REPORT

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# Remedial Investigation

## Richardson Hill Road Municipal Landfill
## Sidney, New York

August 1995

O'BRIEN & GERE
ENGINEERS, INC.

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Report

# Remedial Investigation

*Richardson Hill Road Municipal Landfill*
*Sidney, New York*

Revised: March 1997

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# Executive summary

*Introduction*

The Richardson Hill Road Municipal Landfill Site (RHRMLS) has been identified by the United States Environmental Protection Agency (USEPA) as an abandoned disposal site (EPA ID# NYD980507735). The RHRMLS is located within northern Delaware County in south central New York State. The landfill is located on the west side of Richardson Hill Road approximately 3.3 miles south-southwest of Sidney Center, New York.

*Background*

Field investigations initiated by the USEPA in 1981 revealed that soils within and adjacent to a former waste oil pit at the site contained polychlorinated biphenyls (PCBs) and volatile organic compounds (VOCs) at concentrations which exceeded federal and state standards. In July 1987, the USEPA issued an Administrative Order on Consent (AOC) Index Number II CERCLA-70-205 to Amphenol Corporation and AlliedSignal, Inc. as potentially responsible parties (PRPs). The AOC required that the PRPs conduct a Remedial Investigation/Feasibility Study (RI/FS) at the site. As defined by the AOC, the RHRMLS covers approximately eight acres and consists of two sections, the Southern section (South Area) and the Northern section (North Area).

The South Area consists of the Richardson Hill Road Municipal Landfill (Landfill), a former waste oil pit located on the Landfill, and a man-made surface water body known as the South Pond, which is located downgradient and east of the Landfill. Historic aerial photographs of the South Area indicate the South Pond was constructed between 1963 and 1968. The Landfill was used primarily for the disposal of municipal refuse. The waste oil pit was used by the Landfill owner, from 1964 to 1966, to dispose of waste oils. These oils, some of which contained PCBs and VOCs, were reportedly disposed of in the pit as free liquids and were not containerized.

The North Area is located hydraulically downgradient of the adjacent Sidney Center Landfill on the east side of Richardson Hill Road. A small inactive gravel borrow pit occupies the central portion of the North Area. The northern boundary of the North Area is abutted by a man-made surface water body

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known as the North Pond. Disposal practices in the North Area are not documented.

Remedial investigations (RI) were performed in a number of phases by O'Brien & Gere Engineers, Inc. from September 1988 through September 1995. The objectives of the RI, as refined by the Interim Technical Memorandums (ITM) prepared at the conclusion of each RI phase and summarized in this report, were to identify and evaluate the extent of site contaminants to support the completion of a Risk Assessment (RA) and Feasibility Study (FS).

During the initial phase of the RI, a variety of media (including air, surface soils, subsurface soils, surface water, ground water, and sediments) were sampled in accordance with approved RI Work Plans, and analyzed using contract laboratory procedures (CLP). The resulting data indicated that VOCs and PCBs were the primary chemicals of concern at the site. With the exception of the ambient air quality samples, VOCs and PCBs were detected at various concentrations in the media types sampled. Pesticide, semivolatile and inorganic compounds were, in general, not found to be of concern at the site.

The investigations revealed that the primary area of contamination was the Landfill. Fill materials within the Landfill were found to consist of a heterogeneous mix of refuse materials. PCB and VOC contamination was found at varying concentrations in the fill materials and surrounding media. The highest concentrations of PCBs and VOCs were detected in and around the former waste oil pit. Sediments and surface water along the western shore of the South Pond were also impacted.

Detectable concentrations of PCBs and VOCs were found in the North Area, although at significantly lower concentrations than in the Landfill. Specific disposal areas were not found. Surface water and sediments in the North Pond exhibited minor impacts.

During subsequent phases of the RI, the vertical and horizontal extent of the fill materials in the landfill and North Area were delineated using geophysical surveys, soil vapor surveys, surface soil sampling, and soil borings. Ground penetrating radar surveys and test pits were used to confirm that the fill materials did not contain caches of buried drums. Investigations performed in the North Area identified scattered quantities of municipal refuse. The alleged disposal trenches discussed in the AOC, or other hot-spot areas, were not observed.

*Hydrology and Hydrogeology*

Surface water from the Landfill drains to the east toward the South Pond. The South Pond, in turn, drains to the south along an intermittent tributary. Discharge from the South Pond is currently controlled by a sediment trap system. Surface water from the North Area drains to the south toward the South Pond, and to the north toward the North Pond, which also receives runoff from the Sidney Center Landfill. The North Pond discharges to the north into Carr's Creek.

The subsurface geologic conditions at the RHRMLS are characterized by unconsolidated deposits overlying bedrock, which consist of interbedded layers of shale, siltstone, and sandstone. The unconsolidated overburden deposits are predominantly low permeability glacial till, composed of silt, clay, sand and rock fragments. Sediment cores retrieved from the South Pond indicate that the bottom is composed of low permeability silt and clay.

Ground water in the overburden and shallow bedrock flows east from the Landfill to the South Pond, and generally follows site topography. In a similar fashion, ground water from the Sidney Center Landfill flows from east to west toward both the North and South Ponds.

The overburden and shallow bedrock ground water flow regimes at the Landfill are hydraulically connected. However, large differences in vertical hydraulic head suggest that the deeper bedrock ground water is isolated from the overburden and shallow bedrock ground water zones. Ground water elevation data indicate that overburden and shallow bedrock ground water in the North Area emanates from the Sidney Center Landfill flowing west to the North Pond and southwest to the South Pond.

*Extent of Contamination*

In the South Area, ground water in the overburden contained detectable levels of VOCs and PCBs which emanated from the landfill materials and former waste oil pit. The VOC and PCB plume extends in an easterly direction from the RHRMLS to the South Pond. The highest concentrations of VOCs and PCBs were detected in the vicinity and downgradient of the former waste oil pit. Shallow bedrock ground water also contained detectable levels of PCBs and VOCs. The shallow bedrock ground water plume is, however, smaller in extent and magnitude than the overburden plume. The relative lack of change in the extent and magnitude of the contaminant plume over the course of the RI would suggest that the system is in equilibrium.

Neither VOCs or PCBs were detected in the deep bedrock ground water at the site. The absence of these constituents in the deep bedrock ground water

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indicates this zone is isolated from the overburden and shallow bedrock ground water zones.

Surface water and sediment samples collected along the west shoreline of the South Pond contained elevated levels of VOCs and PCBs. The presence of these contaminants is attributed to the seepage and discharge of leachate and ground water from the Landfill into the pond. Low levels of VOCs and PCBs were also detected downstream of the South Pond. These levels generally decreased with increasing distance from the pond. Low concentrations of VOCs and PCBs were also detected in surface water and sediment samples collected along the southern and eastern shoreline of the North Pond. These impacts are attributed to surface water runoff and ground water discharge from the adjacent Sidney Center Landfill.

Elevated concentrations of VOCs were also detected in two of three springs located hydraulically downgradient of the Sidney Center Landfill. Two springs located to the south of the Richardson Hill Road Landfill, sampled by the New York State Department of Health (NYSDOH), did not contain site contaminants.

*Interim Remedial Measures and Response Actions*
During the course of the RI several interim remedial measures (IRM) and time critical response actions were completed at RHRMLS and included:

- the installation of temporary high visibility fencing around the waste pit and the East side of Richardson Hill Road along the South Pond.
- the posting of signs and locked gates to discourage unauthorized access to the RHRMLS.
- the construction of a sediment trap at the outlet from the South Pond to minimize downstream migration of sediments.
- the removal of sediments along the western portion of the South Pond.
- the installation of whole-house water supply treatment systems on spring water supplies located downgradient of the Sidney Center Landfill.

*Sidney Center Landfill RI/FS*
From 1991 to 1995, Malcolm Pirnie, Inc., on behalf of the USEPA, conducted an RI/FS at the Sidney Center Landfill. The results of the RI indicated the presence of VOCs in ground water. Similar contaminants were also detected in residential springs located downgradient of the landfill. In addition, organic compounds such as PCBs, benzene, toluene, ethylbenzene, and xylene, semivolatiles, and pesticides were detected in on-site surface soils, subsurface soils, and leachate seeps.

Based on fish tissue samples collected from the North Pond, the study concluded that fish were not being adversely affected in the vicinity of the site. The human health evaluation indicated that contaminated (untreated) spring water presented a potential risk to current and/or future residents, and that the presence of PCBs and pesticides in the site media were likely to have an adverse effect on local wildlife if unremediated.

Due to the mixed nature and extent of the waste materials present at the Sidney Center Landfill, the FS proposed the use of a presumptive remedy. The selected remedy outlined in the draft record of decision (ROD) recommends the capping of separate waste disposal areas, the installation of a focused "hot-spot" ground water recovery and treatment system, ground water monitoring, and the implementation of institutional controls to discourage unauthorized access.

*Risk Assessment and Feasibility Study*

The data collected during the RHRMLS RI were validated and found to be of sufficient quality to support a Risk Assessment and prepare a Feasibility Study. The extent and heterogeneous nature of the waste materials in the Landfill, and similarities to the Sidney Center Landfill, should allow for the use of the presumptive remedy for this portion of the RHRMLS. The use of the presumptive remedy for this portion of the site will be addressed in the FS, which will identify environmentally sound and cost effective remedial alternatives for the RHRMLS.

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# 1. Introduction

## 1.1. Purpose of report

The Richardson Hill Road Municipal Landfill Site (RHRMLS) has been identified by the United States Environmental Protection Agency (USEPA) as an abandoned disposal site that requires a Remedial Investigation/Feasibility Study (RI/FS). Pursuant to provisions of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) the site is currently on the USEPA's National Priorities List (NPL) as site #NYD980507735.

In 1981 investigations at the RHRMLS were initiated under direction of the USEPA. The investigations identified the presence of a waste oil pit, as well as the presence of volatile organic compounds (VOCs) and polychlorinated biphenyls (PCBs) at the site. In July 1987 the USEPA issued an Administrative Order on Consent (AOC) Index Number II CERCLA-70-205 to Amphenol Corporation and AlliedSignal, Inc. as potentially responsible parties (PRPs).

From September 1988 to September 1995, O'Brien & Gere Engineers, Inc. performed several phases of field investigations as components towards completion of the RI. The objectives of the RI were to evaluate:

1. the extent and chemical characteristics of waste materials at the RHRMLS; and
2. the nature, extent, and distribution of contaminants released from the waste materials to the surrounding environment.

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This RI report presents the findings and conclusions of the work tasks that have been approved by the USEPA to complete the RI at the RHRMLS.

## 1.2. Site background

### 1.2.1. Site description

The RHRMLS is located in northern Delaware County in south central New York State (Figure 1). The site consists of two sections designated as the Northern section (North Area) and Southern section (South Area). The South Area is comprised of the municipal landfill called the Richardson Hill Road Municipal Landfill (Landfill) which contains a former waste oil pit, and a pond called the South Pond (Figure 2).

The Landfill is located on the west side of Richardson Hill Road approximately 3.3 miles south-southwest of Sidney Center, New York (Figure 1). The Landfill covers approximately eight acres and is situated along a hillside above a drainage ditch, a marsh, and a man-made pond designated as the South Pond (Figure 2). Historic aerial photographs indicate that the South Pond was constructed between 1963 and 1968.

The Landfill was used primarily for the disposal of municipal refuse. A former waste oil pit was used to dispose of waste oils, some of which contained VOCs and PCBs. Municipal and liquid waste disposal practices are not well documented. The oils were reportedly disposed in the pit as free liquids and were not containerized. The former waste oil pit is approximately 25 ft wide by 105 ft long by 14 ft deep.

Landfill surface water drains into the marsh and South Pond through a drainage ditch which is parallel with the road. Water from the South Pond drains through a sediment trap weir system and a beaver



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dam, then south into Herrick Hollow Creek, and eventually into the west branch of the Delaware River and the Cannonsville Reservoir.

The North Area is located about 1,000 ft northeast of the Landfill and directly west of the Sidney Center Landfill (Figure 2). The North Area is situated on a drainage divide between the Susquehanna and Delaware River basins, with the primary drainage towards the Delaware basin. The North Area is abutted by a man-made surface water body called the North Pond. Water from the North Pond drains through a series of beaver dams, then north into Carr's Creek, which is a tributary to the Susquehanna River.

The North Area was identified by the USEPA through use of aerial photographs as having two suspected trenches (USEPA, 1985). Trench locations were not identified in field reconnaissance or investigations. A gravel borrow pit may have been identified as one of the alleged trenches. Surface debris was identified in small sporadic areas. Disposal practices in the North Area are not documented.

### 1.2.2. Site history

Disposal of spent oils from the Scintilla Division of The Bendix Corporation, Sidney, New York occurred at the Landfill from mid-1964 until mid-1966. In June 1964, the New York State Department of Health (NYSDOH) notified the Town of Sidney that it agreed with the disposal of spent oil wastes from Scintilla at the proposed Landfill. In July 1964, the Town of Sidney entered into a contract with Devere A. Rosa for disposal of town wastes at the Landfill, including spent oils from Scintilla. On July 14, 1964, the DOH issued an operating permit to Devere A. Rosa for the Landfill. Pursuant to his contract with the Town, Mr. Rosa collected and disposed of spent oil from Scintilla at the Landfill until June 1966. On June 28, 1966, Mr. Rosa was directed by the NYSDOH to cease collecting spent oils from Scintilla. In July 1966, the NYSDOH reported that spent oil collection from Scintilla had ceased.

It is known that garbage and waste oils were collected from municipal and commercial establishments in a number of local jurisdictions in the area for disposal at the Landfill. There are no existent records which outline the quantities of spent oil collected by Mr. Rosa from Scintilla or from any other generator.

**1.2.3. Summary of site investigations**

In December 1980, a preliminary inspection of the RHRMLS was made by Fred C. Hart Associates, Inc. on behalf of the USEPA under Technical Direction Document (TDD) #02-8011-27. Based on this inspection, Fred C. Hart Associates, Inc. performed a field assessment and sampling at RHRMLS in August 1981. This assessment revealed the presence of PCBs (Aroclor 1248) and VOCs in sediment and water samples collected from the waste oil pit and downgradient of the pit.

To establish and define RI objectives, a Site Operations Plan (SOP) was developed by Environmental Resources Management in February 1987. A Quality Assurance Project Plan (QAPP) was included in the SOP. In March 1987, Amphenol Corporation (formerly Bendix Corporation) retained O'Brien & Gere Engineers, Inc. to incorporate USEPA Region II comments into the SOP and to implement its scope. The revised SOP was submitted and approved by USEPA in July 1988 (O'Brien & Gere Engineers, Inc., 1988). The necessary property access agreements were obtained from July to September 1988. Phase I site investigative work commenced in September 1988 and was completed in November 1988.

Phase I field investigations included an on-site air monitoring program to evaluate the nature and magnitude of airborne transport of potential volatile and fugitive dust emissions from the waste oil pit; a test boring and soil sampling program to define and characterize the area affected by waste oil disposal; and the installation and sampling of ten overburden and one bedrock monitoring well to assess baseline ground water quality.

The results of the Phase I investigations were presented in an Interim Technical Memorandum (ITM) prepared by O'Brien & Gere Engineers, Inc. and submitted to USEPA in May 1989 (O'Brien & Gere Engineers, Inc., 1989 - Exhibit I). The Phase I investigations provided an evaluation of the chemistry of waste pit materials and general ground water flow conditions, but did not fully identify the extent of soil and ground water which was affected by the former waste oil pit. The ITM proposed additional Phase II work efforts.



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Phase II work efforts were designed to expand the RHRMLS database to support an Endangerment Assessment and Feasibility Study. Non-CLP laboratory methods were proposed and utilized for VOC and PCB sample analyses. Samples submitted for VOC and PCB analyses were analyzed using gas chromatography (GC) methods to improve method detection limits.

Subsequent to USEPA approval in August 1990, Phase II field investigations commenced in August 1990 and were completed in December 1990. The Phase II work tasks included confirmatory ground water sampling, surficial geophysical surveys, a soil vapor survey, additional surface and subsurface soil sampling, additional monitoring well installations and associated ground water sampling and analysis, surface water and sediment sampling and analysis, a cultural resource assessment, and wetland delineation. Site field activities were observed and approved by the USEPA oversight contractor VERSAR, Inc.

The results of the Phase II work efforts are summarized in an Interim Technical Memorandum (O'Brien & Gere Engineers, Inc., 1991-Exhibit II). An evaluation of data collected during these investigations indicated that the horizontal and vertical extent of contamination in various media had not been completely evaluated. Additional work tasks, designated as Phase III, were required to collect the additional data necessary to support an Endangerment Assessment and Feasibility Study.

The additional Phase III work tasks included a ground water users survey to locate residential ground water users; monthly ground water and surface water monitoring for a one-year period to evaluate hydrology and variations in seasonal ground water flow conditions; bedrock monitoring well installations to further evaluate site hydrogeology and the horizontal and vertical extent of site-related parameters; ground water sampling and analysis; surface water and sediment sampling and analysis; and the installation of three test wells to evaluate the hydrogeologic characteristics of the overburden water bearing unit.

Subsequent to the Phase III work efforts, a Revised Draft RI Report was prepared describing all investigative activities and results. The RI Report was submitted to the USEPA in September, 1993. Based upon USEPA review, additional Supplemental Remedial Investigations (SRI) were requested at RHRMLS.

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The SRI was performed at RHRMLS to further evaluate the nature, vertical, and horizontal extent of identified site contaminants present in the bedrock aquifer at the site. The SRI was completed in accordance with the work tasks and procedures presented in the SRI Work Plan, which was approved by the USEPA in December 1994.

The SRI work tasks included the following:

1. Performance of a complete round of ground water sampling and analysis of existing site monitoring wells. Samples were analyzed for VOCs and PCBs.
2. Installation and sampling of two additional shallow bedrock and two deep bedrock monitoring wells.
3. Performance of a complete round of ground water elevation monitoring concurrent with ground water elevation monitoring at the adjacent Sidney Center Landfill Superfund Site.

As discussed in Section 1.2.5., additional "Response Actions" were performed at the RHRMLS to satisfy requirements of a second USEPA AOC and Unilateral Administrative Order. Investigations performed as components of the "Response Actions" provided further characterization of the waste oil pit, "hot spot areas", and residential water supply springs. Specifically, characterization included the following:

- installation of soil borings to evaluate the extent of waste debris, contaminated soil, and dense-non aqueous phase liquids (DNAPLs) in the vicinity of the waste oil pit and "hot spot" areas,
- installation of monitoring wells to evaluate the presence of light non-aqueous liquids (LNAPLs) and migration potential, if any, within and downgradient of the waste oil pit as well as adjacent to the South Pond.
- baseline water quality monitoring of three water supply springs

The SRI and additional work efforts, in addition to the previous phases of investigations completed at RHRMLS, are presented in this final RI Report.

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**1.2.4. Interim remedial measures**
In association with the RI, interim remedial measures were performed in November 1991 to prevent unauthorized access to the RHRMLS. These measures involved the installation of fencing and the posting of signs around the former waste oil pit and runoff area which read "CAUTION: CONTAMINATED SOIL - DO NOT CROSS FENCED AREA." The 4-ft high construction barrier fence was installed to restrict access to this area.

The entire perimeter of the Landfill was posted, at 50-ft intervals, with signs which read: "NO TRESPASSING - FEDERAL SUPERFUND SITE - RICHARDSON HILL ROAD LANDFILL - FOR INFORMATION CALL TOLL FREE: UNITED STATES ENVIRONMENTAL PROTECTION AGENCY AT: 1-(800)-346-5009."

**1.2.5. Response actions**
Amphenol Corporation and AlliedSignal, Inc. (Respondents) implemented a "Response Action" at the RHRMLS in accordance with the requirements of the USEPA AOC, Index Number II CERCLA-93-0214 dated September 22, 1993 and UAO, Index Number II-CERCLA-93-0217 dated September 30, 1993. The AOC "Response Action" involved the characterization of baseline water quality conditions of three shallow dug well residential water supply springs and the installation and performance monitoring of two whole-house supply water treatment systems.

The UAO ordered that a time-critical response action be performed at RHRMLS to remove an actual or threatened release of hazardous substances from the landfill to the South Pond. As such, the UAO involved two separate phases and associated work plans. Work Plan I involved additional characterization within and downgradient of the former waste oil pit and associated hot spot areas. Work Plan II addressed the potential migration of contaminants and LNAPL into the South Pond and also addressed the potential transport of sediment from the South Pond.

Work Plan I Investigations indicated that the former waste oil pit and associated hot spots are not presently a reservoir of free-phase oil and do not constitute a continuing source of LNAPL. In addition, PCB concentrations in the hot spot areas were detected below the UAO action limit of 100 ppb. Continuing sources or recoverable quantities of LNAPL are not currently present in the waste oil pit, immediately downgradient from the waste oil pit, or, with the exception of the former monitoring well MW-5S location, beneath Richardson Hill Road. LNAPL in the vicinity of the former MW-5S is currently being controlled with a passive collection system. DNAPL was not observed at the overburden/bedrock interface downgradient from the waste oil pit or in the vicinity of the hot spot areas.

Work Plan II activities involved the construction of a sediment trap weir system at the outlet from the South Pond. The system was designed to control the pond elevation and reduce the potential for sediment migration at the pond outlet. A focused sediment removal program was performed in South Pond to reduce the potential ecological risks associated with site contaminants in the sediments. Approximately 2200 cubic yards of sediment were removed from the South Pond and consolidated on-site in HDPE lined and covered storage cells.

A summary of the AOC response actions are presented in the approved AOC Final Report dated July 1995 (Exhibit III). The UAO response actions are summarized in the approved UAO Final Report dated November 1994 (Exhibit IV).

**1.2.6. Related Investigations**

Sidney Center Landfill is a CERCLA NPL site located approximately 1500 feet northeast of Landfill and due east and adjacent to the North Area (Figure 2). The landfill is situated east of Richardson Hill Road on the western slope of Richardson Hill.

The landfill site comprises approximately 74 acres of which 12 to 15 acres contain municipal and commercial landfill waste. Landfilling operations occurred between 1968 and 1972. Waste disposal practices are not well documented.

Investigations performed at Sidney Center Landfill detected the presence of LNAPL, chlorinated solvents, PCBs, and volatile petroleum hydrocarbons in the overburden and shallow bedrock

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ground water zones. Total concentrations of VOCs ranged up to 300 ppb. Residential spring samples contained similar contaminants at total concentrations of VOCs up to 750 ppb.

The overburden deposits were unsaturated across most of the site. Ground water occurrence and flow at the site is within the fractured bedrock underlying the unsaturated overburden. Flow occurs primarily along bedding planes and fractures. Rapid changes in bedrock ground water elevations during precipitation events and snow melt indicate that the shallow bedrock ground water is hydraulically connected to the overburden. Ground water flow across Sidney Center Landfill is west towards the North Pond and RHRMLS North Area and South Pond.

Contaminants detected in site surface soils were primarily PCBs and pesticides. Volatile petroleum hydrocarbons, PCBs, semi-volatile organic compound (SVOCs), and VOCs were detected in subsurface soils. Leachate seep samples exhibited concentrations of total organic compounds up to 695 ppb.

The human health evaluation found that trichloroethene and manganese in spring water and manganese, arsenic, antimony, barium, beryllium, vinyl chloride, and PCBs in ground water pose risks to human health if left untreated. However, the whole-house water treatment systems installed by Amphenol and AlliedSignal, Inc. under the AOC treat the spring water to levels below Federal and State standards. The naturally high turbidities of the ground water samples were found to be the reason that inorganic analytes were at concentrations large enough to be identified as chemicals of potential concern.

The Environmental Assessment found that several pesticides, PCBs, and several inorganic compounds are likely to have adverse effects on the wildlife using the site vicinity if left unremediated. A feasibility study (FS) has been performed at the Sidney Center Landfill to identify an environmentally sound and cost effective remedial alternative. A draft Record of Decision (ROD) has been issued by the USEPA for the site.

# 2. Field investigations

## 2.1. Topographic mapping

In July 1987, Lockwood Support Services was retained by O'Brien & Gere Engineers, Inc. to perform topographic mapping services at the RHRMLS. The services included:

- a fly-over for site aerial photography which was used for topographic map preparation
- development of a site topographic map at a scale of 1 inch equals 100 ft and a contour interval of 5 ft.

## 2.2. Contaminant source investigation

### 2.2.1. Geophysical surveys

*Electromagnetic and Magnetometer Surveys*

Electromagnetic and magnetic geophysical surveys were conducted at the Landfill and the North Area from September 9 to 14, 1990. USEPA oversight for the surveys was provided by VERSAR, Inc. The purpose of the surveys was to identify fill and waste oil disposal areas, including areas containing subsurface metallic materials.

The surveys were performed by traversing an established site-wide 100-ft grid with a spacing of 20 ft between measuring stations

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(Figures 3 and 4). The surveys were performed using a Geonics EM-31 Terrain Conductivity Meter and a EG & G Model 846 Proton Magnetometer. The terrain conductivity meter was used in the quadrature phase to identify conductivity variations within natural soil and fill materials. In areas where soil conductivity variations were detected, EM-31 surveying was conducted using the in-phase mode of operation to identify areas containing buried metallic materials. Additionally, the magnetometer was utilized to detect the presence of subsurface metallic materials.

Terrain conductivity and proton magnetometer survey data results are included in Appendix A and discussed in Section 4.1.2 of this report.

*Ground Penetrating Radar Surveys*
Ground penetrating radar (GPR) surveys were completed at the Landfill and North Area to address three objectives:

1. further evaluate the nature of the magnetic anomalies identified with the EM-31 and proton magnetometer surveys.
2. satisfy a NYSDEC request to evaluate the anomalously high PCB concentration detected in soil boring SB-10 located in an area southwest of the former waste oil pit.
3. Evaluate the extent of LNAPL beneath Richardson Hill Road.

The first two objectives were addressed on May 28, 1992. The GPR surveys were completed in accordance with the March 1992 work plans which were approved by the USEPA in correspondence dated April 28, 1992 (O'Brien & Gere Engineers, Inc., 1992a and O'Brien & Gere Engineers, Inc., 1992b).

The third objective was addressed on August 23, 1994. The GPR survey was observed by the USEPA On-Scene Coordinator.

GPR surveys were performed by Detection Sciences, Inc. The surveys was completed using a Geophysical Surveys Systems Inc. Subsurface Interface Radar System - 8. GPR is capable of detecting foreign objects in the ground in either the solid or liquid form. Solid and liquid materials produce characteristic radar signatures which can usually be differentiated from native soils and ground water (Appendix B; Section 4.1.2.)

Eight GPR transects, 100 ft in length, were completed at the Landfill and radiated away from grid point N400-W100, which is the location of soil boring SB-10. Six GPR transects, 100 ft in length or less, were completed in the North Area and radiated away from grid point N1800-E900, which is located in a fill area adjacent to the North Pond. The locations of the May 1992 GPR transects are included in the Detection Sciences, Inc. Report in Appendix B.

Three parallel GPR transects were run along Richardson Hill Road in August 1994. The transect survey lines were approximately 1000 foot. The location of the survey lines are presented in the Detection Sciences Report included as Appendix C in Exhibit IV.

### 2.2.2. Soil vapor survey

A preliminary soil vapor survey was completed from August 29 to September 6, 1990 prior to initiating a comprehensive soil vapor survey at Landfill and the North Area. Sampling was conducted in accordance with Appendix D of the Phase I ITM (O'Brien & Gere Engineers, Inc., 1989 - Exhibit I ) and correspondence with USEPA Region II dated January 26, 1990 and March 26, 1990.

During the preliminary survey, soil vapor samples were collected adjacent to monitoring wells MW-1 through MW-10. Total VOCs from the soil vapor survey were compared with total VOCs from ground water headspace samples to evaluate the feasibility of completing a site-wide soil vapor survey. Additionally, a soil vapor depth analysis was completed at three locations to evaluate the optimum depth for conducting a site-wide survey. Soil vapor samples were extracted and analyzed in the field using a portable gas chromatograph (GC).

Subsequent to completing the preliminary soil vapor survey, comprehensive site testing was initiated at Landfill and the North Area from September 27 to October 3, 1990. The purpose of the survey was to evaluate the spatial distribution of VOCs in subsurface soil and ground water.

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A total of ninety-two soil vapor samples were collected from a depth of 3 to 4 feet below the ground surface at locations shown on Figure 5. Sampling was conducted on a 100-foot grid that was established prior to the survey. Vapor samples were collected in dedicated syringes and injected directly into a portable GC. Soil vapor data were reduced to total peak area in volt-seconds, which in turn were converted to total VOCs in parts per million (ppm). Total VOC concentrations detected at the sampling points are shown in Table 1 and on Figure 5.

## 2.3. Soil investigation

### 2.3.1. Surface soil sampling

During Phase I of the RI, surface soil sampling was conducted directly down slope from the former waste oil pit to evaluate the nature, magnitude, and distribution of PCBs in this area. Ten samples, designated as SS-1 through SS-10, were collected and analyzed for PCBs and oil & grease at the location shown on Figure 6. Samples were collected in accordance with the protocols outlined in the SOP (O'Brien & Gere Engineers, Inc., 1988). The results of the laboratory analyses are summarized on Table 2.

Surface soil sampling during Phase II of the RI was conducted on a site-wide basis, at the Landfill and the North Area, to further evaluate direct contact exposure pathways and to evaluate the areal distribution of surface soil contamination. The sampling was also completed to identify and quantify areas of contamination associated with the disposal of waste oils, and to direct the soil boring program.

Thirty-eight surface soil composites designated as SSC-1 through SSC-38 were collected at the locations shown on Figure 7. Samples SSC-1 through SSC-32 were collected at the Landfill and SSC-33 through SSC-38 were collected in the North Area. Within the sample locations three grab samples were collected from the apex of each sampling triangle and one at the midpoint of each hypotenuse.

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Sample collection was completed in accordance with procedures in the SOP (O'Brien & Gere Engineers, Inc., 1988). The four sub-samples were composited and homogenized in a stainless steel bowl. The composited samples were extracted in the field and screened using a field GC for PCB concentrations in accordance with protocols in Appendix D of the Phase I ITM (O'Brien & Gere Engineers, Inc., 1989-Exhibit I). The results of the screening are summarized in Table 3 and illustrated on Figure 7.

Based on the results of the site-wide PCB screening, four of the composites, SSC-7, SSC-16, SSC-18 and SSC-24, were submitted to O'Brien & Gere Laboratories, Inc. in Syracuse, New York for PCB analyses and two composites (SSC-11 and SSC-31) were submitted for PCBs and Target Analyte List (TAL) metals. Laboratory results for PCB data are shown on Table 2 and on Figure 7.

Four additional surface soil samples (ASS-1 to ASS-4) were collected during implementation of the UAO in May 1994 (Figure 6). The samples were collected from surface water runoff pathways down slope from the former waste oil pit to supplement existing RI data. Each sample was analyzed for PCBs and oil & grease using EPA Methods 8080 and 418.1, respectively. Laboratory data are included in Table 2.

### 2.3.2. Subsurface soil borings

Subsurface soil characterization was conducted at the site to evaluate the vertical extent of site-related compounds that were detected in the surface soils. During Phase I of the RI, twenty-eight soil boring locations (designated B-series) were concentrated in and around the former waste oil pit to evaluate the horizontal and vertical extent of contaminants in that area (Figure 6).

A total of 20 soil borings, designated as SB-1 through SB-20, were completed during Phase II of the RI in November 1990. Locations of the soil borings are shown on Figure 8. Three borings were completed in the North Area.

Soil borings were completed by Parratt-Wolff, Inc. under the observation of an O'Brien & Gere Engineers, Inc. geologist. A USEPA oversight contractor (VERSAR, Inc.) was present during the soil boring program and approved the boring locations prior to

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initiation. The borings were completed in accordance with protocols outlined in the SOP (O'Brien & Gere Engineers, Inc., 1988).

Soil samples were collected at 2-ft intervals from the ground surface to a depth of 16 feet using split spoon sampling methods in accordance with ASTM Method D-1586-84. Immediately upon retrieval, Phase II soil samples were placed in pre-weighed eight-ounce jars for VOC screening using photoionization (PID) and flame ionization (FID) detectors. Subsequent to collection, the jars were weighed to obtain mass-corrected standardized VOC data for each 2-foot depth interval (Table 4). Standardized VOC values were obtained by dividing the VOC screening value by the sample weight. Soil descriptions, including moisture content, color, particle size distribution, degree of roundness, lithology, and any structure present, were included on the soil boring logs (Appendix C).

A composite sample, representing a 4-foot depth interval, was also collected for in-field PCB GC screening according to procedures described in the approved SOP (O'Brien & Gere Engineers, Inc., 1988). The PCB screening samples were homogenized in stainless steel bowls prior to analysis. Based on the results of field screening, eight soil samples which exhibited the highest levels of VOCs and/or PCBs were submitted for laboratory analyses of PCBs using EPA Method 8080.

After soil boring completion the lower portion of each borehole which encountered potentially contaminated materials was backfilled a minimum of 8 feet with a cement/bentonite mixture to inhibit the downward migration of potentially contaminated fill materials. The remaining upper portion of the borehole was backfilled with cuttings mixed with dry bentonite. In areas where landfill materials were not encountered, boreholes were backfilled with cuttings mixed with dry bentonite.

Subsequent to backfilling, drilling and sampling equipment were decontaminated between boring locations using a portable steam cleaner. Sampling equipment was decontaminated following procedures described in the SOP (O'Brien & Gere Engineers, Inc., 1988), and included a non-phosphate detergent wash followed by a potable water rinse, acetone rinse, and a final deionized water rinse.

Twenty-two additional soil borings were completed during the UAO response action in May, 1994 to further assess the horizontal and vertical extent of the former waste pit, to evaluate where substantial drop-offs in contaminant levels exist, to evaluate whether a continuing source of LNAPL or free-phase oil was present in the pit, and to identify the presence of DNAPL downgradient from the pit. Three of the UAO soil borings were completed adjacent to Phase II soil boring SB-10 (designated "hot-spot") and two were completed adjacent to Phase I boring B-16 (designated "hot-spot") to further evaluate PCB concentrations at these locations. The location of the soil borings are presented on Figure 3 of the approved UAO Final Report (Exhibit IV).

## 2.4. Surface water and sediment sampling

Surface water and sediment sampling was completed at RHRMLS to evaluate the areal extent of site-related compounds. During Phase II of the RI, a total of 42 sediment samples were collected and analyzed for PCBs using the in-field screening procedures outlined in Appendix D of the Phase I ITM (O'Brien & Gere Engineers, Inc., 1989-Exhibit I). Screening results are listed in Table 5 and illustrated on Figures 9 and 10. Based on results of the sediment screening, four surface water (SWTCL#1 to SWTCL#4) and four sediment samples (SEDTCL#1 to SEDTCL#4) were analyzed for TCL parameters. Additionally, five surface water (SWPCB#1 to SWPCB#5) and ten sediment samples (SEDPCB#1 to SEDPCB#10) were analyzed for PCBs.

To further evaluate the distribution of site-related constituents within and downstream of the South Pond, additional surface water and sediment samples were collected during Phase III of the RI. In December 1991, 20 surface water and 20 sediment samples were collected and are designated 91-1 through 91-20.

To confirm the results of the December 1991 data and to further evaluate VOC and PCB concentrations downstream of the landfill, ten surface water and ten sediment samples were collected in September 1992 and are designated 92-1 through 92-10. Surface water and sediment samples were collected in accordance with protocols in Appendix D of the Phase I ITM (O'Brien & Gere

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Engineers, Inc., 1989 - Exhibit I). Immediately upon collection, samples were transferred to properly labelled containers and placed in a cooler with ice. Chain of custody documentation accompanied the coolers to O'Brien & Gere Laboratories, Inc. in Syracuse, New York.

## 2.5. Ground water investigations

### 2.5.1. Monitoring well installations

A total of eighteen overburden monitoring wells, eight shallow bedrock, and two deep bedrock monitoring wells were installed to provide site-specific information pertaining to geology, hydrogeology and ground water quality. Additionally, three PVC stilling wells were placed in both the North and South Ponds. The stilling wells were used to measure surface water elevations. Surface water elevations were used to evaluate the ground water recharge or discharge status of the ponds.

The monitoring wells were installed in four phases: September 1988; October 1990; December 1991; and during the SRI in January/February 1995. The wells were installed under the observation of an O'Brien & Gere Engineers hydrogeologist in accordance with the approved protocols outlined in the 1988 SOP (O'Brien & Gere Engineers, Inc., 1988). The SRI wells were installed in accordance with the procedures outlined in the approved SRI Work Plan (O'Brien & Gere Engineers, Inc., December 1994).

The monitoring well locations were selected with concurrence from the USEPA and/or USEPA subcontractor VERSAR, Inc. prior to installation. Locations of the monitoring and stilling wells are shown on Figure 11. Well as-built data, including ground elevation, total well depth, and screened intervals are included on Table 6. Boring logs and well construction details are included in Appendix C.

*Overburden Monitoring Wells*
Overburden monitoring wells were installed using 4 1/4-inch I.D. hollow stem augers. Continuous split spoon soil sampling was conducted, in accordance with ASTM Method D-1586-84, to a depth of 16 ft; below which standard soil sampling at 5-ft intervals was employed until the completion depths of the wells. Soil samples were described and logged in detail by an on site O'Brien & Gere hydrogeologist.

The ground water monitoring wells were constructed of 2-inch I.D. schedule 5, stainless steel riser pipe and flush joint threaded 0.020 inch wire-wound stainless steel well screen. Monitoring well risers and screens were also decontaminated prior to their use. A silica sand filter pack compatible for use with a 0.020-inch slotted screen was installed into the annular space around the well screens a minimum of 2-ft above the screened interval. A minimum 2-ft thick bentonite seal was installed above the filter packs. The remaining annular space was filled with a cement/bentonite grout mixture using a tremie pipe.

*Shallow Bedrock Monitoring Wells*
Shallow bedrock monitoring wells were installed using two methodologies. During the initial three RI phases, hollow stem augers were advanced a minimum of 5 ft into the weathered surface of the bedrock. The augers were then filled with a cement/bentonite grout, pulled, and replaced with permanent 6-inch I.D. steel casing. The casing was allowed to set for a minimum of 24 hours before a roller-bit was advanced to the desired depth inside of the casing. The wells were constructed as described under *Overburden Monitoring Wells*.

During the SRI, the shallow bedrock monitoring wells (MW-3D and MW-9D) were installed using a combination of hollow-stem augers and conventional air rotary drilling to initially set permanent 6-inch diameter casings approximately 5 feet into competent bedrock. The steel casings were installed into the borehole and securely grouted in place using a tremie pipe and a portland cement/bentonite mixture. The grout mixture was allowed to cure for a minimum of 24 hours prior to subsequent core barrel sampling.

The SRI shallow bedrock monitoring wells were advanced at approximate five foot intervals using nominal 2-inch diameter



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diamond tipped NX coring techniques. MW-3D and MW-9D were completed to approximate depths of 45.5 and 68 feet below grade, respectively. Subsequent to coring, each corehole was reamed with a 5-7/8 inch air hammer bit prior to well installations.

The recovered cores were visually described and Rock Quality Designation (RQD) values were calculated by the on-site hydrogeologist.

PVC monitoring wells were installed using shallow bedrock boring. Wells was constructed using a 15-foot length of 2-inch I.D., 0.020-inch machine slotted screen attached to appropriate lengths of riser casing. A silica sand filter pack compatible for use with a 0.020-inch slotted screen was installed into the annular space around the well screens a minimum of 2-ft above the screened interval. A minimum 2-ft thick bentonite seal was installed above the filter packs. The remaining annular space was filled with a cement/bentonite grout mixture using a tremie pipe. MW-3D was completed with a flush mount locking well cover. MW-9D was completed to approximately two feet above grade and finished with a 6-inch diameter locking well cover.

*Deep bedrock monitoring well installation*
The deep bedrock monitoring wells (MW-3DD and MW-18DD) were installed using a combination of mud rotary, air rotary, and air coring methods. Initially, mud rotary drilling techniques were used to advance a nominal 18-inch diameter borehole through the overburden and into the bedrock. The 18-inch boreholes were advanced at MW-3DD and MW-18DD to approximately 48 and 55 ft below grade, respectively. Permanent 12-inch diameter carbon steel casings were installed into the boreholes and grouted using a tremie pipe and a portland cement/bentonite mixture. The grout mixture was allowed to cure for a minimum of 24 hours prior to initiation of further drilling activities.

MW-3DD and MW-18DD were advanced to approximately 95 and 98.5 feet below grade, respectively, using nominal 2-inch diameter diamond tipped NX air coring techniques. Rock core samples were collected at five foot intervals. The recovered cores were visually described and RQD values were calculated by the on-site hydrogeologist. Upon completion of the sampling, the existing 2-inch diameter coreholes were reamed using an 11-7/8 inch diameter air



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hammer bit. To minimize the potential for the vertical migration of contaminants within the borehole, nominal 8-inch diameter carbon steel casings were telescoped through the existing 12-inch casings and grouted in place using a tremie pipe and a portland cement/bentonite mixture.

After allowing the grout to cure for a minimum of 24 hours, NX rock coreholes were advanced from 95 to 138 feet at MW-3DD and 98.5 to 140 feet at MW-18DD using air coring techniques. The corehole was sampled at five foot intervals. Subsequent to coring, the coreholes were reamed using a 5-7/8 inch diameter air hammer bit prior to well installations.

PVC monitoring wells were installed within each deep bedrock borehole. Construction methodologies was similar to those previously described for the SRI shallow bedrock wells. MW-3DD was completed with a flush mount locking well cover. MW-18DD was completed to approximately two feet above grade and finished with an 8-inch diameter locking well cover.

Soils generated during well installation were staged on the top of the landfill near the waste oil disposal pit. Drilling and sampling equipment including augers, drill rods and any other artifact that came in contact with the soils were decontaminated with a portable steam cleaner at the decontamination pad above the waste oil disposal area in accordance with procedures described in the approved SOP (O'Brien & Gere Engineers, Inc., 1988).

Upon completion, monitoring wells were developed by hand bailing to remove fine-grained sediment that may have accumulated in and around the well screen during installation, and to enhance the hydraulic connection between the well and the water-bearing unit. Water purged from the wells was allowed to infiltrate into the ground around the well. After completion of the additional monitoring and stilling wells, a field instrument survey was completed to provide the horizontal location and vertical elevation of the well.

### 2.5.2. Packer testing and sampling

Packer tests were conducted during the SRI in the deep bedrock well (MW-3DD and MW-18DD) boreholes to provide vertical characterization of bedrock ground water quality and a qualitative

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estimate of ground water flow in the fractured bedrock. Packer tests were completed from the following intervals:

| | |
|---|---|
| MW-3DD: | 60.5 to 80 ft, 82 to 95 ft, 95 to 119 ft, and 121 to 138 ft |
| MW-18DD: | 62 to 78.5 ft, 78.5 to 98.5 ft, 102 to 120 ft, and 122 to 140 ft. |

Packer testing was conducted at approximate 20 ft intervals within the NX corehole. After completing the last 5 ft core run within a 20 ft interval, the packer test equipment consisting of a 1-inch diameter packer string with one inflatable rubber packer was lowered into the corehole. The packer was inflated through a high pressure hose attached to a cylinder of compressed nitrogen. The effectiveness of the packer seal in the corehole wall was monitored by confirming that the packers were inflated to the target packer inflation pressure. In addition, the drill rig hoist holding the packer string was released allowing the inflated packer to support the weight of the packer string. It was concluded that the packer interval was effectively isolated or sealed if the packer could support the weight of the packer string.

After establishing a proper seal, a length of 5/8-inch diameter dedicated Waterra® tubing and foot valve was lowered through the packer string into the isolated interval. A minimum of 10 corehole volumes of water were removed from each packer test interval. After removal, ground water samples were collected from the Waterra® discharge tube. The samples were submitted to O'Brien & Gere Laboratories, Inc. for VOC and PCB analysis using EPA Method 8010/8020 and 8080, respectively, following appropriate chain of custody procedures.

### 2.5.3. *In situ* hydraulic conductivity tests

*In situ* hydraulic conductivity tests were performed on monitoring wells to evaluate the permeability of the screened overburden and bedrock units. Hydraulic conductivity values were calculated using rising or falling head test methodologies. The method required measurement of static water level followed by well evacuation with a decontaminated stainless steel bailer and measurement of the recovery rate at regular time intervals with a water level probe graduated every 0.01 ft. Measurements were collected until the water in the well had returned to approximately 75% of the measured static levels. Tests were not performed at monitoring wells MW-2 and MW-17 because the length of the water column within these wells was not sufficient to provide recovery data. Because the water column in these wells is generally less than 3 ft throughout the year, hydraulic conductivities would not accurately reflect the screened interval, but only the portion of the formation that is saturated.

Data were analyzed using the Bouwer and Rice method (Bouwer and Rice, 1976) or Hvorslev's method (Hvorslev, 1951). A summary of the results of these analyses are presented on Table 6. Data and calculations are presented in Appendix D.

### 2.5.4. Ground water sampling

Ground water samples from shallow and deep ground water monitoring wells were collected on several occasions throughout Phases I, II, III and the SRI between November 1988 and March 1995 to assess the vertical and horizontal extent of site-related compounds. During Phase I of the RI (November 1988), monitoring wells MW-1, MW-2, MW-6 and MW-8 were analyzed for full TCL parameters while the remaining wells (MW-3, MW-4, MW-5S, MW-5D, MW-7, MW-9, and MW-10) were analyzed for TCL VOCs, PCBs, and Oil & Grease by EPA Methods 624, 608, and 9070, respectively.

During Phase II of the RI, two sampling rounds were conducted. The initial round in August 1990 consisted of confirmatory sampling for VOCs and PCBs from the 11 ground water monitoring wells on-site. The samples were analyzed using the same methods for VOCs and PCBs as used during the November 1988 round. The second round Phase II sampling event was conducted in December 1990 and included eight new overburden wells (MW-11S through MW-18S) and two new bedrock wells (MW-11D and MW-18D), as well as the 11 existing monitoring wells. This round consisted of analyzing the 18 overburden monitoring wells for VOCs and PCBs, and the three bedrock monitoring wells for full TCL parameters.

During Phase III of the RI, two sampling rounds were conducted. The initial sampling event in December 1991 was a confirmatory sampling round of the three bedrock wells (MW-5D, MW-11D, and MW-18D) for VOCs and PCBs. The second Phase III sampling round was conducted in May 1992, and included sampling the newly installed monitoring wells (MW-4D, MW-7D, and MW-19) and MW-4S, MW-7S, MW-5S, MW-5D, MW-11S, MW-11D, MW-18S, and MW-18D. Water samples from the Phase III events were analyzed for VOCs and PCBs using EPA Methods 8010/8020 and 8080, respectively. VERSAR, Inc. performed oversight during sampling for the USEPA.

A full round of ground water samples were collected in the SRI. Seventeen overburden and six shallow bedrock wells were sampled in December, 1994 and two shallow and two deep bedrock wells were sampled in late February/early March 1995. Monitoring well MW-5S was not sampled due the presence of a free-phase floating product in the well. Ground water samples were analyzed for VOCs and PCBs.

For the first three RI Phases, ground water samples were collected in accordance with procedures outlined in the approved SOP (O'Brien & Gere Engineers, Inc., 1988). The methodologies employed required the collection of a complete round of static ground water elevations prior to sampling. Wells were purged a minimum of three well volumes or until the well went dry to confirm that a representative ground water sample was collected. Ground water samples were collected using a decontaminated bottom loading stainless steel bailer and dedicated polypropylene rope. Field measurements of pH, specific conductance, and temperature were collected during sampling and are included on field sampling logs presented in Appendix E. Ground water samples along with appropriate QA/QC samples were placed in properly labelled and preserved containers and placed in a cooler with ice for delivery to O'Brien & Gere Laboratories, Inc. for the requested analyses. Chain of custody documentation was begun in the field and accompanied the samples to the laboratory.

Ground water samples collected during the first and third sampling events submitted for inorganic sampling consisted of unfiltered and filtered samples and are reported as total and dissolved, respectively. Filtering was performed in the field with a disposable in-line 0.45

micron filter and teflon tubing. The sample water was drawn through the filter with a peristaltic pump.

SRI ground water samples were collected using low flow purging techniques in accordance with procedures in the approved SRI Work Plan. Low flow purging involved placing a decontaminated stainless steel Grundfos® pump within the screened interval of a well and purging at a rate less than 0.1 liters/min. During purging, ground water quality parameters including pH, conductivity, temperature, Eh, and dissolved oxygen were monitored continuously using an in-line meter. The ground water sample was collected after stabilization of the parameters, as defined in the SRI Work Plan. Specific well purge rates and ground water monitoring parameters are included in Appendix E. Decontamination of bailers and other equipment that contacted ground water followed the approved protocols and consisted of washing with a non-phosphate detergent, followed by a potable water rinse, acetone rinse and a final deionized water rinse.

**2.5.5. Test well and piezometer installations**

Nine overburden piezometers (PZ-1 through PZ-9) and three overburden test wells (TW-1 through TW-3) were installed downgradient of the Landfill along Richardson Hill Road. Test well and piezometer locations were equally spaced along Richardson Hill Road as shown on Figure 11. Three piezometers were located at distances 5 ft, 15 ft, and 30 ft from the test wells to provide monitoring points for test well pump tests.

Piezometers and test wells were installed to the top of bedrock using the protocols outlined in the SOP (O'Brien & Gere Engineers, Inc., 1988). Piezometers and test wells were installed in December 1991 by Parratt Wolff, Inc. using a track mounted CME-850 drill rig and 10¼-inch I.D. hollow stem augers. Split spoon soil samples were collected at 5-ft intervals and logged by an on-site O'Brien & Gere Engineers, Inc. hydrogeologist. Well and piezometer as-built data, including ground elevation, total depth, and screen length are included in Table 6.

The test wells were constructed using between 10-ft to 22-ft lengths of 6-inch I.D. stainless steel, 0.020-inch wire wound slotted screen. Six-inch I.D. riser sections were attached with flush threaded joints to the screen and extended approximately 2 ft above grade. TW-3 was completed with a flush mount due to its proximity to Richardson Hill Road. A grade #0 Morie silica sand pack was placed around the well screen and extended to approximately 2 ft above the top of the screen for TW-1 and TW-2 and 1 ft for TW-3. A bentonite pellet seal was placed above the sandpack. Portland cement/bentonite grout was used to completely fill the remaining annular space to grade. An 8-inch diameter carbon steel protective locking cover was placed over the top of the diameter 6-inch riser to secure the wells.

Piezometers were constructed of 1-inch I.D. schedule 40 PVC, 0.010-inch slot screen with flush threaded joints. Riser sections were used to extend the piezometer to approximately 2 ft above grade. A grade #00 Morie sand pack was emplaced around the screened PVC section and extended to about 1 ft above the top of the screen. A bentonite pellet seal was placed above the sand pack and portland cement/bentonite was installed to ground surface. Locking 6-inch steel protective covers were placed over each piezometer, except PZ-7 through PZ-9, which were finished with flush curb boxes.

Drilling and sampling equipment were decontaminated prior to their use in accordance with procedures described in the approved SOP (O'Brien & Gere Engineers, Inc., 1988). Test well construction diagrams are included in Appendix C.

Upon completion, test wells were developed by pumping to remove fine-grained sediment that had accumulated in and around the well screen during installation and to enhance the hydraulic connection between the well and the water-bearing unit. After completion of the test wells, a field instrument survey was completed to provide the horizontal location and vertical elevation of wells.

**2.5.6. Pumping tests**

Twenty-four hour pump tests were completed on test wells TW-1, TW-2, and TW-3 between August 3 through August 6, 1992. The tests were performed to evaluate overburden ground water discharge rates into the South Pond, and to evaluate design criteria for a ground water recovery and treatment system.

Short term yield tests were completed at the test wells prior to conducting the pumping tests to select an optimum flow rate that



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could be continuously sustained for twenty-four hours and would adequately stress the aquifer.

Pumping test flow rates area were as follows:

| Well | Concentration |
|---|---|
| TW-1 | 0.87 gpm |
| TW-2 | 0.25 gpm |
| TW-3 | 2.6 gpm |

Prior to initiating the test a complete round of static ground water levels was collected from the test wells, nearby monitoring wells, and piezometers. Water level drawdown was measured at regular time interval during the pump tests to assess overburden aquifer response.

Drawdown data collected from the tests were analyzed using the Cooper-Jacob straight line method, Neuman type curve matching method, and distance-drawdown method. Recovery data were analyzed by plotting the ratio of time since pumping started (t) to the time since pumping stopped (t') versus the residual drawdown (s'). Hydraulic coefficients and characteristics obtained from the pump tests are summarized on Table 7. Data, calculations, and plots are included in Appendix F.

Discharge water from the pump tests was contained and treated using a portable treatment system comprised of two temporary holding tanks, bag filter, carbon adsorption unit, and an aerator. The USEPA approved use of this treatment system in correspondence dated June 17, 1992, provided discharge water concentrations were within limitations established by the NYSDEC. The treated water was discharged into a bermed area on the landfill to enhance infiltration.

## 2.6. Residential water supply survey

A survey was completed to evaluate if residences with water supply systems located downgradient and downstream of the RHRMLS have been affected by the landfill. The survey was performed by reviewing

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the United States Geological Survey well information database to obtain information locating well size and depth, lithology, discharge rates, and primary use of the water. Residential well and spring sampling information was also obtained from the New York State Department of Health (NYSDOH).

The data review indicated that in 1985, the NYSDOH initiated water supply sampling at several residences near the site (Figure 12). Sampling was performed at three shallow dug wells designated Spring 1, Spring 2, and Spring 3, located between the North Area and the Sidney Landfill, and two springs which are shared by residents (6/7 and 8), that are located downgradient of RHRMLS. In addition, a deep bedrock residential supply well located to the south of the Landfill was also sampled.

As a component of the "Response Action" required under the AOC discussed in Section 1.2.5., baseline sampling events were performed on Springs 1, 2, and 3. A total of four sampling events, three wet period and one dry period, were performed between April to June, and October, 1994, respectively. The baseline samples were analyzed for VOCs, SVOCs, PCB/pesticides, and total coliform, using EPA Methods 8010/8020, 8270, 8080, and 9131, respectively.

## 2.7. South pond wetlands delineation

A wetlands delineation limited to the areas immediately adjacent to the South Pond was completed in October 1990. The delineation was completed in accordance with the "Federal Manual for Identifying and Delineating Jurisdictional Wetlands" dated December 1989. Field data forms for the routine on-site determination method are included in Appendix G. The data forms provide information on the vegetation, soils, hydrology, and the jurisdictional determination and rationale.

## 2.8. Test pits

Six shallow test pits were completed at the RHRMLS to visually examine and sample the landfill materials. Five test pits (TP-1 through TP-5) were completed at the Landfill in the vicinity of SB-10 and one test pit (TP-6) was completed in the North Area (Figure 13).

Test pits were requested by the NYSDEC specifically to evaluate subsurface conditions in the vicinity of boring SB-10 at grid node N400, W100 (Figu.e 13). The concentration of PCBs at SB-10 (4 to 6 ft) was notably higher than at other locations on the landfill, and a geophysical survey magnetic high was also noted in this area of the landfill. Prior to initiating the test pit program, a work plan was developed and submitted to the USEPA and NYSDEC. The work plan was approved by the agencies in March 1992.

The test pits were completed using a Rupp Model 1400B backhoe in accordance with the protocols outlined in the work plan dated March 1992 (O'Brien & Gere Engineers, Inc., 1992a and 1992b). Excavations were completed to a depth of about 8 feet. Upon removal, soil and landfill materials were temporarily placed on plastic sheeting to minimize the potential for surface soil contamination. Following completion of the test pits, the excavated soil and landfill materials were placed back in the respective test pits and covered with the plastic used for temporary stockpiling. The backhoe was decontaminated with a steam cleaner prior to starting the excavations and between excavation locations. Detailed logs were completed for each test pit and are included as Appendix H.

One soil sample was collected from each test pit and submitted for analysis of VOCs and PCBs using USEPA methods 8010/8020 and 8080, respectively. The actual sampling locations in each test pit was selected based on visual observations and field screening with a PID. Consistent with the QAPP, soil samples collected for VOC analyses were collected separately, whereas the remaining samples were composited in the field with a stainless steel scoop and bowl.

A continuous test pit, approximately 200 feet in length, was completed immediately downgradient of the waste oil pit in May 1994. Performed as a component of the UAO Response Action, the test pit excavation was used to evaluate potential migration pathways

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and/or seeps, containing free-phase oil, originating from the waste oil pit. The location of the test pit is shown on Figure 13. A description of test pit methodologies is included in Section 2.1.4 of Exhibit IV.

A test excavation was performed in the vicinity of MW-5S in August, 1994 to augment the GPR survey performed on Richardson Hill Road. MW-5S was destroyed during this excavation. The excavation was a component of the UAO response action and is discussed in Section 4.3.1 of Exhibit IV.

## 2.9. Air sampling

Air sampling was conducted at the RHRMLS on September 27, 1988, during Phase I of the RI, using protocols outlined in the SOP (O'Brien & Gere Engineers, Inc., 1988). The objective of the air sampling was to evaluate the nature and magnitude of airborne transport of potential VOC and fugitive dust emissions in the vicinity of the former waste oil pit during quiescent conditions. To achieve this objective, air sampling was conducted at four locations (A through D) surrounding the former waste oil pit, as shown on Figure 6. Sample locations were selected to represent air quality conditions upwind and downwind of the former waste oil pit. Air sampling results are discussed in Section 4.4.

Air samples were collected for analysis of VOCs and PCBs using National Institute of Safety and Health (NIOSH) Methods 1003 and 5503, respectively. Prior to field activities, SKC programmable pumps were calibrated by O'Brien & Gere Engineers, Inc. personnel under the supervision of a certified industrial hygienist. Flow rates were set at about 10 cc/min for VOC samplers, and 50 cc/min for PCB sampling pumps according to NIOSH method requirements. Each pump was labeled with its calibrated flow rate (within 0.01 cc/min).

Sampling trains consisting of appropriate filters and sorbent tubes were connected to pumps via Tygon® tubing, attached to wooden stakes, and installed 2 to 3 ft above the ground at each sampling location. Pumps were turned on at about 11:00 a.m. on

September 27, 1988 and were programmed to operate for 12 hours. Sample pumps were retrieved on September 28, 1988 at about 10:00 a.m. Sorbent tubes and filters were capped and placed in septum sealed vials. A field blank, consisting of a prepared sample train, was included for each analytical methodology. Sampling containers were labeled and preserved according to the SOP and delivered to O'Brien & Gere Laboratories, Inc.

Routine health and safety ambient air quality monitoring was performed with a PID and Draager indicator tubes during field investigations. With the exception of intrusive activities in the waste oil pit, ambient air quality readings in the breathing zone were not above background readings.

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# 3. Physical characteristics of the site

## 3.1. Physiography

Regionally the site is located in the Susquehanna Hills subdivision of the Appalachian Upland landform region of New York State (Cressey, 1981). The region has been glaciated and is underlain by Paleozoic age sedimentary rocks which dip toward the south and west. The area is drained largely by the upper Susquehanna River and its tributaries. There are numerous steep valleys in the region Valley divides are at heights ranging between 1,700 ft to 2,100 ft.

The Landfill and the North Area are located on the west side of Richardson Hill Road in northwestern Delaware County, within the Towns of Masonville and Town of Sidney (Figure 1). The study area is located in a narrow valley where elevations rise abruptly from 1750 feet above mean sea level (amsl) in the valley to more than 2000 feet amsl on the east and west valley walls. Several small ponds are located in the valley adjacent to the RHRMLS (Figure 2). The site is drained mainly by the upper Delaware River and its tributaries.

## 3.2. Meteorology

This area has a humid climate where warm summers are followed by long, cold winters. The mean temperature ranges from about 22.5° F in January to 69.3° F in July (NOAA, 1985). The area receives an average of 40.6 inches of annual precipitation. During the winter months, snowfall amounts generally range from 70 to 100 inches (NOAA, 1985). In general, the ground surface and the smaller surface water bodies remain frozen from December to March each year.

*Surface Water Hydrology*

Richardson Hill forms a drainage divide between the Susquehanna and Delaware basins that runs from the top of the hill to the wetland area on the east side of Richardson Hill Road. The North Area, which is situated within this divide receives sheet flow surface water drainage from the adjacent Sidney Center Landfill.

The Delaware River basin in New York State encompasses about 2,362 square miles, with the principal tributaries being the Neversink and Mongaup Rivers (USGS, 1986). The Pepacton, Cannonsville, and Neversink reservoirs are located within the basin. The Susquehanna River basin in New York State encompasses about 5,450 square miles, with the principal tributaries being the Unadilla, Chenango, and Chemung Rivers (USGS, 1986). The Whitney Point, East Sidney, Almond, and Arkport reservoirs are located within the basin.

A east-west trending drainage divide is located between the North Pond and South Pond. This divide causes surface water drainage from the Landfill to flow in a southerly direction via an intermittent unnamed tributary stream into Herrick Hollow Creek, then into Trout Creek. Trout Creek ultimately discharges into the west branch of the Delaware River and the Cannonsville Reservoir, which are about 8 miles south-southwest of the RHRMLS.

Surface water drainage received by and from the North Area flows a northerly direction into Carr's Creek, which is a tributary to the Susquehanna River. Some surface water from the North Area also drains to the south in culverts along Richardson Hill Road, to the South Pond.

Six surface water staff gauges were installed in October 1990 to evaluate surface water elevations in the North and South Ponds and the hydraulic relationship between the ponds and shallow ground water. The gauges are constructed of 5-ft lengths of slotted 4-inch PVC fastened to post setters. The post setters were driven into the pond substrate with a sledge hammer about 5 ft from the pond shore. The horizontal locations and vertical elevations of the gauges were surveyed using the NYS Plane coordinates and USGS datum elevations. Surface water elevations were measured from the staff gauges between November 1990 and November 1992. The data is summarized in Table 6.

Stream flow measurements were collected concurrently with elevation monitoring to evaluate the average discharge from the South Pond. Stream flow measurements were collected from December 1991 to November 1992 at two locations (VR-1 and VR-2) downstream of the South Pond outlet. Station VR-1 is located about 500 ft downstream of the South Pond outlet, and VR-2 is located about 500 ft downstream of VR-1 (Figure 11). The stream flow measurements are summarized on Table 8 and include data on the stream channel geometry ($ft^2$), velocity (ft/sec), and discharge ($ft^3$/sec and gpm).

High discharge from the South Pond typically occurs from March through May and corresponds to the higher precipitation months. Over the monitoring period, discharge from the South Pond, measured at VR-1, ranged from about 45 gpm (64,800 gpd) in September 1992 to about 382 gpm (550,080 gpd) in May 1992. Data from VR-2 represents the combined flow from the South Pond and a small tributary which originates in the eastern half of the valley.

## 3.3. Hydrogeologic conditions

### 3.3.1. Regional geology

The RHRMLS is located within the Appalachian Uplands Physiographic Province (Van Diver, 1985). The plateau region is characterized by east-west trending ridges of resistant strata that have been dissected by glacial and stream erosion (Coates and King, 1973). Stream valleys within the region have created areas with relief in excess of 1000 ft.

The surficial geology (overburden) of the region is dominated by Pleistocene-age glacial and recent alluvial sediments (Fleisher, 1986). The glacial deposits range from three to ten feet thick, but has been documented in excess of 100 ft, primarily in valleys. The glacial deposits are composed of clay and silt with various percentages of gravel, cobbles, and boulders and typically have low permeability.

Recent alluvial deposits, made up primarily of eroded glacially derived materials, are also present within most of the valleys of the region. Alluvial materials normally consist of well sorted sands and gravels that sometimes exhibit grading. Material deposited within dammed valleys or lakes contains fine grained silts and clays.

Regional bedrock consists of Devonian aged sandstones, siltstones, shales, and conglomerates which comprise the Upper Devonian Walton Formation (Rickard and Fisher, 1970). The materials that make up these rocks were eroded from the eastward lying Taconic mountains and transported to the west as part of the Catskill deltaic-alluvial sedimentary wedge (Van Diver, 1985). The Catskill Wedge is aerially extensive, covering much of New York State with thicknesses reported to range from 300 ft to 1,000 ft. The sedimentary rocks that resulted from lithification of the wedge are made of interlayered sandstones, siltstones, and shales that dip gently southward.

The lower Walton Formation of the Sonyea Group has been identified in the immediate vicinity of RHRMLS. The lower Walton Formation consists of red and green shales and gray sandstones (Sutton, et al, 1970). Although the Sonyea Group is cut by numerous southwest to northeast trending fault blocks to the west of RHRMLS, no brittle structures (faults, shear zones) have been mapped within 2 miles of RHRMLS (Isachsen and McKendree, 1977). Joints in the bedrock including north-south, east-west, and northeast-southwest trending sets have been mapped in the region of the site, however, the north to south is the dominant fracture orientation (Parker, 1942) .

### 3.3.2. Site geology

Site geology has been characterized by soil borings, test pits, and bedrock cores completed during the RI field investigations. Geologic cross-sections of the site are shown on Figure 14. The locations of the cross-section lines A-A' and B-B' are shown on Figure 8.

The site is underlain by up to 44 feet of dense reddish brown to grayish brown glacial till. The till is a heterogeneous mixture of sand, silt, clay, and rock fragments. Sediment samples retrieved for analysis the from South Pond contained fine-grained silts and clays typical of lacustrine sediments.

Within the Landfill, the till is overlain by fill materials which consist of reworked soil mixed with municipal refuse. Visual observations from test pits and soil borings indicate fill cover materials consisted of brown and tan very fine to coarse sand and angular cobbles which were removed from the hill above the Landfill. Test pits and soil boring observations indicate the landfill materials consists of about 65% to 80% brown and gray very fine to very coarse sand and fine to coarse gravel with variable amounts of cobbles, silt, and clay. The soils encountered from grade to a depth of about 8 ft were dry to very moist. Test pit and soil boring observations indicate that disseminated within the soil materials was about 20% to 35% municipal refuse, which was primarily general household trash with a small percentage of office trash.

The aerial extent of fill materials, based on review of aerial photographs, geophysical surveys, soil vapor surveys, and soil borings appears as an approximate area 300 ft wide by 800 ft long that parallels Richardson Hill Road and ranges in elevation from 1760 to 1820 ft amsl (Figure 2).

Fill materials in the waste oil pit extended to a depth of about 6 feet. Fill materials identified in SB-10, SB-11, and SB-17 southwest of the waste oil pit were approximately 16 feet thick.

Bedrock at the site was encountered at depths ranging from 18 feet at MW-7D to 39 feet below the ground surface at MW-9D. The depth to bedrock is less in the center of the valley along Richardson Hill Road. Bedrock elevations at the site decrease from west to east toward the center of the valley. A bedrock contour map is illustrated in Figure 15.

The bedrock immediately beneath the till is comprised of thinly bedded sequences of siltstone and shale interfinger with thicker sequences of sandstone. Bedrock cores from MW-3DD, MW-5D, MW-11D, MW-18D, and MW-18DD indicate bedrock fractures were mainly observed within the siltstone and shale, and at the bedding plane contacts between the siltstone/shale and sandstone. The fracture orientations were parallel to bedding plane surfaces, however, fractures oriented at approximate 45 degrees to bedding planes were observed within the sandstone units. Within the shale bedrock, the fractures were generally lined with clay, whereas within the sandstone; the fractures were lined with silt. The presence of clay and silt in fracture partings suggests the occurrence of recent ground water flow. Fractures encountered in the vicinity of carbonized organic matter exhibited pyritization, indicating a relatively low flow reducing environment and prolonged ground water residence time within the aquifer.

Figures 16 and 17 illustrate bedrock cross-sections in north to south (C-C') and west to east directions (D-D'), respectively. The locations of the cross section lines C-C' and D-D' are shown on Figure 8. From north to south (Figure 16) thicker sequences of siltstone and shale were encountered in the vicinity of MW-3DD and may interfinger with thicker sequences of sandstone in the vicinity of MW-18DD. Much of the sandstone contained carbonized organic matter (plant debris) as well as clay clasts. As indicated from the rock quality designation (RQD) summary (Table 9), the sandstone underlying the site was is more massive and less fractured than the siltstone/shales. RQD averaged approximately 82 percent at MW-3DD and 87 percent at MW-18DD, indicating that the bedrock is competent to the terminal depths of the bedrock borings.

### 3.3.3. Site hydrogeology

Ground water exist at the site in the overburden and the bedrock. The bedrock has been further classified into two zones, shallow and deep.

Ground water elevation data has been collected periodically since the Phase I investigations in 1988. In 1992, elevation data was collected monthly from all site wells concurrently with data collected at the adjacent Sidney Center Landfill by Malcolm Pirnie, Inc. This data was updated in February 1995 during the SRI. Ground water elevation data is summarized in Table 6.

Overburden ground water elevation data collected on February 27, 1995 is illustrated on Figure 18 and indicates that the direction of shallow ground water flow in the overburden follows the slope of the surface topography. Ground water in the vicinity of the Landfill flows east to southeast under an average hydraulic gradient of about 0.15 ft/ft to the South Pond. Overburden flow is topographically controlled and highly heterogenous. Test pits revealed that ground water flow occurs preferentially in till fractures and zones of relatively coarser overburden materials.

Ground water in the North Area appears to flow to the south-southwest toward the South Pond under a hydraulic gradient of about 0.03 ft/ft. Although the North Pond surface water elevation was not monitored in February 1995, previous data (1992) indicate a component of ground water flow in the North Area is to the north towards the north pond. Data from the Malcolm Pirnie well MW-23 (Figure 18) indicate that overburden aquifer flow potential from the Sidney Center Landfill is west towards the North Area and North Pond.

North Area saturated overburden conditions were not observed during the drilling of overburden monitoring well MW-9 and shallow bedrock well MW-9D. Ground water was encountered at the weathered bedrock interface. The water level in MW-9 rose rapidly to approximately 25 ft above that interface (O'Brien & Gere Engineers, Inc., 1989). This condition indicates that the weathered bedrock zone in this vicinity is hydraulically connected to an area of higher hydraulic head, other than the overburden aquifer. A local topographic high area such as Richardson Hill, the location of the Sidney Center Landfill, is likely hydraulically connected to the weathered bedrock zone in the North Area.

Shallow bedrock ground water elevation data is illustrated on Figure 19 and emulate the site bedrock topography (Figure 15) with flow potential from Landfill to the east towards the South Pond under a average flow gradient of 0.12 ft/ft. This flow gradient is primarily influenced by the change in topography across the Landfill. Flow potential from the North Area is to the south to southeast under an average flow gradient of 0.02 ft/ft. A west to southwest flow potential component from the Sidney Center Landfill exists toward both the South and North Ponds.

Ground water elevations exhibited seasonal fluctuations that correlated to precipitation and evapotranspiration demand. The lowest ground water elevations occurred in the late summer to early fall months during low precipitation and high evapotranspiration periods. The highest elevation was noted in the fall (November), when evapotranspirative demands are low and precipitation can be high. The largest fluctuations in ground water elevations were observed in MW-11S and in the vicinity of MW-9 and MW-15 in the north area.

A review of Figure 17 indicates an apparent downward hydraulic flow potential from the overburden to the bedrock at well nests MW-11S/11D, MW-18S/18D/18DD, and Malcolm Pirnie MW-12 well nest. In addition, a large hydraulic head difference (≈30 ft) exists between the shallow and deep bedrock wells in nest MW-3 (Figure 16), MW-18 and Malcolm Pirnie MW-12 wells. This large difference in vertical hydraulic head suggests that the deeper bedrock zones are hydraulically isolated from the upper bedrock zones.

Overburden hydraulic conductivity ranged from 0.1 to 114.6 gpd/ft$^2$ (Table 6). The large range is attributed to the heterogenous nature of the overburden. Based on an assumed porosity of 0.30 for sandy till (Freeze and Cherry, 1979), and a topographically controlled hydraulic gradient of 0.15 ft/ft, estimated ground water flow velocity within the overburden aquifer range from 0.007 ft/day to 7.66 ft/day.

Hydraulic conductivity values in the bedrock wells range from 0.002 gpd/ft$^2$ to 49 gpd/ft$^2$. The degree of hydraulic conductivity in bedrock is dependent on wells intersecting zones of high fracture frequency. Assuming a porosity of 0.20 for bedrock, and a topographically controlled hydraulic gradient of 0.12 ft/ft, estimated ground water flow velocity in the site bedrock range from 0.0002 ft/day to 3.9 ft/day.

#### 3.3.4. Pumping test results

Pumping test data from test wells at TW-1, TW-2 and TW-3 were used to estimate overburden aquifer transmissivity and storativity values. Pumping test data was also compared with the hydraulic conductivity data of adjacent monitoring wells to estimate the typical hydraulic conductivity of the overburden aquifer in the vicinity of each test well. The values are summarized in Table 7. The locations of the wells are shown on Figure 11.

*TW-1*
Transmissivity values ranged from 60 gpd/ft to 1,094 gpd/ft, which correspond to hydraulic conductivities which ranges from 4 to 73 gpd/ft$^2$. The typical overburden estimated hydraulic conductivity in this vicinity is 6.1 gpd/ft$^2$. Storativities ranged from 0.0007 to 0.009.

*TW-2*
Transmissivity values ranged from 28 gpd/ft to 220 gpd/ft, which correspond to hydraulic conductivities which range from 1 gpd/ft$^2$ to 7 gpd/ft$^2$. The typical overburden estimated hydraulic conductivity in this vicinity is 2.1 gpd/ft$^2$. Storativities ranged from 0.0003 to 0.004.

*TW-3*
Transmissivity values ranged from 319 gpd/ft to 4,576 gpd/ft, which correspond to hydraulic conductivities which range from 14 gpd/ft$^2$ to 200 gpd/ft$^2$. The typical overburden estimated hydraulic conductivity in this vicinity is 31 gpd/ft$^2$. Storativities ranged from 0.0003 to 0.006.

The increased transmissivity values observed for TW-3 as compared to TW-1 and TW-2 are likely result of greater aquifer saturated thickness due to the proximity to the South Pond. In addition, the higher hydraulic conductivity is indicative of screening across productive till fractures.

Overburden ground water discharge rates into the South Pond were calculated based on the August 1992 transmissivity values obtained from the pumping tests. For purposes of these calculations, the shoreline along South Pond was divided into three sections corresponding to the areas of TW-1, TW-2 and TW-3. For each of these areas, discharge rates were calculated using high, low and average hydraulic conductivity values obtained from the pump tests each respective test well, an average hydraulic gradient of 0.12 ft/ft and the appropriate cross sectional area. The discharge rates are summarized below.

| Well | High (gpd) | Low (gpd) | Average (gpd) |
|---|---|---|---|
| TW-1 | 45,990 | 2,520 | 9,000 |
| TW-2 | 10,000 | 1,450 | 3,675 |
| TW-3 | 164,000 | 11,500 | 110,000 |

Based on this data, discharge rates from the Landfill to South Pond range from 15,470 gpd to 219,990 gpd, with an average value of 123,275 gpd. This data compares well with the surface water discharge measurements from the South Pond which range from 64,800 gpd to 550,080 gpd.

# 4. Nature and extent of contamination

## 4.1. Source characteristics

### 4.1.1. Residential water supply sampling results

According to NYSDOH records, there is no public water supply currently available to the residences along Richardson Hill Road. The closest potable supply is the Village of Sidney. Ten residences were identified which exist either in close proximity or within 1 mile downstream of the RHRMLS (Figure 12). The residences are either houses or mobile homes which are generally only occupied seasonally. The identified residences all obtain their water supply from shallow dug wells called springs. These springs receive water from the combination of overburden seepage and through overland surface water flow. A summary of the analytical results obtained from the NYSDOH sampling is included on Table 10. In addition, the results of AOC Response Action (Section 1.2.5) spring baseline sampling are presented in Exhibit III.

Spring #1 is located on the west side of Richardson Hill Road and directly south of the North Area (Figure 12). Water from spring #1 contained trichloroethylene (TCE) with low concentrations of 1,1,1-trichloroethane (1,11, TCA) and naphthalene which were detected during only one sampling event. Total VOC concentrations ranged from 48 ppb to 64 ppb. AOC Response Action baseline sampling results corroborated the RI sampling results.

Springs #2 and #3 are located on the east side of Richardson Hill Road directly west and downgradient of the Sidney Center Landfill. Samples from spring #2 contained TCE, 1,2-dichloroethene (1,2 DCE), and 1,1-dichloroethane (1,1 DCA). Total VOC concentrations ranged from 11 ppb to 99 ppb. AOC Response Action baseline sampling demonstrated Spring #2 total VOC concentrations below 5 ppb and all specific constituent concentrations below Federal and State drinking water Maximum Concentration Limits (MCLs).

Spring #3 exhibited higher concentrations of VOCs. Constituents detected include TCE, 1,1 DCA, 1,2 DCE, 1,2-dichloroethane (1,2 DCA), 1,1,1-TCA, tetrachloroethylene (PCE), and 1,1,2,2-tetrachloroethylene. Total VOC concentrations ranged from 651 ppb to 1134 ppb (Table 10). AOC Response Action baseline sampling results corroborated the RI sampling results.

A dissimilarity in spring water chemistry is noted between Spring #1 and Springs #2 and #3. TCE without the associated degradation compounds was detected in Spring #1 , whereas springs #2 and #3 exhibited TCE and associated degradation compounds.

Two springs (6/7 and 8) were sampled downstream and south of the RHRMLS (Figure 12). With the exception of TCE detected at 1 ppb in spring 6/7 during one sampling event, no contaminants were detected in the springs downstream of the RHRMLS.

The residences #4 and #5 are mobile homes which have been abandoned and are no longer in use. The residences indicated #9 and #10 were not sampled by the NYSDOH because the upstream springs 6/7 and 8 did not contain landfill contaminants.

Based on the results of baseline spring sampling whole-house water treatment units were installed in the Spizziri (Spring #1) and Wyatt (Spring #3) residences (Figure 12). The units treat spring water and supply each residence with domestic water that meets Federal and State drinking water MCLs. Spring #2 meets MCLs without treatment.

### 4.1.2. Geophysical survey results

Surficial geophysical surveys were completed at the RHRMLS and North Area using methodology discussed in Section 2.2.1. The purpose of the surveys was to identify fill areas, the limits of the waste oil disposal area, and locate subsurface metal materials.

Fill areas were identified at the Landfill and North Area using both EM-31 terrain conductivity and a proton magnetometer. Figure 3 presents the results of the EM-31 quadrature phase mode of operation and indicates the most prominent fill area extends in a south-southwesterly direction away from the waste oil disposal area. Apparent bulk soil conductivity values are significantly less in fill materials than in the native soil. Soil conductivity values ranged from about 4 to 8 umhos/m for native soils, whereas the conductivity of fill materials is generally 0 umhos/m. The low conductivity



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values identified as fill materials indicate the presence of non-conductive oils or non-metallic materials buried in the fill.

The areal extent of fill material at both the Landfill and the North Area was further delineated using a proton magnetometer (Figure 3). The magnetometer data were reduced by subtracting a regional magnetic field gradient value of 55,000 gammas. Background magnetic values at the site in non-fill areas ranged from 400 gammas to 800 gammas above the regional magnetic field.

A magnetic high area was identified at Landfill southwest and west of the former waste oil pit disposal area and is oriented in a north-south direction. Magnetic readings from this area ranged from 1500 gammas to 5097 gammas above the regional magnetic field (Figure 3). A magnetic low area was identified at the Landfill which extends from about 100 feet north to 440 feet south of the waste oil disposal area (Figure 3). This magnetic low is oriented north-south and parallels the magnetic high. Magnetic readings from this area ranged from -970 gammas to 400 gammas below and above the regional magnetic field. These dipolar (high to low) magnetic areas generally indicate the presence of buried ferrous materials.

Areas which likely contain buried metallic materials were also identified using the EM-31 terrain conductivity meter in the in-phase mode of operation. These areas are illustrated on Figure 4.

The results of the terrain conductivity surveys and magnetometer survey correlate well and indicate that the fill at the Landfill is present in an area about 800 ft long by 300 ft wide and extends in a southwesterly direction away from the former waste oil pit. Two isolated fill sections in the North Area were identified which are both about 70 ft by 70 ft.

GPR survey results of May 1992 are summarized in a report prepared by Detection Sciences, Inc. (Appendix B). The report concluded that, although metallic waste materials are likely distributed throughout the fill, no tightly clustered hyperbola signatures indicative of a cache of buried drums were detected in the vicinity of SB-10 during the surveys. The report also indicates that concentrations of scrap metal which would be suggestive of crushed drums were not identified in the surveys.

GPR survey results of August 1994 are included in a Detection Sciences Report in Appendix G of Exhibit IV. GPR signatures suggest non-ionic responses. The non-ionic responses are the function of the difference of dielectric constants of two materials and may be indicative of solvents or petroleum products. The areas of non-ionic responses were noted in the vicinity and north of MW-5S. Observation points installed in these areas did not reveal the presence of recoverable quantities of LNAPLs. As such, the GPR survey did not provide conclusive results concerning LNAPLs.

#### 4.1.3. Soil vapor survey results

Results of the soil vapor performance testing indicated good correlation between total VOCs in ground water and total VOCs in soil vapor (R-square = 0.87). This correlation indicated that soil vapor surveying could effectively be used to evaluate the areal distribution of VOCs in subsurface soil and ground water. Subsequently, a comprehensive soil vapor survey was completed using methodology discussed in Section 2.2.2. Sampling locations are shown on Figure 5 and results are shown in both Table 1 and on Figure 5.

Figure 5 indicates VOC were pervasive in fill areas. Total VOC concentrations at the Landfill ranged from less than 0.005 ppm to 96.3 ppm. In general, areas where VOCs were detected above 1 ppm correlated with the fill areas identified by the geophysical surveys. The highest concentrations of VOCs were detected west and southwest of the waste oil pit. The most prevalent chromatographic pattern observed site-wide was chlorinated VOCs followed by chlorinated VOCs associated with volatile petroleum hydrocarbons. VOC concentrations in the North Area ranged from non-detect to 7.2 ppm which was located along the edge of the North Pond.

#### 4.1.4. Surface soil analyses

*Volatile Organic Compounds (VOCs)*

VOCs were not detected in surface soil samples SSC-11 and SSC-31.

*Semivolatiles*

Semivolatiles were not detected in surface soil composites SSC-11 and SSC-31.

*Pesticides/Polychlorinated Biphenyls (PCBs)*

Pesticides were not detected in surface soil composites SSC-11 and SSC-31. Surface soil samples SS-1 through SS-10 and ASS-1 through ASS-4 were collected downhill from the former waste oil pit and exhibited PCB concentrations that ranged from below the method detection limit at SS-7 and ASS-1 and ASS-2 to 950 ppm at ASS-3 (Figure 20). PCB concentrations are highest in samples collected in close proximity to the pit or in surface water runoff pathways from the pit. PCB concentrations decrease away from the former oil disposal pit indicating that the pit is the source of the PCBs in downslope surface soil.

The in-field PCB GC screening results of thirty-eight surface soil composites are listed in Table 3 and shown on Figure 7. Screening results from samples SSC-1 through SCC-32 collected at the Landfill indicated PCBs are prevalent site-wide and ranged from less than 0.5 ppm to 626.7 ppm. The areas where the highest levels of detectable PCBs were identified correlate with the fill areas delineated by the geophysical and soil vapor surveys. PCB screening results from samples SSC-33 through SSC-38 collected in the North Area ranged from non-detect (0.5 ppm) to 42.2 ppm.

Based on PCB screening results, six surface soil composites were collected and submitted for laboratory analysis. Laboratory data indicated PCBs ranged from 3.8 ppm to 480 ppm as shown in Table 2 and on Figure 7. With the exception of one sample, laboratory PCB analytical results were approximately an order of magnitude lower than the field GC results indicating that the field GC provides conservative characterization of the site.

*Inorganics*

Inorganic results from SSC-11 and SSC-31 composite samples are summarized below.

***Surface soil composite (ssc) inorganic analytical data (10/9 - 24/90)***

| | NYS Range | SSC-11 | SSC-31 |
|---|---|---|---|
| Aluminum | 1,000-25,000 | 12000 | 12000 |
| Arsenic | 3-12 | 11 | 9 |
| Barium | 15-600 | 63 | 110 |
| Beryllium | 0-1.75 | 0.8 | <0.7 |
| Cadmium | 0.01-0.88 | 0.7 | <0.7 |
| Calcium | 130-35,000 | <590 | 2100 |
| Chromium | 1.5-40 | 15 | 15 |
| Cobalt | 2.5-60 | 12 | 10 |
| Copper | 37.5-112.2 | 70 | 39 |
| Iron | 17,500-25,000 | 25000 | 22000 |
| Lead | 1-36 | 24 | 46 |
| Magnesium | 1,700-6,000 | 3800 | 2600 |
| Manganese | 50-5,000 | 800 | 1200 |

*Surface soil composite (ssc) inorganic analytical data (10/9 - 24/90)*

| | NYS Range | SSC-11 | SSC-31 |
|---|---|---|---|
| Nickel | 8.5-25 | 25 | 18 |
| Potassium | 8,500-43,000 | 630 | 750 |
| Vanadium | 25-60 | 16 | 20 |
| Zinc | 37-60 | 70 | 91 |
| Percent Total Solids | | 85 | 68 |

Notes: NYS concentration range in uncontaminated soils from background concentrations of 20 elements in soils with special regard for New York State by E. Carol McGovern
All values reported in mg/kg (ppm)

With the exception of zinc at SSC-11 and zinc and lead at SSC-31, concentrations of inorganics are within the natural range of New York State soils (McGovern, E.C.). The zinc concentrations in SSC-11 and SSC-31 and lead in SSC-31 are slightly higher than the background range.

*Surface Soil Summary*
VOCs and pesticides were not detected in RHRMLS surface soils. Metal concentrations were within or slightly exceeded NYS background metal concentration ranges.

Surface soil PCB concentrations were typically detected in the areas underlain by fill materials at the RHRMLS. The lateral extent of surface soil PCB contamination at the Landfill is generally defined by non-detectable concentrations north, west, and south along the landfill limits. PCBs concentrations are highest in the areas in close proximity to the former waste oil pit, and decrease in concentration away from the pit.

PCB concentrations detected through field GC screening in surface soil from the North Area ranged in concentration from less than 5 ppm to 42.2 ppm, and were detected in the two isolated fill areas identified by the geophysical and soil vapor surveys.

### 4.1.5. Subsurface soil analyses

*Oil & Grease*

An evaluation of oil & grease concentrations was performed on subsurface soils within and in close proximity to the waste oil pit. Samples were collected during Phase I of the RI, when the investigations were primarily focused in and around the former waste oil pit. Eighty-six samples from depth intervals between 2 ft and 16 ft, were analyzed by O'Brien & Gere Laboratories, Inc. Soil boring and monitoring well locations from which these samples were collected are shown on Figure 20. An additional thirty-five samples from the waste oil pit were analyzed for oil & grease during UAO (Exhibit IV). The summary of RI and UAO oil & grease concentrations is presented in Table 11 of this report and Table 1 of Exhibit IV, respectively.

Concentrations ranged from below the method detection limit to 94,000 ppm. Oil & grease concentrations were highest in the former waste oil pit area, but did not appear to correlate with PCB concentrations or show trends corresponding to depth intervals. Oil was noted in the intersticial pore space of the soil, and in uncompacted and unbound fill materials. However, as indicated in the UAO Report (Exhibit IV), this discrete oil was present in non-recoverable quantities.

*Volatile Organic Compounds (VOCs)*
Sixty-nine Phase I subsurface soil samples were screened and weighed in the field using a PID (or FID) and triple beam balance to obtain standardized VOC concentrations (Table 12). VOC screening concentrations ranged from non-detectable to greater than 1,000 ppm. High VOC screening concentrations were noted in the borings in and around the former waste oil pit.

Five samples were submitted for TCL VOC analysis based on the VOC screening results. The samples and depth intervals included:

| Sample | Depth |
|---|---|
| B6-2 | 3-5 ft |
| B7-2 | 2-5 ft |
| B8-A | 2-4 ft |
| B34-A | 3-5 ft |
| B36-A | 3-5 ft |

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The indicator VOCs detected in the vicinity of the former waste oil pit and the concentrations were as follows:

| VOC | Concentration Range (ppm) |
|---|---|
| 1,2-DCE | 1.02 to 17 |
| TCE | 2.3 to 220 |
| xylene | 2.7 to 5.2 |
| ethylbenzene | 1.6 to 3.9 |
| toluene | 35 to 110 |

Twenty soil borings (SB-1 through SB-20) were completed during Phase II of the RI, at the locations shown on Figure 8. The Phase II soil boring program focused on Landfill areas rather than the former waste oil pit and the North Area. Soil boring samples were screened in the field with a PID and FID for VOCs. Based on screening five samples listed below were submitted to O'Brien & Gere Laboratories, Inc. for TCL VOC analysis.

| Well | Depth |
|---|---|
| SB-5 | 4-6 ft |
| SB-10 | 4-6 ft |
| SB-12 | 6-8 ft |
| SB-17 | 14-16 ft |
| SB-20 | 14-16 ft |

Four samples were collected from the Landfill and one sample from the North Area (SB-5). Screening and laboratory analytical results are included in Table 4 and 13, respectively.

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VOCs and concentrations detected in the Landfill subsurface soils from areas other than the former waste oil pit are as follows:

| VOC | Concentration Range (ppm) |
|---|---|
| acetone | 0.4 to 4.0 |
| 1,2-DCE | 0.033 |
| toluene | 2.2 |
| chlorobenzene | 0.24 |
| ethylbenzene | 1.2 |
| xylenes | 3.0 |

VOCs detected in areas other than the waste oil pit were scattered throughout the Landfill area and exhibited a different chemical pattern. BTEX constituents rather than chlorinated solvents were the indicator VOC parameters from these areas. In addition, the chlorinated solvents detected were typically at concentrations up to an order of magnitude less than concentrations detected in the waste oil pit.

VOCs were not detected in the North Area subsurface soil (SB-5).

Four UAO (Exhibit IV) soil borings designated OP-15 through OP-18 were completed downgradient from the waste oil pit to the top of bedrock. The borings were used to evaluate the presence of DNAPL in the vicinity of and downgradient from the waste oil pit. Two subsurface samples from each soil boring, one from the top of bedrock and the second exhibiting the highest PID screening were analyzed for VOCs, PCBs, and oil & grease. The results are summarized in Table 14.

DNAPLs were not observed in UAO subsurface soil samples. Analytical results indicate that chlorinated solvent constituents detected included TCE, 1,2 DCE, and PCE. The highest concentrations were detected in the samples exhibiting the highest PID reading and are as follows:

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| VOC | Location | Concentration (ppm) |
|---|---|---|
| TCE | OP-18<br>14-16 ft | 1.6 |
| 1,2-DCE | OP-17<br>4-6 ft | 0.17 |
| PCE | OP-15<br>14-16 ft | 0.38 |

One constituent, 1,2 DCE was detected at the top of bedrock in OP-17 (22-24 ft) and OP-18 (20-22 ft), at a concentration of 0.15 ppm and 0.43 ppm, respectively.

*Semivolatiles*
Seven Phase I and II subsurface soil samples were analyzed for semi-volatile organic compounds. Semivolatile compounds, location, and concentration are summarized below:

| Semivolatile | Location | Concentration (ppm) |
|---|---|---|
| butybenzylphthalate | SB-10<br>4-6 ft | 700 |
| naphthalene | SB-17<br>14-16 ft | 0.1 |
| 2-methyl-naphthalene | SB-17<br>14-16 ft | 0.22 |

Bis(2-ethylhexyl)phthalate, a common sampling and laboratory artifact, was detected in the five samples but was also detected in the laboratory blank (Table 15).

*Pesticides/Polychlorinated Biphenyls (PCBs)*
Pesticides and PCB data are summarized in Table 16. A review of Table 16 indicates that pesticides were not detected in RHRMLS subsurface soil samples.

Based upon field screening, 74 Phase I subsurface samples from within and in close proximity to the waste oil pit were analyzed for PCB analyses. Concentrations of PCBs ranged from below the method detection limit (0.080

ppm) to 7000 ppm at boring B36-A. The boring locations shown on Figure 20.

An additional thirty-five samples from the waste oil pit were analyzed for PCBs during the UAO (Exhibit IV). The summary of PCB concentrations is presented in Table 11 of this report and Table 1 of Exhibit IV.

Concentrations of PCBs were highest in the immediate vicinity of the former waste oil pit and generally decreased in concentration away from the pit (Figure 20). The UAO findings indicated that PCBs exhibit substantial decrease in concentrations, defined as an order of magnitude or greater, and/or contain PCB levels below 100 ppm, at depths greater than 8 feet below grade in the waste oil pit.

Based on PCB screening results, thirteen Phase II subsurface soil samples were analyzed by O'Brien & Gere Laboratories, Inc for PCBs. The Phase II soil boring program focused on areas outside the limits of the former waste oil pit and in the North Area. Laboratory analytical results are shown in Table 16 and illustrated on Figure 21.

PCB concentrations ranged from less than the method detection limit of 0.13 ppm to 14,000 ppm at SB-10 (4 to 6 ft). The highest PCB concentrations were found in borings SB-10, SB-12, and SB-17, which are located southwest of the waste oil pit (Figure 20). UAO soil borings were completed in the vicinity of SB-10 to further evaluate this designated "hotspot". UAO subsurface results indicated that substantial drop-offs between the UAO samples and SB-10 exist and that the elevated PCB concentrations detected at SB-10 are isolated. UAO results are presented in Exhibit IV.

Detectable concentrations of PCBs from soil borings were predominantly Aroclor 1248 with the exception of PCB aroclor 1254 in SB-20 (14-16 ft) at 0.055 ppm.

PCBs were detected in the North Area, however, the PCB was Aroclor 1254. The PCB was detected in SB-5 (4-6 ft) at 0.062 ppm.

*Inorganics*
Seven subsurface soil samples were analyzed for TAL inorganic parameters by O'Brien & Gere Laboratories, Inc. The results are presented in Table 17. A summary of metal constituents and concentrations that exceeded typical New York State soil concentrations is provided below.

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| Soil Boring | Iron | Nickel | Zinc |
|---|---|---|---|
| NYS Range | 17,500 to 25,000 | 8.5-25 | 37-60 |
| B6-2 | 27600 | 28.2 | 116 |
| B7-2 | 19600 | 22.8 | 129 |
| SB-5 | 33400 | 32.5 | 74.4 |
| SB-10 | 53100 | 35.1 | 413 |
| SB-12 | 32000 | 37.6 | 80 |
| SB-17 | 25300 | 26.3 | 65.7 |
| SB-20 | 35600 | 33.2 | 64.3 |

Landfill and North Area inorganic concentrations are predominantly within an order of magnitude of both NYS background ranges and other soil samples across the site. This trend suggests that landfill activities did not involve substantial impacts to site inorganic concentrations.

*Soil Borings Summary*

Pesticides were not detected in RHRMLS subsurface soils. Metal concentrations are within an order of magnitude of NYS background metal concentration ranges which suggests that Landfill activities have not impacted metals concentrations in the soil.

Concentrations of VOCs and PCBs at the site were detected in the Landfill fill areas. The highest total VOC and PCB concentrations were noted in the vicinity of the former waste oil pit, where concentrations ranged up to 287 ppm and 7,000 ppm, respectively.

The most prevalent VOCs were 1,2 DCE, TCE, xylene, toluene, and ethylbenzene. PCB concentrations were generally less than 200 ppm. The anomalous PCB concentration of 14,000 ppm at SB-10 was resolved to be an isolated occurrence.

**4.1.6. Test pit characterization**

*Landfill Test Pits*

Five test pits were completed to further evaluate geophysical survey anomalies and to characterize the waste materials. Test pit locations are shown on Figure 13 and test pit logs are included in Appendix H.

Test pits soil samples were analyzed for VOCs and PCBs. The following summarizes detected VOCs and range of concentrations.

| VOC | Concentration (ppm) |
|---|---|
| 1,2-DCE | 0.12 to 0.59 |
| TCE | 0.66 |
| toluene | 1.3 to 21 |
| ethylbenzene | 1.3 to 9.3 |

Test pit #5, which was completed at the location of SB-10, also contained xylene (7 ppm) and generally exhibited the highest VOC concentrations among the test pit samples.

PCBs were detected in the test pit samples at concentrations ranging from 4.4 ppm to 3600 ppm, which is notably less than the PCB concentration (14,000 ppm) detected in soil boring SB-10. This data suggests the detection in SB-10 is an isolated occurrence. The PCB Aroclor detected in the five samples was Aroclor 1248. Test pit analytical results are summarized in Tables 18 through 20.

Findings from the continuous test pit installed downgradient of the former waste oil pit are discussed in the UAO Report (Exhibit IV). It was concluded that the waste oil pit is not presently acting as a reservoir of free-oil and currently does not constitute a continuing source for downgradient LNAPL migration.

A test excavation was performed during implementation of the UAO to evaluate the origin and extent, if any, of LNAPL seeps emanating from the vicinity of MW-5S. The excavation revealed the presence of discrete non-recoverable droplets of LNAPL flowing from till fractures and zones of coarse interbedded overburden. The seeps were subsequently controlled with passive LNAPL collection systems (Exhibit IV). MW-5S was destroyed during the excavation and subsequent installation of the passive LNAPL collection system.

*North Area Test Pits*
Test pit TP-6 was completed in the North Area at the location shown on Figure 13. The subsurface materials encountered at TP-6 consisted of about 5 ft of brown and tan very fine to very coarse sand and angular gravel cover materials overlying gray very moist and wet clayey silt. Disseminated within the clayey silt was about 10% municipal refuse. A detailed log is included in Appendix H.

Soil samples collected from TP-6 indicated the presence of VOCs as follows:

| VOC | Concentration (ppm) |
|---|---|
| 1,2-DCE | 0.077 |
| tetrachloroethene | 0.10 |
| toluene | 0.87 |
| ethylbenzene | 1.1 |
| xylene | 1.7 |

PCB Aroclor 1248 was detected at a concentration of 1.5 ppm. Concentrations of VOCs and PCBs detected in the North Area were substantially lower than levels measured at the Landfill. Test pit analytical results are summarized in Tables 18 through 20.

## 4.2. Ground water quality assessment

### 4.2.1. Landfill overburden ground water quality

*Field Analyses*
Ground water samples have been collected from RHRMLS monitoring wells on six occasions between November 1988 and February 1995 as discussed in Section 2.5. The overburden wells in the Landfill area include MW-1, MW-2, MW-3, MW-4S, MW-5S, MW-6, MW-11S, MW-12, MW-17, and MW-18S.

Temperature of the overburden ground water has fluctuated depending on the month that sampling was conducted. The temperature of ground water in November 1988 ranged from 8°C to 9°C, whereas the temperature in August 1990 ranged from 11°C to 14°C. The pH of ground water ranged from 6.5 to

8.7, with an average pH of 7.6. The specific conductance of ground water has ranged from 28 uS/cm to 670 uS/cm.

LNAPL has been detected on the water table in MW-5S throughout the RI monitoring period at a thickness ranging from 0.22 ft to 0.60 ft. The source of the LNAPL is discussed in Section 4.1.6. LNAPL has not been detected in any other site monitoring wells.

*Oil & Grease*
Oil and grease analyses were completed during Phase I in monitoring wells MW-1 through MW-7. Oil and grease concentrations were highest in the following monitoring wells:

| Well | Concentration (ppb) |
|---|---|
| MW-1 | 20,000 |
| MW-2 | 13,000 |
| MW-3 | 1,100 |
| MW-5D | 30,000 |
| MW-6 | 8,200 |

*Volatile Organic Compounds (VOCs)*
The distribution of VOCs within the overburden ground water is illustrated on Figure 22. The highest concentrations of VOCs were detected in the monitoring wells adjacent to and downgradient of the former waste oil pit including MW-1, MW-2, MW-3 and MW-5S (Tables 21 to 22). Total VOC concentrations were also notably high in MW-6 and MW-18S, which are located in the southern portion of the Landfill area.

The range of total VOCs detected in overburden ground water is from 1 ppb in well MW-11S to 29,860 ppb detected in MW-2 (Figure 22). The most prevalent VOCs present in the ground water are primary chlorinated organic compounds such as TCE, PCE and 1,1,1-TCA along with secondary degradation compounds 1,2-DCE, 1,1-dichloroethene (1,1-DCE), 1,1-DCA, and vinyl chloride. TCE can occur either as a primary compound or secondary degradation compound of PCE. The biotransformation of chlorinated organic compounds under anaerobic conditions will result in dechlorination of these organic compounds (Vogel et al, 1987).

The most common chlorinated organic compounds detected at the Landfill were 1,2-DCE (ND to 24,000 ppb), 1,1,1-TCA (ND to 1300 ppb), and TCE (ND to 8,400 ppb), summarized as follows:

| Well No. | Compound | Concentration (ppb) |
|---|---|---|
| MW-1 | 1,2-DCE | 1,000 to 5,000 |
| | 1,1,1-TCA | 200 to 530 |
| | TCE | 4,700 to 8,400 |
| MW-2 | 1,2-DCE | 21,000 to 24,000 |
| | 1,1,1-TCA | 630 to 1,300 |
| | TCE | 490 to 2,800 |
| MW-3 | 1,2-DCE | 5,700 to 12,000 |
| | 1,1,1-TCA | 160 to 440 |
| | TCE | 3,000 to 4,400 |
| MW-4 | TCE | 1 to 10 |
| MW-5S | 1,2-DCE | 2,200 to 6,000 |
| | 1,1,1-TCA | ND to 430 |
| | TCE | ND to 2,000 |
| MW-6 | 1,2-DCE | 390 to 1,500 |
| | 1,1,1-TCA | ND to 81 |
| | TCE | ND to 430 |
| MW-7 | 1,2-DCE | 2 to 4 |
| | TCE | 2 to 6 |
| MW-11S | TCE | ND to 2 |
| MW-17 | TCE | 6 |
| | 1,2-DCE | 4 to 26 |
| | 1,1,1-TCA | ND to 5 |
| MW-18S | 1,2-DCE | 1,200 to 2,100 |
| | TCE | ND to 140 |

The VOC concentrations in overburden ground water exceeded the New York State Class GA ground water standards for each detected compound.



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However, other than the ten shallow dug well springs identified and evaluated in the vicinity of RHRMLS, overburden ground water is not used as a potable water source.

*Semivolatile Compounds*
With few exceptions, semivolatiles were not detected in the shallow ground water above the method detection limits (Table 23). Bis (2-ethylhexyl) phthalate, a common sampling artifact, was detected in the overburden ground water at concentrations ranging from 21 ppb to 54 ppb. The concentration detected in MW-1 (54 ppb) exceeds the NYS Class GA standard of 50 ppb. However, this compound was also found in the field blank sample at a concentration of 13 ppb, and is therefore not considered a site-related contaminant.

*Pesticides/Polychlorinated Biphenyls (PCBs)*
Pesticides were not detected in overburden ground water (Table 24). Ground water PCB concentrations were highest in the wells located in close proximity to and downgradient of the former waste oil pit. The distribution of PCBs within the overburden ground water is illustrated on Figure 23. PCB concentrations are shown in Tables 24 and 25.

The concentrations of PCBs have been consistently high in the following wells:

| Well | Concentration |
|---|---|
| MW-1 | 94 to 560 ppb |
| MW-2 | 2 to 1,400 ppb |
| MW-5S | 120 to 500 ppb |

PCB concentrations were less in wells located toward the periphery of the plume, such as:

| Well | Concentration |
|---|---|
| MW-3 | 1 to 1.4 ppb |
| MW-6 | 1 to 110 ppb |

All other overburden wells in the Landfill area exhibited PCB concentrations of 1 ppb or less. The primary PCB detected in shallow ground water was Aroclor 1248, with an altered pattern of Aroclor 1242 detected in MW-18S during May 1992, exclusively.

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*Inorganics*

Results of the unfiltered inorganics analyses from MW-1, MW-2, and MW-6 are summarized below:

| | NYS Class GA Standards | MW-1 | MW-2 | MW-6 |
|---|---|---|---|---|
| Aluminum | NE | 17.6 | 43.1 | 745.0 |
| Antimony | 3* | <0.06 | <0.06 | 0.68 |
| Arsenic | 25 | 0.014 | 0.0.34 | 0.086 |
| Barium | 1,000 | 0.448 | 0.643 | 7.72 |
| Beryllium | 3* | <0.005 | <0.005 | 0.028 |
| Cadmium | 10 | 0.010 | 0.006 | 0.105 |
| Calcium | NE | 31.0 | 41.9 | 70.7 |
| Chromium | 50 | 0.154 | 0.125 | 1.09 |
| Cobalt | NE | <0.05 | <0.05 | 0.899 |
| Copper | 200 | 0.375 | 0.113 | 1.66 |
| Iron | 300 | 88.8 | 106.0 | 1690.0 |
| Lead | 25 | 0.211 | 0.064 | 0.51 |
| Manganese | 35,000 | 33.0 | 27.2 | 43.2 |
| Magnesium | 300 | 18.8 | 37.2 | 281.0 |
| Mercury | 2 | 0.0012 | <0.0002 | 0.0012 |
| Nickel | NE | 0.182 | 0.179 | 2.17 |
| Potassium | NE | <5.0 | 5.79 | 36.4 |
| Selenium | 10 | <0.005 | <0.05 | <0.05 |
| Silver | 50 | <0.01 | <0.01 | 0.047 |
| Sodium | 20,000 | 7.31 | 13.2 | 11.8 |
| Thallium | 4* | <0.01 | <0.01 | <0.01 |
| Vanadium | NE | <0.05 | 0.065 | 1.07 |
| Zinc | 300 | 0.511 | 0.297 | 4.14 |
| Cyanide | 100 | <0.01 | <0.01 | <0.01 |

Note: * - NYS Class GA Guidance Value

Concentrations from the filtered samples, which are more likely representative of metals capable of being transported in solution by the ground water,



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indicated only barium, iron, and manganese were detected at concentrations exceeding the Class GA ground water standards. In addition, similar to the Sidney Center Landfill finds, these are likely the result of naturally high ground water turbidities in this region.

*Landfill Overburden Ground Water Summary*
The distribution of VOCs within the overburden ground water shown on Figure 22 indicates that a VOC plume about 1,200 ft wide and 400 ft in length extends from the Landfill to the South Pond. The northern extent is delineated by low concentrations of VOCs detected in MW-4S (3 ppb).

The overburden ground water VOC plume is not contiguous with the VOCs detected at the North Area. The southern extent of the plume is delineated by the 8 ppb of VOCs detected in MW-7S. The highest levels of VOCs detected within the plume were downgradient of the former waste oil pit at MW-2 and MW-3.

The PCB plume is less aerially extensive than the VOC plume and is centered around the former waste oil pit location (Figure 23). Pesticides were not detected in overburden ground water.

Four overburden ground water sampling events spanning approximately six years has been performed at RHRMLS. The ground water quality data shows a historic similarity in plume geometrics and magnitude of concentrations. This historic similarity suggest the VOCs and PCBs are in equilibrium.

### 4.2.2. Landfill shallow bedrock ground water quality

*Field Analyses*
Temperature of the bedrock ground water in May 1992 ranged from 6.4°C at MW-18D to 11.5°C at MW-4D with a mean temperature of 8.6°C. The pH of ground water has ranged from 6.9 to 10 with an average pH of 8.4. The specific conductance of ground water has ranged from 123 uS/cm to 493 uS/cm.

*Oil & Grease*
MW-5D was the only site shallow bedrock monitoring well during the Phase I sampling event. Oil & grease was not detected in MW-5D.

*Volatile Organic Compounds (VOCs)*
The total VOC concentrations in the shallow bedrock ground water from the December 1994 and February 1995 SRI sampling events are illustrated on

Figure 24. The SRI sampling event confirmed conditions identified in previous ground water sampling rounds. This confirmation corroborates the historic similarity of ground water quality data.

The lateral extent of VOCs in the shallow bedrock ground water is similar to the extent of VOCs in overburden ground water and is delineated by monitoring wells MW-4D to the north and MW-7D to the south. The similarity in extent of overburden and shallow bedrock aquifer VOC plumes indicates these zones are hydraulically interconnected.

With the exception of MW-5D and MW-18D, VOC concentrations in shallow bedrock ground water are typically an order of magnitude less than the concentrations detected in overburden ground water. This trend indicates that the overburden aquifer is the source of VOCs in the shallow bedrock zone.

The highest shallow bedrock VOC concentrations were detected in monitoring well MW-5D, which is downgradient of the former waste oil pit, and MW-18D, which is downgradient of the southern portion of the landfill. Over the duration of the RI, the total VOCs in MW-5D have ranged from 1,970 ppb to 7,770 ppb, and the total VOCs in MW-18D have ranged from 2,510 ppb to 3,380 ppb.

The total VOC concentrations in MW-5D and MW-18D have been consistently higher than in the adjacent nested wells MW-5S and MW-18S, respectively. These two wells are located at the base of the Landfill area slope, and may be indicative of ground water which is upwelling prior to discharge in the South Pond.

Prevalent VOCs detected in shallow bedrock ground water include the following:

| VOC | Well | Concentration Range |
|---|---|---|
| 1,2-DCE | MW-5D | 1,700 to 7,200 |
| | MW-18D | 2,200 to 2,900 |
| TCE | MW-5D | ND to 310 |
| | MW-18D | 270 to 400 |
| | MW-11D | 3 to 9 |
| vinyl chloride | MW-18D | 110 to 280 |
| | MW-5D | ND to 190 |

| VOC | Well | Concentration Range |
|---|---|---|
| methylene chloride | MW-5D | ND to 120 |
| 1,1-DCE | MW-5D | ND to 22 |
| 1,1-DCA | MW-5D | ND to 110 |
| 1,1,1-TCA | MW-5D | ND to 210 |
| toluene | MW-5D | 150 to 360 |
| ethylbenzene | MW-5D | ND to 19 |
| xylene | MW-5D | ND to 24 |

Trace concentrations of VOCs have been detected in upgradient well MW-11D.

*Semivolatile Compounds*
As indicated in Table 23, four semivolatile organic compounds were detected in the bedrock ground water samples. However, these concentrations were estimated and below 10 ppb.

*Pesticides/Polychlorinated Biphenyls (PCBs)*
Pesticides were not detected in the shallow bedrock ground water in the Landfill area. PCBs were detected in the following wells:

| Well | Concentration Range (ppb) |
|---|---|
| MW-5D | 0.2 to 0.36 |
| MW-11D | 0.43 to 1.3 |
| MW-18D | 0.1 to 0.43 |

The primary PCB detected in bedrock ground water was Aroclor 1248, with an altered pattern of Aroclor 1242/1248 detected in MW-5D and MW-18D during May 1992, exclusively (Tables 24 and 25). It should be noted that MW-11D generally exhibited the highest measured concentration of PCBs in bedrock ground water. However, the location of this well with respect to the landfill and lack of disturbances in this area do not suggest the area has been used for disposal of waste materials.

*Inorganics*
Results of the inorganics analyses from the unfiltered samples of MW-5D, MW-11D, and MW-18D indicated that ground water exceeded the NYS Class GA standards for arsenic, barium, chromium, iron, lead, and manganese (Table 26).

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| | NYS Class GA Standards | MW-5D | MW-11D | MW-18D |
|---|---|---|---|---|
| Aluminum | NE | 7,550 | 46,500 | 8,180 |
| Antimony | 3* | 50 U | 50 U | 50 U |
| Arsenic | 25 | 34.3 | 25.4 | 7.3 B |
| Barium | 1,000 | 235 | 412 | 200 |
| Beryllium | 3* | 1 U | 4 B | 1 U |
| Cadmium | 10 | 3 U | 3 U | 3 U |
| Calcium | NE | 11,300 | 43,900 | 61,900 |
| Chromium | 50 | 30 | 564 | 214 |
| Cobalt | NE | 22 B | 95 | 8 B |
| Copper | 200 | 80 | 214 | 40 |
| Iron | 300 | 22,300 | 98,100 | 15,400 |
| Lead | 25 | 30.3 | 61.4 | 11.9 |
| Manganese | 35,000 | 5,980 | 22,400 | 12,800 |
| Magnesium | 300 | 5,320 | 2.470 | 826 |
| Mercury | 2 | 0.2 U | 3.19 | 0.2 U |
| Nickel | NE | 31 B | 323 | 119 |
| Potassium | NE | 2,600 B | 30,700 | 10,600 |
| Selenium | 10 | 3 U | 15 U | 15 U |
| Silver | 50 | 4 B | 4 B | 2 U |
| Sodium | 20,000 | 5,090 | 28,900 | 18,200 |
| Thallium | 4* | 1 U | 1 U | 1 U |
| Vanadium | NE | 16 B | 67 | 13 B |
| Zinc | 300 | 78 | 385 | 109 |
| Cyanide | 100 | NA | NA | NA |

Note: * - NYS Class GA Guidance Value
B - The reported value is less than the Contract Required Detection Limit but greater than the Instrument Detection Limit
U - The analyte was not detected

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In most cases, the highest concentrations were detected in upgradient well MW-11D, which is likely indicative of the naturally high ground water turbidities in the region.

Metal concentrations from the filtered samples, which are more likely representative of metals capable of being transported in solution by the ground water, indicated exceedence of the Class GA ground water standard for only manganese of 300 ppb in downgradient wells MW-5D (1,820 ppb) and MW-18D (333 ppb). Upgradient well MW-11D exhibited an exceedence of the Class GA standards for chromium (50 ppb) and sodium (20,000 ppb), which were detected at concentrations of 111 ppb and 28,100 ppb, respectively.

**4.2.3. Landfill deep bedrock ground water quality**

Site-related contaminants, VOCs and PCBs, were not detected in either of the deep bedrock wells, MW-3DD or MW18DD.

*Landfill Bedrock Ground Water Summary*

The shallow bedrock ground water analyses indicate that a contaminant plume extends in an easterly direction from the Landfill with discharge into the South Pond. The lateral boundaries of the plume are generally defined by the non-detectable VOCs in MW-4D and low TCE concentrations in MW-11D and MW-7D. The total VOC concentrations in MW-5D and MW-18D have been consistently higher than in the adjacent nested wells MW-5S and MW-18S, respectively. The primary VOCs in the bedrock ground water are 1,2-DCE and TCE. Site-related contaminants have not impacted deep bedrock ground water at the Landfill.

**4.2.4. North area overburden ground water quality**

*Field Analyses*

The overburden wells in the North Area include MW-8, MW-9, MW-10, MW-13, MW-14, MW-15, MW-16, and MW-19. Results of the ground water sampling are illustrated on Figures 22 and 23.

Temperature of the overburden ground water has fluctuated depending on the month that sampling was conducted. The temperature of ground water in November 1988 ranged from 8°C to 9°C, whereas the temperature in August 1990 ranged from 12°C to 13.5°C. The pH of ground water has ranged from 7.4 to 8.3 with an average pH of 7.9. The specific conductance of ground water has ranged from 48 uS/cm to 160 uS/cm.

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*Oil & Grease*
Oil and grease was detected in MW-8 (1,400 ppb) and MW-10 (1,400 ppb). Oil and grease was not detected in MW-9.

*Volatile Organic Compounds (VOCs)*
Concentrations of total VOCs in the North Area ranged from non-detectable at MW-8 and MW-15 to 305 ppb at MW-9. The primary VOC detected in the North Area ground water was TCE, which was detected above the NYS Class GA standard of 5 ppb (Table 21). MW-9 exhibited the highest VOC concentrations in the North Area and contained predominately TCE (280 ppb), with much lower concentrations of 1,2-DCE, 1,1,1-TCA, and PCE (Table 21).

The North Area VOC plume extends to MW-13 and MW-19 as shown in Figure 22.

*Semivolatile Compounds*
Bis (2-ethylhexyl) phthalate was detected in the overburden ground water in MW-8 at a concentration of 22 ppb, which is below the NYS Class GA standard of 50 ppb. This compound was also found in the field blank sample at a concentration of 13 ppb, and is therefore not considered a site-related contaminant (Table 23). Butyl benzyl phalate and di-n-butylphthalate were also detected at an estimated concentration of 1 ppb.

*Pesticides/Polychlorinated Biphenyls (PCBs)*
Pesticides were not detected in the overburden ground water in the North Area. PCBs were detected at low concentrations in MW-8 (ND, 0.29 ppb, ND), MW-9 (ND, 0.16 ppb, 0.1 ppb), and MW-10 (0.5 ppb, 0.31 ppb, 0.3 ppb) (Figure 23). The PCB Aroclor 1248 was detected in the overburden ground water (Tables 24 and 25).

*Inorganics*
Results of the inorganics analyses from the unfiltered sample from MW-8 indicated that chromium (139 ppb), iron (13,400 ppb), and manganese (429 ppb) were detected at concentrations above the NYS Class GA standards of 50 ppb, 300 ppb, and 300 ppb, respectively (Table 26). Concentrations from the filtered sample, which are more likely representative of dissolved metals capable of being transported in solution by the ground water, indicated concentrations below the Class GA ground water standards.

*North Area Shallow Ground Water Quality Summary*
A review of the ground water contours and contaminants indicates that the VOC plume present in the North Area does not have the same source as the

Landfill plume. The primary VOC detected in north area ground water was TCE. The North Area ground water did not contain constituents present at the Landfill such as 1,1-DCE, 1,1-DCA, vinyl chloride, methylene chloride, chloroform, xylenes, toluene, chlorobenzene, and ethylbenzene. No distinct source areas were identified in the North Area during field investigations. Soil samples collected from the fill material during completion of exploratory test pit TP-6 contained xylene (1.7 ppm), ethylbenzene (1.1 ppm), and toluene (0.87 ppm) as the primary constituents.

Investigations conducted at the North Area suggest that ground water movement occurs within the unconsolidated glacial till unit and along the till/bedrock interface, which is under confining pressure from the overlying till unit. Contaminants detected in the North Area ground water may be migrating along the till/bedrock interface, from a topographically higher and upgradient source area, as evidenced by the artesian conditions at MW-9.

#### 4.2.5. Vertical ground water quality assessment

The installation and ground water quality analysis of the deep bedrock monitoring wells during the SRI defined the vertical extent of VOC and PCB concentrations at the Landfill. The SRI packer test data, summarized in Table 27 and illustrated on Figures 16 and 17, indicates that VOC contamination does not extend below 60 ft at MW-3DD and was below Federal drinking water maximum concentration limits below 98.5 ft at MW-18S. However, confirmatory laboratory analysis indicated VOCs and PCBs were not detected in MW-3DD and MW-18DD (Table 21).

### 4.3. Sediment and surface water quality assessment

#### 4.3.1. Sediment quality

An in-field sediment screening program was conducted to evaluate the lateral distribution of contaminants in the North Pond and South Pond, and to optimize the placement of surface water and sediment sampling locations. The results of the in-field screening program are presented in Table 5 and illustrated on Figures 9 and 10.

The screening data from the South Pond indicated PCBs ranged in concentration from less than 5 ppm to 175 ppm. The screening data from the North Pond indicated PCB concentrations were 5 ppm or less.

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A review of the figures indicates PCB concentrations are generally highest along the west edge of the South Pond. This may be an indication of overburden and shallow bedrock ground water discharge into the pond. PCB concentrations decrease towards the east away from the shoreline. The screening data indicates that the Landfill area is the primary source of PCBs in South Pond.

*Volatile Organic Compounds (VOCs)*
Total VOCs in sediment from the South Pond ranged from 0.006 ppm to 4.96 ppm. The data is presented in Tables 28 and 29 and shown on Figure 25.

The VOCs detected in the sediment included both chlorinated hydrocarbons and aromatic hydrocarbons. The most prevalent VOCs in the South Pond were 1,2-DCE and toluene. The concentration of 1,2-DCE was highest on the west shoreline at SEDTCL#2 at 3.5 ppm (Table 28), whereas the 1,2-DCE concentrations elsewhere in the pond ranged from non-detected to 0.12 ppm. Toluene was detected in the South Pond sediment at concentrations ranging from non-detectable to 0.27 ppm at SEDTCL#2.

South Pond sediment also contained methylene chloride (0.004 ppm to 0.019 ppm), acetone (0.067 ppm to 0.41 ppm), and 2-butanone (0.018 ppm to 0.12 ppm). Other VOCs detected in the sediment were xylene, ethylbenzene, chlorobenzene, 1,1-DCA, 1,1,1-TCA, TCE, chloromethane, and carbon disulfide. With the exception of chloromethane detected in sample 92-7 (0.008 ppm), VOCs were not detected in sediment downstream of the South Pond (Figure 26).

Total VOCs in sediment from the North Pond are shown on Figure 27. Sediment from the North Pond contained methylene chloride (0.002 ppm and 0.003 ppm), carbon disulfide (0.001 ppm and 0.002 ppm), toluene (0.006 ppm and 0.003 ppm), and xylenes (0.004 ppm and 0.06 ppm).

*Semivolatiles*
Semivolatiles were not detected in the sediment from the North and South Ponds (Table 30) with the exception of benzo(a)pyrene in the North Pond at 0.5 ppm (estimated) in SEDTCL #4.

*Pesticides/Polychlorinated Biphenyls (PCBs)*
Pesticides were not detected in sediments from the North and South Ponds.

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The distribution of PCBs in sediments is illustrated on Figures 28 through 30. Results of laboratory sediment analyses indicated a range of PCBs in the South Pond from below the detection limit to 1300 ppm (Tables 31 and 32 and Figure 28). The data indicate that the highest concentrations of PCBs were in sediments on the west shoreline of the South Pond.

PCBs were detected in sediment downstream of the South Pond at concentrations ranging from non-detect to 6.6 ppm. PCB concentrations decreased with increasing distance from the South Pond at downstream sampling locations (Figure 29). The furthest downstream sample location (92-10). which is located about 7500 feet from the South Pond, exhibited PCBs at 0.059 ppm.

In the North Pond, only sample SEDTCL#3 exhibited a detectable concentration of PCBs (aroclor 1248) at 0.37 ppm (Figure 30).

*Inorganics*
Sediment inorganic analyses are presented in Table 34. The following inorganic constituents were detected above the typical chronic sediment guidance values for the protection of aquatic environments (Long and Morgan, 1990):

| Inorganic | Guidance Value (ppm) | Location | Concentration (ppm) |
|---|---|---|---|
| Lead | 35 | SEDTCL#1 | 380 |
| | | SEDTCL#2 | 54.5 |
| arsenic | 33 | SEDTCL#2 | 35.2 |

**4.3.2. Surface water quality**
Surface water samples have been collected to evaluate the lateral distribution of site-related contaminants in the North Pond and South Pond.

*Volatile Organic Compounds (VOCs)*
Surface water total VOC concentrations in the South Pond ranged from 3 ppb to 1,982 ppb (Figure 31). The highest VOCs were detected on the west shoreline at SWTCL#2 (1,982 ppb) and at the South Pond outlet in SWTCL#1 (66 ppb).

The VOC constituents detected in SWTCL#2 included:

| VOC | Concentration (ppm) |
|---|---|
| 1,2-DCE | 1,600 |
| vinyl chloride | 200 |
| 1,1,1-TCA | 65 |
| TCE | 59 |
| 1,1-DCA | 48 |
| toluene | 10 |

Acetone was detected in (SW92-5) at a concentration of 7 ppb and may have been a residual of the decontamination procedure.

With few exceptions, the only contaminant detected in surface water throughout the South Pond was 1,2-DCE. The concentrations of 1,2-DCE ranged from non-detectable at 92-1 to 1600 ppb at SWTCL#2.
Low concentrations of TCE were detected at SWTCL#1 (4 ppb), SW92-3 (2 ppb), and SW92-4 (1 ppb). Carbon disulfide was detected at SW92-1 (29 ppb) and SW92-5 (4 ppb), and 1,1,1-TCA was detected at SWTCL#1 (2 ppb). Surface water VOC data are summarized on Tables 35 and 36 and illustrated on Figures 27, 31 and 32.

Downstream of the South Pond, 1,2-DCE was detected in SW91-17 (4 ppb) and SW91-18 (1 ppb). Sampling location 92-6, which is located about 1,450 ft downstream of the South Pond exhibited methylene chloride (0.9 ppb) exclusively (Figure 32). Carbon disulfide was detected downstream of the South Pond in SW92-7 (10 ppb), SW92-9 (1 ppb) and SW92-10 (12 ppb). Carbon disulfide was also detected in the trip blank at a concentration of 2 ppb. The low carbon disulfide concentrations indicate that it is naturally occurring.

Surface water sampling locations in the North Pond are shown on Figure 27. Surface water in the North Pond contained TCE in SWTCL#3 (4 ppb) and 1,2-DCE in SWTCL#4 (1 ppb). Sample SW91-20, which was collected from the small pond adjacent to MW-13, exhibited TCE (9 ppb).

*Semivolatiles*



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Di-n-butylphthalate was detected at low concentrations in the South Pond in SWTCL#1 (2 ppb) and SWTCL#2 (1 ppb). Bis(2-ethylhexyl)phthalate was detected at low concentrations in the South Pond in SWTCL#1 (3 ppb) and SWTCL#2 (2 ppb), and in the North Pond in SWTCL#3 (1 ppb). This compound was also found in the field blank sample, and is not considered a site-related contaminant (Table 37).

*Pesticides/Polychlorinated Biphenyls (PCBs)*
Pesticides were not detected in surface water from the North and South Ponds.

Results of laboratory surface water analyses listed in Tables 38 through 40 indicate a range of PCBs in the South Pond from non-detectable to 2.9 ppb. With the exception of SWTCL#2 and SWPCB#3, the South Pond surface water contained less than 1 ppb (Figure 33). The PCB detected in surface water was Aroclor 1248, with the exception of SW91-5 which contained both Aroclor 1248 and Aroclor 1260.

Downstream of the South Pond, PCBs were detected at concentrations ranging from 0.15 ppb to 4.6 ppb (Figure 34). PCBs were not detected at sampling points beyond SW91-19, which is about 2,600 ft downstream of the South Pond.

PCBs in the North Pond ranged from non-detect in SWTCL#3 and SWTCL#4 to 0.13 ppb in SWPCB#5 (Figure 30). The small pond adjacent to MW-13 did not contain PCBs.

*Inorganics*
Surface water inorganic analyses are summarized in Table 41. Results of the inorganics analyses indicates that SWTCL#2, which is directly downgradient of the landfills, exhibited above background concentrations of aluminum (609 ppb), iron (8,990 ppb), and manganese (2,850 ppb).

## 4.4. Air quality characterization

Air sampling was conducted at four locations (A through D) surrounding the former waste oil pit, as shown on Figure 6. Sample locations were selected to represent air quality conditions upwind and downwind of the former waste oil pit. Air samples were collected for analysis of VOCs using NIOSH Method 1003, and PCBs with NIOSH Method 5503. Air sampling results are summarized in Tables 42 and 43.

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No VOCs or PCBs were detected in the air samples or in the field blank. The NIOSH detection limits for Methods 1003 and 5503 were 2.0 mg/m$^3$ and 0.001 mg/m$^3$, respectively.

## 4.5. Data validation

The analytical data collected in RI Phases I, II, and III has been validated by Paladin Associates, Inc. The SRI data was validated by H2M Corporation. The results of the validation are included in Appendix I. In summary, the data validation reveals that the data are useable; however, selected analytical results are classified as estimated values. The analytical data are usable to characterize the magnitude and extent of PCB and VOC concentrations within the soils, ground water and surface water, and the data are acceptable to meet the objectives of the RI.

The analytical data for the RI was analyzed using various analytical methods and quality control procedures dependent upon the data quality objectives at the time of sampling. During Phases I and II of the RI, chemical characterization was performed using gas chromatography/mass spectrometry (GC/MS) methods consistent with the QAPP included in the approved Site Operations Plan dated February 1988. As an element of the Phase II program and during the Phase III program, the use of alternative GC analytical methods was proposed and approved by the USEPA for the analysis of VOCs and PCBs. To maximize the quality of data generated by the USEPA, a Standard Operating Procedure (SOP) for the validation of these data was prepared by O'Brien & Gere Engineers, Inc. and approved by the USEPA. The SOP is also included in Appendix I.

## 4.6. Wetlands delineation

A wetlands delineation was completed in the areas immediately adjacent to the South Pond and directly east of the landfill. The delineation was completed in accordance with the "Federal Manual for Identifying and Delineating Jurisdictional Wetlands", dated December 1989. A wetlands delineation map was prepared and is included as Figure 35. The wetlands extend north along Richardson Hill Road and encompass the smaller pond northwest of the Wyatt residence. The delineation extended south and was discontinued in the clearing for the power lines. North Pond wetlands delineation was performed separately during the Sidney Center Landfill.

# 5. Cultural resource assessment

In January 1990, a Stage 1A Cultural Resource Assessment (CRA) was performed at the Landfills by the Department of Anthropology at the State University of New York (SUNY) at Binghamton. The assessment included site files checks, a literature review, and site inspection. The Stage 1A Public Archaeology Facility Report is included as Appendix J and recommended the following:

1. The portion of the site in the Town of Masonville exhibits a low to moderate potential for containing prehistoric resources and a low potential for historic sites, therefore a Stage 1B CRA was recommended;

2. The portion of the site in the Town of Sidney exhibits a high potential for containing prehistoric sites, therefore a Stage 1B CRA was recommended;

3. Stage 1B CRA should be completed in compliance with OSHA regulations; and

4. The Field Services Bureau of the Office of Parks, Recreation and Historic Preservation in Albany be consulted prior to initiating Stage 1B CRA investigations.

In October 1992, a Stage 1B CRA was performed at the RHRMLS by the Department of Anthropology at SUNY - Binghamton. The assessment was designed to locate, identify, and inventory all historic and prehistoric cultural resources in the vicinity of the RHRMLS. The Stage 1B Public Archaeology Facility Report is included as Appendix K and concluded the following:

1. Twenty-nine shovel test pits were excavated and screened, and no pre-1945 historic artifacts or prehistoric artifacts were recovered;

2. Background research indicated two 19th century residences just north of the Landfills, but suggested that no pre-1945 historical resources would be located within the RHRMLS;

3. There were no standing structures within the project area; and

4. Based on the Stage 1B work completed by SUNY - Binghamton, it was recommended that no further archaeological work be completed at the Landfills.

# 6. Contaminant fate and transport

The objective of the environmental fate and transport evaluation is to identify the physical, chemical and biological changes that occur to a compound as it exists and migrates through media. These changes can affect the mobility of a compound, as well as its bioavailability. An array of physical, chemical, and biological processes control the movement and degradation or transformation of chemicals in soils, sediments, ground water, surface water, air, and biota. The following sections include general profiles of the fate and transport of inorganic and organic constituents of potential concern (COPCs) detected at the site in different media. The discussions are organized according to general chemical classes, as defined by common physical, chemical, or structural properties. These sections are summaries of information taken from several sources (Howard 1989, Howard 1991, ATSDR 1993, U.S. EPA 1985). A site specific fate and transport evaluation, based on the site environment and field analytical data, is presented below.

## 6.1. General profiles

*PCBs*

The dynamics, fate, and distribution of PCBs in the environment depend on a number of chemical, physical, and biological properties including solubility, octanol/water partition coefficient ($K_{ow}$), vapor pressure, Henry's Law constant, adsorption to soils and sediments, bioaccumulation in fish and wildlife, atmospheric and aqueous oxidation and photolysis, hydrolysis, and biodegradation.

The major factor affecting the environmental fate of PCBs is their low solubility in water and their affinity for organic rich media. PCBs are a mixture of chlorinated biphenyl congeners whose solubility in water is isomer-specific, determined by the degree of chlorination of the individual components. Measured water solubilities of PCBs range from a low of approximately 0.02 μg/l for an octachlorinated congener and increase to 1 μg/l for a hexachlorinated congener, 60 μg/l for a tetrachlorinated congener, and about 800 μg/l for a dichlorinated congener. It should be noted that different isomers within each of the congener groups can exhibit water solubilities which vary by an order of magnitude. Consistent with their low solubilities, PCBs exhibit relatively high $K_{ow}$s[1], ranging from about 5.0 for a dichlorinated congener to about 7.0 for an octachlorinated congener. This implies that at equilibrium, a 500,000- to 1,000,000-fold excess of PCBs would be partitioned into the organic phase compared to what might be present in the aqueous phase.

Due to their high $K_{ow}$s and low solubilities, PCBs in organic soils and sediments systems tend to be strongly sorbed to the particles in these media. The leaching potential is generally considered to be low. However, the presence of oils or other organics may facilitate the movement of PCB residues to ground water.

As a class, PCBs have relatively low Henry's Law constants, indicating that the tendency to volatilize from surface water bodies is relatively low, particularly if the PCBs are strongly sorbed to sediments. Likewise, volatilization of PCBs from soil sources is likely to be an insignificant transport pathway. However, airborne PCB concentrations may occur if dust with PCB adsorbed residues is generated due to wind and/or mechanical erosion.

Due to their affinity for lipids, PCBs have a tendency to accumulate in aquatic and terrestrial biota depending on the degree of metabolism within the organism to more water-soluble and more readily excreted structures. Lipophilic compounds, including PCBs, will be readily absorbed through gill tissue (in fish) and the gastrointestinal tract, and, once in the body, will favorably partition into adipose and other lipid-rich tissues. Thus, it has frequently been observed that residues of such compounds may accumulate in continuously or intermittently exposed organisms (Spacie and Hamelink 1984). Such an event is termed bioconcentration or bioaccumulation. The extent of bioconcentration depends on a

[1]$K_{ow}$ or the Octanol Water partition coefficient is an important parameter used to quantify this tendency. It is used in predicting biological uptake, lipophilic storage, and adsorption to sediments.

variety of factors, including the type of Aroclor mixture, the bioavailability of the PCB residues, and the exposure pattern of the exposed organism.

The tendency of PCBs to bind to organic rich media could also limit their bioavailability following ingestion by organisms. For example, activated charcoal and other high organic content particulates may inhibit the uptake and accumulation of PCBs by mice. This suggests that the bioavailability of PCBs adsorbed to soils and sediments is decreased due to the strength of adsorption, which in turn is a direct function of the organic content of the soil or sediment. This relationship is reasonable, in light of the known adsorptive behavior of PCBs on soils of varying organic content and the empirical relationships between water solubility, $K_{ow}$, $K_{oc}$, and soil binding.

The microbial degradation of the PCBs in aquatic media such as natural waters, activated sludge systems or sediments, is relatively extensive. The general trend which is observed is that PCBs, especially the mono-, di-, and trichloro-compounds could undergo biodegradation in sediments under aerobic conditions with half lives of 2 to 40 days. Half-lives for the tetra- and the pentachloro- (and higher) biphenyls range from one week to greater than one year. Information on PCB degradation in soils is considerably more limited, but a comparable range of persistence in soil (under aerobic conditions) was estimated for the various PCB congener groups (U.S EPA 1983). Microbial communities can also anaerobically degrade PCBs. Recent data suggests that even higher order congeners may be degraded by anaerobic mechanisms.

*Chlorinated Aliphatics*
In general, chlorinated aliphatic compounds can be described as relatively dense, highly volatile, and moderate to highly soluble compounds. Volatilization is the most important transport process for aliphatics in surface soils and surface waters. Chlorinated aliphatics present in exposed surface soils and surface water tend to rapidly volatilize to the ambient air. In sub-surface soils and ground water, chlorinated aliphatics residues may volatilize to soil gas in the vadose zone. Vapors in subsurface soils have the potential to migrate through soil gas and, depending on site specific conditions, may migrate to indoor air of homes in the vicinity of the soil of ground water sources. Additionally, since many chlorinated aliphatics have relatively high water solubility, VOCs in subsurface soils may

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leach to and migrate via advective and diffusive mechanisms in ground water.

In general, volatile chlorinated aliphatics are not known to bioconcentrate or bioaccumulate in terrestrial or aquatic biota, and therefore do not biomagnify through trophic levels in terrestrial or aquatic ecosystems.

Chlorinated aliphatics have been shown to degrade by both biological and non-biological mechanisms in surface water, ground water, and sediments. However, in most environmental settings, non-biological transformations are relatively minor processes compared with biological degradation. Chlorinated alkanes and alkenes in water may undergo hydrolysis to form chlorinated alcohols, and chlorinated alkanes can undergo dehydrohalogenation to form alkenes. Degradation can occur under anaerobic and aerobic conditions in water or soils. Anaerobic degradation of chlorinated aliphatics has been documented in field studies, and there is evidence of aerobic degradation as well. Biodegradation can occur through several mechanisms including reductive dehalogenation, enzymatic oxidation, and hydrolysis. In some situations, chlorinated aliphatics can be completely mineralized to water and $CO_2$, without the formation of persistent intermediates. However, formation of persistent intermediates, often more toxic, can also occur.

*Benzene and Substituted Benzenes [Benzene, Ethylbenzene, Toluene, Xylene (BTEX)]*
The properties of BTEX that exert the greatest influence on environmental fate are their high solubilities (approximately 1000 mg/l at 25 C) and high volatilities (approximately 95 mm Hg at 25 C, Henry's Law constant 5.5 E-3 atmosphere -cubic meter/mole). As such, when these compounds are released to surface water or surface soils, volatilization is the major fate and transport mechanism. For example, a half life of 4.81 hours for benzene in a 1m deep body of water has been calculated. Benzene released to the surface soil can be transported to the air through volatilization, to surface water through runoff, and to ground water as a result of leaching. For benzene in subsurface soil, the most likely transport mechanism will be leaching to ground water. Generally, based on relatively low organic carbon partition coefficients ($K_{oc}$ = 60 to 83), non-chlorinated aromatics are considered to be moderately to highly mobile in soils and ground water. Other parameters that determine

the leaching potential for BTEX in the soil include the soil type, amount of rainfall, depth of ground water, and potential for biological degradation.

The most significant atmospheric degradation reaction for BTEX vapors is photooxidation by the hydroxyl radical. It appears, however, that the photooxidation reaction occurs at a much slower rate in water than in air. As stated earlier, the most important loss mechanism for BTEX from water is volatilization. In addition, BTEX can be biodegraded by microorganisms in soils under both aerobic and anaerobic conditions. Generally, biodegradation will occur most rapidly in aerobic conditions, though BTEX compounds have been shown to biodegrade at slower rates under anaerobic conditions. Half lives of 68 to 110 days have been reported for the aerobic degradation of benzene in naturally occurring soil-ground water systems.

*Chlorinated Aromatics*
Chlorinated aromatics (CA) such as chlorobenzene and dichlorobenzene exhibit different chemical and physical characteristics than other aromatics, since they are relatively less volatile, have higher molecular weights, and are less mobile in media. CAs are relatively resistant to biodegradation in soils, sediments, and water. Unlike non-chlorinated aromatics, CAs have a potential to bioconcentrate.

In surface waters, volatilization is an important removal process for CAs. For example, DCB has a reported half-life of 4 hours from a river system one meter deep flowing at 1 meter per second. There is evidence that CAs can be biodegraded in surface waters and soils under aerobic conditions. CAs are not expected to biodegrade in anaerobic ground waters. In general, the less substituted (less chlorinated) aromatic VOCs are less resistant to biodegradation. Abiotic degradation is not a significant fate process in media.

## 6.2. Site specific evaluation

The following section is a site specific discussion of fate and transport processes that will govern the migration of chemicals of potential concern at the site.

**PCBs**

Available site data indicates that PCB residues are present in the shallow ground water beneath the waste oil pit, and are migrating in the ground water to the South Pond. As discussed in Section 6.1., the single most important property governing the fate, transport, and distribution of PCBs in surface water systems is their tendency to preferentially partition into organic rich media. As such, most of the PCB residues in the South Pond are expected to be strongly sorbed to sediment particles.

Based on the data collected during the RI, PCB concentrations in the South Pond sediments averaged about 150 ppm on dry solids basis. If left unimpeded, the primary transport mechanism of PCBs from the South Pond would be by advection of sediments through the outlet creek to a specific series of four beaver ponds situated from 50 to approximately 5000 feet downstream.

However, a sediment trap has been installed at the South Pond outlet to impede this transport mechanism. Analytical data downstream of the South Pond indicate that migration of PCB containing sediment has previously occurred (Figure 29). The detected sediment concentrations suggest, however, that the beaver ponds downstream of South Pond behave like settling basins, causing concentrations in the sediment phase to be significantly reduced within 1500 feet. This is supported by field data which showed that (5000) feet downstream, the average concentration in sediment was 0.107 ppm[2].

[2] The detected downstream sediment concentrations reflect the average steady state sediment PCB concentrations at this location. However, temporal variation in the PCB flux, corresponding with temporal variability in stream flow and sediment suspension rates, would be expected. Considering that the PCBs in South Pond would be strongly sorbed to sediment particles, the fate and transport processes dominating their migration would be settling and resuspension. Events that increase the rate of resuspension will enhance the transport of PCB laden sediment e.g., strong winds that create shear currents in the water column, activity of fish and humans that perturbs the sediment, etc. Storm events, while increasing resuspension, will also produce a compensatory diluting effect by contributing runoff and surface sediment washload.

Further, it is reasonable to conclude that migration through air will not present a significant pathway for transport. The high $K_{oc}$ of these compounds renders them virtually non volatile when they are sorbed to sediments. The aqueous PCB fraction will volatilize to a small extent, but surface water concentrations would be orders of magnitude lower than those in the sediment, so that the mass flux across the air water interface is expected to be negligible. This is consistent with air monitoring data collected during initial phases of the RI showing that ambient air PCB residues were not detected at a detection limit of 0.001 $mg/m^3$.

Some dissipation of PCBs is likely to occur as a result of microbial degradation. As reviewed in Section 6.1., recent data indicates that PCBs may degrade in anaerobic sediments similar to those in the South Pond. The strong affimity for lipids of PCBs will cause them to bio-accumulate in the tissue of fish present in South Pond. However, since the fish population in the South Pond is dominated by minnows and other small species which are not consumed by humans, the risk of human exposure via fish ingestion from the South Pond is low. Field observations suggest that recreational fishing is unlikely to occur in the large beaver pond located approximately 5,000 ft downgradient. In addition, sediment PCB concentrations decline to less than 0.5 ppm prior to this pond.

*Volatile Organic Compounds*
Volatile organic compounds (chlorinated aliphatics, BTEX compounds) are present in surface and sub-surface soils at the waste oil pit. Due to their high volatility, VOC residues in surface soils are likely to primarily evaporate to the ambient air. However, the resulting ambient air concentrations are expected to be low, due to dispersion and dilution mechanisms. This is supported by air sampling data collected during the initial phase of the RI which indicated that the site is not contributing detectable concentrations of site-related chemicals to the ambient air.

VOC residues in sub-surface soils are leaching to ground water and are migrating in the ground water to the South Pond. Temporal data from monitoring wells at the site indicate that the VOC ground water plume at the site is under steady state conditions. This implies that, if left unimpeded, VOC residues will continue to load to the South Pond at a constant rate. In addition, hydrogeologic data suggest that the shallow ground water at the site is contiguous with residential drinking water springs. However, VOCs have not been detected in the residential drinking water springs downgradient from the Landfill.

Due to their high solubility, high volatility, and relatively low $K_{oc}$ values, VOCs are not expected to significantly partition to sediments following discharge to the South Pond. Most of the VOC residues are expected to dissolve in the water column, and subsequently evaporate from the surface water. Consistent with this, analysis of South Pond sediment showed low VOC concentrations ranging from 0.009 ppm to 4.96 ppm. Based on the previously reviewed information, it is likely that some additional dissipation of sediment VOCs, particularly chlorinated aliphatics, will occur as a result of microbial degradation. Subsequent advective transport from the South Pond to downstream locations is unlikely to occur past the sediment trap. As such, most of the VOC residues are expected to volatilize.

*Chlorinated Aromatics*
Chlorinated aromatic (CA) compounds (e.g chlorobenzene) are present in surface and sub-surface soils at the waste oil pit. Due to their relatively high volatility, CA residues in surface soils are likely to primarily evaporate to the ambient air. However, the resulting ambient air concentrations are expected to be low, due to dispersion and dilution mechanisms. This is supported by air sampling data collected during the initial phase of the RI which indicated that the site is not contributing detectable concentrations of site-related chemicals to the ambient air.

As with the VOCs, CA compounds in sub-surface soils are migrating to the South Pond via leaching and ground water transport. The chlorinated aromatic compounds are likely to partition to South Pond sediments to a greater degree than the VOCs. However, dissolution in the surface water and subsequent evaporation will also be an important fate mechanism. Microbial degradation rates of CAs in the sediments is likely to be low. Subsequent advective transport from the South Pond to downstream is unlikely to occur past the sediment trap. In addition, most of the CA residues are expected to volatilize rather than partition to downstream sediments. This is supported by the fact that CA residues were not detected in down stream sediment samples analyzed during the RI.

# 7. Risk assessment

## 7.1. Human health risk assessment

### 7.1.1. Introduction

This section presents a baseline human health risk assessment related to chemical residues detected at the Richardson Hill Road Municipal Landfill Site (RHRMLS, the Site). The assessment was conducted in accordance with USEPA guidelines and procedures, as presented in the following guidance documents:

- *Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual* (Part A) (USEPA 1989)
- *Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual Supplemental Guidance "Standard Default Exposure Factors"* (USEPA 1991).
- *Guidance on Risk Characterization for Risk Managers and Risk Assessors* (USEPA 1992a).
- *Guidelines for Exposure Assessment* (USEPA 1992b) (57 FR 104, May 29 1992).

### 7.1.2. Methodology

The methodology utilized in this assessment first considers potential circumstances under which humans might become exposed to site related materials, and then evaluates the potential consequences of those exposures thought to be feasible. A chemical may pose a risk to human health only if receptor populations have the potential to be exposed to a chemical in sufficient quantities to affect their health. As such, a site specific risk assessment (RA) involves the identification of potentially hazardous chemicals at a site, the estimation of the concentrations of the chemicals that were detected at locations where receptors may contact them (exposure point concentrations), and the evaluation of potential adverse effects that may result from the estimated dose of the chemicals absorbed by receptors.

Consistent with the cited guidance, the RA was conducted in the following phases:

1. *Site Characterization.* The first step in the assessment process was to characterize the Site with respect to its physical characteristics as well as those of the human populations at or near the Site. The output of this step was a qualitative evaluation of the Site and surrounding populations with respect to those characteristics that influence exposure.

2. *Data Evaluation.* The objective of the data evaluation step was to organize the data into a form appropriate for use in the assessment, to evaluate the quality of the data for RA purposes and identify chemicals of potential concern (COPCs).

3. *Exposure Assessment.* In the exposure assessment, the pathways by which receptors may be exposed to on-Site chemicals were identified, and exposure point concentrations of the chemicals were estimated for each complete exposure pathway. Exposure point concentrations were estimated directly from analytical data collected at the Site as well as through the use of chemical fate and transport models.

4. *Toxicity Assessment.* In the toxicity assessment, available toxicological data for Site related compounds were gathered and reviewed. Dose-response relationships between the extent of exposure and the potential occurrence/severity of potential adverse health effects were evaluated.

5. *Risk Characterization.* In the risk characterization step, the toxicity and exposure assessments were integrated into quantitative expressions of potential human health risk. The resultant estimates of potential carcinogenic and non-carcinogenic health effects were characterized.

6. *Uncertainty Analysis*. In this section, the major uncertainties in the calculated risk estimates are discussed. The uncertainty analysis included a Monte Carlo Analysis of the potential cancer risk and Hazard Index (HI) associated with the dominant exposure pathways and chemical analytes.

These steps are presented and discussed in the remainder of this section.

### 7.1.3. Site Characterization

A detailed description of the Site conditions is presented in Section 3 of the RI report. The key elements of the Site with respect to the human health considerations are summarized below.

*Chemical and Physical Site Characteristics*. The RHRMLS consists of an 8-acre landfill formerly used for the disposal of municipal waste. It is located in a rural, sparsely populated region of Delaware County, New York. Operations at the landfill ceased in 1968. From 1964 to 1966, a portion of the landfill was used for the disposal of oils, some of which contained PCBs. The waste oils were disposed in a pit, approximately 50 ft long by 25 ft wide, that is located near the eastern perimeter of the landfill, about 200 ft west of Richardson Hill Road. Temporary fencing has been placed around the pit to restrict access by trespassers, while signs have been placed around the landfill to warn trespassers of the landfill's existence and boundaries.

The principal contaminant of concern at the Site and the focus of the RI is Aroclor 1248. Based on data collected during the RI, the average Aroclor concentration within the pit is 1203 ppm, with minimum and maximum values of 12 and 7000 ppm, respectively. Within the landfilled areas away from the pit, the average PCB concentration is 324 ppm. Other compounds detected in subsurface soils in the pit were 1,2 dichloroethylene (1,2-DCE), trichloroethylene (TCE), xylene, toluene, and ethylbenzene. In general, similar compounds were found in the landfill away from the pit, but at lesser concentrations.

Most of the land in the vicinity of the Site is characterized by steep topography and undeveloped woodland. There are approximately ten residences, generally occupied only seasonally, located within one mile of the Site. Each of these residences obtain their water from springs. There is no public water supply available in the area. Volatile organic compounds, (primarily TCE and 1,1,1,-TCA) have been detected in the springs designated as 1, 2, and 3 (see Figure 12). As discussed in Section 4.1.1, Amphenol has installed carbon filters and ultraviolet oxidation systems to treat the spring water at the Spizziri and Wyatt residences (springs 1 and 3, respectively). The residences are located hydraulically upgradient of the RHRMLS and hydraulically downgradient of the Sidney Landfill. Although contamination of springs 1, 2, and 3 is attributable to the Sidney Landfill, the risk to current residents (as well as the risk to future residents) is presented herein at the recommendation of USEPA. For purposes of the HHRA, the area in which these residences are located is called the North Area. Water supplies within this area have been designated in the HHRA as "spring" water and "ground water" to distinguish between different risk scenarios related to the types of water that may be used within the North Area under current and future use scenarios.

Available Site data indicate that PCB, VOCs, and other analytes are migrating via ground water discharge from the Site and, in particular the waste oil pit, to the South Pond. PCBs, VOCs, and certain elevated inorganic compounds have been detected in sediments and surface water from the South Pond. During implementation of the RI, sediments were removed from the western portion of the South Pond as an IRM.

The South Pond drains into Herrick Hollow Creek which may be used for recreational activities. Analytical data from sediments collected downstream of the South Pond indicate that migration of PCB containing sediment has occurred. However, for the most part, the PCBs migrating from the South Pond settle out in ten downstream ponds that are part of Herrick Hollow Creek.

Recreational fishing may occur in the downstream ponds. However, it is highly unlikely that fish caught from these ponds could be eaten since no edible fish were obtained from these ponds during fish collection activities.

*Future Site Conditions*. Land use at the Site in the reasonably foreseeable future is expected to remain the same as the current usage. To satisfy USEPA requirements for the HHRA, however, a future use scenario associated with residential ground water usage in the North Area is presented. The scenario assumes that a residence would be built within the North Area, that a drilled well would be installed, and that water from the well would be used as a potable source of water to the residence.

*Potential Receptor Populations*. The potential human receptors identified at the Site are:



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- construction workers potentially active at the landfill
- adolescent recreational users of the stream, ponds and woods around the Site
- current adult and child residents of the homes located in the North Area vicinity
- future residents in the North Area who consume ground water

Construction workers may be exposed to subsurface soils at the landfill if these soils are excavated in the future. Recreators may be exposed to sediments and surface water in the ponds, as well as the stream channel comprising Herrick Hollow Creek to its confluence with Trout Creek, and surface soils at the landfill. Current residents may be exposed to chemicals via ingestion and domestic use of spring water. Within the North Area, it is conceivable, although highly unlikely, that the ground water may be used by a future homeowner as a source of potable water.

### 7.1.4. Data Evaluation

The results of the risk assessment are used to identify remedial action objectives and medium specific remedial goals. Therefore, it is important that the risk assessment be based on high quality, technically defensible analytical data with respect to the identification and quantification of Site related residues. Furthermore, the collected data should provide a reasonably conservative representation of Site conditions, in order to make remedial decisions that are protective of human health and the environment.

The objectives of the data evaluation were to:

- evaluate the quality of existing analytical data,
- evaluate the quantity, spatial coverage, and appropriateness of sample locations,
- evaluate the appropriateness of analytical methods and detection limits,
- identify data gaps, and
- identify COPCs for the Site.

The data evaluation process was conducted in three steps :

A) data compilation and general review
B) data useability assessment
C) identification of chemicals of potential concern.

*Data Compilation and General Review.* The data collected during the RI were used as the input data for the risk assessment. A brief discussion of available data from each of the sampled media is presented below. Summaries of the analytes detected in surface soil, subsurface soil, ground water, sediment, surface water, and spring water are presented on Tables 44 to 49.

*Surface Soil (Table 44).* Thirty-four surface soil samples were collected to a depth of six inches below ground surface and analyzed for PCBs from the landfill area. Sixteen samples were also collected and analyzed for VOCs. In accordance with the USEPA-approved work plan, three samples were collected and analyzed for metals with the exception of selenium and sodium. One sample was analyzed for these latter analytes.

PCBs/Aroclor 1248 were detected on 29 of the 34 samples at concentrations ranging from 0.016 to 480 mg/kg, with an average concentration of 39 mg/kg. No VOCs or semi-VOCs were detected in surface soils.

*Subsurface Soils (Table 45).* One hundred and thirty seven subsurface soil samples were collected from the landfill area and analyzed for PCBs. PCBs/Aroclor 1248 were detected in 118 of the 137 samples at concentrations ranging from 0.08 to 14,000 mg/kg, and an average concentration of 370 mg/kg. VOCs and semi VOCs were also detected in subsurface soils. Ranges were from 0.2 mg/kg (1, 1, 2, 2 TCA) to 220 mg/kg (TCE) for VOCs and from 0.1 mg/kg (butylbenzylphthalate) to 700 mg/kg (naphthalene) for semi VOCs.

*Site Ground Water (Table 46).* For purposes of the HHRA, Site ground water is defined as the ground water beneath the RHRML as well as between the RHRML and the South Pond. The discharge of this ground water is to the South Pond.

During the RI, ten wells were installed in the landfill overburden ground water, three wells were installed in the shallow bedrock aquifer, and two wells in the deep bedrock. Chlorinated VOCs (predominantly 1,2 DCE, 1,1,1, TCA and TCE) were detected in shallow overburden ground water. PCBs were detected in shallow overburden wells MW-1, MW-2, and MW-5S, which are located immediately downgradient of the waste oil pit, whereas lower concentrations of PCBs were detected in wells located toward the periphery of the plume. Chlorinated VOCs and PCBs were detected in the shallow bedrock ground water. No Site related analytes were detected in the deep ground water.

*North Area Ground Water (Table 46A).* North Area ground water was evaluated using representative ground water data collected from wells MW-8, MW-9S, MW-13, MW-14, MW-15, and MW-16. Predominant analytes detected in the ground water from these wells were 1, 1, 1 TCA, PCBs, TCE, and trans-1, 2 DCE.

*Sediment (Table 47).* The sediment samples collected during the RI from the South Pond as well as Herrick Hollow Creek to its confluence with Trout Creek were used in the HHRA. Seventy-six sediment samples were collected to a depth of six inches and analyzed for PCBs. PCBs/Aroclor 1248 were detected in 65 of 76 samples. In addition, VOCs and metals were detected in sediments at the concentrations indicated on Table 47. Sediment samples downstream of the South Pond were collected in response to USEPA comments on the 1/96 draft Ecological Risk Assessment (ERA) and have been utilized in developing the HHRA.

*Surface Water (Table 48).* Surface water samples collected from the South Pond and in Herrick Hollow Creek during the RI were used in the HHRA. PCBs/Aroclor 1248, VOCs, and metals were detected in the surface water. Surface water samples downstream of the South Pond were also collected in response to USEPA comments on the draft ERA and have been used to develop the HHRA.

*Spring Water (Table 49).* Analytical data from Spring # 1 before treatment were used to evaluate potential human exposures to spring water. The chemical compounds associated with Spring # 1 are representative of the spring water in the North Area.

*Data Useability Assessment.* The data useability assessment was conducted in general conformance with guidance presented in *Guidance for Data Useability in Risk Assessment* (USEPA 1992a). Consistent with USEPA guidance, the following factors were evaluated in the data evaluation process:

1) Data Reports and Site Information,
2) Analytical Methods and Detection Limits, and
3) Data Quality Indicators.

*Data Reports and Site Information.* The sources of data used in developing the risk assessment were:

- RI report,
- Site visits by risk assessment and project staff, and
- interviews/communication with Amphenol personnel.

From these sources, specific Site characteristics relevant to the risk assessment were identified, including surface features, topography, vegetation, access and usage, surrounding land use, and surface water bodies.

*Analytical Methods and Detection Limits.* As discussed in the RI report, samples were analyzed according to approved USEPA methods under Level IV Data Quality Objectives, which is appropriate for risk assessment data. The sample-specific practical quantitation limits were compared with applicable, relevant and appropriate requirements (ARARs) and risk based screening concentrations to evaluate if the sensitivity of the analysis was adequate to address potential human health considerations at the Site. Samples for which the PQLs exceeded the ARARs or screening levels were noted. If a non-detected analyte had consistently elevated detection limits in a given medium, and there was a reasonable likelihood that the analyte may be present at elevated concentrations at the Site, then the analyte was included as a chemical of potential concern for the risk assessment. If the detection limit for a given analyte exceeded ARARs or risk based preliminary remediation goals (PRGs), but the analyte was unlikely to be present at the Site based on historical information and other Site specific data, then the analyte was not included as a COPC.

*Data Quality Indicators.* The data collected during the RI were validated prior to use in the risk assessment. The detailed data validation report for the analytical data is presented in Appendix I of the RI report. During the data validation, data quality indicators reflective of the uncertainty and useability of the reported result were assigned to specific analytical results. The following actions were taken during the risk assessment in response to the assigned data quality indicators:

R - The data were determined to be unusable for qualitative and quantitative purposes. Rejected data were not utilized in the risk assessment.

J - The analyte was positively identified; the associated numerical value was the approximate concentration of the analyte in the sample. While J indicated that the reported result is approximate, the analytical data were not adjusted to compensate for potential bias in the analytical result, due to uncertainty regarding the magnitude and direction of the bias.

B - The compound was detected in the associated blank, as well as in the sample, at a concentration less than the action limit. Consistent with USEPA guidance, it was assumed that "B" qualified data may be attributable to extraneous contamination. As such, "B" qualified data were treated as non-detects, and for COPCs the reported concentration was assumed to be the detection limit for the sample.

*Selection of Compounds of Potential Concern.* A list of Site COPCs was developed based on the data evaluation. The following sequence of comparisons was performed to identify the COPCs for the Site:

*Comparison with Risk Based Screening Values.* USEPA Region III has published analyte specific risk based soil and water screening concentrations (USEPA 1995). If the maximum detected concentration of a chemical in Site media is less than the published screening value, then it is highly unlikely that chemical represents a significant risk to human health, and was eliminated as a COPC for that media. Pursuant to discussions with USEPA Region II, the Region III screening values corresponding to an excess lifetime cancer risk of $10^{-6}$ for carcinogens, and a Hazard Quotient of 0.1 for non-carcinogens were used as a basis for comparison. If the maximum detected on-Site concentration was less than the selected Region III screening level then the analyte was eliminated as a COPC for that media. For soils and sediments, the residential surface soil screening values were used. For surface water, ground water, and spring water, the residential tap water screening values were used. The chemical specific health based screening values applied in selecting the COPCs are presented on Tables 44 thru 49.

*Comparison with Background.* Inorganic constituents which exceeded the risk based screening values were compared with their background concentrations. Background samples of soil, sediment and surface water were collected from control areas identified in the vicinity of the Site. A detailed discussion of the selection and sampling of control areas and the detected medium specific background concentrations is presented in Section 8.4.2 of the ecological risk assessment. Pursuant to discussions with USEPA Region II, the maximum detected Site concentration was compared with 2 times the arithmetic mean of the background concentrations. Analytes for which the maximum concentration was less



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than 2 times the mean background concentration were eliminated from the list of COPCs.

*Evaluation of Essential Nutrients*. Naturally occurring compounds were eliminated from the COPC list if they were essential nutrients, were present at concentrations only slightly elevated above naturally occurring levels, and were toxic only at very high doses. Consistent with the Sidney Landfill Risk Assessment (Malcolm Pirnie, 2/95), aluminum and essential inorganic nutrients (calcium, iron, magnesium, potassium, and sodium) were not included as COPCs.

*Evaluation of Detection Frequency and Concentration*. Certain analytes may have been eliminated as COPCs if they were detected at a low frequency (< 5 %) and at low concentrations the concentration of the detected analyte is less than that at which no adverse health effect would be anticipated.

Analytes not eliminated based on the considerations listed above were identified as COPCs and were carried through the quantitative risk assessment process. A full list of medium specific COPCs is presented on Tables 44 to 49 for surface soil, subsurface soil, ground water, sediment, surface water, and spring water respectively. A summary of the COPCs across environmental media is presented as Table 50.

**7.1.5. Fate and Transport Assessment**

A detailed description of the environmental fate and transport of COPCs at the Site has been presented as Section 5 of the RI report. The key elements of the fate and transport assessment with respect to the human health considerations are summarized below.

PCBs/Aroclor 1248, VOCs and metals are present in surface and subsurface soils at the landfill, and are migrating to the South Pond as dissolved ground water constituents . However, based on an evaluation of ground water flow direction and interpretation of hydrogeological data, ground water COPCs at the RHRML are not migrating to the potable water springs in the North Area. They are attributable to the Sidney Landfill. Similarly, the North Area ground water contaminants are also associated with the Sidney Landfill as opposed to the RHRML.

In the South Pond, the Aroclors and metals principally partition to the sediment phase, whereas the VOCs tend to volatilize from the surface water. PCB containing sediments have been detected at downstream locations. The ten ponds downstream of the South Pond would act as settling basins, limiting the extent of PCB transport downstream of the South Pond. Analytical data indicate that the sediment PCB levels decline to less than 1 mg/kg within 0.80 miles of the South Pond, and would not exceed 1 mg/kg at locations beyond the 10th downstream pond.

### 7.1.6. Exposure Assessment

In the exposure assessment, the mechanisms by which receptors may be exposed to the COPCs present at or migrating from the Site are identified, and the concentrations of the chemicals to which receptors may be exposed are estimated. The exposure assessment is conducted in the following sections:

- Exposure Pathway Analysis
- Quantification of Exposure
  - Estimation of Exposure Point Concentrations
  - Identification of Exposure Scenarios
  - Calculation of Chemical Intakes by Receptors

*Exposure Pathway Analysis*. An exposure pathway describes the course a chemical takes from the source to the exposed individual. An exposure pathway analysis links the sources, locations, and types of environmental releases with population locations and activity patterns to determine the significant pathways of human exposure.

An exposure pathway consists of four elements:

- a source and mechanism of chemical release,
- a retention or transport medium,
- a point of potential human contact with the contaminated medium (referred to as the exposure point), and
- an exposure route (e.g., ingestion) at the contact point.

A pathway is considered to be **complete** if the conditions listed above exist for that pathway. If one or more of these conditions are not met, there is no physical means by which a receptor may be exposed to the compounds of potential concern, and the pathway is classified as **incomplete**. Incomplete pathways were not considered further in the RA.

A summary of the Exposure Pathway Analysis for the RHRML Site is presented as Table 51, and is described below.

**Contact with Surface Soils** - The surface soil pathway is classified as **complete** under current and reasonably foreseeable future Site uses. COPCs have been detected in surface soils at the landfill. Potential recreators may contact surface soils at the landfill.

**Contact with Subsurface Soils** - The subsurface soil pathway is classified as **complete** under current and reasonably foreseeable future Site uses. COPCs have been detected in subsurface soils at the landfill. Potential construction workers may be exposed to subsurface soils via incidental ingestion and dermal contact if the fill materials are excavated for remediation.

**Contact with Surface Water/Sediments** - The surface water/sediment pathway is classified as **complete** under current and reasonably foreseeable future Site uses. COPCs have been detected in surface water and sediment in the South Pond and in Herrick Hollow Creek downstream of the South Pond. Potential recreators may be exposed to COPCs in surface water and sediment in the South Pond and downstream areas via incidental ingestion and dermal contact.

**Inhalation of Outdoor Air** - The outdoor air inhalation pathway is classified as **complete**. Recreators may be exposed to VOC vapors or particulate sorbed COPCs. However, no VOCs or Aroclors were detected in ambient air during air monitoring conducted during the RI. Therefore, while this pathway is theoretically complete, airborne exposures at the Site are likely to be negligible. Therefore, the ambient air exposure pathway is not quantified in this assessment.

**Use of Spring Water as a Domestic Water Supply** - This pathway is classified as **complete**. Spring water is currently used as a domestic water supply. Amphenol has installed water treatment systems which reduces analyte concentrations in the water to acceptable levels. However, pursuant to discussions with USEPA, the household use of spring water, assuming no remediation, has been quantified in this risk assessment. The exposure routes associated with this pathway are ingestion, dermal contact, and inhalation while showering.

**Use of Shallow Site Ground Water as a Potable Water Supply** - Exposure to shallow ground water at the Site through potable uses has been determined to be **incomplete**. This determination is based on factors related to the topography of the area, the limited yield of the shallow ground water zone, the small area represented by the RHRML, and the overall poor quality of this ground water.

The Site topography is characterized by steep relief, and would be unsuitable for residential use. There are no current residential uses of the area between the landfill and the South Pond, which is the point of discharge for shallow ground water from the Site. This area is very small and is bisected by Richardson Hill Road.

It is recognized, however, that the shallow ground water is mixed with leachate which contains elevated levels of Site COPCs, the remediation of which, irrespective of the absence of an exposure risk, will be established as a remedial objective.

**Use of North Area Ground Water as a Potable Water Supply** - Ground water in this area is not hydraulically connected to that of the RHRMLS. However, at the recommendation of USEPA, this area is being considered within the HHRA. Because this area contains residences, residential usage of the North Area ground water as a potable water supply could occur in the future. Therefore, this pathway has been classified as **complete** should residences utilizing ground water be established.

**Ingestion of Fish** - This pathway is classified as **incomplete**. No edible fish were obtained from the South Pond and down gradient ponds during seining and electroshocking activities. As such, it is unlikely that the South Pond and down gradient ponds contain sufficient edible fish populations to support sustained household consumption. Therefore, the fish ingestion pathway is not likely to be a significant exposure mechanism at the Site.

*Quantification of Exposure*. The next step in the exposure assessment was to quantify the magnitude, frequency, and duration of exposure for the complete exposure pathways. This information was used to estimate the chemical daily intake (CDI) for the exposed population. The quantification of exposure was performed in accordance with the following USEPA guidance for exposure assessment activities:

- *Guidelines for Exposure Assessment* (USEPA 1992b)
- *Risk Assessment Guidance for Superfund* (USEPA 1989).

The cited guidance states that the exposure/risk assessments should: 1) present a full and complete picture of risk, 2) be consistent and comparable across risk assessments, and 3) incorporate professional scientific judgement in the overall statement of risk (USEPA 1992b). To achieve these aims, the guidance specifically directs USEPA risk assessors to present a "full characterization" of the calculated risk including the potential central tendency, range, and some descriptor of the probability distribution of the calculated risk estimate. Risk assessments would, therefore, allow more informed risk management decisions which are based on a range of plausible risk calculations.

There may be considerable uncertainty relating to the estimated CDI for a given receptor group. This results from the random variability in exposure parameters, including the estimated exposure point concentrations. Variability and/or uncertainty in the *exposure point concentration* may arise from spatial and temporal variations in the chemical concentration at the exposure point. Variability and/or uncertainty in the *exposure parameters* occurs because each individual in the population has differences in activity patterns, behavior, and physical and biological characteristics. The current USEPA guidance requires that the CDI calculations include the "*average*","*high end*", and "*upper bound*" estimates in order to better communicate this uncertainty in the calculated risk estimate (USEPA 1992b).

Average (or central tendency) estimate refers to the expected value of the risk estimate for a randomly selected individual in the exposed population. The average estimate can be derived by using average values (usually the median value) for all the exposure factors.

High end risk refers to the top 95th percentile of the calculated risk distribution. The high end risk value is intended to be a plausible estimate of the risk level for those individuals at the upper end of the risk distribution. Theoretically, the 95th percentile high end risk estimate would represent the risk level which would be expected to be exceeded by only 5% of the population, and 95% of the population would have individual risks below that estimate. The high end risk estimate may be derived by conducting a mathematical simulation of the risk levels that may occur by randomly combining the exposure parameters according to the probability distribution for each of the exposure parameters. This technique is known as a Monte Carlo simulation. The high end risk estimate may be selected as the upper 95th percentile of the final risk distribution generated by the Monte Carlo simulation. Further, by using input distributions which are reflective of age specificivity, the effects of factor interdependence that contribute to uncertainty are minimized.



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Upper bound estimates represent a "worst case" scenario for the risk estimate. The bounding estimate may be used to develop a statement that the risk "does not exceed" the calculated bounding value. The upper bound estimate is calculated by applying "highly conservative" assumptions in calculating the risk value. "The actual probability that any individual in the population would be subject to the combination of conservative exposure assumptions is extremely small, and usually so small so as to be a practical impossibility" (USEPA 1992b).

To address the central tendency analysis required by USEPA Region II, a Monte Carlo simulation is presented in Appendix L. Appendix L describes the input parameters and output from the analysis. Risk calculations are summarized in Sections 7.1.9 and 7.1.10.

Current USEPA guidance (USEPA 1989) also recommends the calculation of the "Reasonable Maximum Exposure " (RME) when evaluating exposures. The RME estimate is derived by combining a series of default average and upper bound exposure factor estimates in calculating the CDI. However, in most cases, application of the default RME parameters results in a CDI estimate which is reflective of the "upper bound" exposures, rather than the "high end" exposures.

For the exposure assessments at the RHRML Site, and pursuant to discussions with USEPA Region II, the 'RME" estimates, as defined by USEPA were calculated and are discussed in this risk assessment. In addition, consistent with current guidance, the probability distribution for the calculated risk values was characterized via a Monte Carlo simulation. The methods, results and conclusions of the Monte Carlo simulation are presented in Appendix L, and are discussed in the Uncertainty Analysis section of this report.

The calculations were performed in the following steps:

- Estimation of exposure point concentrations
- Calculation of chemical intakes by receptors.

*Estimation of Exposure Point Concentrations*. An exposure point is a location where receptors may contact Site related chemical residues. Exposure point concentrations were estimated based on the analytical data collected from the Site. Specific methods for deriving exposure point concentrations are discussed below.



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Exposure point concentrations were estimated from Site specific analytical data. In calculating exposure point concentrations, a value of one-half the detection limit was applied for non-detect results. For each COPC, the number of samples, proportion of non-detects, and frequency distributions of the data and log-transforms of the data were examined. Consistent with USEPA guidance, the arithmetic mean of the detected COPCs was utilized as the average exposure point concentration. USEPA guidance states that the upper 95% confidence level on the arithmetic mean (UCL), or the maximum detected concentration, whichever is lower, should be used as the RME exposure point concentration. Therefore, the following approach was applied in estimating the RME exposure point concentrations:

For small data sets, the derived UCL is dependent on the shape of the underlying distribution of the data (eg. normal, lognormal, other). However, it is difficult to infer the shape of the underlying distribution for a given COPC if the number of detected analytes is less than 15. Therefore, for COPCs for which there were fewer than 15 detects, the maximum detected concentration was used as the RME exposure point concentration.

For analytes with greater than 15 but less than 25 detects, the data distribution was examined, evaluated for consistency with a normal or log-normal distribution, and the appropriate distributional assumptions used in calculating the UCL. However, if a normal or log-normal distribution did not adequately characterize the data, then, by default, normality assumptions were applied in calculating the UCL. The RME value was taken as the lower of the UCL or the maximum concentration.

If there are greater than 25 detects for a given COPC, and the total fraction of detects vs non-detects is high, then, based to the central limit theorem, the UCL may reasonably be estimated using an assumption of normality. This is based on the Central Limit Theorem which states that, for large data sets, the sampling distribution of the arithmetic mean will be normally distributed regardless of the shape of the underlying distribution. Therefore, for COPCs for which there were greater than 25 detects, the UCL was derived based on an assumption of normality. The RME value was taken as the lower of the UCL or the maximum concentration.

Summary statistics for each COPC and the calculated average and RME exposure point concentration, and the basis for the calculated RME concentration, are presented in Table 52. The above approach was applied for all analytes. Vinyl chloride, however, merits specific discussion.

Although vinyl chloride in surface water would not have been retained as a COPC because of the infrequency of detections, the single detection of 200 ug/l was retained by averaging the detection with one-half the detection limit of the non-detects to acknowledge its possible presence in the South Pond. Based on prior experience, it is unreasonable to assume that the RME concentration would be equal to the maximum detected concentration, since the single detection was an outlier and could not be confirmed. The resulting RME concentration calculated and applied in this assessment is 7 ug/l.

*Calculation of Chemical Intakes by Receptors.* Based on the identified exposure scenarios and intake parameters, chemical specific chronic daily intake (CDI) estimates were developed for the construction worker, adolescent recreator, current adult residential user of spring # 1, current child residential users of spring #1, and future North Area resident according to the equations presented on Table 53 and the parameter estimates presented on Tables 54, 55, 5, 56a, and 57, respectively.

### 7.1.7. Toxicity Assessment

In the toxicity assessment, available toxicological data summaries for Site related compounds are reviewed and the relationship between the extent of exposure to a specific contaminant and the increased likelihood and/or severity of adverse effects are estimated. The potential toxicologic effects induced by a given dose of a chemical are classified according to two criteria: carcinogenic effects, and non-carcinogenic effects.

*Non-Cancer Effects.* A non-cancer health effect occurs as a result of damage to cells in one or more human organs, which causes the organ(s) to function less efficiently. Due to the body's ability to cope with small doses of a chemical, a non-cancer health effect will not occur if intake of a chemical is less than a certain critical dose. This is referred to as a No Observed Adverse Effect Level (NOAEL) for a chemical. If the calculated intake of a chemical is less than the NOAEL for that chemical in a given species, then no adverse non-cancer health effects are expected as a result of that exposure.

The specific non-carcinogenic toxic effects that may be elicited depend on the exposure concentration and the duration of exposure. If an individual is exposed to very high concentrations of a chemical, severe organ dysfunction can occur in a short period of time. This is termed an acute toxic effect. If an individual is exposed to lower levels of a chemical regularly for a long period of time, smaller amounts of repeated damage to the organ can accumulate and ultimately cause the organ to malfunction. These are termed sub-chronic and chronic toxic effects (depending on the exposure duration).

Examples of non-cancer health effects on the body can be illustrated by the effects of alcohol. At very high doses, alcohol can have an acute toxic effect on the central nervous system. The symptoms of this effect may be dizziness, nausea and loss of consciousness. If smaller doses of alcohol are consumed regularly for a long period of time, it can result in toxicity to the liver which can lead to impaired liver function and ultimately cirrhosis of the liver. This is a chronic toxic effect of alcohol. Alcohol will have no toxic effects, however, if it is consumed in very small quantities, even for a long period of time. This quantity would represent the chronic NOAEL for alcohol.

*Reference Doses (RfDs).* In order to evaluate potential non-carcinogenic effects following exposure of human populations to chemicals, USEPA derives chemical specific "reference doses" (RfDs). If the calculated intake of a chemical is less than the published RfD, then no adverse non-carcinogenic effects are expected in the exposed population. A brief discussion of the methods by which RfDs are derived is presented below.

For some chemicals, RfDs are derived directly from data on human exposures. Such data include data relating to occupational exposures that are known to have no adverse effects, normal dietary levels of certain chemicals (eg. magnesium), therapeutic doses of certain chemicals (e.g. silver), and epidemiologic data relating to populations with background exposures (eg. selenium). For most chemicals, USEPA derives RfDs based on laboratory studies in which experimental animals were exposed to different concentrations of a chemical, and a NOAEL is estimated. If data from several animal studies are available, USEPA seeks to identify the species that is most comparable to humans based on a knowledge of specific biologic properties. However, if adequate comparative data are not available, USEPA selects the study on the most sensitive animal species as the critical study for the basis of the NOAEL. The NOAEL is then used to derive a RfD for potential adverse effects in human populations.

*RfD Uncertainty Factors.* In most cases, there is considerable uncertainty regarding the extension of toxicologic data from animal studies to humans. In other words, the actual RfD for humans or sensitive sub-populations of humans would not be precisely known based on data in animal species.

This uncertainty arises because there may be differences between the animal and human species regarding factors such as the metabolism of the chemical, the distribution and clearance rate of the chemical from the body, and the sensitivity of specific organ systems to the chemical. Therefore, the USEPA derives RfDs that are designed to be protective of the public at large, *including sensitive sub-populations*. The USEPA applies a series of "uncertainty" factors to calculate the final RfD values. Depending on the data, the NOAEL may be divided by an uncertainty factor ranging from 0 to 10,000. For human data, in most cases, the uncertainty factor of 10 is applied for the application of data from the public at large to sensitive sub-populations. For animal data, a minimum uncertainty factor of 100 (10 for sensitive sub-population, and 10 for animal-human extrapolation) is applied for deriving the human RfD.

*Carcinogenic Effects.* The other health effect of concern in the exposure of humans to chemicals in the environment is the induction of cancer.

*Weight of Evidence.* USEPA classifies chemicals according to their potential to induce cancer in humans. In general, USEPA reviews and evaluates available data regarding the potential carcinogenic effects of a chemical, and assigns a "carcinogenicity" classification according to a weight of evidence classification scheme (USEPA 1986). A chemical may be classified into one of five groups with respect to the weight of evidence for human carcinogenicity. The categories are:

- Group A - Known Human Carcinogen. A chemical is classified in group A if there is sufficient[3] evidence from human observations (epidemiological studies) to support an association between exposure to a chemical agent and cancer in humans.

- Group B - Probable Human Carcinogen. A chemical may be classified as a B1 or a B2 carcinogen. An agent is classified as a B1 carcinogen if there is sufficient evidence for carcinogenicity based on animal studies, and limited (suggestive but not conclusive) evidence based on human observations. A B2 carcinogen is an agent for which there is sufficient evidence for carcinogenicity in animals, and inadequate evidence for carcinogenicity in humans.

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[1] The definition of the terms "sufficient", limited, and adequate are given in USEPA 1986, Guidelines for Carcinogen Risk Assessment, (51 FR 33992).



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- Group C - Possible Human Carcinogen. An agent is classified as a group C carcinogen if there is limited evidence for carcinogenicity in animals and inadequate evidence for carcinogenicity in humans.

- Group D - An agent is classified as a group D agent if there is insufficient data available with which to evaluate the carcinogenicity of the chemical.

- Group E - An agent is classified as a group E agent if there is no evidence for carcinogenic effects based on at least two technically adequate animal studies.

*Slope Factors*. For group A, B, or C chemicals, USEPA derives chemical specific cancer slope factors. A cancer slope factor is a number which, when multiplied by the estimated chemical specific CDI, provides an estimate of the "excess cancer risk" associated with that exposure. Theoretically, the excess cancer risk represents lifetime the probability (greater than background) that a carcinogenic event would occur in an individual as a result of a given exposure or pattern of exposures. However, it is important to note that for many chemicals, the excess cancer risk as calculated by USEPA's procedure may result in a highly conservative estimate of the potential cancer risk. Indeed, as acknowledged by USEPA 1986, the procedure "does not necessarily give a realistic prediction of the risk. The true value of the risk is unknown, and may be as low as zero".

*Toxicity Profiles*. For each chemical of concern, a brief synopsis of the human toxicological effects, including acute effects, and chronic RfDs and cancer slope factors published by USEPA, was compiled from the following hierarchy of sources:

- USEPA's Integrated Risk Information System (IRIS) database (USEPA 1994),
- Health Effects Summary Tables (HEAST),
- Environmental Criteria Office (ECAO); and
- Agency for Toxic Substances and Disease Registry (ATSDR).

This information is summarized on Table 58 and is presented in Appendix M of this report.

*Adjustment for GI Absorption*. In some instances, it was necessary to use oral toxicity data to evaluate dermal exposures. The dermal CDI represents the absorbed dose of the analyte. However, for some analytes, the oral toxicity data were based on the *administered* dose rather than the absorbed dose. Therefore, in order to assess dermal exposures, the oral toxicity data were adjusted to reflect the absorbed dose in accordance with USEPA guidance (USEPA 1989) as follows:

Adjusted dermal RfD = Oral Rfd x gastrointestinal absorption efficiency

Adjusted dermal slope factor = Oral slope factor/gastrointestinal absorption efficiency

The adjusted toxicity data for evaluating dermal exposures is presented as Table 59.

**7.1.8. Risk Characterization**

In this section of the RA, the toxicity and exposure assessments are summarized and integrated into numerical values which may be used to evaluate the likelihood of adverse health effects in populations potentially exposed to Site related chemicals. RME estimates of potential health risks were developed for each potentially exposed receptor at the Site.

Chronic non-cancer health effects were evaluated by comparing the chemical specific CDIs with the respective chronic RfD as given below.

1. For each receptor identified during the exposure assessment, and each individual exposure pathway, the chemical specific hazard quotients (HQs) were calculated where HQ is given by:

$$HQ = CDI/RfD$$

2. For each exposure pathway, the chemical specific hazard quotients were summed to calculate the Hazard Index (HI) for that pathway. For each receptor, the pathway specific hazard indexes were summed to obtain the total HI for that receptor. A total HI of less than one indicates that it is highly unlikely that chronic non-cancer toxic effects would occur for the given receptor.

To evaluate carcinogenic effects, the incremental cancer risk associated with exposure to chemicals of concern was calculated using chemical specific slope factors as described below.

1. For each receptor, identified during the exposure assessment and each exposure pathway, the chemical specific risk is given by:

$$\text{Cancer risk} = \text{CDI} * \text{slope factor.}$$

2. For each receptor, the total incremental excess cancer risk was calculated by summing the pathway specific cancer risk. This calculated risk estimate was then compared with an acceptable excess cancer risk. A total Site cancer risk that is less than $10^{-4}$ to $10^{-6}$ is considered to be an acceptable exposure level (NCP 1990).

Summaries of the chemical specific, pathway specific and total HIs for each receptor are given on Table 60, while supporting information and calculations are provided in Appendix N. Chemical specific, pathway specific and total excess cancer risk estimates associated are summarized on Table 61. A brief discussion of the RME risk estimates associated with each receptor is presented below.

*Construction Worker.* The calculated HI for the construction worker is 52 and the calculated cancer risk is 1.1 x $10^{-4}$. The only analyte for which the HQ exceeds 1 is Aroclor 1248. This suggests that unprotected construction workers may experience calculated non-cancer health risks due to PCB exposure outside the range of acceptability defined by USEPA if they repeatedly contact landfill subsurface soils in the future.



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The reviewer should note that the RfD for Aroclor 1254 was used to derive the HQ because there is no published RfD for Aroclor 1248 currently available and because of the structural similarity between Aroclors 1248 and 1254. The HQ could be considerably more conservative than would otherwise be anticipated based on the toxicological differences between these Aroclors. It is generally recognized that toxicity decreases with decreasing chlorine content. Consequently, Aroclor 1248 with a chlorine content of 48% would be considered less toxic in its non-cancer effects than Aroclor 1254 with a chlorine content of 54%. For a similar reason, the cancer risks provided in the risk calculations may also be over estimated. To calculate the cancer risk, the slope factor for Aroclor 1260 was used because there is currently no published slope factor for Aroclor 1248. For cancer effects, Aroclors containing a higher chlorine content are generally believed to be of higher tumorigenic potency than those of lower chlorine content; a positive indication of carcinogenicity by Aroclor 1248 has not been shown (Chase et al. 1989). The uncertainty introduced by the use of the slope factor for Aroclor 1260 is further discussed in Section 9.

*Adolescent Recreator.* The calculated total RME HI for the adolescent recreator is 14 and the calculated cancer risk is $1.8 \times 10^{-4}$. As shown on Tables 60 and 61, the majority of the calculated HQ and cancer risk is associated with PCBs in soils, sediments, and surface water.

*Current Residents* The calculated HI for the adult and child resident of the houses in the North Area is 0.3 and 0.73, respectively. This is supportive of a conclusion that domestic use of untreated water from Spring #1 would not result in unacceptable health risks to residents. The total lifetime cancer risk for the adult and child resident is $1.9 \times 10^{-5}$ and $1.8 \times 10^{-5}$, respectively, which is within USEPA's target risk range.

*Future Residents.* The calculated HI for an adult and child resident ingesting ground water from the North Area is 2.2 and 5.2 , respectively, while the total lifetime cancer risk for an adult and child resident is $1.0 \times 10^{-4}$ and $5.9 \times 10^{-5}$, respectively. These risks for ground water, originating from the Sidney Center Landfill, are outside the USEPA's target risk range.

### 7.1.9. Uncertainty Analysis

The risk measures used in this RA are not precise, deterministic estimates of risk, but conditional estimates controlled by a considerable number of consecutive upper-bound assumptions regarding exposure and toxicity. They are designed to overestimate the true risk value, as opposed to present a precise, realistic estimate of actual health risks. This is done by convention, consistent with USEPA protocols. There are several categories of uncertainties associated with RAs: calculation of exposure point concentrations, toxicity values for each substance, and exposure assessment.

The main sources of uncertainty in the RA are the estimation of exposure point concentrations and calculation of CDIs. Uncertainties related to these sources are discussed below.

**Calculation of Exposure Point Concentrations** - Conservative judgements were applied in calculating RME exposure point concentrations. For most of the analytes, the maximum detected concentration was applied. For PCBs, the RME exposure point concentration was based on the UCL on the arithmetic mean. As such, it is highly unlikely that the estimated exposure point concentrations underestimate representative concentrations in the Site media.

**Calculation of Chronic Daily Intakes** - Most of the input parameters applied in estimating the RME CDIs were conservative estimates, consistent with USEPA's RME methodology. In particular, the dermal contact calculations for sediment and surface water are particularly conservative, since the methods assume that 25% of the total body surface area would be in continuous contact with surface water/sediment over a period of 2 hours per day exposed. While this scenario is possible, it is likely that surface water/sediment contact rates would be significantly less than this most of the time. This is particularly important because the dermal contact pathway accounts fora large percentage of the total HI and cancer risk associated with recreational exposures.

**Selected Toxicity Values** - Current USEPA policy dictates that all Aroclor mixtures be evaluated based on the cancer toxicity data for Aroclor 1260, which have been shown to significantly increase cancer incidence rates when administered at high doses to laboratory animals. The cancer slope factor published by USEPA of 7.7 $(mg/kg/day)^{-1}$ is based on dose response for Aroclor 1260 in rats. However, as discussed by USEPA (1992) available data indicate that lower chlorinated PCB mixtures, such as Aroclor 1248, have not been shown to be carcinogenic in laboratory animals, and may be less potent as compared with Aroclor 1260. Most of the PCB concentrations detected at the Site were Aroclor 1248. Therefore, use of USEPA's toxicologic reference data for Aroclor 1260 likely results in a significant overestimation of potential cancer risks from exposure to Site related residues.

**Monte Carlo Simulation** - Probability distributions presented in Appendix L were derived using the RfDs for Aroclors 1016 as well as 1254 in order to establish the degree of uncertainty of the RME point estimates for Aroclor 1248. Predominant pathways providing the majority of the risk that may be associated with the Site based on RME calculations - exposure to surface soil, surface water, and sediments by an adolescent recreator - were used in the analysis. For non cancer risks, the output of the Monte Carlo simulations presents that: 1) there is a moderate to high degree of uncertainty in the RME point calculations, 2) the actual HQ for these pathways is substantially less than the RME point estimates and 3) the HQ for 95% of the individuals within the exposed population is less than 4.3. The Monte Carlo simulations for possible cancer effects from exposure to Aroclor 1260 in the same matrices indicates that the uncertainty is moderate (Hoffmann and Hammonds 1994) and that the calculated increased cancer risk for 95% of the individuals in the population exposed to Aroclor 1260 in the same matrices would be less than $4 \times 10^{-5}$.

Additional uncertainty arises from the USEPA's use of the linearized multi-stage model for estimating the cancer slope factor for PCBs. The linearized multi-stage model is one mathematical model which may describe the carcinogenic process. The model assumes that at low doses, there is a linear relationship between the dose of a chemical and the excess cancer risk. In addition, the model assumes that there is no threshold dose for the induction of cancer. However, there are many instances in which this assumption may not hold true for the carcinogenic process. Examples of such effects are chemicals that cause an increased high dose cancer incidence due to stimulated cell proliferation rates (Cohen and Ellwein 1990, Ames and Gold 1990), chemicals for which metabolic parameters limit the delivered dose to the target organ (discussed in Zeisse 1987), promoters, or in instances where DNA repair mechanisms may be dose dependant (discussed in Zeisse 1987). In these instances, the linearized multi-stage model may provide a "plausible upper limit" of potential cancer risk (USEPA 1986). However, as acknowledged by USEPA 1986, "such an estimate does not necessarily give a realistic prediction of the risk. The true value of the risk is unknown, and may be as low as zero". The basic toxicologic effects of PCBs, (i.e. induction of cell injury at high doses, no significant mutagenic effects) suggest that the linearized multistage model may result in the overestimation of risks from low level exposures to PCBs.

### 7.1.10. Discussion and Conclusions

The results of the HHRA calculations suggest that subsurface soil exposure to unprotected construction workers would be outside the range of acceptability established by USEPA, primarily due to PCB exposure. Implementation of appropriate health and safety protective measures at the time of soil excavation would mitigate this potential risk.

For current residents, the potential risks associated with household use of spring water are within reference levels of acceptability for cancer risks, as well as non cancer risks. At the recommendation of the USEPA, and since the levels of specific analytes in spring water exceed Federal MCLs, Amphenol Corporation has, however, provided water treatment measures for the spring water.

For future residents in the North Area who could possibly use the ground water as a source of potable water in the future, cancer and non cancer regulatory risk levels outside the range of acceptability defined by USEPA were calculated. Thus, in the event that North Area ground water is used as a source of potable water, the ground water would require treatment prior to such usage.

The RME calculations present the risk indices for adolescent recreators (the group of individuals at the greatest risk according to the hypothesized exposure scenarios provided in the HHRA). The RME derived risk for non cancer risks and cancer risks exceed regulatory reference levels and are largely associated with exposure of the adolescent recreator to sediments and landfill surface soils.

The Monte Carlo simulation, used to analyze the central tendencies associated with the exposure of an adolescent recreator to PCBs in soil, sediment, and surface water, suggests a moderate to high degree of uncertainty in the RME calculations. This statement is further supported by the degree of uncertainty of the RME calculations involving the use of reference doses for Aroclor 1254 and cancer slope factors for Aroclor 1254, which although similar in structure to Aroclor 1248, would be expected to drive the derived risk values upward. Quantitation of the uncertainty, using a Monte Carlo approach, presents a non cancer risk value to the adolescent recreator substantially less than RME values. The calculated HQ is less than 4.3. Monte Carlo simulations of the cancer risks present a cancer risk of less than $4 \times 10^{-5}$ to 95% of the population, which is within the risk range considered acceptable by USEPA (NCP 1990).

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## 7.2. Ecological risk assessment

### 7.2.1. Introduction

This section of the RI Report is an Ecological Risk Assessment (ERA) of conditions related to the RHRMLS. This ERA was designed and conducted in accordance with USEPA methodologies presented in the following documents: *Risk Assessment Guidance for Superfund Volume II Environmental Evaluation Manual* (USEPA, 1989a), *Ecological Assessment of Hazardous Waste Sites: A Field and Laboratory Reference* (USEPA, 1989b), *Ecological Assessment of Superfund Sites: An Overview* (USEPA, 1991), *Framework for Ecological Risk Assessment* (USEPA 1992a), and *Developing a Work Scope for Ecological Assessments* (USEPA, 1992c). The ERA was performed in accordance with the ERA Work Plan (O'Brien & Gere, 1995), prepared in consultation with and approval by USEPA.

Responses of the South Pond ecosystem to releases from the RHRMLS could not be directly correlated to or quantified through direct field methods. Due to the physical disturbances to the South Pond ecosystem that occurred in the Fall of 1994, related to a USEPA Unilateral Administrative Order (UAO) which included the temporary drainage and removal of sediments from the South Pond, quantitative community comparisons could not be used in this ERA . Therefore, a hazard quotient (HQ) model was used as a theoretical approach to estimate the potential impact of residuals remaining following the Fall 1994 remedial measures to the ecological receptors of the South Pond and downstream areas.

The ERA utilizes a quantitative, receptor-model approach where uptake of compounds of potential concern (COPCs) by representative wildlife species are modeled for comparison to literature-derived toxicity values in a hazard quotient. The theoretical approach was complemented by field observations, which assisted in the formulation of conclusions regarding potential ecological impacts.

Consistent with the USEPA guidance documents, this ERA is organized and presented as six tasks: Problem Formulation, Data Collection, Exposure Assessment, Ecological Effects Assessment, Risk Characterization and Conclusions.

- **Task 1 - Problem Formulation** presents a description of the RHRMLS, a summary of existing information, and defines the objectives of the ERA.

- **Task 2 - Data Collection** summarizes field observations and presents the analytical results of media samples and toxicity tests.

- **Task 3 - Exposure Assessment** identifies the COPCs and links the contaminant fate and transport characteristics to ecological receptors to develop complete exposure pathways.

- **Task 4 - Ecological Effects Assessment** involves the evaluation of literature or toxicity testing derived effects of contaminant exposures on ecological receptors.

- **Task 5 - Risk Characterization** presents the interpretation of the results of the ERA tasks and the calculation of risk utilizing the HQ methodology.

- **Task 6 - Conclusions** presents the interpretation of the results of the ERA.

Detailed discussions of the approach and results of the five tasks are presented in the following sections.

### 7.2.2. Task 1 - Problem formulation

Scoping of the ERA consisted of the preparation of a detailed Conceptual Site Model and the design of a sampling and analysis plan to address the potential ecological concerns of the RHRMLS. The USEPA-approved ERA Work Plan is presented as Appendix O, and should be referred to for additional scoping information. Detailed information regarding the history of the RHRMLS is presented in Section 1.2.2 of this RI Report. This section of the ERA presents pertinent elements of that site information within the context of the primary question: Does the site represent a significant impact on local habitat and wildlife resources?

During RI scoping, the RHRMLS was identified as warranting further evaluation within the ERA framework. The site is in a sparsely populated rural area. Most of the land in the vicinity of the RHRMLS is characterized by steep topography and mixed-hardwood woodlands. Surface water bodies in the immediate vicinity of the RHRMLS consist of two shallow ponds, the North Pond and the South Pond; a smaller pond at the outlet of the South Pond (the outlet pond); and a small stream running south from the outlet of the outlet pond (Figure 36). The ponds support fish, amphibians, reptiles, aquatic furbearers and migratory waterfowl. The South Pond is relatively shallow (1 to 4 ft) and supports only one, non-game, fish species: fathead minnow. The North Pond is deeper than the South Pond and supports a greater diversity of fish species than the South Pond (USEPA, 1993a). Water from the South Pond drains through a series of connected ponds and wetland areas, south into Herrick Hollow Creek, Trout Creek, and eventually into the Cannonsville Reservoir, approximately 15 miles downstream of the South Pond.

The RHRMLS also includes the North Area, which is situated hydraulically upgradient (west) of the North Pond. The Sidney Landfill is located on the opposite side of the valley from the North Area (east). The Sidney Landfill is hydraulically upgradient of, and drains into, the North Pond. A Remedial Investigation of the Sidney Landfill was performed by Malcolm Pirnie, a USEPA subcontractor, in 1995. The USEPA-approved RI report concluded that there were no risks to ecological receptors of the North Pond associated with contaminants being released into and present in the North Pond. Therefore, an evaluation of potential effects of the North Area on the North Pond was not performed. However, the Malcolm-Pirnie report was used for reference.

Data collected under O'Brien & Gere's Remedial Investigation and ERA activities were utilized for quantitative calculations presented in this report. Data collected by USEPA's Emergency Response Team (ERT) were used to identify target sampling points and media. However, ERT data was not used in the ERA due to the lack of QA/QC information required for risk assessment purposes.

As presented in Section 4 of this RI Report, components of materials within the RHRMLS landfill have been detected in the South Pond and Herrick Hollow Creek. As described in the introduction, a remedial program was conducted in the Fall of 1994, which established a barrier between the landfill and the South Pond, and removed pond sediments. The objective of this ERA was, therefore, to consider post-remedial conditions with respect to potential impacts.

Based on the RI and ERA analytical data review, potential COPCs for the site consist of VOCs, PCBs, and inorganic compounds that had been detected in surface water, sediment, and soil. Site-related exposures by ecological receptors evaluated included exposures to: 1) surface soils in portions of the landfill, 2) surface water and sediments in the South Pond, 3) surface water and sediments in Herrick Hollow Creek, and 4) through the food chain.

Questions considered within this ERA were:

- Does exposure to RHRMLS releases to surface water of the South Pond have the potential to affect fish?
- What is the bioavailability of PCBs in the South Pond to fish?
- Does exposure to site-related chemicals in sediments of the South Pond have the potential to affect aquatic invertebrates?
- Does consumption of aquatic biota by piscivorous (fish-eating) wildlife represent a potential food chain impact?
- Does consumption of terrestrial invertebrates represent a potential food chain impact to terrestrial wildlife?

The following methods were utilized to address these questions,:

- Chronic toxicity tests were performed using South Pond surface water and larval fathead minnows
- An *in-situ* PCB bioaccumulation study was performed in the South Pond using caged fish,
- A chronic toxicity test was performed using South Pond sediments and Chironomid larvae,
- Resident aquatic and terrestrial biota were collected and analyzed to estimate exposures for hazard quotient modeling of the food chain.

### 7.2.3. Task 2 - Data collection

Data collection activities conducted in support of the ERA are discussed below.

***Site characterization***

The physical, chemical, and ecological characteristics of the RHRMLS were identified through field reconnaissance, map review, literature review, and consultation with state and federal agencies. The focus of this task was to identify potential ecological receptors in the RHRMLS and to characterize the ecological communities in the vicinity of the site.

**Site reconnaissance**



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Site reconnaissance events were performed by O'Brien & Gere biologists in the spring and summer of 1995. The reconnaissance events were performed to identify potential ecological receptors and to prepare the covertype map for the site.

**Ecological receptor survey**

The objective of this subtask was to identify wildlife species that inhabit the study area that could be exposed to site-related compounds. Ecological receptors were identified during the site reconnaissance by traversing the site and recording actual sightings; observations of wildlife indicators such as nesting places, burrows, tracks, scat or browse; or audible indicators such as bird songs. Terrestrial surveys were conducted by walking transects adjacent to both the site and a control area (Discussed in Section 7.4.2) at the following dates and times:

| | | |
|---|---|---|
| Dawn: | 04:30 - 06:00 | 6/30/95 |
| Midday: | 10:00 - 14:00 | 6/19-23/95 |
| Dusk: | 20:00 - 21:30 | 6/29/95 |
| Night: | 22:00 - 23:30 | 6/29/95 |

Tables 62 and 63 present the observed wildlife for the both the study area and control pond and vicinity, respectively.

Aquatic vertebrate surveys were performed by seining portions of the South Pond, outlet pond, and a control pond. The sampling effort required to collect aquatic biota samples for analysis was recorded to provide an estimate of population size. The results of aquatic surveys are presented in Table 64.

**Covertype analysis**

Vegetative communities of the areas surrounding the ponds were characterized for the development of a covertype map for the RHRMLS. The covertype map includes the area within 0.5 miles of the RHRMLS and illustrates the types, locations, and relative sizes of the available wildlife habitat in the area. The Covertype Map is presented as Figure 37.

Covertypes were classified in accordance with *Ecological Communities of New York State* (NYSDEC, 1990). A "covertype" is defined as an area characterized by a distinct pattern of natural (e.g. forest) or cultural (e.g. residential) land use. Covertype designations were applied based on the dominant vegetation and other physical features observed during the study area reconnaissance (6/27/95, 6/28/95, and 7/25/95) and from the site aerial



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photographs (dated 5/16/87). A list of dominant woody and herbaceous plant species observed during the field reconnaissance is presented in Table 65.

The covertypes within the study area consist of *terrestrial, palustrine, riverine, and lacustrine* covertypes. Based on Natural Heritage Program State and Global rankings (NYSDEC 1990), none of the covertypes within the study area represent rare ecological communities. A total of ten habitat covertypes were identified. Six of the eleven identified covertypes are considered natural covertypes and five are considered cultural covertypes (NYSDEC 1990). The identified covertypes of the study area are discussed below.

## Terrestrial Covertypes

Mixed Hardwood Forest.
Mixed hardwood is the dominant covertype for the RHRMLS study area. This covertype comprises three forest habitats: beech-maple mesic forest, successional northern hardwoods, and hemlock-northern hardwood. Typical species found in this covertype are: maple, beech, oak, poplars, cherry, ash, birch, hickory, and elm. Hemlock, pine, and spruce are found sporadically throughout this covertype.

Successional Old Field/Successional Shrubland.
Abandoned croplands and pastures of grasses and shrubs were identified within the study area. These covertypes were either characterized as successional old field or successional shrubland, depending on the percent cover of shrub/tree species. Areas with less than 50 percent shrub cover were considered successional old field. Typical species of the successional old field covertype include: goldenrod, aster, hawkweed, rose, sumac, raspberries, and dogwood.

Areas with more than 50 percent shrub cover were considered successional shrubland. Within the study area, this covertype is usually found within close proximity to successional old field. Species found in successional shrubland are similar to those found in successional old field. Successional shrublands tend to have a higher frequency of tree species ranging from saplings to mature specimens. Tree species associated with this covertype within the study area include maples, ash, oak, cherry, pine, and hawthorn.

Cropland/Field Crops.
Although a majority of the successional old field and successional shrubland covertypes may have been historically cultivated, currently, only

one area within the RHRMLS study area was in active cultivation. This area was a hayfield.

Pine Plantation.
Six areas in the Study Area are considered pine plantation. These even-aged stands of pine were planted in symmetrical rows.

Mowed Pathway.
The study area is bisected by a New York State Electric and Gas (NYSEG) overhead electric powerline right-of-way (ROW). Within the study area, the vegetative community of the NYSEG ROW varies from grasses and forbs to scrub shrub. Typical species found in these vegetative communities include: grasses, goldenrods, hawkweeds, ferns, rose, blackberry, sumac, dogwood, ash, oak, and maple.

Landfill/Dump.
Five areas on the covertype map are noted as landfill/dump. These areas include the RHRMLS South Area, the RHRMLS North Area, and three areas associated with the Town of Sidney Landfill. The RHRMLS South Area is currently void of vegetation as a result of landfill capping activities. The majority of the other four landfill/dump areas resemble successional old field and successional shrubland type habitats.

Other Cultural.
NYSDEC 1990 habitat descriptions used to classify this covertype include mowed lawn, mowed lawn with trees, paved road/path, unpaved road/path, and rural structure exterior. This covertype is characterized by vegetative communities that have been extensively manipulated and are currently maintained for human use. The covertype is represented by residences and their associated yards and driveways, Richardson Hill Road, and farm/access roads. Vegetation associated with this covertype include native and ornamental shrubs and trees, maintained lawn grasses, and maintained roadside vegetation.

<u>Palustrine, Lacustrine, and Riverine Systems</u>
Based on the wetland classification system derived by the USFWS (Cowardin et al., 1980), wetland habitat types within the study area include: palustrine emergent; palustrine scrub-shrub; and riverine upper perennial. Descriptions of these systems are provided below.

Palustrine Systems.

Palustrine wetland covertypes of the RHRMLS study area consist of: shrub swamp, shallow emergent marsh, sedge meadow, and deep emergent marsh. Typical species found in these wetlands include: maple, ash, spirea, dogwood, willow, sedges, bulrushes, cattail, goldenrod, purple loosestrife, reeds, and duckweed. Site wetlands are further discussed in Section 4.6.

Lacustrine Systems.
Within 2 miles of the site there are approximately 116 acres of open water ponds. Areas of open water within 0.5 miles of the site consist of the South and North Ponds, a pond within the NYSEG ROW, and three man-made farm ponds. These open water areas are listed on the US Fish and Wildlife Service's NWI maps (Figure 38) as palustrine emergent wetlands with unconsolidated bottoms; however, for the purposes of this covertype analysis, these areas are classified as open water.

Riverine Systems.
Within 2 miles of the site there are approximately 14 miles of flowing water streams. Two riverine systems are found within the RHRMLS study area: marsh headwater stream and ditch/artificial intermittent stream. Both the North and South Ponds are part of headwater systems for the Susquehanna and Delaware River Drainage Basins, respectively. A drainage basin divide occurs within the study area. These headwater streams are listed on the NWI maps (Figure 38).

The North Pond outlet is designated as a palustrine emergent habitat. Approximately 300 ft downstream (north) of the study area boundary, the North Pond outlet designation changes to riverine upper perennial (unconsolidated bottom) habitat. This stream is tributary to Carrs Creek approximately two miles to the north.

The South Pond and its outlet are part of the headwaters of Herrick Hollow Creek, which is tributary to Trout Creek, a riverine upper perennial (unconsolidated bottom) habitat. There are also two artificial ditches within the study area that are associated with farm ponds. These ditches flow towards Herrick Hollow Creek.

**Review of available literature**
State and federal agencies were contacted for information regarding the presence of special resources in the vicinity of the RHRMLS. According to USEPA (1989a), special resources include environmentally sensitive areas; significant habitats; rare, threatened or endangered species (RTE); regulated wetlands; and federally classified wild and scenic streams, lakes and rivers.

The presence of special resources within 2 miles of the RHRMLS was evaluated through literature searches and in consultation with the New York State Department of Environmental Conservation (NYSDEC) and the U.S. Fish and Wildlife Service. Terrestrial features and natural waterways were inventoried through review of wetland, soil and topographic maps, and research of soil and surface water classifications. The presence of special resources within the area of inquiry was further evaluated by performing the qualitative vegetation and wildlife censusing discussed above. The environmentally sensitive areas identified within the Study Area consisted of wetlands and streams, as discussed in the following sections.

Threatened or endangered species

Neither the Natural Heritage Program of the NYSDEC nor the U.S. Fish and Wildlife Service identified any threatened, endangered, or species of special concern within the project area (NYSDEC 1995; USFWS 1995).

Fish populations

The New York State Department of Environmental Conservation (DEC) was contacted regarding the presence of fish species in the surface waters within two miles of the site. The DEC stated that, although no fish collection data were available for review, brook trout and brown trout were the primary sport fish that inhabited typical streams of the region. The most common non-trout species include; white sucker, blacknose dace, common shiner, creek chub and fallfish (NYSDEC, 1995). Two of these species, blacknose dace and creek chub, were observed in the control pond during the field sampling activities. No trout species were observed in either the control pond or the South Pond.

Regulated wetlands

State Regulated Wetlands.

The presence of NYS regulated freshwater wetlands in the study area was identified through a review of NYS Freshwater Wetland Maps (Figure 39). No NYS regulated wetlands are located within the 0.5 miles of the site. One NYS Regulated Wetland is within 2 miles of the RHRMLS. This wetland, designated as FR-4 by the NYSDEC, is approximately 1.3 miles northeast of the RHRMLS. This wetland is associated with a tributary and sub-tributary of Carr Creek. The distance between the site and Wetland FR-4 and the movement of surface and ground water from RHRMLS to the south, indicate that the wetland could not be impacted by site-related contaminants.

Federally Regulated Wetlands.

Jurisdictional wetland boundary delineations were performed for the RHRMLS South and North Pond areas. In October 1990, a wetland delineation was performed by O'Brien & Gere Engineers on the area immediately adjacent to the South Pond. This delineation was completed in accordance with the *Federal Manual for Identifying and Delineating Wetlands*, 1989. The 1989 Manual presented the federally approved delineation methodology at the time of the delineation. The delineated wetland boundaries are presented on Figure 35.

The presence of USFWS NWI wetland and deepwater habitats in the study area was identified through a review of NWI maps for the study area (Figure 38). Although these maps assist in the preliminary identification of regulated wetlands, they do not represent federally regulated wetlands in the United States. NWI wetland habitat types within the 0.5 mile of the site include: palustrine emergent, forested, scrub-shrub, and riverine (Figure 38).

<u>Regulated streams and navigable waterbodies</u>
Regulated streams and navigable water bodies were identified through United States Geological Survey (USGS) topographic map interpretation and a review of New York Code of Rules and Regulations 6 NYCRR Part 875.

Herrick Hollow Creek, which originates from the South Pond, is listed as a Class C, C(T) stream. The "C" classification indicates the waters are suitable for fishing and fish propagation, and primary and secondary contact recreation. The "T" designation indicates suitability for trout populations. Trout Creek, which Herrick Hollow Creek is tributary to, is a Class C, C(TS) stream. The "S" designation indicates suitability for trout spawning. Trout Creek empties into the Cannonsville Reservoir, approximately 15 miles downstream of the South Pond.

*Identification of reference and control ponds*
Reference ponds and a control pond were identified for the RHRMLS through visual and chemical interpretation. Five reference ponds were identified and sampled to provide a range of inorganic reference concentrations to be used to identify COPCs for the ERA. One of these ponds was selected to serve as a control for fish and wildlife population comparisons, biological tissue concentration comparisons, surface water and sediment bioassays, and for the caged fish study. USEPA approved the selected control and reference ponds prior to the initiation of field activities (USEPA 1995).

**Selection methodology and data gathering efforts**

Aerial photographs and regional maps covering the vicinity of the RHRMLS were reviewed to identify potential reference ponds based on similarity in size, proximity to the site, and potential for impact from the site. On May 16, 1995, several potential reference ponds, were visited and evaluated for similarity to the South Pond. Five of these ponds designated as sites 1 through 5, were selected as reference sites for further evaluation. Figure 40 presents the location of the sites with respect to the RHRMLS. Data were collected from each site regarding its physical and biological characteristics and samples of surface water and sediment were collected for inorganic analyses. Summaries of the physical and biological characteristics of the reference sites are presented in Table 66. Analytical results of the surface water and sediment samples from the ponds are summarized in Tables 67 and 68, respectively.

**Control pond selection**

Of the five reference sites, Site 4 was selected as the control pond. Site 4 is located on Pine Swamp Road approximately 1 mile east of South Pond. This pond was formed by a series of beaver dams on a small headwater stream that originates just north of the northernmost pond and is tributary to Trout Creek (See Figure 40). This water body is outside the influence of the Sidney landfill as evidenced by the analytical results and by its location, 0.5 miles east of the opposite side of the Sidney Landfill hill. The pond exhibited similar water quality parameters, depth, and size. Its location near the headwater of a small stream and its origin as a result of beaver activity compare favorably to conditions at the South Pond.

**Reference media concentrations**

Reference pond surface water and sediment inorganic compound concentrations were used to establish background concentrations for comparison to site media concentrations to identify elevated detections. Site concentrations that exceeded the range of detections in the reference ponds were further evaluated in the ecological assessment as COPCs.

Surface water and sediment samples were collected from the five reference locations . The analytical results of four of the samples compared well to each other. The analytical results of the sediment sample from one site, Site 5 (located along Route 88), were generally an order of magnitude higher than the other four locations (Table 68). The concentration differences could be attributed to the proximity to a major roadway, which is likely contributing road salt and runoff to the pond. It should be noted that the South Pond is bounded to the west by Richardson Hill Road, a maintained roadway. The analytical results of Site 5 provide an indication of the range of inorganic concentrations that are observed in the environment as a result of anthropogenic activities. Therefore, only the results of the samples from Sites 1, 2, 3, and 4 were combined to develop the range of inorganic background concentrations. However, the analytical results from Site 5 were retained as a plausible upper bound of inorganic environmental concentrations. The ranges of background concentrations are presented in Table 69.

**Additional control pond activities**
The selected control pond was also the subject of the following assessments:

- Qualitative vegetation survey
- Qualitative wildlife survey during crepuscular, diurnal and nocturnal periods
- Organic analyses of surface water and sediments
- Surface water bioassay
- Caged fish study
- Fish sampling and analysis for inorganics, volatiles and PCBs.

*Environmental media sampling*
Environmental media samples were collected under ERA activities to evaluate the detected concentrations for 1) toxicity evaluations, 2) criteria comparisons, and 3) to assist in interpreting study results. Table 70 presents a summary of the numbers of samples collected of each medium and the analyses performed on each. Sampling locations for surface water, sediment, and biota samples from the South Pond and outlet pond are presented on Figure 41. Figure 42 presents the surface soil and earthworm sample locations. Sediment sample locations for the downstream characterization effort are presented in Figure 43.

**Surface water and sediments**
Surface water and sediment samples were collected from the following locations:

- the western (SW-02, SED-02), southern (SW-03, SED-03), and eastern (SW-04, SED-04) portions of the South Pond to characterize exposure concentrations and for surface water and sediment bioassays.

- the outlet pond immediately downstream of the South Pond (SW-05, SED-05) for chemical analysis to evaluate bioconcentration in downstream fish,

- the first encountered pond downstream of the outlet pond that is capable of supporting fish (SW-06, SED-06) to evaluate bioconcentration in downstream fish,

- the control pond (SW-01, SED-01) for toxicity testing and chemical analysis,

As part of downstream characterization efforts, performed in May 1996, 59 sediment samples were collected from 21 segments of Herrick Hollow Creek from the South Pond to Trout Creek. Details of the downstream characterization effort are presented in the Downstream Characterization Report (Appendix P).

In-field water quality measurements (pH, temperature, dissolved oxygen, and specific conductance) were collected at the time of surface water sample collections to evaluate the ability of the pond to support aquatic life. Water quality data are presented in Table 71.

Surface water and sediment samples were analyzed for inorganics (CLP SOW ILM02.1), volatile organics (EPA Method 8240) and PCBs (EPA Method 8080). Surface water samples were also analyzed for total alkalinity (Method 310.1). Analytical results for surface water and sediment samples are presented in Tables 72 and 73, respectively. Explanations of data qualifiers used in the data tables are presented in Table 72d. Sediment samples were collected from the 0 to 6 inch interval and were analyzed for total organic carbon (EPA Method 415.1) (See Table 73a) for sediment criteria calculations, and grain size (ASTM D422, D1140) (See Table 74) to provide a physical description of sediments. Sediment samples collected as part of the downstream characterization effort were analyzed for PCBs (EPA Method 8080), total organic carbon (SW846 Method 9060), and percent solids (ASTM Method D2216). The results of these analyses are presented in Table 73d. Sample collection and handling methods are presented in the approved Site Operations Plan (O'Brien & Gere, 1988) for the RHRMLS.

**Aquatic vertebrates**

Four adult aquatic vertebrate samples were targeted for collection from each of the following locations:

- The South Pond - to evaluate vertebrate body burdens and exposure concentrations for carnivorous wildlife. Adult vertebrate samples from the South Pond consisted of composite samples of red-spotted newt (NEWTA-01), fathead minnow (FHMA-01), and green frog (FROGA-01).

- The outlet pond immediately downstream of the South Pond - to evaluate uptake of COPCs by off-site aquatic vertebrates. Adult vertebrate samples from the Outlet Pond consisted of composite samples of red-spotted newt (NEWTA-02), fathead minnow (FISHA-02), and green frog (FROGA-02).

- The first encountered pond downstream of the outlet pond that is capable of supporting fish - to evaluate uptake of COPCs by off-site aquatic vertebrates. Adult vertebrate samples from the downstream pond consisted of composite samples of red-spotted newt (NEWTA-04), fathead minnow (FHMA-04), and green frog (FROGA-04).

- The control pond - to establish background COPC concentrations of biota in the region. Adult vertebrate samples from the control pond consisted of composite samples of red-spotted newt (NEWTA-03), fathead minnow (FHMA-03), and green frog (FROGA-03).

In addition to adult vertebrate samples, four young-of-the-year (YOY) vertebrate samples were targeted for collection from each of the following locations:

- The South Pond - to evaluate body burdens resulting from parental inheritance and exposures under current conditions. Larval vertebrate samples from the South Pond consisted of composite samples of spotted salamander (NEWTY-01), fathead minnow (FHMY-01, FHMY-02), and tadpole (FROGY-01).

- The outlet pond immediately downstream of the South Pond - to evaluate exposures under current conditions. Larval vertebrate samples from the Outlet Pond consisted of composite samples of tadpole (FROGY-02). No other larval vertebrates were observed.



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- The control pond for comparison purposes. Larval salamanders and fathead minnows were not present in sufficient quantities for sampling.

The objectives of evaluating site-specific biota COPC concentrations were to determine if exposure and uptake are occurring in resident biota, and to identify the COPC body burdens in resident biota for evaluations of impacts to higher trophic level organisms. Aquatic receptors may be exposed to site-related COPCs by contact and uptake from surface water and sediments. Dietary exposure may occur directly through ingestion of soil, water, sediment, or lower food chain organisms. Biota can bioconcentrate (the process of concentrating a chemical in tissue from the uptake of non-living substrate i.e. fish: water) some chemicals above their background levels in the environment and can accumulate chemicals via dietary uptake of biota. In contrast, the situation where chemical levels increase in the tissue of organisms by several orders of magnitude for each trophic level of a food chain is called biomagnification.

Fish sampling was performed primarily with seines after electroshocking efforts proved unsuccessful due to the small size of the fish in the pond. The sampling effort and duration were recorded as an indicator of population status in each sampled water body. The physical condition of collected fish was evaluated at the time of collection, noting physical abnormalities, such as tumors or lesions. Biota samples were shipped on ice to the laboratory for whole body analysis for inorganics, volatiles, PCBs, and percent lipid.

Analytical results of biota samples are presented in Table 75. No physical abnormalities were observed in the collected fish. Many of the collected fish, including those from the Control Pond, were infected with numerous black spot (*Neascus* spp.) parasites. Black spot is a common Trematode which appears as a pigmented cyst in the fish integment and is not lethal to fish (Lagler, 1956).

**Soils**

Soil samples were collected from the zero to twelve inch depth interval at the one location on the RHRMLS (Figure 42). The soil sample was collected from the earthworm sample location. The purpose of the soil sample was to evaluate the potential bioconcentration of PCBs in earthworm samples.

Three additional soil samples were collected from the vicinity of the control pond to establish reference soil concentrations for inorganic compounds. Analytical results of soil samples are presented in Table 76.

**Soil invertebrates**
One composite sample of earthworms was collected from the RHRMLS for PCB analysis using EPA Method 8080. Earthworm analytical results are presented in Table 76. The results were used to model exposures to earthworm consumers such as small mammals. During the risk characterization task (Section VI), the bioconcentration factor for the worms and soil from the sampled location were used to calculate a reference soil concentration for worm-eating wildlife.

Due to the dry summer of 1995, earthworm collection proved difficult. The drought-like conditions hardened the soil and drove worms deep into moister, cooler soil. Therefore, earthworms could only be collected on-site where they were available. The composite earthworm sample was collected by digging and hand picking.

*Toxicity testing*
Toxicity testing was performed on surface waters and sediment from the South Pond to evaluate the chronic toxicity of site media to resident fish and invertebrate species. Literature derived toxicity information may not accurately portray actual site conditions, especially when more than one compound is present. Therefore, bioassays were performed using site surface waters and sediment to estimate cumulative toxicity. The bioassay results also serve to assist in evaluating the spatial extent of contaminant effects within the South Pond.

**Surface water toxicity testing**
Three chronic bioassays were conducted utilizing fathead minnows in dilutions of surface water from three South Pond locations: the leachate discharge area from the landfill to the pond on the northwest portion of the pond; at a location directly in front of the dam where contaminants may have accumulated in the sediments; and on the eastern portion of the pond (Figure 41). A bioassay was also conducted with a surface water sample from the control pond for comparative purposes and to evaluate extraneous test-specific mortality.

The chronic toxicity test for surface water samples was performed in accordance with USEPA Test Method 1000.0 (USEPA, 1989c) utilizing larval fathead minnows, exposed for seven days in varying concentrations of pond water. Test mortalities and fish growth were used to evaluate the effects of site surface water exposures.

**Sediment toxicity testing**
Sediments of the western portion of the South Pond, adjacent to identified seep locations, were collected for sediment toxicity testing. One composite sediment sample was submitted for a 10-day chironomid survival and growth test; USEPA Method 100.2 (USEPA, 1994). A composite sediment sample from the control pond was also submitted for the 10-day toxicity test to evaluate environmental factors that could influence toxicity test results. The sediment sample collected from the western portion of the South Pond was also used to characterize sediment quality in this portion of the pond for the PCB bioaccumulation study discussed below.

*PCB accumulation study*
The objective of the PCB accumulation study was to estimate the bioavailability of residual PCBs in surface water and sediments of the South Pond and its outlet pond. This objective was accomplished through the performance of an *in situ* caged fish study utilizing fathead minnows. Fathead minnows were selected for this study because they are natural inhabitants of the pond and, as consumers of algae, bottom detritus, insect larvae, and zooplankton, they represent sediment and water column exposures.

One set of four cages were set up at three locations in the South Pond, one location in the outlet pond, and one location in the control pond (Figure 41). The cages were placed in direct contact with undisturbed sediments to evaluate PCB uptake by the fish from the water column and the sediments following the performance of UAOs, and to provide an indication of the bioavailability of PCBs in the sediment and surface water.

Adult fish of approximately 60 mm in length, obtained from a commercial supplier, were acclimated to pond water prior to being placed in the pond for the accumulation study. Approximately twenty-five fish were placed into each cage. Surface water and sediment samples were collected from each of the four study sites and analyzed for PCBs by GC method 8080 on the first day of the study. For quality control purposes, a random fish sample was collected on the first day of the study from the fish stock and submitted for PCB and percent lipid analyses to identify any initial PCB concentration.

The fish from one cage at each location were collected for PCB and percent lipid analyses on days 9, 14, 21, and 34. Water quality parameters were measured on each collection day.

### 7.2.4. Task 3 - Exposure assessment

Exposure is defined as the contact of a receptor with a chemical or physical agent. An exposure pathway is a mechanism by which a receptor may be exposed to a chemical or physical agent at or originating from a source. Exposure pathways may be classified as being **complete** or **incomplete**. An exposure pathway is complete when an ecological receptor contacts a physical or chemical agent under the site-specific conditions. The pathway is incomplete when biotic receptors are not exposed under the specified conditions. Incomplete exposure pathways are not considered further in the ERA. Complete exposure pathways were further evaluated in Task 4 (Section V) to assess the potential effects of exposure to RHRMLS receptors.

In order to evaluate chemical exposure scenarios for ecological receptors, the COPCs were identified and their environmental fate evaluated to determine the overlap of the contaminant plume with receptor habitats. Under this task, existing data generated during the RI and data collected under the ERA are coupled with environmental fate and transport information (presented in Chapter 6) to determine the exposure point contaminant concentrations for ecological receptors. The significance of the exposure point concentrations was then evaluated by comparison to appropriate regulatory criteria, standards, and guidance values. Compounds that exceed screening criteria are considered COPCs for the site. The COPCs were then linked to the ecological receptor information gathered under Task 1 to develop site-specific exposure scenarios for resident ecological receptors.

This task is subdivided into two subtasks: 1) Identification of COPCs, and 2) Exposure Pathway Analysis.

*Identification of COPCs*

The objective of this task is to differentiate site-related compounds which may be present at elevated levels from other inert, inactive, or low concentration chemicals present at the RHRMLS. The identification of COPCs is based on detected concentrations, fate and transport potential, and potential toxicity to ecological receptors. The initial list of COPCs consists of organic chemicals detected on-site or in site-related media, and inorganic chemicals detected at elevated levels on a medium-specific basis. Compounds were then eliminated from the list of COPCs based on the following considerations:

- Naturally occurring compounds were eliminated from further consideration if they are essential nutrients, were present at concentrations less than maximum background levels, or are toxic only at very high doses.

- Certain compounds were also eliminated from potential consideration if they were detected at low concentrations or below applicable media-specific criteria, standards, or guidance values.

- A toxicity concentration screening procedure was performed to eliminate from consideration those compounds that were present at concentrations that do not pose a potential risk to ecological receptors.

Elevated detections in site media were initially screened by comparisons with regulatory criteria. Maximum surface water detections were compared to New York State Water Quality Standards and Guidance Values (NYSDEC, 1991) or USEPA Ambient Water Quality Criteria (USEPA, 1992d), whichever was lower. When neither NYSDEC nor USEPA criteria were available, secondary chronic toxicity values were obtained from Suter and Mabrey (1994).

Maximum sediment concentrations were evaluated against proposed NYS sediment quality criteria (SQC) and guidance values developed by the Ontario Ministry of The Environment's (MOE) Sediment Quality Guidelines (Persuad et al., 1992). These guidelines were developed based on the Equilibrium Partition (EqP) approach that estimates contaminant concentrations in sediment that might result in pore water concentrations above the ambient water quality criteria, based strictly on technical grounds. Because the EqP assumes direct substitution of organic carbon constants with octanol water constants, the method is highly conservative and not generally scientifically accepted. However, the guidance values were used in this ERA as a preliminary screening tool. In the absence of sediment screening values from either of the above sources, sediment screening benchmarks developed by Hull and Suter (1994) were used to screen sediment COPCs.

Maximum soil detections were screened against cleanup levels presented in the New Jersey Cleanup Responsibility Act, as cited in USFWS's *Evaluating Soil Contamination* document (Beyer, 1990). The comparison with regulatory screening criteria resulted in a list of COPCs for each medium. Compounds for which neither comparison criteria nor toxicity values exist were not further evaluated in the ERA.

Although concentrations of chemicals that are below regulatory criteria indicate no further evaluation is necessary, exceedance of the conservative regulatory criteria does not necessarily indicate a potential for site-related effects. Actual effects in a natural ecosystem can be diminished or enhanced by the influences of naturally occurring site conditions. Discussions of the COPCs in surface water, sediments, and soil are presented below.

**Surface water**
Background inorganic compound levels in surface water were based on the range of detections in four surface water samples collected from four reference ponds. Maximum inorganic compound detections in site surface waters that exceeded the range of detections in the reference pond consisted of aluminum, barium, calcium, copper, iron, magnesium, manganese, mercury, potassium, sodium, and zinc.

Screening of the elevated inorganic compounds and the detected organic compounds against regulatory comparison values resulted in the identification of the following compounds as COPCs in surface water: aluminum, barium, copper, manganese, mercury, zinc, vinyl chloride, 1,1-dichloroethane, 1,2-dichloroethene, 1,1,1-trichloroethane, carbon disulfide, and PCB Aroclor 1248 (See Table 77).

Due to the prevalence of elevated levels of iron naturally occurring in the environment, a toxicity value has not been developed for iron in surface water. Therefore iron was not identified as a COPC for further evaluation using the HQ methodology.

**Sediment**
Background inorganic compound concentrations in sediments were based on the range of reference pond detections. Maximum inorganic compound detections in site sediments that exceeded the range of those detected in the reference pond consisted of aluminum, arsenic, barium, calcium, chromium, cobalt, copper, iron, lead, magnesium, manganese, nickel, potassium, and vanadium.

Screening of the elevated inorganic compounds and the detected organic compounds against regulatory comparison values resulted in the identification of the following compounds as COPCs in sediments: arsenic, barium, chromium, copper, lead, manganese, nickel, vinyl chloride, carbon disulfide, 1,1-dichloroethane, 1,2-dichloroethylene, 1,1,1-trichloroethane, toluene, PCB Aroclor 1248, and PCB Aroclor 1254 (See Table 78).

Although the detected iron concentration exceeds the MOE's sediment quality guidance of 2%, Hull and Suter (1994) conclude that 3% is a more accurate iron guidance value in sediments. Therefore, the maximum site detection of 2.4% is not considered significant and iron was not identified as a COPC.

**Soil**

Background inorganic compound concentrations in soil were based on the range of detections in three soil samples collected from the vicinity of the control pond during ERA field activities (Table 79). Maximum surface soil detections from the landfill area that exceeded background ranges consisted of cadmium, calcium, chromium, copper, lead, magnesium, manganese, nickel, and zinc.

None of the detected metals were found to exceed the NJDEP soil cleanup levels (manganese had no level established). However, the elevated soil inorganic compounds (except for the nutrients calcium and magnesium) and PCB Aroclors 1248 and 1254, were further evaluated in the ERA because of their potential to bioconcentrate and because they were detected in site earthworms.

*Exposure pathway analysis*

The concurrent processes of the Site Characterization efforts and the Fate and Transport Analysis converge in the exposure pathway analysis. Site characterization efforts identified important habitats and ecological receptors with the potential for site-related impact. The fate and transport analysis characterized the vehicle of impact. Under this task, potential exposure pathways for ecological receptors were identified and characterized based on consideration of the physical or chemical agents, their sources, release mechanisms, migration pathways, and environmental media. A summary of the exposure pathway status for the site is presented in Table 80. Receptors that can contact environmental media resulting in exposures at concentrations exceeding regulatory criteria, standards, or guidance values, or chemicals for which no comparison criteria have been established, were considered complete exposure pathways and were further evaluated in the ecological effects and risk characterization discussions.

### 7.2.5. Task 4 - Ecological effects assessment

The exposures to the COPCs for the RHRMLS by identified ecological receptors were evaluated utilizing techniques that included comparisons with the control pond, comparisons with ecological risk-based media criteria, surface water and sediment toxicity tests, and a PCB bioaccumulation study.

*Control site comparisons*

Control data were collected for comparison with activities conducted on-site to evaluate contaminant exposures and to serve as points of comparison for in-situ testing.

**Fish community structure**

Based on the similarity in structure and location of the control and South Pond, similar fish communities were expected. Although the fish sampling efforts of O'Brien & Gere were not designed to provide a quantitative evaluation of the fish community of the South Pond, the consistent use of seining techniques at the control pond and the South Pond provided an indication of the fish community at each pond.

The control pond was found to be dominated by a mixed-age sunfish population and also contained fathead minnow, creek chub and blacknose dace. Only fathead minnow were found in the South Pond (See Tables 62, 63, and 64). The reasons for the fish community differences in the two ponds can be attributed to physical or chemical differences. The age of the control pond could also influence the state of the fish community. Lotic species, such as trout, could survive in newly created ponds for only a short term if water temperatures and turbidity increase.

The South Pond fish community was impacted when the South Pond was drained during the UAOs performed in the Fall of 1994. The draining of the pond left numerous fish stranded as the water receded. It is likely that the draining had a significant effect on the fish community and that fathead minnows were the first species to re-establish.

Prior to the UAOs, USEPA's Emergency Response Team found only fathead minnow during limited sampling activities for analytical purposes in the Spring of 1993. ERT collected live and dying fish during their investigation of fish kills in the North and South Ponds (USEPA, 1993a).

ERT's limited sampling indicates that fathead minnow may have dominated the fish community prior to the pond draining.

Fathead minnow were the only fish encountered during the fish sampling activities conducted in the outlet pond and in other ponds further downstream, to the pond that intersects Richardson Hill Road approximately 0.75 miles downstream of the South Pond. This pond was larger and deeper than the South Pond. A few informal seine hauls in this pond revealed a fish community similar to the control pond with sunfish, darters, chubs, and fathead minnows.

**Other control pond uses**
Water from the control pond was used as a control for the chronic toxicity tests and the caged fish study to evaluate mortalities, or other effects, related to the test methodology.

As previously discussed, surface water, sediments, and soils from the control pond location provided reference concentrations of inorganic compounds for evaluation of elevated site detections. Results of bioassays on surface waters and sediments used as controls for the respective toxicity tests fell within the range of laboratory control results.

*Toxicity testing*
Toxicity testing, performed on surface waters and sediments of the South Pond and control pond provided reference information regarding the toxicity of site-related contaminants to test organisms in laboratory conditions. Toxicity test summary reports are presented in Appendices O and P.

**Surface water toxicity testing results**
Chronic toxicity tests using larval fathead minnows were performed on surface water samples collected from the western, southern, and eastern portions of the South Pond, and the control pond. Surface water bioassay test results are presented in Appendix Q. Of the four locations, only the water sample from the western portion of the South Pond exhibited a chronic effect and only in the 100% water sample. These results suggest a spatial variability of potential impacts within the South Pond, with observed bioassay toxicity localized within the western portion. Although, the results of the bioassay might suggest that there is a potential for toxicity in the western portion of the South Pond, numerous fathead minnow fry were observed throughout the South Pond, including the western portion, during sampling activities in the Summer of 1995. The

presence of minnow fry indicate that minnow populations in the pond are reproducing and surviving.

**Sediment toxicity testing results**
Chronic toxicity tests using a larval aquatic invertebrate were performed on South Pond and control pond sediments. The site sediments were collected from the same location on the western portion of the South Pond as the surface water and sediment samples for analysis, water toxicity test sample, and the caged fish study. Sediment toxicity test results are presented in Appendix R. The control pond test results were consistent with the laboratory control sample, having acceptable differences in mortality and weight gain.

The sample from the South Pond showed a 13% survival rate and observed weight losses in the laboratory study compared to the 95% survival in the control pond and laboratory control samples. Although an impaired invertebrate community would be expected in the South Pond based exclusively on the results of the laboratory sediment toxicity test, numerous invertebrate species, including Chironomids, were observed in sediments throughout the South Pond during sampling activities. The presence of numerous benthic invertebrates in the South Pond is an indication that a viable benthic invertebrate community exists in the South Pond and the results of the laboratory toxicity test may not be an accurate portrayal of potential impacts of sediments on invertebrate populations within the South Pond.

*Bioaccumulation studies*

**Caged fish study**
The results of the accumulation study were used to evaluate the bioavailability of PCBs in the South Pond and outlet pond based on PCB uptake by the caged fish. The results of the caged fish PCB concentrations are presented on Figure 43. As shown on the graph, caged fish PCB body burdens increased during the duration of the test. Equilibrium in the fish was not confirmed. Analytical data of PCB concentrations in surface water and sediments were correlated with fish tissue PCB concentrations collected on days 0, 9, 14, 21, and 34 of exposure in the South Pond and outlet pond (Table 81). After 34 days of confined exposure, fish accumulated approximately 9 ppm PCBs in the southern portion of the South Pond, 11 ppm in the western portion of the pond, and 13 ppm in the outlet pond. PCBs were only detected in surface water at the southern South Pond location at an estimated (below the detection limit)

concentration of 0.77 ppb. Sediment PCB concentrations in the caged fish locations were 12 ppm at the west South Pond location, 21.5 ppm at the south location, and 28 ppm in the outlet pond. The similar uptake rates at all locations indicate that PCBs in the South Pond media are bioavailable to fish.

Several difficulties were encountered at the South Pond east location, which ultimately resulted in the elimination of this location from the study. At the end of seven days, three of the four cages had been disturbed (possibly by wildlife or wind), so that the three cages were largely submerged in the sediments. This resulted in the mortality of all fish in the three cages, likely as a result of low oxygen levels. In order to continue the test at the eastern location, minnows from two cages at the control pond were transferred to the eastern portion of the South Pond. The intact cage at the eastern South Pond location was left in place to serve as the long term exposure cage. After five and fourteen days of exposure, cages (originally from the control pond) were collected from the eastern South Pond location and submitted for analysis. However, on the final sample collection day, all fish were found to be dead in the remaining cage at this location. Potential causes of mortality at the eastern location are likely related to this location being the most shallow, unvegetated location which also received direct sunlight. Mortalities are assumed to be related to low oxygen levels and are supported by the toxicity test from this location, which showed no chronic toxic effects.

The quality and overall health of the stock fish population was evaluated by maintaining a portion of the population under laboratory conditions and by performing a caged fish study in the control pond. The caged fish mortalities in the South Pond and outlet pond were compared to control pond mortalities and the mortality data from the laboratory maintained portion of the stock population. An approximate 10% mortality was observed in the laboratory control, control pond caged fish, and site caged fish locations (except for the South Pond east location discussed above), indicating a healthy fish stock.

**Young-of-the-year fish sampling**

YOY fish samples, representing a composite of fish that were exposed for up to 60 days (since time of spawning) to the South Pond media, were also analyzed for PCBs (Table 75). The South Pond YOY analytical results indicate PCB body burdens between 6.2 ppm and 8.4 ppm. These results support the conclusion of the caged fish study that the PCBs in the South Pond media are bioavailable to fish.

**Adult fish sampling**
Adult fish samples were collected from the South Pond and downstream waterbodies and analyzed for PCBs (Table 75). Analytical results of adult fish indicate PCB body burdens ranging between 5.6 ppm and 33 ppm. Adult fish from the control pond had no detectable levels of PCBs.

### 7.2.6. Task 5 - Risk characterization

*Objective*
The objective of risk characterization is to evaluate the potential effects of site-related activities or chemical exposures on ecological receptors. The potential for risks to ecological receptors through complete exposure pathways was characterized based on the results of the previous tasks. A first level of risk screening was performed during the comparison with risk-based screening levels for the environmental media. Exceedances of the criteria and/or guidance values indicate the potential for risk to aquatic life through exposure. Under this task, potential risks to ecological receptors that are not directly exposed through the environmental media but through the food chain, are evaluated by hazard quotient calculations.

*Hazard quotient calculations*
Hazard quotient (HQ) calculations are screening techniques often used as a quantitative component of ecological risk assessment. The hazard quotient method evaluates potential risk to wildlife by comparing estimated total daily intakes of COPCs from environmental media to toxicologic end points (Barnthouse et al., 1986). The quotient is expressed as a unitless ratio of total daily intake of chemical compound (mg/kg/day) to the No Observed Effect Level (NOEL) or the Lowest Observed Adverse Effects Level (LOAEL) (mg/kg/day) for a receptor organism. HQ results of less than 1 indicate no risk to the modeled organism; HQ results between 1 and 10 are considered low risk levels; HQ results between 10 and 100 are moderate risk levels; and HQ results greater than 100 are considered high risk.

Total daily intake of a chemical is the amount of chemical consumed through forage, water consumption and incidental soil or sediment ingestion. Forage types are specific to the modeled receptor and could include small mammals, fish, amphibians, reptiles, invertebrates, and/or vegetation. Exposure through ingestion of forage is the primary exposure pathway for higher food chain organisms due to the larger percentage in their diet.

The total daily intake (TDI) of a COPC by a receptor is a function of several factors including the COPC concentrations in the ingested media, the daily ingestion rate of the modeled organism; the amount of time a receptor forages on-site (the area use factor (AUF)), and the receptor's bodyweight. The following equation was used to calculate the daily intake of each COPC in each dietary component to the modeled receptors:

$$DI_{forage} = [CONCN]_{forage} \times \%d \times AUF \times DIR \times 1/BW$$

| | | |
|---|---|---|
| Where DI | = | the daily intake of the COPC through forage expressed in mg/kg/day. |
| [CONCN] | = | the COPC concentration in the medium in mg/kg. |
| %d | = | the % of diet composed of the specific medium |
| AUF | = | the area use factor (unitless) |
| DIR | = | the dietary ingestion rate expressed in kg/day. |
| BW | = | the body weight in kg, which is used to normalize the daily intake on a unit mass basis. |

The total daily intake of a COPC is calculated by adding the daily intakes of the COPC through each applicable intake medium (fish, reptiles/amphibians, invertebrates, water, and soil/sediments) based on the receptor's ingestion rate and diet percentages.

Hazard quotients for each COPC for each receptor were derived by dividing the TDI of each COPC by the literature derived No Observed Adverse Effect Level (NOAEL) for the specific receptor as presented in the equation below:

$$\text{Hazard Quotient} = \frac{\text{Total Daily Intake(mg/kg/day)}}{\text{NOAEL (mg/kg/day)}}$$

**Modeled Species**

The species selected to evaluate potential exposures of site releases to ecological receptors through the aquatic food chain were the great blue heron and the mink. The aquatic food chain modeled for these species assumes uptake through ingestion of surface water, sediments, aquatic invertebrates, reptiles/amphibians and fish. The species selected to evaluate exposures of site releases to ecological receptors through the terrestrial food chain was the deer mouse. The terrestrial food chain modeled for the mouse assumes uptake through ingestion of surface water, soil, terrestrial invertebrates (worms), and plants. The following sections briefly describe the life histories of the modeled species.

Mink Life History

The mink is a small nocturnal mammal that lives along rivers, creeks, lakes, ponds, and marshes. The mink may either use old muskrat burrows, abandoned beaver dens, or hollow logs for dens, or it may dig out its own den along streambanks (Whitaker, 1980). Mink hunt in and along the water and kill their prey by biting the neck (Whitaker, 1980). The preferred food of mink is muskrats, but they also eat fish, frogs, young snapping turtles, snakes, small mammals, and marsh-dwelling birds. Mink may eat on the spot or cache excess prey in their den (Whitaker, 1980). The home range of the mink has been estimated to range between 8 ha and 20 ha in Montana rivers and from 1 km to 5 km along streams in Sweden (USEPA, 1993b).

Although no mink were observed during the wildlife survey of the study area, the study area habitat is suitable for mink. Sightings of mink downstream of the South Pond have been reported by area residents. The mink is believed to be one of the most sensitive mammals to PCB exposures. Selection of mink as a reference mammal therefore provides for a very conservative evaluation of mammalian risk. For this reason, the mink is preferred by regulatory agencies for PCB exposure modeling.

Great Blue Heron Ecology

Great blue heron are the most widespread of all the North American herons. They live in both salt water and fresh water and tend to frequent shallow waters of lakeshores, ponds, bays, oceans, marshes, tidal flats, sandbars, and streams (Terres, 1980). The great blue heron perch and nest in trees, but spend the majority of time ashore or in shallow water. The great blue heron tends to be solitary except during breeding when it nests communally in heronries in the tops of tall cypresses and pines (Terres, 1980).

Great blue heron are reported to forage up to 24 km from its nest site (USEPA, 1993b), most actively fish just before dawn and dusk. They fish either by standing in shallow water and waiting for prey to come within striking distance of their sharp bill or can come from perch or flight and drop into deep water to strike at schools of fish (Terres, 1980). Small fish are swallowed whole, while large fish are speared.

Heron may also hunt away from the water in meadows and fields. Besides fish, the great blue heron will eat frogs, salamanders, snakes, shrimp, crabs, crayfish, grasshoppers, dragonflies, aquatic insects, small mammals, and occasionally rails and phalarope (Terres, 1980).

Great blue heron were observed feeding in the South Pond during the wildlife survey. Their consumption of fish and amphibians make their dietary exposures representative of a variety of piscivorous birds that could frequent the study area.

Deer Mouse Life History

The deer mouse is a small nocturnal mammal that can be found in a variety of habitats within its range, including conifer forests, mixed forests, field borders and stone walls (Degraaf and Rudis, 1987). Nuts, seeds, grains, fruits, mushrooms, worms, snails, insect larvae, and occasionally carrion are food for the deer mouse (Degraaf and Rudis, 1987). It caches food during the fall for winter use when the deer mouse remains active. The home range of the deer mouse is reported to be 0.014 ha and 0.01 ha (USEPA, 1993b). Deer mouse are torpid during the peak winter season.

The deer mouse was selected to represent exposures to small mammals in the study area. Although the deer mouse was not observed during the wildlife survey of the study area, it is a likely inhabitant. Its varied diet provides a representation of several dietary exposures.

**HQ Input Values**

HQ input values for ingestion rate, body weight, home range, and diet percentages for each receptor species were obtained from published literature. These values are presented in Table 82. Brief discussions of the AUF, forage concentrations, and selected toxicity values are presented in the following sections.

Area Use Factor

An Area Use Factor (AUF) provides an indication of the amount of time an ecological receptor would be expected to spend foraging in contaminated media of the site based on the receptors life history. Different receptors have different feeding patterns, forage bases, and territory sizes. An AUF of 1 indicates that a receptor spends all of its life foraging in site media, which is inappropriate for organisms with large home ranges or feeding territories. In reality, wildlife foraging strives for minimal energy expenditures for maximum food return. Therefore, receptors favor those areas that provide a plentiful and accessible food source. Once an area requires more energy to obtain food than the organism is getting from the food, it will move on to a better foraging area. The South Pond was found to contain relatively few fish and mostly very small minnows which would not make the pond or the downstream areas attractive to piscivores such as the great blue heron and the mink.

To best represent the amount of time a receptor would spend on the site, the amount of available surface water, in the form of ponds and streams, within a 2-mile radius of the site was identified from National Wetland Inventory (NWI) Maps. One hundred and sixteen acres of ponded water and 19 miles of stream were identified as a result of this research. (It should also be noted that the South Pond and the downstream beaver ponds were not present on the NWI maps because the ponds were not present at the time of the mapping. Beaver ponds created on streams throughout the region since the preparation of the NWI maps, would create even more available habitat than identified from the mapping).

The total acreage of the South Pond and Herrick Hollow Creek (10.5 acres) was divided by the total acreage of ponded water in the vicinity of the site (116.2 acres) to develop the AUF of 9% for pond-feeding wildlife receptors (great blue heron). The total length of the South Pond and Herrick Hollow Creek (1.5 miles) was divided by the total length of streams in the vicinity of the site (13.9 miles) to provide an indication of the proportion of the site to the available habitat. The result of this calculation was an AUF of 0.11 for stream-feeding wildlife (the mink).

The home range of the deer mouse, as reported in the literature, ranges from 0.04 acres to 0.32 acres. Since the landfill size is 8 acres, it is possible for a mouse to feed exclusively within the landfill area. Therefore, the AUF used for the deer mouse is 1.

<u>Forage COPC Concentrations</u>

Concentrations of the COPCs in each medium were obtained from analytical results of samples collected from the South Pond, when available. Mean analytical results of surface water and sediment samples collected under the RI, ERA, and downstream characterization effort were used in the exposure assessment. Mean fish COPC concentrations were calculated using adult and young-of-the-year fish collected from the South Pond and Herrick Hollow Creek. Mean COPC concentrations were also calculated for amphibians using frogs, tadpoles, and salamanders collected from the South pond.

COPC concentrations in aquatic invertebrates and vegetation were not measured in the South Pond or Herrick Hollow Creek. Therefore, concentrations in these media were modeled using literature derived bioconcentration factors (BCFs) and plant uptake factors (PUFs), respectively. Sediment BCF were only available in the literature for chromium and PCBs. BCFs used for other COPCs in sediments were for water and therefore provide a conservative estimate of COPC concentrations in aquatic invertebrates.

Mean soil concentrations were calculated from analytical results of RI and ERA soil samples. A site-specific bioconcentration factor for terrestrial invertebrates was calculated using the analytical results of worm and soil samples collected from the site. This site-specific BCF was applied to the mean site-wide soil COPC concentrations to estimate the associated invertebrate COPC concentrations. COPC concentrations in vegetation were calculated using a literature derived plant uptake factor, which was applied to the mean site-wide soil concentration to estimate associated plant COPC concentrations.

Biomagnification factors for terrestrial plants and soil/sediment invertebrates, when not available in the literature, were modeled using regression-derived equations based on $K_{ow}$ and $K_d$ values (Baes et al. 1984, Lyman et al. 1990, and Travis and Arms 1988). A Plant Uptake Factor (PUF) or Biomagnification Factor (BMF) is a unitless function that expresses the proportion of chemical in biotic tissue relative to levels in environmental media. Terrestrial PUFs for inorganic and organic chemicals were estimated using the following regression equation (Baes et al. 1984, Travis and Arms 1988):

$$\log PUF = 1.588 - 0.578 \log K_{ow} \text{ or } K_d.$$

The PUF regression equation is based on dry weight relationships. To apply the calculated PUF to the wet weight media concentrations a factor of 0.1 was applied to correct for the percent moisture assumed in the plant (10%).

Chemical uptake by sediment invertebrates was estimated using the following regression equation based on the relationship between the log $K_{ow}$ value and bioconcentration factors in aquatic invertebrates (Lyman et al. 1990):

$$\text{Log } BMF = 0.819 \log K_{ow} - 1.146.$$

Use of this equation for aluminum resulted in an invertebrate concentration that is unrealistic (380,000 mg/kg). Since a literature-derived BCF was not available for aluminum, the site-specific BCF of soil invertebrates to soil was used to estimate sediment invertebrate aluminum body burdens.

Selected Threshold Values

The selected threshold values (STV) are the NOAELs used for the HQ calculations. STVs are chemical and organism specific and are based on laboratory toxicity studies. Laboratory toxicity studies have only been performed on relatively few species that are found in the wild. Therefore, STVs are often extrapolated to species other than those tested. Opresko et. al (1994) provided the STVs utilized in this ERA. This document was the result of research conducted by the Oak Ridge National Laboratory related to toxicity of chemicals to wildlife. It is based on and expands upon the existing toxicity database. Although there is always uncertainty associated with extrapolations of laboratory data to conditions in the wild or to other species, the values provided in Opresko et. al present the most comprehensive values available in the literature. The STVs selected for HQ modeling are presented in Table 83.

*HQ results*
COPCs in surface water, sediment and soil that could bioconcentrate through the food chain were modeled for exposures to the great blue heron, mink and deer mouse using mean media concentrations detected in the South Pond. The chosen receptors are representative of trophic levels potentially exposed to site-related releases, and therefore, calculated risks are representative of risks to other receptors at the same trophic level.

The results of the HQ modeling are presented in Tables 84, 85, and 86 for the great blue heron, mink, and deer mouse, respectively. As shown on Table 84, only PCB Aroclor 1248 and zinc resulted in a calculated HQ of greater than 1 for the great blue heron, however, the level of risk is considered to be low (HQ = 2.8 and 1.2, respectively). As shown on Table 85, aluminum, arsenic, and PCB Aroclor 1248 resulted in high, low and low risk to the mink, respectively (HQ = 93, 9.1, and 3.2, respectively). As shown on Table 86, cadmium and PCB Aroclor 1248 resulted in low and moderate risk to the deer mouse, respectively.

The results of the HQ modeling indicate a potential for low levels of risk to higher food chain ecological receptors of the South Pond related to media concentrations of PCB Aroclor 1248 and aluminum. In the terrestrial component, higher food chain ecological receptors are potentially at risk as a result of exposures to cadmium and PCB Aroclor 1248, which resulted in moderate level HQ values. The uncertainties associated with the HQ methodology and results are discussed in the following section.

*Uncertainty*
ERAs are not quantitative probablistic estimates of risk. They are conditional estimates based on assumptions regarding exposure and toxicity. Therefore, it is important to identify and evaluate site-specific uncertainty factors to place the assessment in the proper perspective. Uncertainty in an ERA is due, in part, to natural variability, sampling errors, measurement errors, and estimation errors.

Sampling uncertainty is based on the fact that samples cannot be collected over all geographical space at all times and collected samples may not be representative of actual exposure point concentrations. Measurement uncertainty results from sample processing or laboratory analyses. The use of standard analytical methods and the incorporation of QA/QC duplicates and spikes minimize the uncertainties associated with the analytical data.

A moderate level of uncertainty is associated with the input values used in the HQ model. Although the selected values are the best available values from the literature, the quality of information in the literature has not yet reached the level available for human health. This is primarily due to the diversity of ecological receptors as opposed to only humans. However, as previously mentioned, the Oak Ridge National Laboratory has invested significant efforts towards developing a useable ecological toxicity database (Opresko et.al 1994).

Even with the numbers available from Opresko et.al (1994) the use of a single NOAEL and a sole hazard quotient for each receptor results in a high level of uncertainty. The NOAEL is a conservative toxicological endpoint based on chronic effects observed under laboratory conditions in a single species. It is typically determined by single factor studies in laboratory settings and expressed in mg/kg/day. Single factored, laboratory-derived toxicologic data are not applicable to multifaceted field conditions where stressors, other than chronic toxicity, are dominant. Chemical toxicity is a function of complex interactions between an organism and its environment (Loomis, 1978). Biotic factors that influence toxicity include chemical translocation, reverse functional capacity, storage and tolerance. Chemical factors that influence toxicity include non-specific chemical action, selective chemical action, ionization and lipid solubility effects on translocation, and biotransformation mechanisms (Loomis, 1978). Consequently, laboratory toxicity studies may not accurately represent *in situ* toxicity and, more importantly, population or ecosystem impacts.

To evaluate the level of uncertainty in the HQ calculations, ecological hazard profiles were developed for the COPCs that resulted in HQs greater than 1. The ecological hazard profile is a histogram of possible values of the HQ for a given exposure pathway based on literature-derived ranges of life history input parameter values. The profile of potential ecological hazard incorporates natural ecosystem variability into the hazard assessment for the modeled receptor. Life history parameters and exposure point concentrations are highly variable in a natural system. The use of single values for these parameters results in a high degree of uncertainty. Therefore, the minimum and maximum values for media concentrations, body weight, and dietary ingestion rate were used to represent the natural variability of the parameters in the ecosystem. The result of the hazard profile is a histogram of the relative proportions of HQs in each risk category. The hazard profiles and input values are presented in Appendix S.

A comparison of the single value calculated HQ with the hazard profile output provides an assessment of where the HQ result falls in relation to the range of exposures possible in the natural system.

The resulting HQs for Aroclor 1248 for each receptor fell in the highest frequency risk category, indicating likely scenarios.

The HQ result for mink exposures to aluminum were in the low end of risk frequencies. However, higher concentrations of aluminum at the Sidney Landfill were not considered to be of concern due to the ubiquitos presence of aluminum in the environment. Therefore, consistent with the Sidney Landfill ERA, the presence of aluminum in environmental media will not be considered as significant.

The deer mouse HQ result for cadmium fell within the highest frequency risk range, while the PCB result fell within the lower risk category of two equal frequencies.

### 7.2.7. Task 6: Conclusions

This section presents an interpretation of the results of the ERA tasks as they relate to the assessment endpoints identified in Section 1 of the ERA.

- **Does exposure to RHRMLS releases to surface water of the South Pond have the potential to affect fish?**

Site-related chemicals are present in surface water at concentrations that exceed ecological screening criteria. A chronic bioassay conducted on surface water from the western portion of the South Pond indicated effects on survival and growth within the controlled conditions of the laboratory test. Application of these results would suggest that conditions in the western portion of the South Pond may be a factor in the mortality of fish fry in this area of the South Pond. However, laboratory test results may not provide an accurate portrayal of population impacts in a natural system. Larval fish were observed throughout the South Pond, indicating that they are being reproduced and the fry are surviving.

- **What is the bioavailability of PCBs in the South Pond to fish?**

The results of the caged fish study indicate that PCBs in the surface water and sediments of the South Pond and outlet pond are bioavailable to fish residing in these areas. Although resident fish would not be limited to constant exposures in a specific area of the pond, uptake of PCBs is likely.

YOY fish exhibited PCB concentrations similar to the caged fish. It is concluded that under current conditions of the South Pond and outlet pond, fish could bioaccumulate PCBs to concentrations observed in adult fish. The food chain exposure model indicates that fish with elevated PCB body burdens present only a low level of risk to modeled piscivorous wildlife.

- **Does exposure to site-related chemicals in sediments of the South Pond have the potential to affect aquatic invertebrates?**

A sediment bioassay conducted on Chironomid larvae in sediments from the western portion of the South Pond resulted in 87% mortality. Although the laboratory test result indicates a potential for invertebrate community impact, several invertebrate species were observed in the South Pond, indicating that the mortality levels observed in the laboratory study may not be occurring in the pond to the extent observed in the bioassay.

Invertebrate chemical body burdens were not measured in the South Pond due to the sediment disturbances associated with the UAO activities. Therefore, invertebrate COPC body burdens were conservatively modeled, using water BCFs in some cases, to include invertebrate ingestion exposures for modeled receptors. In spite of the conservative assumptions, invertebrate COPC concentrations contributed to only low levels of risk for modeled invertebrate consumers.

- **Does consumption of South Pond and Herrick Hollow Creek biota by piscivorous wildlife represent a potential food chain impact?**

HQ modeling of piscivorous receptors of the South Pond and Herrick Hollow Creek indicate a high risk potential to the mink due to aluminum media concentrations. It should be noted however, that the USEPA-accepted RI for the Sidney Landfill did not consider elevated concentrations of aluminum in environmental media to be an ecological risk concern. Their conclusion was based on the ubiquitous presence of aluminum in the environment. Consistent with this precedent, exposures to aluminum in site media will not be considered significant.

- **Does consumption of terrestrial invertebrates by terrestrial wildlife represent a potential food chain impact?**

HQ modeling of exposures to the deer mouse through the consumption of worms and other media from the RHRMLS indicate low risk potentials associated with cadmium and Aroclor 1248.

*Conclusions*

Based on the information presented in this ERA, wildlife exposures to site releases present in South Pond and Herrick Hollow Creek environmental media may represent a potential for impact based on exceedances of ecologically based screening criteria. Further evaluation of the potential for impact was performed by modeling risks through the food chain using the hazard quotient methodology. The HQ results indicated only low risk potentials to higher food chain organisms related primarily to PCB, arsenic, zinc, and cadmium exposures.

The aquatic communities of the affected water bodies appeared to be different from nearby similar habitats. The South Pond and Herrick Hollow Creek comprise only a small part of the available suitable habitat for ecological receptors in the vicinity of the site. Suitable feeding habitats are available throughout the study area that minimize the uptake of site-related chemicals. Although great blue heron were seen feeding in the South Pond, the lack of large fish in large quantities probably do not make the South Pond an efficient or attractive feeding area. Infrequent consumption of South Pond fish would result in much lower dietary intakes than those modeled, resulting in a lower risk calculation. For similar reasons, the mink would not find the South Pond as a preferred habitat. Estimated risks to the deer mouse are not high. Impacts to a mouse population as a result of chemical exposure would be minimal as compared to natural controls on the population, such as predation. In addition, the prolific reproduction exhibited by mice would compensate for stressor impacts.

In conclusion, the greatest potential for ecological impact associated with site releases is bioaccumulation through the food chain, resulting in upper food chain PCB residues. The low levels of risk estimated for ecological receptors of the site indicates that population impacts through the food chain are not likely.

Exceedances of ecologically based criteria or guidance values for surface water and sediments may present a potential for organism effects. However, observations of invertebrate and fish populations in the South Pond indicate the conservative nature of these values and the differences between laboratory derived criteria and effects in a natural system.

The results of the surface water and sediment toxicity tests indicate localized areas where conditions were toxic to caged fish. These results were inconsistent with the observations of mature minnows and invertebrates in the same areas. These differences may be related to acclimation or avoidance by the native populations.

Potential downstream impacts associated with site releases will be contained by the natural wetland and pond systems currently in place. The ponds and the densely vegetated wetlands in the downstream areas function to remove sediment contaminants and prevent downstream migration. Disruptions to the system that is currently functioning as a natural control, are likely to mobilize residual contamination and result in increased bioavailability and transport further downstream.

Surface soil PCB and cadmium concentrations represent low potentials for risk to the deer mouse. Potential risks related to soil exposures will be eliminated when the landfill area is capped.

The remedial objective to be considered for the RHRMLS is to substantially reduce or eliminate continuing sources of site releases to and within the South Pond and Herrick Hollow Creek. Natural processes, such as sedimentation and biodegradation, will function to reduce the bioavailability of sediment chemicals once no additional inputs are occurring. The reduced sediment loading will result in decreases in water and biota concentrations, over time, further reducing the potential for exposure to ecological receptors of the South Pond and Herrick Hollow Creek.



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# 8. Summary and preliminary conclusions

Based on field investigations completed during the RI and Response Actions, the summary of physical characteristics and nature and extent of contamination associated with the site and each area is discussed below.

*Overall Site*

- RHRMLS site media were analyzed using contract laboratory procedures (CLP) during the initial phase of the RI. The CLP data indicated that VOCs and PCBs were the primary chemicals of concern at the site. Pesticides were not detected. Semivolatile and inorganic compound concentrations were not found to be of concern at the site.

- The subsurface geology at the RHRMLS consists of up to 44 ft of glacial till overlying sedimentary bedrock. Ground water occurs in both the glacial till and the bedrock zones.

- Elevated concentrations of VOCs were also detected in two of three shallow overburden spring wells located hydraulically downgradient of the Sidney Center Landfill. Two springs located to the south of the Landfill, sampled by the New York State Department of Health (NYSDOH), did not contain site contaminants.

- In association with the RI, interim remedial measures (IRM) were performed in November 1991 to prevent unauthorized access to the Landfill. These measures involved the installation of fencing and posting of signs around the former waste oil pit and runoff area, and the posting of the entire perimeter of the Landfill with signs spaced at 50-ft intervals.

  As part of the UAO Response Action, a sediment trap was installed at the outlet of the South Pond to minimize the potential for sediment migration from the pond. In addition, a focused sediment removal program was performed along the western portion of the South Pond. Approximately 2200 cubic yards of sediments were removed from an area extending 450 feet along and 100 feet out from the west bank of the South Pond.

- Two whole house water treatment units were installed at the residences utilizing Springs #1 and #3. The treatment units provide continuous, on-demand treatment of the water supply to these dwellings

- Air quality samples collected at four locations in the vicinity of the former waste oil pit did not contain VOCs or PCBs. Quantitative air monitoring performed during the RI field program did not indicate that detectable concentrations of VOCs were present in the ambient air at the site.

- A Stage 1B Cultural Resource Assessment was conducted by SUNY - Binghamton, in October 1992, and concluded the following: twenty-nine shovel test pits were excavated and screened, and no pre-1945 historic artifacts or prehistoric artifacts were recovered; background research indicated two 19th century residences just north of the RHRMLS, but suggested that no pre-1945 historical resources were located within the RHRMLS; there were no standing structures within the project area. It was recommended that no further archaeological work be completed at the RHRMLS.

- Three mechanisms of transport (evaporation, surface water runoff and ground water migration) were evaluated to assess whether residues were migrating off-site. It was concluded that the site is not contributing detectable concentrations of site-related compounds to off-site air. Surface water transport has resulted in the detection of compounds that may be site-related downstream of the site. Ground water transport of contaminants to off-site areas has occurred.

- The data collected during the RHRMLS RI was validated and found to be of sufficient quality to support the Risk Assessment and to prepare a Feasibility Study.

*North Area*

1. Physical Characteristics

- The weathered bedrock zone in the vicinity of MW-9 well nest is likely hydraulically connected to the Sidney Center Landfill weathered bedrock zone. Saturated overburden conditions were not observed at this location.

- Ground water elevation data indicate that overburden and shallow bedrock ground water in the North Area emanates from the Sidney Center Landfill flowing west to the North Pond and southwest to the South Pond.

- Two isolated fill sections (approximately 70 ft by 70 ft) were identified in the North Area.

2. Surface and Subsurface Soils

- PCB screening results from surface soils in the North Area ranged from non-detect to 42.2 ppm and were typically detected in the areas underlain by fill materials.

- Soil borings completed in the North Area did not contain elevated concentrations of VOCs. Soil boring SB-5 did contain 0.062 ppm of PCBs.

- Analysis of soil collected from a test pit completed in the North Area exhibited substantially lower concentrations of VOCs and PCBs than the Landfill. Detected VOCs included; 1,2-dich'--oethene (0.077 ppm), tetrachloroethene (0.1 ppm), toluene (0.87 ppm), ethylbenzene (1.1 ppm), and xylene (1.7 ppm). PCBs were also detected at a concentration of 1.5 ppm.

3. Ground Water

- Contaminants detected in the North Area ground water may be migrating along the till/bedrock interface from a topographically higher and upgradient source area, and is evidenced by artesian conditions at MW-9.

  - Concentrations of total VOCs in the North Area ground water have ranged from non-detectable at MW-8 and MW-15 to 310 ppb at MW-9. The compounds with the highest VOC concentrations in the North Area consists of predominately TCE, with much lower concentrations of 1,2-DCE, 111-TCA, and PCE.

  - PCBs were detected at low concentrations (ND to 0.31 ppb) in ground water from the North Area.

4. Surface Water and Sediment

- Surface water from the North Area drains to the south toward the South Pond, and to the north toward the North Pond which also receives runoff from the Sidney Center Landfill. The North Pond discharges to the north into Carr's Creek.

- Low levels of VOCs and PCBs were detected in surface water and sediment samples collected along the southern and eastern shoreline of the North Pond, and are attributed to surface water runoff and ground water discharge from the adjacent Sidney Center Landfill.

  - Sediment from the North Pond contained low concentrations of methylene chloride, carbon disulfide, toluene, and xylenes. In the North Area, one sample (SEDTCL#3) exhibited a detectable concentration of PCBs at 0.37 ppm.

  - Surface water in the North Pond contained TCE (4 ppb) and 1,2 DCE (1 ppb). PCBs in North Pond surface water ranged from non-detect to 0.13 ppb.

*South Area*

1. Physical Characteristics

- Landfill fill materials are present in an area approximately 300 ft wide by 800 ft long that extends southwest from the waste oil pit and parallels Richardson Hill Road.

- Surface water from the Landfill drains to the east toward the South Pond. The South Pond drains to the south along an intermittent tributary. Discharge from the South Pond is currently controlled by a sediment trap system.

- Overburden and shallow bedrock ground water flow generally follows the site topography and flows in an easterly direction from the landfill to the South Pond. Overburden ground water moves under an average hydraulic gradient of 0.15 ft/ft. Shallow bedrock ground water flows under an average hydraulic gradient of 0.12 ft/ft.

- Overburden ground water flow occurs preferentially in till fractures and zones of relatively coarser overburden materials.

- The similarity in extent of the overburden and shallow bedrock aquifer VOC plumes indicates these zones are to some degree hydraulically interconnected.

- The large vertical hydraulic head difference (greater than 30 ft) that exists between the shallow bedrock and deeper bedrock ground water zones at the landfill site suggests that they are not hydraulically connected.

- Based on pump test data and *in situ* hydraulic conductivity test data, the estimated hydraulic conductivity of the overburden ranges from 2.1 gpd/ft$^2$ and 31 gpd/ft$^2$.

- Hydraulic conductivity values in the bedrock wells ranged from 0.002 gpd/ft$^2$ to 49 gpd/ft$^2$.

2. Surface and Subsurface Soils

- Soil vapor surveys completed on a 100-ft by 100-ft site wide grid indicated that total VOC concentrations at the Landfill ranged from less than 0.005 ppm to 96.3 ppm. VOCs were pervasive in fill areas, and the highest concentrations were detected west and southwest of the waste oil pit. Soil vapor VOC concentrations greater than 5 ppm exhibited good correlation with the areas underlain by fill materials.

- VOCs and semi-volatiles were not detected in Landfill surface soils.

- Surface soil screening of thirty-eight composite samples for PCBs indicated a range of concentrations from less than 0.5 ppm to 626.7 ppm. PCBs were primarily detected in fill materials at the Landfill.

- Surface soil PCB concentrations were highest in samples collected in close proximity to the former waste oil pit or in surface water runoff pathways from the pit. PCB concentrations decreased away from the former oil disposal pit.

- Pesticides were not detected in subsurface soil samples.

- Soil borings completed at the Landfill indicated that subsurface materials contain elevated levels of VOCs and PCBs. VOCs and PCBs were detected in and around the former waste oil pit, and in the fill materials southwest of the former oil pit. The highest VOC concentrations were detected in the vicinity of the former waste oil pit, where concentrations ranged up to 287 ppm. The most prevalent VOCs were 1,2-DCE, TCE, xylene, toluene and ethylbenzene. PCB concentrations were also highest in the former waste oil pit, with concentrations ranging up to 7,000 ppm. Most of the PCB concentrations were, however, generally less than 200 ppm.

- An anomalous PCB concentration of 14,000 ppm was measured at SB-10, which is located in a separate area southwest of the former waste oil pit. Subsequent investigations conducted in this area during the course of the RI and focused efforts completed in accordance with the provisions of the UAO did not confirm this level of contamination.

- DNAPLs were not observed in subsurface soils downgradient from the waste oil pit.

- GPR survey transects and test pits were completed at the Landfill in the immediate vicinity of SB-10, and along Richardson Hill Road. The investigations concluded that although some scrap metal is disseminated throughout the fill, no cache of buried or crushed drums exist. Soil samples collected from the five test pits indicated the presence of 1,2-DCE (0.12 ppm to 0.59 ppm), TCE (0.66 ppm), toluene (1.3 ppm to 21 ppm), and ethylbenzene (1.3 ppm to 9.3 ppm). PCBs (Aroclor 1248) were detected in the subsurface soil at concentrations ranging from 4.4 ppm to 3,600 ppm.

- A continuous test pit, approximately 200 feet in length, was completed immediately downgradient of the waste oil pit as part of the UAO Response Action Investigation. The investigation revealed that the waste oil pit did not constitute a continuing source of free oil and that free oil was not migrating to downgradient areas.

- The presence of discrete, non-recoverable droplets of LNAPL in seeps flowing from till fractures and zones of coarse interbedded overburden the vicinity of MW-5S were subsequently controlled with passive LNAPL collection systems in the UAO.

3. Ground Water

- Laboratory analyses of ground water samples collected from overburden monitoring wells located in the Southern section indicate that concentrations of VOCs and PCBs have migrated from the fill materials, southeast towards the South Pond. The relative lack of change in the size and magnitude of the contaminant plume over the course of the RI would suggest that it is in equilibrium.

  - The lateral boundaries of the plume are generally defined by the low or non-detectable VOC concentrations in MW-4S, MW-12, MW-11S, and MW-7S, and low PCB concentrations at MW-4S, MW-12, MW-11S, and MW-7S. The plume is about 1,200 ft wide by 400 ft long and extends from the Landfill to the South Pond.

- The highest concentrations of VOCs were detected in the monitoring wells adjacent to and downgradient of the former waste oil pit (MW-1, MW-2, MW-3, and MW-5S). Elevated concentrations of VOCs were also detected in MW-6 and MW-18S, which are located downgradient of the southern portion of the Landfill. The most frequently detected VOCs in the overburden ground water were TCE, PCE, 1,1,1-TCA, 1,2-DCE, 1,1-DCE, 1,1-DCA, and vinyl chloride.

- PCB concentrations (ranging from 35 ppb to 1,400 ppb) were highest in monitoring wells MW-1, MW-2, and MW-5S, which are located in close proximity to, and downgradient of, the former waste oil pit. PCB concentrations were lower in wells located towards the periphery of the plume, such as MW-3 and MW-6, located to the north and south, respectively. The other overburden ground water wells at Landfill exhibited PCB concentrations of 1 ppb or less.

- The shallow bedrock ground water at the Landfill contains similar VOC and PCB constituents as the overburden. Shallow bedrock ground water analyses indicate that a VOC plume extends in an easterly direction from the Landfill, with bedrock ground water discharge into the South Pond. The lateral boundaries of the plume are generally defined by the non-detectable VOCs in MW-4D and MW-11D, and low TCE concentrations in MW-11D. The relative lack of change in the size and magnitude of the contaminant plume over the course of the RI would suggest that it is in equilibrium.

  - The primary VOCs in shallow bedrock ground water are 1,2-DCE and TCE, and have been detected in monitoring wells directly downgradient of the former waste oil pit (MW-5D) and directly downgradient of the southern portion of the Landfill

(MW-18D). Total VOCs in these wells have ranged from 2,510 ppb to 7,770 ppb.

- PCBs were detected at low concentrations (0.2 ppb to 1.3 ppb) in the shallow bedrock wells MW-5D, MW-11D, and MW-18D.

- Neither VOCs or PCBs were detected in the deep bedrock ground water downgradient of the Landfill. The absence of these constituents in the deep bedrock ground water indicates this zone is isolated from the overburden and shallow bedrock ground water zones.

4. Surface Water and Sediment

- Pesticides were not detected in surface water or sediment samples collected from the South Pond.

- Surface water and sediment samples collected along the west shoreline of the South Pond contained elevated levels of VOCs and PCBs. The presence of these contaminants is attributed to the seepage and discharge of leachate and ground water from the landfill into the pond. Low levels of VOCs and PCBs were also detected downstream of the South Pond. However, these levels generally decreased with increasing distance from the pond.

  - Total VOCs in sediment from the South Pond ranged from 0.013 ppm to 4.96 ppm. The most prevalent VOCs in the South Pond were 1,2-DCE (ND to 3.5 ppm) and toluene (ND to 0.27 ppm). South Pond sediment also contained low concentrations of methylene chloride, acetone, 2-butanone, xylene, ethylbenzene, chlorobenzene, 1,1-DCA, 1,1,1-TCA, TCE, chloromethane, carbon disulfide, and vinyl chloride.

  - PCB concentrations in the South Pond sediments ranged from less than 0.6 ppm to 1300 ppm. The highest concentrations were measured in sediments collected along the west shoreline of the South Pond prior to removal during the UAO.

- With the exception of chloromethane detected in sample SW92-7 (0.008 ppm), VOCs were not detected in sediment downstream of the South Pond. PCBs were detected in sediment downstream of the South Pond at concentrations ranging from 0.059 ppm to 6.6 ppm. At downstream sampling locations, PCB concentrations decreased with increased distance from the South Pond.

- Total VOC concentrations in surface water samples collected in the South Pond ranged from 3 ppb to 1,982 ppb. The highest VOC concentrations were measured adjacent to leachate seeps along the west shoreline of the South Pond. The VOC constituents included: 1,2-DCE, vinyl chloride, 1,1,1-TCA, TCE, 1,1-DCA, and toluene.

- PCBs detected in surface water samples collected in the South Pond ranged in concentration from non-detect to 100 ppb. With the exception of sample SWTCL#2, located along the west shoreline directly downgradient of the former waste oil pit, the surface water in the South Pond contained less than 1 ppb of PCBs.

- Downstream of the South Pond, VOCs were detected at low concentrations including 1,2-DCE (1 ppb to 4 ppb), methylene chloride (0.9 ppb), and carbon disulfide (10 ppb to 12 ppb). PCBs were also detected at concentrations ranging from 0.15 ppb to 0.42 ppb. PCBs were not detected at sampling points beyond SW91-19, located approximately 2,600 ft downstream of the South Pond.

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