subsections.

5.2.1 Lab Sample Area

Subsurface Soil. Analytical results from subsurface soil samples at three additional locations at the Lab Sample Area show relatively low concentrations of site-related chemicals. Results from these borings and from the Phase I RI suggest this area is not a high-concentration source of site-related chemicals in groundwater.

Surface Soil. Mercury analysis results for two additional surface soil samples (SS-116 and SS-117) were one or more orders of magnitude less than the previously reported high concentration sample (SS-103), and show that the higher mercury concentration is isolated.

Groundwater. Carbon tetrachloride and methylene chloride were detected at high concentrations in one boring (SB-3). However, based on the soil sample results from this area, the presence of these chemicals is not likely related to the Lab Sample Area. These detections were interpreted to be part of an area-wide plume rather than the result of a chemical source in the Lab Sample Area.

5.2.2 Downgradient Investigations

Downgradient well installations and sampling, and surface water sampling provided additional information about the off-site extent of site-related chemicals in groundwater and surface water.

5.2.2.1 Groundwater.

Overburden. The areal distribution of site-related chemicals in overburden groundwater has, in general, been delineated. Site-related chemicals are interpreted to have not migrated beyond the new overburden well to the southeast (MW-114). To the west of the Olin Plant site, the overburden becomes unsaturated. Here the limit of saturation marks the western extent of chemicals in overburden groundwater. The Phase II analytical results support the findings of the Phase I RI, that concentrations of pyridines are distributed more widely than any other group of site related chemicals. Based on the analytical results, no additional overburden groundwater investigations are recommended.

Bedrock. Analytical results show site-related chemicals, specifically pyridines and selected VOCs, are present south and southwest of the Olin Plant site. Pyridines are distributed as far west as the Dolomite Products Quarry in the Town of Gates, where they were detected in groundwater seep samples. Pyridines have not been detected in the water that is pumped from the quarry to the Erie Barge Canal.

Neither the southern nor the western extent of site-related chemicals in bedrock groundwater has been fully delineated. Additional bedrock groundwater sampling in each of the these directions would be required to characterize the areal extent of these constituents.

5.2.2.2 Surface Water Surface water analytical results from the Erie Barge Canal show detections of chloropyridines in two out of five quarterly sampling events at upstream and downstream locations. Each detection is at an estimated concentration (less than the detection limit of 10 µg/L). The estimated concentrations were detected at SW-1, located upstream and upgradient of the Olin Rochester Plant site, as well as at the two downstream sampling locations conducted as part of the Phase II investigations. Monitoring conducted subsequent to the Phase II investigations showed detections of chloropyridines at SW-1, SW-2, SW-3, and locations both upstream and downstream. These results have been provided to NYSDEC in Olin's quarterly reports.

5.3 HUMAN HEALTH AND ECOLOGICAL RISK ASSESSMENT

The baseline risk assessment is summarized in the following subsections:

5.3.1 Human Health Risk Assessment

Health risks were evaluated for potential exposures to off-site media at the Olin Plant, including groundwater (and associated inhalation exposures), surface water, and groundwater seeps. Chemicals of potential concern (CPCs) were selected on a medium- and location-specific basis. Generally, the CPCs identified were VOCs (particularly chlorinated compounds), SVOCs (primarily chloropyridines), and inorganics.

The exposure scenarios quantitatively evaluated include:

- current and potential future recreational exposures to surface water in the ErieBarge Canal,

- current and potential future industrial worker exposures to groundwater seeps at the Dolomite Products Quarry, and
- potential future industrial/commercial worker exposures to groundwater;

The USEPA acceptable risk level for noncarcinogenic risk of an HI of 1 was used to characterize potential noncancer risks. The USEPA risk range of 1×10^{-4} to 1×10^{-6} for carcinogenic risk was used to characterize potential cancer risks. Potential health risks characterized as exceeding the USEPA-acceptable risk level or range are considered significant, whereas those exceeding a level of 1×10^{-6} are considered by the NYSDOH to require additional evaluation (i.e., determine whether specific remedial goals need to be developed). The results of the risk assessment are as follows:

- No significant human health risks were identified for potential exposures to surface water in the Erie Barge Canal or to groundwater seeps in the Dolomite Products Quarry under current or potential future land use conditions; cancer risks did not exceed an excess lifetime cancer risk of 1×10^{-6}, and non-cancer risks did not exceed a hazard index of 1.
- Evaluation of potential future worker exposure to off-site groundwater used as industrial process water identified cancer risks above 1×10^{-6} but below 1×10^{-4} (i.e., within the USEPA acceptable cancer risk range) for mean groundwater concentrations. For maximum concentrations, however, carcinogenic risks were calculated to be 9×10^{-4}, above the upper end of the USEPA acceptable cancer risk range. Calculated non-cancer risks were elevated, but the majority of that risk appears to be attributable to iron and zinc associated with turbidity in groundwater samples. In addition, the iron and zinc concentrations detected in off-site wells were higher than any detections on-site. Therefore, risks are unlikely to be related to Olin operations. Because risks for potential future exposures to off-site groundwater used as industrial process water exceeded an excess lifetime cancer risk of 1×10^{-6} and a non-cancer hazard index of 1, remedial goals will be calculated for the Feasibility Study.
- Modelling a hypothetical future release of VOCs from groundwater used as process water to industrial facility air did not result in any exceedances of workplace air standards.

- Groundwater exceeded MCLs and New York Standards for several CPCs. No domestic use of the groundwater is anticipated. For aesthetic reasons, groundwater in the vicinity of the Olin Study Area is not used as a drinking water source. Naturally-occurring sulfide and dissolved gases preclude use of bedrock groundwater for drinking water.

5.3.2 Ecological Risk Assessment

A supplemental ERA was conducted to address certain information gaps identified during the Phase I RI, in particular, the potential ecological risks associated with off-site surface water exposures in the vicinity of the Olin Plant. This assessment focused on aquatic receptor and semi-aquatic wildlife exposures to selected pyridine compounds in the Erie Barge Canal. Measured, rather than modeled, surface water analytical data were used to assess the likelihood of adverse impacts to ecological receptor populations that exist in this habitat. Aquatic toxicity benchmarks were developed for all surface water analytes and were compared to the detected estimated surface water concentrations. Food chain-related exposures by semi-aquatic receptors were evaluated using bioconcentration factors to estimate fish tissue concentrations. Finally, potential risk associated with on-site surface soil exposure was reexamined based on a consideration of regional background conditions. The conclusions are listed below.

- Estimated concentrations of the three surface water analytes detected in the Erie Barge Canal were lower than all toxicity benchmarks for aquatic receptors. Consequently, no adverse impacts to these receptors would be anticipated.

- Due to the low-magnitude, low frequency detections of estimated concentrations of pyridines, and the low uptake potential of the surface water analytes, bioconcentration hazards to semi-aquatic wildlife are considered insignificant.

- Based on concentrations of pyridines detected in Phase II wells adjacent to the Erie Barge Canal, no adverse effects to ecological receptors were identified in the ERA should undiluted groundwater discharge into the canal.

- The on-site surface soil constituents chromium and zinc exceed regional background concentration ranges and available toxicological benchmarks. However, habitat constraints most likely limit plant and soil invertebrate populations at the Olin Plant site. The site cover is sparsely vegetated and provides poor habitat for these populations.

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6.0 NEXT STEPS

Based on the information collected during the Phase I and II RIs and previous investigations, the following tasks are planned as future work:

- Continue to monitor surface water from the Erie Barge Canal for pyridines, to assess impacts to the Erie Barge Canal.
- Periodically collect and analyze groundwater seeping into the quarry and surface water discharging from the quarry into the Erie Barge Canal for the presence of pyridines at the Dolomite Products Quarry.
- Sample for pyridines from selected monitoring and/or industrial wells located west of the Erie Barge Canal and south of the Ness site to further assess the off-site distribution of pyridines in bedrock groundwater.
- Based on off-site results from the selected monitoring and industrial wells located west of the Erie Barge Canal, consider the installation of new bedrock monitoring wells to further evaluate the distribution of chemicals in groundwater.
- Further evaluate performance of the on-site groundwater recovery system and consider options for adjusting pumping rates to optimize capture.

GLOSSARY OF ACRONYMS AND ABBREVIATIONS

ACGIH	American College of Governmental and Industrial Hygienists
AQUIRE	aquatic toxicity database
ASP	Analytical Services Protocol
ATSDR	Agency for Toxic Substances and Disease Registry
BCF	bioconcentration factor
bgs	below ground surface
BTEX	benzene, toluene, ethylbenzene, xylenes
CAS	Chemical Abstract Service
cm/sec	centimeters per second
CPCs	chemicals of potential concern
CSFs	cancer slope factors
2-CPL	2-chloropyridine
2,6-CPL	2,6-dichloropyridine
3-CPL	3-chloropyridine
4-CPL	4-chloropyridine
CSOAP	Combined Sewer Overflow Abatement Program
CV	chronic values
1,1-DCA	1,1-dichloroethane
1,1-DCE	1,1-dichloroethene
1,2-DCE	1,2-dichloroethene
ECOSAR	ecological structure activity relationship
EPC	exposure point concentration
ERA	ecological risk assessment
FID	flame ionization detector
ft	feet
ft/ft	feet per foot
gpm	gallons per minute
GPR	ground-penetrating radar
HEAST	Health Effects Assessment Summary Tables
HI	hazard index
HQ	hazard quotient
ID	inside diameter

IRIS	Integrated Risk Information System
J	estimated
K_{OC}	organic carbon partition coefficient
LOAEL	lowest observed adverse effects level
mg/kg	milligrams per kilogram
MS	matrix spike
MSB	matrix spike blank
MSBD	matrix spike blank duplicate
MSD	matrix spike duplicate
NCEA	National Center for Environmental Assessment
NYSDEC	New York State Department of Environmental Conservation
Olin	Olin Corporation, Chemicals Division
PCE	tetrachloroethene
PEL	permissible exposure limits
P-FAE	p-fluoroaniline
PID	photoionization detector
POTW	publicly-owned treatment works
PQL	practical quantitation limit
PVC	polyvinyl chloride
QA	quality assurance
QAPP	Quality Assurance Project Plan
QC	quality control
RfD	reference dose
RI	remedial investigation
R_{ij}	risk factor
RME	reasonable maximum exposure
RTVs	reference toxicity values
S	coefficient
SAR	structure activity relationship
SQL	Sample Quantitation Limit
SVOCs	semivolatile organic compounds

TAGM	Technical and Administrative Guidance Memorandum
TAL	Target Analyte List
1,1,1-TCA	1,1,1-trichloroethane
TCE	trichloroethene
TCL	Target Compound List
TIC	tentatively identified compounds
USEPA	U.S. Environmental Protection Agency
μg/kg	micrograms per kilogram
μg/L	micrograms per liter
VC	vinyl chloride
VOCs	volatile organic compounds

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TABLE 2-2
BACKGROUND COMPARISON OF INORGANICS CONCENTRATIONS
SURFACE AND SUBSURFACE SOIL SAMPLES

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, N.Y.

ANALYTE	FREQUENCY OF DETECTION	RANGE		EASTERN U.S. BACKGROUND (ppm)(1)	USEPA REGION III RBC for INDUSTRIAL SOIL (mg/kg)(4)	LOCATIONS EQUAL TO OR GREATER THAN BACKGROUND																				
		MINIMUM	MAXIMUM			SS-101	SS-102	SS-103	SS-104	SS-105	SS-106	SS-107	SS-108	SS-109	SS-110	SS-111	SS-112	SS-113	SS-114	SS-115	SS-116	SS-117	SB-1	SB-2	SB-3	# Detections > Background
Aluminum	18 / 18	2700	18600	33000	NA																-	-				
Antimony	0 / 18	NA	NA	NA	410																-	-				
Arsenic	18 / 18	1.7	12	3 - 12 (1)	310													X			-	-				1
Barium	18 / 18	18.2	210	15 - 600	72000																-	-				
Beryllium	0 / 18	NA	NA	0 - 1.75	0.67																-	-				
Cadmium	15 / 18	0.1	1.8	0.1 - 1	510			X	X												-	-				2
Calcium	18 / 18	4900	97700	130 - 35000(2)	NA				X	X							X			X	-	-	X	X	X	7
Chromium	18 / 18	5.3	180	1.5 - 40(2)	5100		X		X												-	-		X		3
Cobalt	14 / 18	4.55	16.6	2.5 - 60(2)	61000																-	-				
Copper	18 / 18	2	300	1 - 50	38000				X		X										-	-				2
Cyanide	1 / 18	59.7	59.7	NA	20000																-	-				
Iron	18 / 18	6900	35800	17500 - 25000(2)	NA																-	-		X		1
Lead	18 / 18	4.9	530	200 - 500(3)	NA				X												-	-				1
Magnesium	18 / 18	2700	58900	100 - 5000	NA		X		X	X	X		X	X	X	X	X		X	X	-	-	X	X	X	14
Manganese	18 / 18	240	1300	50 - 5000	5100																-	-				
Mercury	14 / 20	0.04	214	0.001 - 0.2	310	X	X	X	X	X	X			X		X				X		X		X		11
Nickel	17 / 18	4.1	49	0.5 - 25	20000	X	X		X						X					X	-	-		X		6
Potassium	18 / 18	590	1900	8500 - 43000(2)	NA																-	-				
Selenium	1 / 18	0.8	1.4	0.1 - 3.9	5100																-	-				
Silver	16 / 18	0.1	0.95	NA	5100																-	-				
Sodium	18 / 18	222	2050	6000 - 8000	NA																-	-				
Thallium	0 / 18	NA	NA	NA	NA																-	-				
Vanadium	18 / 18	3.3	37.5	1 - 300	7200																-	-				
Zinc	18 / 18	24.1	640	9 - 50	310000	X	X	X	X	X	X	X		X	X	X	X	X		X	-	-		X		14
Number of analytes exceeding background -						3	5	3	9	4	4	1	1	3	3	3	3	2	1	5	0	1	2	7	2	

NOTE:

Frequency and range calculated from 15 surface soil samples collected during the Phase I RI (SS-101 to SS-115), 3 subsurface soil samples collected during the Phase II RI (SB-1, SB-2, and SB-3), and 2 surface soil samples (SS-116 and SS-117) collected during the Phase II RI for mercury only.

(1) NYSDEC, Division of Technical and Administrative Guidance Memorandum on Determination of Soil Cleanup Objectives and Cleanup Levels, January, 1994.
(2) New York State background
(3) Typical average background levels in metropolitan or surburban areas or near highways, NYSDEC, 1994.
(4) USEPA, Region III, Risk-Based Concentration (RBC) Table, 3rd Quarter, 1994.

NA Background data or risk-based concentration not available

TABLE 2-1
NON-NUTRIENT INORGANICS DETECTED IN SUBSURFACE SOIL LAB SAMPLE DISPOSAL AREA

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

INORGANIC	FREQUENCY OF DETECTS	MEAN	RANGE (mg/kg) MINIMUM	MAXIMUM	LOCATION OF MAXIMUM
Aluminum	3 / 3	8865	3720	18600	SB-2
Arsenic	3 / 3	5.12	1.7	11.3	SB-2
Barium	3 / 3	44.1	18.2	92.3	SB-2
Cadmium	1 / 3	1	1	1	SB-3
Chromium	3 / 3	21.5	5.3	52.6	SB-2
Cobalt	2 / 3	10.6	4.4	16.6	SB-2
Copper	3 / 3	15.1	2	24.8	SB-2
Cyanide	1 / 3	59.7	59.7	59.7	SB-2
Lead	3 / 3	29.8	4.9	77.7	SB-2
Manganese	3 / 3	740	350	1300	SB-2
Mercury	3 / 3	0.17	0.04	0.42	SB-2
Nickel	3 / 3	17.6	4.1	41.1	SB-2
Silver	1 / 3	0.95	0.95	0.95	SB-2
Vanadium	3 / 3	10.5	3.3	24.1	SB-2
Zinc	3 / 3	110	24.1	272	SB-2

Note:

(1) Analytes that are essential human nutrients include: calcium, iron, magnesium, potassium, and sodium)

TABLE 2-3
HYDRAULIC CONDUCTIVITY TEST RESULTS

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, N.Y.

WELL	TYPE			K (cm/sec)	Kave(cm/sec)
SHALLOW BEDROCK WELLS					
BR-111	FALLING HEAD			1.53E-02	
	RISING HEAD			5.26E-02	
					3.4E-02
BR-112A	FALLING HEAD			1.40E-04	
	RISING HEAD			1.40E-04	
					1.4E-04
BR-113	FALLING HEAD			7.33E-02	
	RISING HEAD			8.00E-02	
					7.7E-02
BR-114	FALLING HEAD			2.80E-02	
	RISING HEAD			3.40E-02	
					3.1E-02
RANGE		**8.0E-02**	**to**	**1.4E-04**	
DEEP BEDROCK WELLS					
BR-111D	FALLING HEAD			3.20E-03	
	RISING HEAD			3.00E-03	
					3.1E-03
BR-112D	FALLING HEAD			7.32E-03	
	RISING HEAD			6.60E-03	
					7.0E-03
BR-113D	FALLING HEAD			6.50E-03	
	RISING HEAD			6.90E-03	
					6.7E-03
RANGE		**7.3E-03**	**to**	**3.0E-03**	
OVERBURDEN WELLS					
MW-114	INSUFFICIENT WATER DEPTH AVAILABLE FOR TESTING				

NOTES:

K (cm/sec)	Hydraulic conductivity calculated in centimeters per second
Kave (cm/sec)	Average (arithmetic mean) hydraulic conductivity from rising and falling he

TABLE 2-4
SELECTED CHLOROPYRIDINE CONCENTRATIONS (1) IN OVERBURDEN GROUNDWATER

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

WELL	DATE	2,6-CPL	2-CPL	3-CPL	4-CPL	Pyridine	p-FAE	SUM OF VALUES (2)
ONSITE WELLS								
B-1	12-Sep-95	24	12	-	NA	NA	-	36
B-17	13-Sep-95	19000 J	120000	6200	NA	NA	400 E	145600
B-6	12-Sep-95	11000	42000	770	NA	NA	180 J	53950
E-1	12-Sep-95	350	1400	79	NA	NA	6 J	1835
E-3	12-Sep-95	120	82	-	NA	NA	29	231
OFFSITE WELLS								
MW-103	11-Sep-95	1 J	23	-	NA	NA	-	24
MW-104	11-Sep-95	51	130	-	NA	NA	-	181
MW-106	11-Sep-95	15000 J	84000	4000	NA	NA	320	103000
MW-107	11-Sep-95	1 J	14	-	NA	NA	-	15
MW-114	7-Dec-95	-	-	-	-	-	-	ND

Notes:

(1) Concentrations reported in micrograms per liter (ug/l); "-" represents not detected.
(2) Sum of 2,6-DCPE, 2-CPE, 3-CPE, and p-FAE used in contour plots shown in Section 3.0.

NA = Not analyzed

2,6-DCL = 2,6-Dichloropyridine
2-CPL = 2-Chloropyridine
3-CPL = 3-Chloropyridine
4-CPL = 4-Chloropyridine
p-FAE = p-Fluaroaniline

TABLE 2-5
SELECTED CHLOROPYRIDINE CONCENTRATIONS (1) IN BEDROCK GROUNDWATER

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

WELL	TYPE	DATE	2,6-CPL	2-CPL	3-CPL	4-CPE	Pyridine	p-FAE	SUM OF VALUES (2)
ONSITE WELLS									
BR-1		12-Sep-95	-	14	-	NA	NA	-	14
BR-101		11-Sep-95	2300	6700	560	NA	NA	-	9560
BR-102		12-Sep-95	110	660	10	NA	NA	17	797
BR-2		12-Sep-95	250	1600	38	NA	NA	24	1912
BR-2D		14-Sep-95	4 J	46	3 J	NA	NA	-	53
BR-3		12-Sep-95	9000	69000	4600	NA	NA	250	82850
BR-3D		14-Sep-95	1 J	91	6 J	NA	NA	-	98
BR-4	Duplicate	12-Sep-95	68	190	8 J	NA	NA	-	266
BR-4		12-Sep-95	77	220	8 J	NA	NA	-	305
BR-5A		12-Sep-95	82	230	2 J	NA	NA	37	351
BR-6		12-Sep-95	8800 J	74000	3300	NA	NA	25 J	86125
BR-8	Duplicate	13-Sep-95	1100	4400	72 J	NA	NA	210	5782
BR-8		13-Sep-95	1200	4900	130	NA	NA	180	6410
OFFSITE WELLS									
BR-103		11-Sep-95	0.5 J	15	-	NA	NA	-	15.5
BR-104		11-Sep-95	59	140	-	NA	NA	-	199
BR-105		11-Sep-95	750	12000	210	NA	NA	14	12974
BR-105D		11-Sep-95	32	1300	15	NA	NA	4 J	1351
BR-106		11-Sep-95	810	5800	250	NA	NA	24	6884
BR-107		11-Sep-95	-	17	-	NA	NA	-	17
BR-111		7-Dec-95	-	-	-	-	-	-	ND
BR-111D		7-Dec-95	-	-	-	-	-	-	ND
BR-112A		7-Dec-95	-	-	-	-	-	-	ND
BR-112D		7-Dec-95	-	4 J	-	-	-	-	4
BR-113	Duplicate	7-Dec-95	-	2 J	-	-	-	-	2
BR-113		7-Dec-95	-	2 J	-	-	-	-	2
BR-113D		7-Dec-95	1 J	76	-	-	-	-	77
BR-114		7-Dec-95	6 J	12	8 J	-	-	-	26
NESS-E		12-Sep-95	140	2200	8 J	NA	NA	-	2348
NESS-E		7-Dec-95	97	1300	-	-	-	2 J	1399
NESS-W		12-Sep-95	12	600	14	NA	NA	1 J	627
NESS-W		7-Dec-95	11	-	150	6 J	-	-	161

Notes:

(1) Concentrations reported in micrograms per liter (ug/l); "-" represents not detected.
(2) Sum of 2,6-DCPE, 2-CPE, 3-CPE, and p-FAE used in contour plots shown in Section 3.0.
NA = Not analyzed

2,6-CDL = 2,6-Dichloropyridine
2-CPL = 2-Chloropyridine
3-CPL = 3-Chloropyridine
4-CPL = 4-Chloropyridine
p-FAE = p-Fluaroaniline

TABLE 2-6
SELECTED VOC CONCENTRATIONS (1) IN OVERBURDEN GROUNDWATER

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

WELL	DATE	1,1,1-TCA	1,1-DCA	1,2-DCE	CCL4	CHCL3	MECL	PCE	TCE	SUM OF VALUES (2)
ONSITE WELLS										
B-1	12-Sep-95	-	-	-	-	-	-	-	-	ND
B-17	13-Sep-95	-	-	-	100000	35000	2800	3100	-	140900
B-6	12-Sep-95	-	-	2 J	-	-	-	-	-	2
E-1	12-Sep-95	-	-	10 J	420	680	33 J	17 J	-	1160
E-3	12-Sep-95	-	-	7 J	-	-	-	-	2 J	9
OFFSITE WELLS										
MW-103	11-Sep-95	-	-	-	-	-	-	-	-	ND
MW-104	11-Sep-95	-	-	-	-	-	-	-	1 J	1
MW-106	11-Sep-95	-	-	9 J	-	89	-	-	-	98
MW-107	11-Sep-95	-	2 J	-	-	-	-	-	-	2
MW-114	7-Dec-95	-	0.85	-	-	1.8	-	3.1	5.1	10.85

Notes:

(1) Concentrations reported in micrograms per liter (ug/l); "-" represents not detected.
(2) Sum of values used in contour plots shown in Section 3.0.

1,1,1-TCA = 1,1,1-Trichloroethane
1,1-DCA = 1,1-Dichloroethane
1,2-DCE = 1,2-Dichloroethene (Total)
CCL4 = Carbon Tetrachloride
TCE = Trichloroethene

CHCL3 = Chloroform
MECL = Methylene Chloride
PCE = Tetrachloroethene
TCE = Trichloroethene

TABLE 2-7
SELECTED VOC CONCENTRATIONS (1) IN BEDROCK GROUNDWATER

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

WELL	SAMPLE TYPE	DATE	1,1,1-TCA	1,1-DCA	1,2-DCE	CCL4	CHCL3	MECL	PCE	TCE	SUM OF VALUES (2)
ONSITE WELLS											
BR-1		12-Sep-95	-	-	-	-	-	-	-	-	ND
BR-101		11-Sep-95	-	-	-	-	5400	31000	-	-	36400
BR-102		12-Sep-95	-	-	-	19 J	340	220	17 J	5 J	601
BR-2		12-Sep-95	-	-	73 J	14000	14000	9000	180	63 J	37316
BR-2D		14-Sep-95	-	-	1 J	-	5 J	-	-	-	6
BR-3		12-Sep-95	-	-	-	1700 J	14000	24000	-	-	39700
BR-3D		14-Sep-95	-	-	-	-	-	170	-	-	170
BR-4	Duplicate	12-Sep-95	-	-	2 J	-	-	3 J	-	-	5
BR-4		12-Sep-95	-	-	2 J	-	-	3 J	-	-	5
BR-5A		12-Sep-95	-	-	610	-	65	-	-	75	750
BR-6		12-Sep-95	-	-	-	750	6700	1400	190 J	-	9040
BR-8	Duplicate	13-Sep-95	-	-	18 J	-	-	-	-	-	18
BR-8		13-Sep-95	-	-	17 J	-	-	-	-	-	17
OFFSITE WELLS											
BR-103		11-Sep-95	-	-	24	-	-	-	-	-	24
BR-104		11-Sep-95	-	-	-	-	-	-	-	-	ND
BR-105		11-Sep-95	-	2 J	5 J	-	-	-	2 J	4 J	13
BR-105D		11-Sep-95	-	6 J	59	-	-	-	-	-	65
BR-106		11-Sep-95	-	28 J	490	-	7 J	-	-	9 J	534
BR-107		11-Sep-95	-	6 J	360	-	-	-	-	-	366
BR-111		26-Oct-95	-	-	-	-	-	-	-	-	ND
BR-111D		26-Oct-95	-	-	-	-	-	-	-	-	ND
BR-112A		27-Oct-95	-	-	-	-	0.41 J	-	-	-	0.41
BR-112D		27-Oct-95	0.89	35	48	-	-	-	-	2.4	86.29
BR-113	Duplicate	26-Oct-95	-	-	-	-	-	-	-	-	ND
BR-113		26-Oct-95	-	-	-	-	-	-	-	-	ND
BR-113D		26-Oct-95	0.85	35	36	-	1	-	-	1.6	74.45
BR-114		27-Oct-95	-	0.95	-	-	-	-	-	-	0.95
NESS-E		12-Sep-95	-	6 J	4 J	-	-	-	3 J	9 J	22
NESS-E	Duplicate	20-Nov-95	-	4.8	4.9	-	1	-	3.9	11	25.6
NESS-E		20-Nov-95	-	4.5	4.9	-	1.2	-	3.8	12	26.4
NESS-W		12-Sep-95	-	7 J	14	-	-	3 J	2 J	2 J	28
NESS-W		20-Nov-95	-	6.3	46	-	1.4	2.8 J	1.3	1.5	59.3

Notes:

(1) Concentrations reported in micrograms per liter (ug/l); "-" represents not detected.
(2) Sum of values used in contour plots shown in Section 3.0.

1,1,1-TCA = 1,1,1-Trichloroethane
1,1-DCA = 1,1-Dichloroethane
1,2-DCE = 1,2-Dichloroethene (Total)
CCL4 = Carbon Tetrachloride

CHCL3 Chloroform
MECL Methylene Chloride
PCE Tetrachloroethene
TCE Trichloroethene

TABLE 2-8
CHLORINATED ETHENE CONCENTRATIONS (1) IN OVERBURDEN GROUNDWATER

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

WELL	DATE	1,2-DCE	PCE	TCE	VC	SUM OF VALUES (2)
ONSITE WELLS						
B-1	12-Sep-95	-	-	-	-	ND
B-17	13-Sep-95	-	3100	-	-	3100
B-6	12-Sep-95	2 J	-	-	-	2
E-1	12-Sep-95	10 J	17 J	-	-	27
E-3	12-Sep-95	7 J	-	2 J	-	9
OFFSITE WELLS						
MW-103	11-Sep-95	-	-	-	-	ND
MW-104	11-Sep-95	-	-	1 J	-	1
MW-106	11-Sep-95	9 J	-	-	8 J	17
MW-107	11-Sep-95	-	-	-	-	ND
MW-114	7-Dec-95	-	3.1	5.1	-	8.2

Notes:

(1) Concentrations reported in micrograms per liter (ug/l); "-" represents not detected.
(2) Sum of values used in contour plots shown in Section 3.0.

1,2-DCE = 1,2-Dichloroethene (Total)
PCE = Tetrachloroethene
TCE = Trichloroethene
VC = Vinyl Chloride

TABLE 2-9
CHLORINATED ETHENE CONCENTRATIONS (1) IN BEDROCK GROUNDWATER

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

WELL	TYPE	DATE	1.2-DCE	PCE	TCE	VC	SUM OF VALUES (2)
BR-1		12-Sep-95	-	-	-	-	ND
BR-101		11-Sep-95	-	-	-	-	ND
BR-102		12-Sep-95	-	17 J	5 J	-	22
BR-2		12-Sep-95	73 J	180	63 J	17 J	333
BR-2D		14-Sep-95	1 J	-	-	-	1
BR-3		12-Sep-95	-	-	-	-	ND
BR-3D		14-Sep-95	-	-	-	-	ND
BR-4	Duplicate	12-Sep-95	2 J	-	-	17	19
BR-4		12-Sep-95	2 J	-	-	18	20
BR-5A		12-Sep-95	610	-	75	47	732
BR-6		12-Sep-95	-	190 J	-	-	190
BR-8	Duplicate	13-Sep-95	18 J	-	-	-	18
BR-8		13-Sep-95	17 J	-	-	-	17
OFFSITE WELLS							
BR-103		11-Sep-95	24	-	-	3 J	27
BR-104		11-Sep-95	-	-	-	-	ND
BR-105		11-Sep-95	5 J	2 J	4 J	2 J	13
BR-105D		11-Sep-95	59	-	-	30	89
BR-106		11-Sep-95	490	-	9 J	350	849
BR-107		11-Sep-95	360	-	-	280	640
BR-111		26-Oct-95	-	-	-	-	ND
BR-111D		26-Oct-95	-	-	-	-	ND
BR-112A		27-Oct-95	-	-	-	-	ND
BR-112D		27-Oct-95	48	-	2.4	67	117.4
BR-113	Duplicate	26-Oct-95	-	-	-	-	ND
BR-113		26-Oct-95	-	-	-	-	ND
BR-113D		26-Oct-95	36	-	1.6	33	70.6
BR-114		27-Oct-95	-	-	-	-	ND
NESS-E		12-Sep-95	4 J	3 J	9 J	4 J	20
NESS-E	Duplicate	20-Nov-95	4.9	3.9	11	4.9	24.7
NESS-E		20-Nov-95	4.9	3.8	12	4.3	25
NESS-W		12-Sep-95	14	2 J	2 J	9 J	27
NESS-W		20-Nov-95	46	1.3	1.5	60	108.8

Notes:

(1) Concentrations reported in micrograms per liter (ug/l); "-" represents not detected.
(2) Sum of values used in contour plots shown in Section 3.0.

1,2-DCE = 1,2-Dichloroethene (Total)
PCE = Tetrachloroethene
TCE = Trichloroethene
VC = Vinyl Chloride

TABLE 2-10
BTEX CONCENTRATIONS (1)
IN OVERBURDEN GROUNDWATER

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

WELL	DATE	Benzene	Ethylbenzene	Toluene	Total Xylenes	SUM OF VALUES
ONSITE WELLS						
B-1	12-Sep-95	-	-	-	-	ND
B-17	13-Sep-95	-	-	280 J	-	280
B-6	12-Sep-95	30	3 J	70	2 J	105
E-1	12-Sep-95	-	-	12 J	31 J	43
E-3	12-Sep-95	44	-	-	-	44
OFFSITE WELLS						
MW-103	11-Sep-95	-	-	-	-	ND
MW-104	11-Sep-95	-	-	-	-	ND
MW-106	11-Sep-95	190	8 J	2300	27 J	2525
MW-107	11-Sep-95	-	-	-	-	ND
MW-114	7-Dec-95	0.64	-	0.55	0.59	1.78

Notes:

(1) Concentrations reported in micrograms per liter (ug/l); "-" represents not detected.
(2) Sum of values used in contour plots shown in Section 3.0.

TABLE 2-11
BTEX CONCENTRATIONS (1) IN BEDROCK GROUNDWATER

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

WELL	TYPE	DATE	Benzene	Ethylbenzene	Toluene	Total Xylenes	SUM OF VALUES (2)
ONSITE WELLS							
BR-1		12-Sep-95	-	-	-	-	ND
BR-101		11-Sep-95	400 J	790 J	26000	3800	30990
BR-102		12-Sep-95	30	-	21 J	-	51
BR-2		12-Sep-95	28 J	-	84 J	-	112
BR-2D		14-Sep-95	3 J	-	4 J	7 J	14
BR-3		12-Sep-95	-	-	310 J	-	310
BR-3D		14-Sep-95	5 J	1 J	11	10	27
BR-4	Duplicate	12-Sep-95	-	-	-	-	ND
BR-4		12-Sep-95	-	-	-	-	ND
BR-5A		12-Sep-95	82	-	26 J	-	108
BR-6		12-Sep-95	-	-	260 J	-	260
BR-8	Duplicate	13-Sep-95	66	-	45 J	-	111
BR-8		13-Sep-95	68	-	43 J	-	111
OFFSITE WELLS							
BR-103		11-Sep-95	2 J	-	-	-	2
BR-104		11-Sep-95	-	-	-	-	ND
BR-105		11-Sep-95	10	-	2 J	-	12
BR-105D		11-Sep-95	10	-	6 J	2 J	18
BR-106		11-Sep-95	86	4 J	230	6 J	326
BR-107		11-Sep-95	110	-	6 J	6 J	122
BR-111		26-Oct-95	1.6	1.1	0.59	2.8	6.09
BR-111D		26-Oct-95	240	38	14	41	333
BR-112A		27-Oct-95	-	-	0.24 J	0.28 J	0.52
BR-112D		27-Oct-95	22	3.2	0.54	1.8	27.54
BR-113	Duplicate	26-Oct-95	30	62	140	340	572
BR-113		26-Oct-95	31	61	130	340	562
BR-113D		26-Oct-95	24	1.3	1.1	2	28.4
BR-114		27-Oct-95	0.58	1.1	0.25 J	2.1	4.03
NESS-E		12-Sep-95	4 J	-	-	-	4
NESS-E	Duplicate	20-Nov-95	2.8	0.26 J	0.32 J	0.25 J	3.63
NESS-E		20-Nov-95	2.3	0.27 J	0.32 J	0.23 J	3.12
NESS-W		12-Sep-95	34	10	4 J	5 J	53
NESS-W		20-Nov-95	35	7.3	3.1	5.2	50.6

Notes:

(1) Concentrations reported in micrograms per liter (ug/l); "-" represents not detected.
(2) Sum of values used in contour plots shown in Section 3.0.

TABLE 2-12
INORGANICS DATA SUMMARY (1)
FOR GROUNDWATER

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

INORGANIC	FREQUENCY	RANGE (ug/l) MINIMUM	MAXIMUM	MEDIAN [2] CONCENTRATION
Aluminum	10 / 10	86.6	16600	2105
Antimony	2 / 10	5.4	57.6	BDL
Arsenic	6 / 10	6.2	371	8
Barium	10 / 10	36.7	174	133
Beryllium	5 / 10	0.21	2.1	BDL
Cadmium	3 / 10	0.72	98.4	BDL
Calcium	10 / 10	135000	1220000	202500
Chromium	6 / 10	2	102	6
Cobalt	6 / 10	1.7	34.6	5
Copper	9 / 10	1.6	70700	15
Iron	10 / 10	2540	864000	18300
Lead	9 / 10	2.3	4750	18
Magnesium	10 / 10	28800	357000	77850
Manganese	10 / 10	54.1	8640	808
Mercury	3 / 10	0.54	7.7	BDL
Nickel	8 / 10	1.6	514	13
Potassium	10 / 10	4040	171000	12800
Selenium	2 / 10	7.1	17.7	BDL
Silver	3 / 10	1.1	33.4	BDL
Sodium	10 / 10	12200	6490000	124100
Vanadium	8 / 10	3.9	74.8	6
Zinc	10 / 10	17.6	2780000	110

Notes:

(1) Data reported for 9 offsite shallow and deep bedrock wells and 1 offsite overburden well; Samples collected November 20, 1995.

(2) Median concentration selected from average of 6th and 7th ranking values from population of 10 samples and 2 duplicate samples. "BDL" or below detection limit is reported for frequencies of 50% or less.

Table 2-13
BR-7A and BR-6A Pumping Test Results

Olin Chemicals Phase II RI Report
Rochester, NY

BR-7A Pumping Test						
Well	**Distance to BR-7A (ft)**	**Apparent Drawdown**	**Interpreted Maximum Drawdown**	**Analysis Method**	**T (ft^2/ day)**	**S**
BR-7A	0.25	7.2 ft at 1890 min	7.6	Harrill	350	N/A
PZ-105	145	0.44 ft at 1683 min	0.63	Theis (Jacob)	330 (400)	0.004 0.003
PZ-106	480	0.35 ft at 1688 min	0.46	Theis (Jacob)	300 (310)	0.001 0.0009
PZ-107	530	0.21 ft at 1685 min	0.41	Theis (Jacob)	300 (470)	0.0009 0.0007
PZ-103	155	0.10 ft at 1380 min	0.14	Theis (Jacob)	610 (880)	0.02 0.02
PZ-104	205	None	None	N/A		
BR-3A	265	None	None	N/A		
BR-8	265	None	None	N/A		

40' [illegible]

BR-6A Pumping Test						
Well	**Distance to BR-6A (ft)**	**Apparent Drawdown (ft)**	**Interpreted Maximum Drawdown**	**Analysis Method**	**T (ft^2/day)**	**S**
BR-6A	0.25	8.67 at 1060 min	8.34	Theis (Jacob)	250 (300)	NA
PZ-105	134	1.55 at 1060 min	1.23	Theis (Jacob)	700 (900)	0.0006 (0.0004)
PZ-106	245	1.00 at 1060 min	0.66	Theis (Jacob)	700 (870)	0.004 0.003
BR-2A	350	0.34 at 1028 min	0.20	Theis (Jacob)	1300 NA	0.009 NA
PZ-107	270	1.00 at 1060 min	0.7 (est)	No fit		
BR-7A	375	4.5 at 840 min	4.3 (est)	Insufficient data		
BR-3A	125	None	None	N/A		
PZ-103	400	None	None	N/A		

Notes:

T = Transmissivity
S = Storativity

TABLE 3-1
PHYSIO-CHEMICAL PROPERTIES OF SELECTED CHEMICALS

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, N.Y.

Chemical Name	CAS Number	Specific Gravity	Water Solubility (mg/L)	Vapor Pressure (mmHg)	Henry's Law (1) (atm-m3/mol)	Koc (ml/g) (2)
VOCs						
1,1,1-Trichloroethane	71-55-6	1.3492	1330	1.20E+02	2.80E-02	152
1,1-Dichloroethane	75-34-3	1.175	5500	2.30E+02	5.70E-03	30
1,1-Dichloroethene	75-35-4	1.218	2250	6.00E+02	1.54E-01	65
1,2-Dichlorobenzene	95-50-1	1.35	118	1.47E+00	1.90E-03	1700
1,2-Dichloroethane	107-06-2	1.25	7990	8.56E+01	1.10E-03	14
1,2-Dichloropropane	78-87-5	1.16	2700	4.20E+01	1.54E-01	51
2-Hexanone	591-78-6	0.83	35000	1.00E+01	3.78E-05	14.8
4-Methyl-2-pentanone	108-10-1	0.8006	19000	6.00E+00	6.77E-05	19
Acetone	67-64-1	0.791	miscible	2.70E+02	3.67E-05	2.2
Benzene	71-43-2	0.871	1750	9.52E+01	5.46E-03	65
Bromoform	75-25-2	2.89	3010	5.00E+00	5.32E-04	116
Carbon Disulfide	75-15-0	1.263	2940	3.60E+02	1.23E-02	54
Carbon Tetrachloride	56-23-5	1.59	758	1.10E+02	2.30E-02	439
Chlorobenzene	108-90-7	1.1	491	1.17E+01	3.40E-03	330
Chloroform	67-66-3	1.489	7220	1.51E+02	3.80E-03	44
Chloromethane	74-87-3	0.991	6500	7.60E+02	9.90E-03	5.5
Dibromochloromethane	124-48-1	2.38	4540	7.60E+01	4.59E-03	107
Ethyl Benzene	100-41-4	0.867	153	7.00E+00	8.43E-03	220
Methyl ethyl ketone	78-93-3	0.805	268000	7.75E+01	5.14E-05	4.51
Methylene chloride	75-09-2	1.325	18000	4.11E+02	2.60E-03	8.8
Tetrachloroethene	127-18-4	1.626	484	1.85E+01	2.30E-02	364
Toluene	108-88-3	0.867	1550	2.84E+01	6.60E-03	120
Trichloroethene	79-01-6	1.4679	1470	7.43E+01	8.90E-03	126
Vinyl Chloride	75-01-4	0.912	2670	7.60E+02	6.90E-01	8.2
p-Xylene	106-42-3	0.86	198	8.82E+00	7.04E-03	238
trans-1,2-Dichloroethene	156-60-5	1.26	6300	3.24E+02	6.60E-03	59
SVOCs						
1,2,4-Trichlorobenzene	120-82-1	1.574	48.8	2.90E-01	1.42E-03	9200
1,3-Dichlorobenzene	541-73-1	1.288	133	2.28E+00	3.60E-03	1700
1,4-Dichlorobenzene	106-46-7	1.458	73.8	1.18E+00	1.60E-03	1700
2,4,6-Trichlorophenol	88-06-2	1.49	434	1.20E-02	4.82E-06	2000
2,4-Dimethylphenol	105-67-9	1.036	7870	2.60E-02	5.31E-07	96
2,4-Dinitrotoluene	121-14-2	1.521	280	2.17E-04	1.86E-07	251
2,6-Dichloropyridine	2402-78-0	ND	ND	ND	ND	ND
2-Chloroethyl vinyl ether	110-75-8	1.048	18900	2.68E+01	1.99E-04	11.7

TABLE 3-1
PHYSIO-CHEMICAL PROPERTIES OF SELECTED CHEMICALS

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, N.Y.

Chemical Name	CAS Number	Specific Gravity	Water Solubility (mg/L)	Vapor Pressure (mmHg)	Henry's Law (1) (atm-m3/mol)	Koc (ml/g) (2)
2-Chlorophenol	95-57-8	ND	11400	2.35E+00	3.49E-05	73
2-Chloropyridine	109-09-1	1.205	2000	1.00E+00	ND	ND
2-Methylnaphthalene	91-57-6	0.994	25.4	4.50E-02	3.31E-04	7940
3-Chloropyridine	626-60-8	1.194	ND	ND	ND	ND
4-Chloropyridine	7379-35-3	ND	ND	ND	ND	ND
Acenaphthene	83-32-9	1.069	3.93	2.15E-03	2.40E-04	4600
Anthracene	120-12-7	1.25	0.073	6.00E-06	5.90E-05	14000
Benzo(a)anthracene	56-55-3	ND	0.014	2.10E-07	4.50E-06	1380000
Benzo(a)pyrene	50-32-8	ND	0.00005	5.60E-09	3.72E-05	5500000
Benzo(b)fluoranthene	205-99-2	ND	0.014	5.00E-07	1.18E-05	550000
Benzo(g,h,i)perylene	191-24-2	ND	0.0003	1.03E-10	1.25E-07	1600000
Benzo(k)fluoranthene	207-08-9	ND	0.0043	5.10E-07	3.94E-05	550000
Benzoic acid	65-85-0	1.265	2900	7.05E-03	3.92E-07	54.4
Butylbenzylphthalate	85-68-7	1.1	2	2.12E-05	4.35E-06	17000
Chrysene	218-01-9	1.274	0.002	6.40E-09	9.60E-07	200000
Di-n-butylphthalate	84-74-2	1.046	9.2	1.00E-05	1.30E-06	1390
Di-n-octylphthalate	117-84-0	0.986	0.34	1.40E-04	5.50E-06	19000
Dibenzo(a,h)anthracene	53-70-3	ND	0.014	1.00E-10	2.61E-09	3300000
Dibenzofuran	132-64-9	1.089	10	3.37E-05	7.45E-07	9120
Diethylphthalate	84-66-2	1.12	680	3.50E-03	1.50E-06	69
Dimethylphthalate	131-11-3	1.189	2120	4.19E-03	5.05E-07	17.4
Fluoranthene	206-44-0	1.252	0.26	9.20E-06	9.41E-06	38000
Fluorene	86-73-7	1.203	1.98	6.00E-04	8.40E-05	7300
Hexachlorobenzene	118-74-1	2.044	0.006	2.58E-03	1.70E-03	3900
Hexachlorobutadiene	87-68-3	1.675	3.23	2.00E+00	1.03E-02	29000
Hexachloroethane	67-72-1	2.09	50	4.00E-01	3.89E-03	20000
Indeno(1,2,3-c,d)pyrene	193-39-5	ND	0.00053	1.00E-10	6.85E-08	1600000
N-Nitrosodiphenylamine	86-30-6	1.23	35	6.69E-04	5.00E-06	648
Naphthalene	91-20-3	1.152	31.7	7.80E-02	4.20E-04	940
Phenanthrene	85-01-8	1.025	1.29	1.20E-04	3.90E-05	14000
Phenol	108-95-2	1.07	93000	3.41E-01	3.95E-07	14.2
Pyrene	129-00-0	1.271	0.135	4.50E-06	8.86E-06	38000
Pyridine	110-86-1	0.982	miscible	2.00E+01	8.85E-06	3.02
bis(2-Chloroethyl)ether	111-44-4	1.22	10200	7.10E-01	1.31E-05	13.9
bis(2-Ethylhexyl)phthalate	117-81-7	0.99	0.4	2.00E-07	4.40E-07	87400

Chem. Properties

TABLE 3-1
PHYSIO-CHEMICAL PROPERTIES OF SELECTED CHEMICALS

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, N.Y.

Chemical Name	CAS Number	Specific Gravity	Water Solubility (mg/L)	Vapor Pressure (mmHg)	Henry's Law (1) (atm-m3/mol)	Koc (ml/g) (2)
p-Fluoroaniline	371-40-4	1.1725	ND	ND	ND	ND
p-Nitroaniline	100-01-6	1.424	800	1.50E-03	3.41E-07	15.1

NOTES:

VOC = Volatile Organic Compounds
SVOC = Semivolatile Organic Compounds
Koc = Organic carbon partion coefficient
mg/L = milligrams per liter
mmHg = millimeters of mercury
atm-m3/mol = Atmosphere-cubic meters per mole
ml/g = milliliters per gram
ND = No Data

1)

Range of H (atm-m3/mol)	Degree of Volatility
$H < 3.0E-07$	non-volatile
$3.0E-07 < H < 1.0E-05$	low volatility
$1.0E-05 < H < 1.0E-03$	moderate volatility
$H > 1.0E-03$	high volatility

2)

Degree of Adsorption	Koc	Degree of Mobility
very weak	< 10	very high
weak	10 - 100	high
moderate	100 - 1000	moderate
moderate to strong	1000-10000	low
strong	10000-1000000	very low
very strong	>100000	extremely low

TABLE 4-1
CHEMICALS OF POTENTIAL CONCERN FOR THE HUMAN HEALTH RISK ASSESSMENT GROUNDWATER

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, N.Y.

Compound	Range of SQLs	Frequency of Detection	Detected Concentration Minimum	Detected Concentration Maximum	Mean of all Samples**	Federal MCL	CPC?	COMMENTS
OFFSITE GROUNDWATER* (mg/L)								
VOLATILE ORGANIC COMPOUNDS								
1,1,1-Trichloroethane	0.0005 – 0.05	2 / 19	0.0008	0.0009	0.0050	0.2	No	Toxicity Screen[1]
1,1,2,2-Tetrachloroethane	0.0005 – 0.05	1 / 19	0.0009	0.001	0.0050	–	No	Toxicity Screen[1]
1,1-Dichloroethane	0.0005 – 0.05	10 / 19	0.0009	0.035	0.0092	–	No	Toxicity Screen[1]
1,2-Dichloroethene (Total)	0.0005 – 0.01	10 / 19	0.004	0.49	0.0573	0.07	Yes	Class[2]
2-Butanone	0.001 – 0.05	1 / 19	0.055	0.055	0.0066	–	No	Toxicity Screen[1]
Acetone	0.01 – 0.1	4 / 19	0.014	1.2	0.0774	–	No	Toxicity Screen[1]
Benzene	0.0005 – 0.01	14 / 19	0.0006	0.24	0.0413	0.005	Yes	
Carbon Disulfide	0.0005 – 0.05	9 / 19	0.0003	0.036	0.0070	–	No	Toxicity Screen[1]
Chlorobenzene	0.0005 – 0.01	5 / 19	0.001	1.4	0.0856	0.1	No	Toxicity Screen[1]
Chloroethane	0.001 – 0.05	1 / 19	0.003	0.003	0.0054	–	No	Toxicity Screen[1]
Chloroform	0.0005 – 0.01	6 / 19	0.0004	0.089	0.0077	0.08	No	Toxicity Screen[1]
Ethylbenzene	0.0005 – 0.01	10 / 19	0.0003	0.062	0.0089	0.7	No	Toxicity Screen[1]
Methylene Chloride	0.003 – 0.05	1 / 19	0.003	0.003	0.0065	0.005	No	Toxicity Screen[1]
Tetrachloroethene	0.0005 – 0.05	3 / 19	0.001	0.004	0.0048	0.005	Yes	Class[2]
Toluene	0.01 – 0.01	14 / 19	0.0002	2.3	0.1435	1	No	Toxicity Screen[1]
Total Xylenes	0.01 – 0.01	13 / 19	0.0002	0.34	0.0247	10	No	Toxicity Screen[1]
Trichloroethene	0.0005 – 0.05	7 / 19	0.001	0.012	0.0047	0.005	Yes	Class[2]
Vinyl Acetate	0.005 – 0.05	1 / 9	0.025	0.025	0.0064	–	No	Toxicity Screen[1]
Vinyl Chloride	0.001 – 0.01	10 / 19	0.002	0.35	0.0442	0.002	Yes	
SEMIVOLATILE ORGANIC COMPOUNDS								
1,2-Dichlorobenzene	0.01 – 0.01	1 / 9	0.003	0.004	0.0048	0.6	No	Toxicity Screen[1]
2,4-Dimethylphenol	0.01 – 0.01	1 / 9	0.002	0.002	0.0047	–	No	Toxicity Screen[1]
2,6-Dichloropyridine	0.01 – 0.01	15 / 23	0.0005	15	0.7384	–	Yes	
2-Chloropyridine	0.01 – 0.01	19 / 23	0.002	84	4.6343	–	Yes	
2-Methylnaphthalene	0.01 – 0.01	2 / 9	0.002	0.014	0.0054	–	No	Toxicity Screen[1]
2-Methylphenol	0.01 – 0.01	1 / 9	0.0009	0.0009	0.0045	–	No	Toxicity Screen[1]
3-Chloropyridine	0.01 – 0.01	8 / 23	0.007	4	0.2023	–	Yes	
4-Chloropyridine	0.01 – 0.01	1 / 13	0.006	0.006	0.0051	–	Yes	Class[2]
4-Methylphenol	0.01 – 0.01	3 / 9	0.001	0.008	0.0046	–	No	Toxicity Screen[1]
bis(2-Chloroethyl)ether	0.01 – 0.01	1 / 9	0.006	0.009	0.0053	–	No	Toxicity Screen[1]
bis(2-Ethylhexyl)phthalate	0.01 – 0.01	6 / 9	0.001	0.016	0.0052	0.006	No	Toxicity Screen[1]
Butylbenzylphthalate	0.01 – 0.01	1 / 9	0.0007	0.0007	0.0045	–	No	Toxicity Screen[1]
Di-n-butylphthalate	0.01 – 0.01	2 / 9	0.0008	0.002	0.0042	–	No	Toxicity Screen[1]
Naphthalene	0.01 – 0.01	3 / 9	0.001	0.016	0.0052	–	No	Toxicity Screen[1]
p-Fluoroaniline	0.01 – 0.01	7 / 23	0.001	0.32	0.0197	–	No	Toxicity Screen[1]
INORGANICS								
Aluminum		9 / 9	0.0866	16.6	4.0034	0.2#	No	Toxicity Screen[1]
Antimony	0.0051 – 0.0051	2 / 9	0.0054	0.0576	0.0060	0.006	No	Toxicity Screen[1]
Arsenic	0.0053 – 0.0053	5 / 9	0.0062	0.371	0.0332	0.05	Yes	
Barium		9 / 9	0.0367	1.54	0.2033	2	No	Toxicity Screen[1]
Beryllium	0.0002 – 0.0002	4 / 9	0.0003	0.0021	0.0004	0.004	No	Toxicity Screen[1]
Cadmium	0.0004 – 0.0004	2 / 9	0.0012	0.0984	0.0062	0.005	Yes	
Calcium		9 / 9	135	1220	322.7778	–	Yes	
Chromium	0.001 – 0.001	6 / 9	0.002	0.102	0.0128	0.1	No	Toxicity Screen[1]

TABLE 4-1
CHEMICALS OF POTENTIAL CONCERN FOR THE HUMAN HEALTH RISK ASSESSMENT
GROUNDWATER

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, N.Y.

Compound	Range of SQLs	Frequency of Detection	Detected Concentration Minimum	Detected Concentration Maximum	Mean of all Samples**	Federal MCL	CPC?	COMMENTS
Cobalt	0.0016 – 0.0016	5 / 9	0.0017	0.0346	0.0046	–	No	Toxicity Screen[1]
Copper	0.0011 – 0.0011	8 / 9	0.0016	70.7	4.4574	1.3*	Yes	
Iron		9 / 9	2.54	864	110.1289	0.3#	Yes	
Lead	0.0014 – 0.0014	8 / 9	0.0023	4.75	0.3136	0.015*	Yes	Toxicity Value[3]
Magnesium		9 / 9	28.8	357	111.5889	–	Yes	
Manganese		9 / 9	0.0541	5.97	0.8991	0.05#	No	Toxicity Screen[1]
Mercury	0.0002 – 0.0002	2 / 9	0.00054	0.0077	0.0010	0.002	No	Toxicity Screen[1]
Nickel	0.0014 – 0.0014	7 / 9	0.0016	0.514	0.0407	0.1	No	Toxicity Screen[1]
Potassium		9 / 9	4.04	171	29.7333	–	Yes	
Selenium	0.005 – 0.005	2 / 9	0.0071	0.0177	0.0039	0.05	No	Toxicity Screen[1]
Silver	0.0011 – 0.0011	2 / 9	0.0011	0.0334	0.0027	0.1#	No	Toxicity Screen[1]
Sodium		9 / 9	12.2	6490	873.5000	–	Yes	
Vanadium	0.0016 – 0.0016	7 / 9	0.0039	0.0748	0.0111	–	No	Toxicity Screen[1]
Zinc		9 / 9	0.0176	2780	174.5142	5#	Yes	
MISCELLANEOUS PARAMETERS								
Methanol	1 – 1	1 / 12	0.98	0.98	0.54	–	No	Toxicity Screen[1]

NOTES:
** Mean of all samples is arithmetic average of all detections plus one-half the SQL for non-detects. If mean concentration exceeds maximum concentration, only maximum concentration will be used in evaluation.
* – Action Level
# – Secondary Standard
– = No standard available for this compound
Toxicity Screen[1] – Chemicals with low ratios (i.e., less than 0.01) are not considered chemicals of potential concern (CPCs)
Class[2] – Chemical is a transformation or parent compound of a chemical selected as a CPC;
or belongs to a class of compounds where one member of the class has been selected as a CPC.
Toxicity Value[3] – No toxicity data available.

ACRONYMS:
mg/L – milligrams per liter
SQL – Sample Quantitation Limit
MCL – Maximum Contaminant Level: Drinking Water Regulations and Health Advisories, U.S. Environmental Protection Agency Office of Water, May 1995.
CPC – Chemical of Potential Concern

Sample Locations:
[a] – Based on samples QS-1 through QS-4, September 1995 and QS-4 and duplicate October 1995; BR-111, BR-111D, BR-112A, BR-112D, BR-113 and duplicate, BR-113D, and BR-114 October and December 1995; NESS E, NESS W, September, November, and December 1995 plus NESS E duplicate November 1995; BR-103 through BR-107 and BR-105D, MW-103, MW-104, MW-106, and MW-107 September 1995.

TABLE 4-2
CHEMICALS OF POTENTIAL CONCERN FOR THE HUMAN HEALTH RISK ASSESSMENT ERIE BARGE CANAL SURFACE WATER

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

Compound	Range of SQLs	Frequency of Detection	Detected Concentration		Mean of all Samples[1]	CPC?	Comments
			Minimum	Maximum			
BARGE CANAL SURFACE WATER[a] (mg/L)							
SEMIVOLATILE ORGANIC COMPOUNDS							
2,6-Dichloropyridine	0.01 - 0.017	7 / 34	0.0003	0.005	0.005	Yes	
2-Chloropyridine	0.01 - 0.011	16 / 34	0.0001	0.039	0.008	Yes	
3-Chloropyridine	0.01 - 0.021	5 / 34	0.001	0.002	0.0048	Yes	
QUARRY OUTFALL WATER[b] (mg/L)							
SEMIVOLATILE ORGANIC COMPOUNDS							
2,6-Dichloropyridine	NA	3 / 3	0.007	0.032	0.021	Yes	
2-Chloropyridine	NA	3 / 3	0.018	0.16	0.900	Yes	
3-Chloropyridine	NA	3 / 3	0.0004	0.011	0.0058	Yes	

NOTES:

[1] – Mean of all samples is arithmetic average of all detections plus one-half the SQL for non-detects. If mean concentration exceeds maximum concentration, only maximum concentration will be used in evaluation.

ACRONYMS:
mg/L – milligrams per liter
SQL – Sample Quantitation Limit
CPC – Chemical of Potential Concern

Sample Locations:

[a] – Based on samples SW-1 through SW-12, QO-2D1, QO-2D2, QO-2U1, QO-2U2 collected April, June, and September 1996 and April and June 1997.

[b] – Based on sample QO-2 collected June and September 1996 and June 1997

TABLE 4-3
CHEMICALS OF POTENTIAL CONCERN FOR THE HUMAN HEALTH RISK ASSESSMENT GROUNDWATER SEEPS

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, N.Y.

Compound	Range of SQLs	Frequency of Detection	Detected Concentration Minimum	Detected Concentration Maximum	Mean of all Samples[1]	CPC?	Comments
GROUNDWATER SEEP IN QUARRY[a] (mg/L)							
SEMIVOLATILE ORGANIC COMPOUNDS							
2-Chloropyridine	NA	8 / 8	0.004	3.2	1.174	Yes	
3-Chloropyridine	0.01 - 0.01	6 / 8	0.007	0.12	0.039	Yes	
2,6-Dichloropyridine	0.01 - 0.01	7 / 8	0.004	1	0.334	Yes	
p-Fluoroaniline[b]	0.01 - 0.01	2 / 5	0.002	0.005	0.0044	Yes	

NOTES:

[1] - Mean of all samples is arithmetic average of all detections plus one-half the SQL for non-detects. If mean concentration exceeds maximum concentration, only maximum concentration will be used in evaluation.

ACRONYMS:

mg/L - milligrams per liter
SQL - Sample Quantitation LImit
CPC - Chemical of Potential Concern

Sample Locations:

[a] - Based on samples QS-2 and QS-3 sampled 9/95, and sample QS-4 sampled 9/95, 10/95, 3/96, 6/96, 9/96, and 12/96.

[b] - Based on samples QS-2 and QS-3 sampled 9/95, and sample QS-4 sampled 9/95 and 10/95; elevated reporting limits for QS-4 sampled 3/96, 6/96, 9/96, and 12/96. precluded use in risk assessment.

TABLE 4-4
SUMMARY OF RECEPTORS AND EXPOSURE PATHWAYS

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, N.Y.

RECEPTOR	EXPOSURE PATHWAY
CURRENT AND FUTURE USE	
Recreational boater/swimmer at Barge Canal	Dermal Contact with surface water Incidental Ingestion of surface water
Quarry Worker at the Dolomite Quarry	Dermal Contact with groundwater seeps
FUTURE USE	
Off-site industrial worker	Dermal Contact with groundwater used as industrial processs water Inhalation of volatile compounds released from industrial process water

TABLE 4-5
QUANTITATIVE RISK SUMMARIES BY MEDIA

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, N.Y.

		MEAN		MAXIMUM	
		Total Cancer Risk	Total Hazard Index	Total Cancer Risk	Total Hazard Index
CURRENT AND FUTURE USE					
SURFACE WATER					
Ingestion and Dermal Contact with Surface Water in Barge Canal:	Recreational Child	9E-09	0.00001	1E-07	0.00007
	Recreational Adult	1E-08	0.00001	2E-07	0.00006
TOTAL: Recreational Swimmer		**2E-08**	**NA**	**3E-07**	**NA**
GROUNDWATER SEEP					
Dermal Contact with Groundwater Seep in Dolomite Quarry:	Quarry Worker	7E-08	0.00003	2E-07	0.00007
TOTAL: Quarry Worker		**7E-08**	**0.00003**	**2E-07**	**0.00007**
FUTURE USE					
OFFSITE GROUNDWATER					
Dermal Contact with Offsite Groundwater:	Worker	7E-05	4	9E-04	29
TOTAL: Industrial/Commercial Worker		**7E-05**	**4**	**9E-04**	**29**

NA – Not Additive; child and adult hazard indices are not additive.

TABLE 4-6
QUANTITATIVE RISK SUMMARIES BY RECEPTOR

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, N.Y.

		MEAN		MAXIMUM	
		Total Cancer Risk	Total Hazard Index	Total Cancer Risk	Total Hazard Index
CURRENT AND FUTURE USE					
RECREATIONAL SWIMMER					
Ingestion and Dermal Contact with Surface Water in Barge Canal:	Recreational Child	9E-09	0.00001	1E-07	0.00007
	Recreational Adult	1E-08	0.00001	2E-07	0.00006
TOTAL: Recreational Swimmer		**2E-08**	**NA**	**3E-07**	**NA**
QUARRY WORKER					
Dermal Contact with Groundwater Seep in Dolomite Quarry:	Quarry Worker	7E-08	0.00003	2E-07	0.00007
TOTAL: Quarry Worker		**7E-08**	**0.00003**	**2E-07**	**0.00007**
FUTURE USE					
INDUSTRIAL/COMMERCIAL WORKER					
Dermal Contact with Offsite Groundwater:	Worker	7E-05	4	9E-04	29
TOTAL: Industrial/Commercial Worker		**7E-05**	**4**	**9E-04**	**29**

NA – Not Additive; child and adult hazard indices are not additive.

TABLE 4-7
POTENTIAL SOURCES OF UNCERTAINTY

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

UNCERTAINTY	EFFECT	JUSTIFICATION
Likelihood of exposure pathways	Overestimate	Future pathways may not actually occur.
Exposure assumptions (e.g., frequency, duration)	Overestimate	Parameters selected are conservative estimates of exposure.
Degradation of chemicals not considered	Overestimate	Risk estimates are based on recent chemical concentrations. Concentrations will tend to decrease over time as a result of degradation.
Extrapolation of animal toxicity data to humans.	Unknown, probably overestimate	Animals and humans differ with respect to absorption, metabolism, distribution, and excretion of chemicals. The magnitude and direction of the difference will vary with each chemical. Animal studies typically involve high-dose exposures, whereas humans are exposed to low doses in the environment.
Use of linearized, multistage model to derive cancer slope factors.	Overestimate	Model assumes a non-threshold, linear-at-low-dose relationship for carcinogens. Many compounds induce cancer by non-genotoxic mechanisms. Model results in a 95% upper confidence limit of the cancer risk. The true risk is unlikely to be higher and may be as low as zero.
Summation of effects (cancer risks and hazard indices) from multiple substances.	Unknown	The assumption that effects are additive ignores potential synergistic and/or antagonistic effects. Assumes similarity in mechanism of action, which is not the case for many substances. Compounds may induce tumors or other toxic effects in different organs or systems.
Use of uncertainty factors in the derivation of reference doses	Unknown	Ten-fold uncertainty factors are incorporated to account for various sources of uncertainty. Although some data seem to support the ten-fold factor, its selection is somewhat arbitrary.
Some analytes, such as the chloropyridines, are evaluated using surrogate toxicity values.	Unknown	Although the toxicity values used are from structurally similar compounds, the actual toxicity of the evaluated compound is not established in IRIS or HEAST.

TABLE 4-8
SURFACE WATER SAMPLING RESULTS UPSTREAM OF QUARRY OUTFALL

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

Analyte [a]	Frequency of Detection	Average Concentration (ug/l) [b]	Maximum Concentration (ug/l) [c]	Toxicity Benchmark [d]
2,6-Dichloropyridine	15/36	4.1	5.0	4,700
2-Chloropyridine	23/36	11	45	14,000
3-Chloropyridine	7/36	4.6	3.0	12,900

NOTES:

[a] These analytes were detected in surface water collected above the quarry outfall, from the following sample locations, SW-1, SW-2, SW-3, SW-7, SW-8, SW-9, SW-11, and SW-12.

[b] The average concentration is calculated with one-half the reporting limit used as the value for non-detects.

[c] The maximum concentration represents the highest detected concentration.

[d] The development of aquatic toxicity benchmarks is described in Section 4.2.4.1.

-- = Analyte not detected.

TABLE 4-9
SURFACE WATER SAMPLING RESULTS ADJACENT TO AND DOWNSTREAM OF THE QUARRY OUTFALL

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

Analyte [a]	Frequency of Detection	Average Concentration (μg/L) [b]	Maximum Concentration (μg/L) [c]	Toxicity Benchmark [d]
2,6-Dichloropyridine	--	--	--	4,700
2-Chloropyridine	9/20	3.5	4	14,000
3-Chloropyridine	--	--	--	12,900

NOTES:

[a] These analytes were detected in surface water collected above the quarry outfall, from the following sample locations QO-2N1, QO-2N2, QO-2S1, QO-2S2, SW-4, SW-5, and SW-6.

[b] The average concentration is calculated with one-half the reporting limit used as the value for non-detects.

[c] The maximum concentration represents the highest detected concentration.

[d] The development of aquatic toxicity benchmarks is described in Section 4.2.4.1.

-- = Analyte not detected.

Units are in μg/L

TABLE 4-10
COMPARISON OF SURFACE SOIL INORGANIC CONCENTRATIONS EVALUATED IN THE PHASE I HBA WITH REGIONAL BACKGROUND RANGES [a]

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

Analyte	Frequency of Detection	Minimum Detected Concentration	Maximum Detected Concentration	Mean of Detects	Mean of all Samples	Eastern USA Background [b]
Aluminum	6 / 6	3,900	8,700	6,533	6,533	33,000
Arsenic	6 / 6	2.7	12	5.4	5.4	3 - 12**
Barium	6 / 6	37	110	60	60	15 - 600
Cadmium	6 / 6	0.1	0.8	0.5	0.5	0.1 - 1.0
Calcium	6 / 6	4,900	95,000	42,983	42,983	130 - 35,000**
Chromium	6 / 6	8.2	150	28.0	28.0	1.5 - 40**
Cobalt	3 / 6	5	7.1	5.5	3.9	2.5 - 60**
Copper	6 / 6	8.6	48	20.6	20.6	1 - 50
Iron	6 / 6	12,000	17,000	13,833	13,833	17,500 - 25,000
Lead	6 / 6	12	140	73.7	73.7	[c]
Magnesium	6 / 6	2,700	50,000	17,508	17,508	100 - 5,000
Manganese	6 / 6	240	760	428	428	50, - 5,000
Mercury	4 / 6	0.2	0.4	0.28	0.20	0.001 - 0.2
Nickel	6 / 6	13	62	23.6	23.6	0.5 - 25
Potassium	6 / 6	630	1,200	892.5	892.5	8,500 - 43,000**
Selenium	1 / 6	0.8	0.8	0.8	0.4	0.1 - 3.9
Silver	6 / 6	0.1	0.7	0.4	0.4	NA
Sodium	6 / 6	260	1,400	647.5	647.5	6,000 - 8,000
Vanadium	6 / 6	12	20	16.9	16.9	1 - 300
Zinc	6 / 6	65	240	150	150	9 - 50

NOTES

a. Includes samples SS-102, SS-105, SS-109, SS-112, SS-113, and SS-115. Concentrations in mg/kg.

b. NYSDEC, 1994. "Determination of Soil Cleanup Objectives and Cleanups Levels"; TAGM 4046. Appendix A, Table 4. Ranges indicated with a double asterisk are based on New York State background levels.

c. Average background levels in metropolitan or suburban areas or near highways typically range from 200 - 500 mg/kg.

Shading indicates analytes where the maximum detected concentration exceeds the range of background concentrations.

TABLE 4-11
SEMI-AQUATIC RISK ESTIMATES FOR SURFACE WATER ANALYTES

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

	Maximum Surface Water Conc. (μg/L)	BCF [a]	Tissue Concentration (mg/kg) [b]	Ingestion Rate [c] (kg/day)	Body Weight [d] (kg)	Total Body Dose [e] (mg/kgBW-d)	Reference [f] Toxicity Value (mg/kgBW-d)	Hazard Quotient (unitless)
BELTED KINGFISHER								
(Ceryle alcyon)								
2,6-Dichloropyridine	5	20	0.1	0.067	0.148	0.0453	2.6	1.7E-02
2-Chloropyridine	45	5	0.225	0.067	0.148	0.1019	2.6	3.9E-02
3-Chloropyridine	3	5	0.015	0.067	0.148	0.0068	2.6	2.6E-03
RACCOON								
(Procyon lotor)								
2,6-Dichloropyridine	5	20	0.1	0.214	3.99	0.0054	2.6	2.1E-03
2-Chloropyridine	45	5	0.225	0.214	3.99	0.0121	2.6	4.6E-03
3-Chloropyridine	3	5	0.015	0.067	0.148	0.0068	2.6	2.6E-03

NOTES:

[a] Fish BCFs for 2,6-chloropyridine and 2-chloropyridine are calculated values, obtained using a SAR model.

[b] Tissue concentration is calculated by multiplying the detected surface water concentration by the BCF, units converted to ppm.

[c] Ingestion rates for the belted kingfisher and raccoon were obtained from Nagy (1987) and USEPA (1993), respectively.

[d] Body weights for the belted kingfisher and raccoon were obtained from Opresko et al. (1993) and USEPA (1993), respectively.

[e] Total body dose is calculated by multiplying the tissue concentration by the ingestion rate and dividing by the body weight.

[f] Reference toxicity values are presented in Appendix D, Table D-7, of the Phase I Remedial Investigation.

TABLE 4-12
AQUIRE DATA SUMMARY

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

Name	Reliability [a]	Organism Latin	Organism Common	Age	Exposure Regimen [b]	Test Conditions [c]	Effect [d]	Endpoint	Effect Concentration	Accession Number [e]
Pyridine	1	Cyprinus carpio	Carp	4-5CM	24 H	FW; LAB	MOR	LC50	47500	200530
Pyridine	1	Cyprinus carpio	Carp	4-5CM	48 H	FW; LAB	MOR	LC50	35000	206646
Pyridine	1	Cyprinus carpio	Carp	4-5CM	96 H	FW; LAB	MOR	LC50	26000	206647
Pyridine	2	Crangon septemspinosa	Sand shrimp	6.4-8.3 CM, 2.4-4.5 G	96 H	SW; LAB	MOR	LC50	>50000	207264
Pyridine	3	Xenopus laevis	Clawed toad	Early cleavage to mid-blastula *	96 H	FW; LAB	ABN		10000	213886
Pyridine	2	Gambusia affinis	Mosquitofish	Adult, female	24 H	FW; LAB	MOR	LC50	1350000	215420
Pyridine	2	Gambusia affinis	Mosquitofish	Adult, female	48 H	FW; LAB	MOR	LC50	1350000	215421
Pyridine	2	Gambusia affinis	Mosquitofish	Adult, female	96 H	FW; LAB	MOR	LC50	1300000	215422
Pyridine	2	Gambusia affinis	Mosquitofish	Adult, female	96 H	FW; LAB	MOR		1000000	215423
Pyridine	2	Tetrahymena pyriformis	Ciliate	Log phase	72 H	FW; LAB	GRO	EC50	1193700	215541
Pyridine	3	Lepomis humilis	Orangespotted sunfish	4-6 G	1 H	FW; LAB	MOR	LC100	1477000	215587
Pyridine	4	Scenedesmus quadricauda	Green algae	NR	NR	LAB *	PGR		120000	217623
Pyridine	2	Lepomis macrochirus	Bluegill	Fingerlings, 38-76 MM	96 H	FW; LAB	MOR	LC0	2400000	218058
Pyridine	2	Daphnia magna	Water flea	NR	24 H	FW; LAB	MOR	LC50	2114000	218465
Pyridine	2	Daphnia magna	Water flea	NR	48 H	FW; LAB	MOR	LC50	944000	218466
Pyridine	2	Tetrahymena pyriformis	Ciliate	10 G, growth phase	72 H	FW; LAB	GRO	EC50	1211000	218482
Pyridine	4	Chilomonas paramecium	Cryptomonad	NR	48 H	LAB *	PGR		3900	218799
Pyridine	2	Daphnia magna	Water flea	24 H	24 H	FW; LAB	MOR	LC50	240000	220914
Pyridine	3	Daphnia magna	Water flea	< 24 H	48 H	FW; LAB	MOR	LC50	1140000	225532
Pyridine	3	Daphnia pulex	Water flea	< 24 H	48 H	FW; LAB	MOR	LC50	520000	225533
Pyridine	3	Daphnia pulex	Water flea	< 24 H	48 H	FW; LAB	MOR	LC50	630000	225534
Pyridine	3	Daphnia cucullata	Water flea	< 24 H	48 H	FW; LAB	MOR	LC50	2390000	225535
Pyridine	3	Daphnia cucullata	Water flea	< 24 H	48 H	FW; LAB	MOR	LC50	2550000	225536
Pyridine	3	Daphnia magna	Water flea	< 24 H	48 H	FW; LAB	MOR	LC50	1210000	225537
Pyridine	3	Daphnia magna	Water flea	< 24 H	48 H	FW; LAB	MOR	LC50	1120000	225538
Pyridine	3	Daphnia magna	Water flea	< 24 H	48 H	FW; LAB	MOR	LC50	1570000	225539
Pyridine	3	Daphnia magna	Water flea	< 24 H	48 H	FW; LAB	MOR	LC50	1940000	225540
Pyridine	2	Scenedesmus quadricauda	Green algae	Initial culture turbidity reported	7 D	LAB *	PGR		120000	227293
Pyridine	2	Entosiphon sulcatum	Flagellate	Initial culture turbidity reported	72 H	LAB *	PGR		3500	227294
Pyridine	3	Kuhlia sandvicensis	Aholehole	30-60 mm	0.033	SW; LAB	BEH		20000	229477
Pyridine	2	Ambystoma mexicanum	Salamander	3-4 wk	48 H	FW; LAB	MOR	LC50	950000	235077
Pyridine	2	Xenopus laevis	Clawed toad	3-4 wk	48 H	FW; LAB	MOR	LC50	1400000	235078
Pyridine	3	Oncorhynchus mykiss	Rainbow trout	Fingerling, <= 10 cm	24 H	FW; LAB	STR		5000	236353
Pyridine	3	Lepomis macrochirus	Bluegill	Fingerling, <= 10 cm	24 H	FW; LAB	STR		5000	236354
Pyridine	3	Petromyzon marinus	Sea Lamprey	Larvae, 8-13 cm	24 H	FW; LAB	STR		5000	236355
Pyridine	2	Xenopus laevis	Clawed toad	Mid-blastula embryo	24 H	FW; LAB	MOR	LC50	3800000	238233
Pyridine	2	Xenopus laevis	Clawed toad	Mid-blastula embryo	48 H	FW; LAB	MOR	LC50	2570000	238234
Pyridine	2	Xenopus laevis	Clawed toad	Mid-blastula embryo	72 H	FW; LAB	MOR	LC50	2340000	238235
Pyridine	2	Xenopus laevis	Clawed toad	Mid-blastula embryo	5 D	FW; LAB	MOR	LC50	1620000	238236
Pyridine	2	Xenopus laevis	Clawed toad	Tailbud embryo	24 H	FW; LAB	MOR	LC50	9550000	238237
Pyridine	2	Xenopus laevis	Clawed toad	Tailbud embryo	48 H	FW; LAB	MOR	LC50	3390000	238238
Pyridine	2	Xenopus laevis	Clawed toad	Tailbud embryo	72 H	FW; LAB	MOR	LC50	2820000	238239
Pyridine	2	Xenopus laevis	Clawed toad	Tailbud embryo	96 H	FW; LAB	MOR	LC50	2460000	238240
Pyridine	2	Xenopus laevis	Clawed toad	Tailbud embryo	5 D	FW; LAB	MOR	LC50	1000000	238241
Pyridine	2	Xenopus laevis	Clawed toad	Larvae	24 H	FW; LAB	MOR	LC50	1660000	238242
Pyridine	2	Xenopus laevis	Clawed toad	Larvae	48 H	FW; LAB	MOR	LC50	1590000	238243
Pyridine		Xenopus laevis	Clawed toad	Larvae	72 H	FW; LAB	MOR	LC50	1200000	238244

TABLE 4-12
AQUIRE DATA SUMMARY

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

Name	Reliability [a]	Organism Latin	Organism Common	Age	Exposure Regimen [b]	Test Conditions [c]	Effect [d]	Endpoint	Effect Concentration	Accession Number [e]
Pyridine	2	Xenopus laevis	Clawed toad	Larvae	96 H	FW; LAB	MOR	LC50	1090000	238245
Pyridine	2	Xenopus laevis	Clawed toad	Larvae	5 D	FW; LAB	MOR	LC50	1050000	238246
Pyridine	2	Xenopus laevis	Clawed toad	Mid-blastula embryos	24 H	FW; LAB	ABN	EC50	2190000	238247
Pyridine	2	Xenopus laevis	Clawed toad	Mid-blastula embryos	48 H	FW; LAB	ABN	EC50	1550000	238248
Pyridine	2	Xenopus laevis	Clawed toad	Mid-blastula embryos	72 H	FW; LAB	ABN	EC50	1350000	238249
Pyridine	2	Xenopus laevis	Clawed toad	Mid-blastula embryos	96 H	FW; LAB	ABN	EC50	1200000	238250
Pyridine	2	Xenopus laevis	Clawed toad	Mid-blastula embryos	5 D	FW; LAB	ABN	EC50	850000	238251
Pyridine	2	Daphnia magna	Water flea	NR	24 H	FW; LAB		EC50	520000	242388
Pyridine	2	Daphnia magna	Water flea	NR	24 H	FW; LAB		ECO	266000	244902
Pyridine	2	Selenastrum capricornutum	Green algae	NR	4 H	LAB *	PSE		1000000	250321
Pyridine	2	Scenedesmus quadricauda	Green algae	NR	8 D	LAB *	PGR		120000	257350
Pyridine	2	Anacystis aeruginosa	Blue-green algae	NR	8 D	LAB *	PGR		28000	257351
Pyridine	3	Oryzias latipes	Medaka, high-eyes	2 cm, 0.2 g	24 H	FW; LAB	MOR	LC50	400000	272147
Pyridine	3	Oryzias latipes	Medaka, high-eyes	2 cm, 0.2 g	48 H	FW; LAB	MOR	LC50	330000	272148
Pyridine	2	Selenastrum capricornutum	Green algae	Log phase	96 H	FW; LAB	PGR		50000	275679
Pyridine	2	Chlorella pyrenoidosa	Green algae	Log phase	48 H	FW; LAB	PGR		150000	275680
Pyridine	2	Scenedesmus pannonicus	Green algae	Log phase	48 H	FW; LAB	PGR		280000	275681
Pyridine	2	Aedes aegypti	Mosquito	3rd instar	48 H	FW; LAB	MOR	LC50	130000	275682
Pyridine	2	Culex pipiens	Mosquito	3rd instar	48 H	FW; LAB	MOR	LC50	66000	275683
Pyridine	2	Hydra oligactis	Hydra	Budless	48 H	FW; LAB	MOR	LC50	1150000	275684
Pyridine	2	Lymnaea stagnalis	Great pond snail	3-4 wk	48 H	FW; LAB	MOR	LC50	350000	275685
Pyridine	2	Oncorhynchus mykiss	Rainbow trout	5-8 wk	48 H	FW; LAB	MOR	LC50	560000	275686
Pyridine	2	Oryzias latipes	Medaka, high-eyes	4-5 wk	48 H	FW; LAB	MOR	LC50	1560000	275687
Pyridine	2	Pimephales promelas	Fathead minnow	3-4 wk	48 H	FW; LAB	MOR	LC50	115000	275688
Pyridine	3	Tetrahymena pyriformis	Ciliate	NR	60 H	FW; LAB	PSR	EC50	1.67858	279907
Pyridine	2	Tetrahymena pyriformis	Ciliate	NR	60 H	FW; LAB	PSR	EC50	1678580	281136
Pyridine	2	Daphnia magna	Water flea	< 24 H	24 H	FW; LAB	IMM	EC50	495000	286460
Pyridine	2	Daphnia magna	Water flea	< 24 H	30 D	FW; LAB	MOR		25000	286461
Pyridine	2	Daphnia magna	Water flea	< 24 H	7 D	FW; LAB	GRO		25000	286462
Pyridine	2	Daphnia magna	Water flea	< 24 H	15 D	FW; LAB	REP		25000	286463
Pyridine	1	Artemia salina	Brine shrimp	Nauplii	24 H	SW; LAB	IMM	EC50	1318400	289901
Pyridine	1	Artemia salina	Brine shrimp	Stage II nauplii	24 H	SW; LAB	IMM	EC50	831800	290022
Pyridine	1	Artemia salina	Brine shrimp	Stage II nauplii	24 H	SW; LAB	IMM	EC50	489400	290023
Pyridine	2	Oncorhynchus mykiss	Rainbow trout	10-40 g	96 H	FW; LAB	MOR	LD50	651.87 [f]	293992
Pyridine	4	Brachydanio rerio	Zebrafish	NR	96 H	LAB *	MOR	LC50	> 512000	295635
Pyridine	4	Leuciscus idus	Silver or golden orfe	NR	96 H	LAB *	MOR	LC50	> 512000	295636
Pyridine	2	Tubificidae	Oligochaete	NR	48 H	FW; LAB	MOR	LC50	1300000	296383
Pyridine	2	Chironomus thummi	Midge	NR	48 H	FW; LAB	MOR	LC50	229000	296384
Pyridine	2	Erpobdella octoculata	Leech	NR	48 H	FW; LAB	MOR	LC50	2400000	296385
Pyridine	2	Asellus aquaticus	Aquatic sowbug	NR	48 H	FW; LAB	MOR	LC50	220000	296386
Pyridine	2	Lymnaea stagnalis	Great pond snail	NR	48 H	FW; LAB	MOR	LC50	350000	296387
Pyridine	2	Dugesia lugubris	Flatworm	NR	48 H	FW; LAB	MOR	LC50	1900000	296388
Pyridine	2	Hydra oligactis	Hydra	NR	48 H	FW; LAB	MOR	LC50	1150000	296389
Pyridine	2	Corixa punctata	Water boatman	NR	48 H	FW; LAB	MOR	LC50	30000	296390
Pyridine	2	Gammarus pulex	Scud	NR	48 H	FW; LAB	MOR	LC50	182000	296391
Pyridine	2	Ischnura elegans	Dragonfly	NR	48 H	FW; LAB	MOR	LC50	410000	296392
Pyridine	2	Nemoura cinerea	Stonefly	NR	48 H	FW; LAB	MOR	LC50	254000	296393

TABLE 4-12
AQUIRE DATA SUMMARY

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

Name	Reliability [a]	Organism Latin	Organism Common	Age	Exposure Regimen [b]	Test Conditions [c]	Effect [d]	Endpoint	Effect Concentration	Accession Number [e]
Pyridine	2	Cloeon dipterum	Mayfly	NR	48 H	FW; LAB	MOR	LC50	165000	296394
Pyridine	1	Oncorhynchus kisutch	Coho salmon	4.0(3.5-4.5) cm, 0.5(0.3-0.9)	24 H	FW; LAB	MOR	LC50	4300	300092
Pyridine	1	Oncorhynchus tshawytscha	Chinook Salmon	6.8(5.8-7.5) cm, 2.7(1.4-3.8)	24 H	FW; LAB	MOR	LC50	3200	300093
Pyridine	1	Oncorhynchus keta	Chum salmon	4.5(3.9-5.0) cm, 0.5(0.3-0.8)	24 H	FW; LAB	MOR	LC50	4000	300094
Pyridine	1	Oncorhynchus gorbuscha	Pink salmon	3.5(3.4-3.7) cm, 0.2(0.2-0.2)	24 H	FW; LAB	MOR	LC50	1300	300095
Pyridine	1	Oncorhynchus nerka	Sockeye salmon	3.9(3.5-4.3) cm, 0.5(0.3-0.6)	24 H	FW; LAB	MOR	LC50	6900	300096
Pyridine	1	Oncorhynchus mykiss	Rainbow trout	4.1(3.7-4.5) cm, 0.7(0.4-0.9)	24 H	FW; LAB	MOR	LC50	4600	300097
Pyridine	1	Oncorhynchus kisutch	Coho salmon	4.0(3.5-4.5) cm, 0.5(0.3-0.9)	48 H	FW; LAB	MOR	LC50	4000	300098
Pyridine	1	Oncorhynchus tshawytscha	Chinook Salmon	6.8(5.8-7.5) cm, 2.7(1.4-3.8)	48 H	FW; LAB	MOR	LC50	2900	300099
Pyridine	1	Oncorhynchus keta	Chum salmon	4.5(3.9-5.0) cm, 0.5(0.3-0.8)	48 H	FW; LAB	MOR	LC50	4000	300100
Pyridine	1	Oncorhynchus gorbuscha	Pink salmon	3.5(3.4-3.7) cm, 0.2(0.2-0.2)	48 H	FW; LAB	MOR	LC50	1200	300101
Pyridine	1	Oncorhynchus nerka	Sockeye salmon	3.9(3.5-4.3) cm, 0.5(0.3-0.6)	48 H	FW; LAB	MOR	LC50	6900	300102
Pyridine	1	Oncorhynchus mykiss	Rainbow trout	4.1(3.7-4.5) cm, 0.7(0.4-0.9)	48 H	FW; LAB	MOR	LC50	4600	300103
Pyridine	1	Oncorhynchus kisutch	Coho salmon	4.0(3.5-4.5) cm, 0.5(0.3-0.9)	72 H	FW; LAB	MOR	LC50	3800	300104
Pyridine	1	Oncorhynchus tshawytscha	Chinook Salmon	6.8(5.8-7.5) cm, 2.7(1.4-3.8)	72 H	FW; LAB	MOR	LC50	2900	300105
Pyridine	1	Oncorhynchus keta	Chum salmon	4.5(3.9-5.0) cm, 0.5(0.3-0.8)	72 H	FW; LAB	MOR	LC50	3900	300106
Pyridine	1	Oncorhynchus gorbuscha	Pink salmon	3.5(3.4-3.7) cm, 0.2(0.2-0.2)	72 H	FW; LAB	MOR	LC50	1200	300107
Pyridine	1	Oncorhynchus nerka	Sockeye salmon	3.9(3.5-4.3) cm, 0.5(0.3-0.6)	72 H	FW; LAB	MOR	LC50	6900	300108
Pyridine	1	Oncorhynchus mykiss	Rainbow trout	4.1(3.7-4.5) cm, 0.7(0.4-0.9)	72 H	FW; LAB	MOR	LC50	4600	300109
Pyridine	1	Oncorhynchus kisutch	Coho salmon	4.0(3.5-4.5) cm, 0.5(0.3-0.9)	96 H	FW; LAB	MOR	LC50	3800	300110
Pyridine	1	Oncorhynchus tshawytscha	Chinook Salmon	6.8(5.8-7.5) cm, 2.7(1.4-3.8)	96 H	FW; LAB	MOR	LC50	2900	300111
Pyridine	1	Oncorhynchus keta	Chum salmon	4.5(3.9-5.0) cm, 0.5(0.3-0.8)	96 H	FW; LAB	MOR	LC50	3700	300112
Pyridine	1	Oncorhynchus gorbuscha	Pink salmon	3.5(3.4-3.7) cm, 0.2(0.2-0.2)	96 H	FW; LAB	MOR	LC50	1100	300113
Pyridine	1	Oncorhynchus nerka	Sockeye salmon	3.9(3.5-4.3) cm, 0.5(0.3-0.6)	96 H	FW; LAB	MOR	LC50	6300	300114
Pyridine	1	Oncorhynchus mykiss	Rainbow trout	4.1(3.7-4.5) cm, 0.7(0.4-0.9)	96 H	FW; LAB	MOR	LC50	4600	300115
Pyridine	5	Pimephales promelas	Fathead minnow	31 d, 18.1 mm, 0.100 g	96 H	FW; LAB	MOR	LC50	93800	302666
Pyridine	5	Pimephales promelas	Fathead minnow	32 d, 20.6 mm, 0.140 g	96 H	FW; LAB	MOR	LC50	106000	302715
Pyridine	2	Mytilus edulis	Common bay mussel	1.4-2.6 g	96 H	SW; LAB	PHY		100000	313187
Pyridine	2	Daphnia magna	Water flea	NR	24 H	FW; LAB		EC100	1430000	314412
2-Chloropyridine	2	Tetrahymena pyriformis	Ciliate	NR	60 H	NR; LAB	GRO	EC50	657770	1101077
4-Chloropyridine, H	2	Tetrahymena pyriformis	Ciliate	NR	60 H	FW; LAB	PSR	EC50	826030	1081482
2,6-Dichloropyridin	2	Crangon septemspinosa	Sand shrimp	6.4-8.3 cm, 2.4-4.5 g	96 H	SW; LAB	MOR	LC50	>43000	1006933
3-Chloropyridine	2	Tetrahymena pyriformis	Ciliate	NR	60 H	NR; LAB	GRO	EC50	619680	1101072
4-Fluoroaniline	3	Oncorhynchus mykiss	Rainbow trout,donaldson tr	Fingerling, <= 10 cm	24 H	FW; LAB	STR*		5000(*)	1031586
4-Fluoroaniline	3	Lepomis macrochirus	Bluegill	Fingerling, <= 10 cm	24 H	FW; LAB	STR*		5000(*)	1031587
4-Fluoroaniline	3	Petromyzon marinus	Sea lamprey	Larvae, 8-13 cm	24 H	FW; LAB	STR*		5000(*)	1031588
4-Fluoroaniline	1	Pimephales promelas	Fathead minnow	32 d, 20.4 mm, 0.138 g	96 H	FW; LAB	MOR	LC50	16900	1102473

NOTES:

All units are in ug/l, except where noted.

[a] These values are codes which indicate the reliability of a study as established by the Environmental Protection Agency. 1 = meets all criteria, 2 = meets some criteria, 3 = does not meet criteria, and 4 = not reviewed.

[b] Values describe the exposure period followed in the study; H = hours, D = days.

[c] These codes describe general test conditions; FW = freshwater, SW = salt water, LAB = laboratory study, and NR = not reported.

[d] These are the categories which describe the type of endpoint effects observed in the study; PRG = population growth, MOR = mortality, IMM = immobility, GRO = growth, BEH = behavior, ABN = abnormality, PSE = photosynthisis, PHY = physiological, PSR = population size reduction, STR = stress.

[e] This number is a unique identifyer assigned to each record in the AQUIRE data base.

[f] Units are in ug/kg.

* = Refers to additional information provided in the AQUIRE database.

TABLE 4-13
SUMMARY OF ECOSAR MODEL INPUTS

OLIN CHEMICALS PHASE II RI REPORT
ROCHESTER, NEW YORK

Analyte	CAS Number	Water Solubility (mg/l) [a]	Molecular Weight (g/mole) [b]	Melting Point (deg.C)	Physical State	Log Kow [c]
2,6-Dichloropyridine	2402-78-0	--	147.99 [e]	NA	NA	2.15
2-Chloropyridine	109-09-1	--	113.55 [e]	NA	Liquid [d]	1.33
3-Chloropyridine	626-60-8	--	113.55 [e]	NA	NA	1.38
4-Chloropyridine	626-61-9	--	113.55 [e]	NA	NA	1.28
Pyridine	110-86-1	--	79.11	-41.6 [b]	Liquid [b]	0.67
p-Fluoroaniline	371-40-4	--	111.13	-1.9 [b]	Liquid [b]	1.15

NOTES:

[a] Available chemical data indicate that all analytes are at least slightly miscible.

[b] Values obtained from the The Merck Index (1989), unless otherwise noted.

[c] LogKow values are literature values (Hansch and Leo, 1979). Values for pyridine, 2-chloropyridine, and 3-chloropyridine are an average of the values presented in this reference.

[d] MDL Information System (1994).

[e] Molecular weights obtained from the Dictionary of Chemical Names and Synonyms (Howard and Neal, 1992)

Kow = Octanol-water partition coefficient.

NA = not available

Table 4-14
Summary of ECOSAR Model Equations Utilized in Surface Water Assessment

Olin Chemicals Phase II RI Report
Rochester, New York

Chemical Class	SAR Description [b]	SAR Equation	Reference
Neutral Organics	Daphnid 48-h LC_{50}	Log 48-h $LC_{50} = 1.72 - 0.91 \log K_{ow}$	Hermans et al., 1984.
Neutral Organics	Green Algae 96-h EC_{50}	Log 96-h $EC_{50} = 1.466 - 0.885 \log K_{ow}$	Calamari et al., 1983 Galassi, S., and Vighi, M., 1981 USEPA, 1991.
Neutral Organics	Fish 14-d LC_{50}	Log 14-d $LC_{50} = 1.87 - 0.871 \log K_{ow}$	Konemann, H., 1981.
Neutral Organics	Daphnid 16-d LC_{50}	Log 16-d $LC_{50} = 0.27 - 0.64 \log K_{ow}$	Hermans et al., 1984.
Neutral Organics	Daphnid 16-d LC_{50}	Log 16-d $EC_{50} = 0.05 - 0.72 \log K_{ow}$	Hermans et al., 1984.
Neutral Organics	Earthworm 14-d LC_{50}	Log 14-d $LC_{50} = 1.405 - 0.308 \log K_{ow}$	Neuhauser et al., 1986. Neuhauser et al., 1985.
Neutral Organics	Fish 96-h LC_{50}	Log $LC_{50} = 0.94 \log K_{ow} + 1.75$	Veith et al., 1983.
Neutral Organics	Green Algae Chronic Value	Log $ChV = 0.036 - 0.634 \log K_{ow}$	Calamari et at., 1983. Galassi, S., and Vighi, M., 1981 USEPA, 1991.
Neutral Organics	Fish 28-d BCF	Log $BCF = 0.79 \log K_{ow} - 0.40$	Veith, G.D., and Kosian, P., 1982.
Neutral Organics	Fish Chronic Value	Log $ChV = 0.72 - 0.87 \log K_{ow}$	USEPA, 1991.
Anilines	Fish 96-h LC_{50}	Log 96-h $LC_{50} = 0.956 - 0.739 \log K_{ow}$	Vieth, G.D., and Broderius, S.J., 1987.

See note at end of table

Table 4-14 (Continued)
Summary of ECOSAR Model Equations Utilized in Surface Water Assessment

Olin Chemicals Phase II RI Report
Rochester, New York

Chemical Class	SAR Description [a]	SAR Equation	Reference
Anilines	Daphnid 48-h LC_{50}	$\log 48\text{-h } LC_{50} = -1.623 - 0.271 \log K_{ow}$	Canton, J.H., and Adema, D.M.M., 1978. Kuhn et al., 1989. Sloof et al., 1983.
Anilines	Fish 14-d LC_{50}	$\log 14\text{-d } LC_{50} = 1.02 - 0.988 \log K_{ow}$	Hermans et al., 1984.
Anilines	Green Algae Chronic Value	$\log ChV = -0.411 - 0.588 \log K_{ow}$	Sloof et al., 1983.
Anilines	Fish Chronic Value	$\log ChV = -1.516 - 0.625 \log K_{ow}$	Bresch et al., 1990. Call et al., 1987. USEPA, 1990. USEPA, 1991. Van Leeuwen et al., 1990.
Anilines	Daphnid Chronic Value	$\log ChV = -3.12 - 0.36 \log K_{ow}$	USEPA, 1990.

Note:

[a] USEPA, 1994.

[b] Additional information is provided in Appendix E.