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FINAL

REMEDIAL INVESTIGATION REPORT

GLOVERSVILLE MUNICIPAL LANDFILL
GLOVERSVILLE, NEW YORK

VOLUME I: SECTIONS 1 - 8

Tighe&Bond
Consulting Engineers
Environmental Specialists

3 PHYSICAL CHARACTERISTICS OF STUDY AREA

4 NATURE AND EXTENT OF CONTAMINATION

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APPENDIX

SECTION 1
INTRODUCTION

1.1 PURPOSE OF REPORT

The Gloversville, NY Municipal Sanitary Landfill was placed on the New York State Register of Inactive Hazardous Waste Disposal Sites in 1981. A Work Plan completed in 1989 (33) calls for a multi-phased Remedial Investigation and Feasibility Study (RI/FS) of the Landfill. A Draft Report (29) prepared at the conclusion of the Phase 1 activities presented results of the Phase 1 tasks and recommended additional data gathering. A Scope of Work (31) was developed to address the additional data requirements identified in the Phase 1 Report.

This Report presents the results of the Remedial Investigation portion of the study, including 1989 Work Plan tasks and tasks completed under the Phase 2 Scope of Work. The results presented herein provide the basis for conducting the Feasibility Study. The Feasibility Study Report (FS) considers a range of potential remedial technologies and recommends appropriate remedial actions at the Gloversville Landfill. This FS Report is presented separately.

1.2 SITE BACKGROUND

1.2.1 Site Description - The Gloversville Municipal Landfill is owned by the City of Gloversville. Reportedly, open refuse disposal occurred at the site from near the turn of the century until disposal practices were upgraded to sanitary landfilling procedures in 1958. According to a 1986 report by Wehran Engineering, (42), the site accepted all wastes generated by the City of Gloversville and part of the Town of Johnstown including domestic, commercial and industrial wastes. Several other towns in Fulton County disposed of solid waste at the Gloversville Landfill until it closed in July, 1989.

The Gloversville Landfill occupies approximately 80 acres of a 175 acre property located in the south-central portion of Fulton County in the Town of Johnstown, New York.

Figure 1-1 is a locus map showing the general location of the City's parcel on which the landfill is located.

For the purposes of discussion in this report, the actual landfill area is divided into the "active" and "inactive" areas. The "active" disposal area is the central portion of the site where disposal activities occurred throughout the late 1980's, terminating in July 1989. The "inactive" area refers to older sections of the landfill which are currently overgrown with brush and small to medium size trees. The area surrounding the landfill is densely wooded along the west, north and east sides. South of the landfill, private residences on small open lots predominate beyond a wooded buffer area.

The "active" area is terraced and the waste is up to 70 feet thick. A large open borrow pit exists south of the "active" area from which sand had been excavated for use as cover material. An old scrap metal disposal area surrounds a depression north of the active area which frequently contains standing water. Along the northeast side of the Landfill, demolition debris forms the shoreline of a large beaver pond. On the northwest side of the "inactive" area, glass and metal debris extends to the south bank of Anthony Creek.

The extent of the disposal area is fairly well delineated along the northern, western and northeastern sides while overgrown vegetation obscures the extent of the trash on the eastern and southern sides. The top of the active area of the landfill is relatively flat and covered with a fine sand cover material.

Several other potential sources of environmental contamination exist in the vicinity of the landfill. These include a scrap yard, auto repair shops, a trucking company terminal, a farm, a manufacturer of precast concrete steps and other businesses. While the landfill is the subject of this study, these other potential sources could contribute to contamination of Anthony Creek or private wells in the area.

<u>1.2.2 Previous Investigations</u> - The most widely referenced previous investigation cited for the Gloversville Landfill is a September 1980 report prepared by Dunn Geoscience

Corp. (7). The report was part of the State of New York Open Dump Inventory prepared for the Division of Solid waste Management, New York State Department of Environmental Conservation. Other investigations of the landfill site include: Engineering Investigations at Inactive Hazardous Waste Sites in the State of New York - Phase 1 Investigations - Gloversville Landfill prepared by Wehran Engineering, P.C. for the Division of Solid and Hazardous Waste, New York State Department of Environmental Conservation in April 1986 (42). The Soil Conservation Service prepared a soil resource report for the City of Gloversville in December 1976 (28). SCS Engineers authored a report titled Evaluation of Operations for the Planning Department of Fulton County in April 1981 (25). In addition, many New York State DEC, DOH and DOL files, and City files and records contribute to the background information available for the landfill.

The Dunn Report (7) was based on the most extensive scope of work prior to initiation of the current RI/FS. It included an initial site reconnaissance and file search, the installation of groundwater monitoring wells, sampling and analysis of groundwater, and geophysical mapping. The Dunn Report includes a description of the site geology and hydrogeology, analytical groundwater data which suggest a minimal impact of the landfill on the quality of the groundwater and a determination that the landfill site is in compliance with RCRA Groundwater Criteria as specified by 40 CFR Part 257.3-4.

The Wehran Engineering Phase 1 Investigation (42) presents the findings of an extensive file review and a site inspection. The purpose of the Wehran Report was to develop a preliminary Hazard Ranking System (HRS) score. The HRS score consists of three factors: the Migration Score (M_s), the Direct Contact Score (DC_s) and the Fire and Explosion score (FE_s). The high migration score for this site, in addition to the documentation on industrial waste disposal, was sufficient to warrant a Remedial Investigation/Feasibility Study at the site.

1.3 REPORT ORGANIZATION

The format of this Report follows the suggested outline presented in EPA's "Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA" (Interim Final Draft, October, 1988) (41). The contents of each report section are further described below. In some cases where specific tasks were performed by subconsultants, the descriptions provided in the main text of this Report are summaries of more extensive reports prepared by the subconsultant.

A Draft RI/FS report (29) was prepared at the conclusion of Phase 1 field activities. This report contained detailed accounts of all Phase 1 tasks. The main text of this Report discusses results from all RI/FS tasks, including the Phase 1 activities. However, the detailed subconsultant Reports from Phase 1, which were appended to the Phase 1 Report are not included in this Report.

<u>Section 1</u> provides an introduction and a brief history of the site, including information on the RI/FS team participants.

<u>Section 2</u> describes the field activities conducted during the RI/FS. These descriptions provide a detailed account of the methods used to gather the data used in the preparation of this Report. Phase 1 field tasks were discussed in detail in the Phase 1 Report. These are summarized in Section 2.1.

<u>Section 3</u> describes the geologic and hydrogeologic characteristics of the site. Information on the soil and rock types in the area of the landfill is presented along with specific information on the groundwater flow patterns. Phase 2 includes the development of a computerized groundwater model of site hydrogeology.

<u>Section 4</u> presents results of all the chemical analyses performed during Phase 1 and Phase 2. The data are summarized in tabular form and briefly discussed in the text. Validated laboratory reporting sheets and complete data validation reports for Phase 2

are presented in Appendix A. Validated reports for Phase 1 data are available in the Phase 1 Report (Appendix A and Addendum 1).

Section 5 discusses contaminant migration from the landfill. This evaluation is based on the analytical results explained in Section 4 and the groundwater flow paths discussed in Section 3. Section 5 includes several figures that show contaminant concentrations across the site. These figures provide a graphical presentation of the data and contaminant migration from the site.

Section 6 furnishes the results of a baseline risk assessment which was prepared using all of the analytical data from Phases 1 and 2.

Section 7 summarizes the results of the Remedial Investigation and provides conclusions that will be used to complete the Feasibility Study.

Section 8 tabulates references and abbreviations.

1.4 RI/FS TEAM

The Remedial Investigation consists of several separate yet interrelated studies. Together these studies provide the basis for describing the geography, geology, hydrology, chemistry and biology that affects the generation and transport of potential contaminants from the site. A number of firms contributed to the RI/FS of the Gloversville Municipal Landfill. These firms and their respective RI/FS responsibilities are presented in Table 1-1.

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SECTION 2

STUDY AREA INVESTIGATIONS

2.1 PHASE 1 STUDIES

As described in Section 1.3, this Report contains data generated during both Phases of the RI/FS. Phase 1 tasks that were repeated or continued in Phase 2 are fully described in Sections 2.2 - 2-7. For example, Sections 2.2 and 2.3 provide details of the Phase 1 and 2 data collection activities for the source area and surface water, respectively.

Several Phase 1 tasks provided important background information in support of the overall study. These tasks were detailed in the Phase 1 Report and are summarized in the following subsections.

2.1.1 Mapping - During the planning phase of the Study, it became apparent that maps of the landfill and surrounding areas were outdated. Disposal activity and site re-grading at the landfill had changed the ground contours significantly. Beaver activity in the streams to the north of the site had altered water courses and wetland areas. Tallamy, VanKuren, Gertis & Associates (TVGA) provided aerial photographs of the landfill area and developed a topographic map of the site. Using stereophotogrammetric methods, TVGA prepared a base map from the aerial photographs at a scale of 1" = 50' with 2' vertical contours.

2.1.2 Air Screen Survey - Early in Phase 1, Tighe & Bond and TRC Environmental Consultants conducted a screening survey of the ambient air at the landfill. Ambient concentrations of volatile organic compounds were measured using three types of direct-reading, handheld instruments: a photoionization detector, a flame ionization detector organic vapor analyzer, and a combustible gas indicator.

Measurements were made 2-3 feet above the ground at each point on a 200-foot grid covering the active refuse mound. Samples were also collected from shallow depressions dug into the ground surface at random points. The results of the screening study were used in the development of the Health and Safety Plan for the remainder of the field activities.

2.1.3 Stream Gauging - Leggette, Brashears & Graham personnel conducted three rounds of stream gauging at various points along Anthony Creek on July 10, 18 and October 11, 1990. Where possible, the locations chosen for stream gauging coincided with the surface water sampling locations. Using a Pygmy flow meter, stream velocity was measured in each segment of the stream at a given location. Measurements of the width and depth of each segment of the stream were combined with the velocity information to calculate a flow rate. The sum of the flows from each stream segment provided the total stream flow at a given location.

Stream flow data were summarized in Section 3.4.2 and detailed in Appendix E of the Phase 1 Report. The data provided a basis for estimating groundwater discharge rates from the landfill area.

2.1.4 Geophysical Studies - Geophysical studies are non-intrusive methods used to assess subsurface conditions. These techniques do not involve penetration of the ground and therefore, can be used over a large area relatively quickly. The results of these type of studies are generally confirmed with borings and monitor wells.

Leggette, Brashears & Graham (LBG) used three techniques during Phase 1 to obtain an indication of the depth to bedrock and of the possible presence of contaminant plumes: seismic refraction, earth resistivity and terrain conductivity, each of which is described briefly below. These studies were conducted along the lines shown on Figure 2-1. Detailed operating procedures were presented in the Field Sampling Plan (30) and a full geophysics report was provided in Appendix C of the Phase 1 Report.

Seismic Refraction - This survey was conducted to determine the depth to and topography of the bedrock surface underlying the site. This information is important since the bedrock surface can impact the groundwater flow patterns at the site. It can also assist in understanding how the overlying soils were deposited during glaciation.

The basic principle of a seismic refraction survey is to measure the time it takes sound energy to pass from the surface to the bedrock and reflect back to the surface. This is accomplished by detonating a small explosive charge and recording the time of travel of the seismic wave through the use of geophones positioned in a linear array on the ground surface. The knowledge of the distribution of seismic velocities allows characterization of the types of subsurface materials encountered and their approximate depth beneath ground surface.

Earth Resistivity - LBG made earth resistivity measurements in an attempt to determine the existence and, if present, the lateral extent of a deep leachate plume. The survey was conducted using a vertical electric sounding which involves expanding a horizontal linear array of four electrodes. After recording resistivity between the electrodes at one spacing, the distance between electrodes is increased and another resistivity measurement is made. This procedure causes an electric current to flow to progressively greater depths and allows characterization of earth resistivity with depth.

Areas of low resistivity are sometimes associated with the presence of groundwater contamination. The presence of certain ions in the groundwater (e.g. iron, chloride, sodium) increase the water's ability to conduct electricity, thereby reducing the apparent resistivity of the saturated zone. These same ions are often indicative of a contaminant plume.

Terrain Conductivity - An Electromagnetic (EM) Survey was conducted to measure terrain conductivity. The purpose of this survey was to determine the existence and, if present, the areal extent of a shallow leachate plume. Completion of the survey required the use of a terrain conductivity meter and data recorder. This meter consists of a receiver coil and a separate transmitter coil which induces an electrical source field in the earth. Each one of the current loops generates a magnetic field proportional to the value of the current flow within the loop. A part of the magnetic field from each current loop is intercepted by the receiver coil and converted to an output voltage which is linearly related to terrain conductivity.

While the data obtained in a terrain conductivity survey is subject to several limitations and generalizations, they can yield useful information on the potential location of contaminant plumes. Assuming the soil type and moisture content stay constant, higher concentrations of ions in solution in groundwater increase terrain conductivity. Increased ion concentration is sometimes indicative of contaminant plumes.

2.1.5 Wildlife Studies - New England Environmental, Inc. (NEE) performed the wildlife and wetlands studies. Their report was attached as Appendix E of the Phase 1 Report. Figure 2-2 shows the wildlife sampling locations.

Macroinvertebrate Survey - Macroinvertebrate samples were collected from Anthony Creek, an unnamed tributary to Anthony Creek, and associated beaver ponds located on both streams during October 1989 and May 1990. Sample collection points included seven stream locations, two pond locations, and an additional unconnected control location, Marston's Pond. The unconnected control pond was only sampled in May 1990. Identification of the collected species provided abundance and diversity data for each sampling location.

Fisheries - Fish populations in the streams and ponds adjacent to the landfill were analyzed by NEE during the fall of 1989 and the spring of 1990. Sampling points corresponded to the stations used during the macroinvertebrate survey. Fish captured by gill netting and electroshocking were identified to the species level in the field. Size, age, abundance and diversity data were recorded during both rounds of sampling.

In-Situ Bioassay - To determine the degree of toxicity to fish at each sampling location, NEE conducted two rounds (July and September 1990) of 72-hour in-situ bioassays using fathead minnows as the study organism. During each trial, ten fathead minnows were placed inside minnow traps and located at each of the following stations: SW-2, SW-3, SW-4, SW-5, SW-6, SW-7, SW-8, SW-9, and WL-10. These locations are shown on Figure 2-2. Inspections of the minnows were made at 2, 4, 8, 24, 48, and 72 hours to check for mortality and behavioral changes.

Wildlife and Rare/Protected Species Identification - NEE conducted investigations during the fall of 1989 and the spring of 1990 to determine terrestrial wildlife areas. The investigations included literature reviews to determine the possible occurrence of species based on their range distributions, passive observations of species occurrences, and active trapping of wildlife. NEE compiled a list of observed and captured species of wildlife present at the landfill site. This list was compared to lists of rare and endangered species obtained from the NYSDEC and the New York Natural Heritage Program to document the presence or non- presence of species of concern.

Tissue Analysis - To determine the degree of uptake of contaminants by wildlife populations, specimens of shrews, snapping turtles, and fish collected during the wildlife investigation were frozen for subsequent tissue analysis.

2.1.6 Wetlands Delineation - All wetlands adjacent to the landfill were identified and delineated by NEE through interpretation of 1" = 500' aerial photographs. These photo interpretations were checked for accuracy by comparison with the NYS freshwater wetlands map for the Fulton County, Gloversville Quadrangle. Discrepancies were corrected through field verifications. Subsequent to the field verifications, classification of the wetlands followed the National Wetland Inventory Classification System. The resultant wetland boundaries are plotted on Figure 2-2.

2.2 SOURCE CHARACTERIZATION

The focus of several tasks conducted during Phase 1 and Phase 2 was to define the source area. Disposal areas were delineated through the use of a magnetometer survey, test pits, borings and visual observations. Sampling and analyses of the filled wastes, surface soils, subsurface soils and leachate provided information on the chemical constituency of these source area media. The following subsections detail the methodology of these source characterization tasks.

2.2.1 Delineation of Disposal Boundaries

Magnetometer - The earth possesses an ambient magnetic field which can be modified locally by both naturally-occurring and man-made magnetic materials. Iron or steel objects, such as buried drums or construction debris, act as local dipoles which are generally oriented differently than the earth's magnetic field.

LBG personnel conducted the magnetometer survey at the landfill in July 1989. The purpose of the survey was to attempt to delineate the areal extent of the refuse material by recording the mapping magnetic anomalies produced by buried ferrous objects. The 200-foot grid points established for the air screen survey served as reference points for the magnetometer investigation.

An ABEM Model GSM-19 Overhauser Memory Magnetometer/Gradiometer was used at 10-foot intervals along the 200-foot grid. The magnetometer/gradiometer operates with two sensors, one measures the earths magnetic field and the second measures the local magnetic field. The difference between the two magnetic field is the gradient which can be used to determine disposal area boundaries.

West and north of the refuse mound, the magnetometer survey clearly delineated the extent of the fill material. However, the southeastern boundary of the fill material could only be inferred. Therefore, test pits were excavated along this boundary to better define the lateral extent of the fill material.

Test Pits - On July 25, 1990, Marcor of New York began the excavation of test pits under the supervision of Tighe & Bond personnel. A total of 13 test pits were excavated using a backhoe to further delineate the extent of the refuse material. Soil samples were collected for HNu head-space analysis. Figure 2-3 shows the locations of the test pits.

Steam cleaning of the backhoe bucket removed soil and other residue prior to initiating the excavation of each test pit from which a soil sample was collected for chemical analysis. Test pit excavation proceeded to a depth of approximately 6 feet, maintaining a vertical profile along the sidewalls of the excavation. This aided in the development of a stratigraphic log of the test pit soils. An HNu photoionization detector was utilized during the excavation to monitor for the presence of volatile organic compounds. Soil excavated from 2 foot intervals was segregated into separate stockpiles and sampled for an HNu head-space analysis.

Four soil samples were transferred to proper sample containers and sent to the laboratory for chemical analyses. These were selected to be "worst case" samples based on discoloration, odors and/or elevated HNu headspace readings.

2.2.2 Thickness and Nature of Waste

"Inactive" Area Borings - As part of the Phase 2 drilling program, twenty-two soil borings were completed through the buried refuse in "inactive" areas of the landfill, north of the fire road surrounding the "active" area. The borings were advanced to at least two feet below the bottom of the refuse using a drilling rig and hollow stem augers. Continuous split-spoon samples were taken of the fill material and the underlying native soil. Figure 2-3 shows the locations of the borings.

To characterize the surface soils, 14 samples were collected for chemical analysis. The analyses include volatile organic compounds, semi-volatile organic compounds, pesticides/PCB's, and metals.

To evaluate the contamination in the soils that lie beneath the refuse, 12 subsurface samples were collected for chemical characterization. The subsurface soil samples were also analyzed for volatile and semi-volatile organic compounds, pesticides/PCB's and metals. In accordance with the QA/QC plan established for the Phase 2 sampling program, a single duplicate soil sample was collected. The borings also determined the vertical extent of fill so that waste volume estimates can be made. The analytical results of the soil sampling and a discussion of the contaminants follows in Section 4.

Test Pits - In April 1992, 29 test pits were excavated in the "inactive" areas to characterize the waste types. A tracked excavator was used to dig the trenches and/or pits and to establish a depth profile. Marcor of New York supplied the excavating equipment and an operator. Tighe & Bond personnel inspected the work and recorded observations. Photographs were taken of the different types of waste materials.

Concurrent with the excavation of the pits, ambient air was monitored for the presence of VOC's with a photoionization detector (PID) and for combustible gases with a combustible gas indicator. In addition, the inspector was outfitted with a sampling device for total chromium on those days when dry, windy conditions existed. This protocol was designed to provide an indication of the hazards created with the excavation of waste material.

The potential impact of the leaching effects of percolating rainwater was evaluated by collecting five (5) waste samples from the "inactive" area. These samples, selected to be representative of the entire mass of waste through which the pits were dug, were collected in containers supplied by the contract laboratory and subjected to the Toxicity Characteristic Leaching Procedure. This procedure simulates leaching characteristics of unlined/uncapped landfills. The resultant extract was analyzed for volatile and semi-volatile organic compounds, pesticides/PCB's, metals, nitrogen compounds and other miscellaneous inorganic compounds as indicated in Table 2-1.

Test pit locations were concentrated in the "inactive" area of the landfill, northwest of the fire road and maintenance garage. Other pits were located west of the fire road in the general vicinity of well cluster MW-2 and near boring locations B-21 and B-27. Figure 2-3 shows the exact location of these test pits.

2.2.3 Leachate Characterization - The chemical constituents present in the landfill leachate were determined by sampling and analyzing leachate emanating from the side of the landfill and by sampling leachate collected through monitor wells installed in the waste mass. Table 2-1 shows that the 7 leachate samples and 5 TCLP extract samples were all analyzed for VOC's, semi-volatile organics, pesticides/PCB's, metals, miscellaneous inorganics and field parameters. The latest sampling round also included analyses for treatment parameters and nitrogen compounds.

Leachate Seeps - Visual observation of the north and northeast sides of the "active" portion of the landfill reveals several points where leachate from within the refuse is discharged to the surface. These leachate seeps occur more frequently and flow more readily after significant rainfall events. To aid in characterization of the landfill leachate, Tighe & Bond measured the specific conductance in 6 seeps following a rain event. Figure 2-4 identifies the locations of these seeps.

The two seeps exhibiting the highest specific conductance were sampled for chemical analyses. Since specific conductance typically increases with higher ion concentrations, it serves as an indication of contamination. Leachate samples were collected directly into the sample containers provided by the laboratory. Sediment samples were also collected from these two seeps and submitted for chemical analyses.

Landfill Monitor Wells - The project Work Plan called for the collection of subsurface water from within the "active" disposal area. WTD Environmental Drilling installed MW-7 and MW-8 through the landfilled material. Water-filled hollow stem augers were used to advance the boring to the underlying natural materials. Split spoon samples were not collected until most of the refuse mound had been drilled through, then continuous 24-inch split spoon samples were collected to confirm the bottom of refuse. A split spoon sample of soil was collected from each borehole beneath the waste. These samples were subsequently transferred to appropriate sample containers and shipped to the laboratory for chemical analysis. In MW-8, sampling commenced after the auger had passed completely through the refuse. Therefore, the precise elevation of the bottom of the refuse was not established.

The water level in the augers was allowed to equilibrate with the water table within the waste mass. Once the water table elevation was established, the borehole was grouted with bentonite QUIK GEL from the bottom of the borehole up to approximately 15 feet below the water table. Monitor wells were installed above this point.

Wells MW-7 and MW-8 are constructed of 2-inch diameter, flush jointed, threaded PVC pipe with 20′ long 0.10-inch slotted screens which extend from approximately 15 feet below the water table to 5 feet above it. The borehole annulus is packed with sand from 2 feet below the bottom of the well to at least 2 feet above the top of the well screen. A 2-foot thick bentonite plug seals the annulus above the sand, preventing any surface runoff or percolating water from flowing down the borehole into the screened zone. The remainder of the annulus is filled with the bentonite/cement grout to within 1 foot of the surface. A locking steel casing set in concrete at the surface protects the well from incidental damage.

During the Phase 2 drilling program, one monitor well, MW-12 was installed in fill material. MW-12 is located north of the fire road in the "inactive" area. A borehole was advanced through the fill by the use of 4.25-inch i.d., 8-inch o.d. hollow stem augers. Continuous 2-foot split spoon samples were taken during the borehole advancement. Representative samples of the fill material were placed in jar and covered with aluminum foil, then capped for subsequent head-space analysis. Termination of the boring occurred five feet below the bottom of the refuse so that the soil underlying the landfill could be sampled.

The monitor well installed at this location consists of factory-sealed, 2-inch diameter, 0.020-inch slot, flush-joint PVC screen, 15 feet in length. The riser piper extends from the top of the screen to approximately 2 feet above the ground surface. Prior to the well screen installation, the borehole was backfilled with granular bentonite from the bottom of the borehole to 2-feet from the bottom of the screen. Two feet of No. 2 quartzite sand was then placed on top of the bentonite. The 15-foot well screen is set above the sand, extending from 12 feet below to 3 feet above the water table. A No. 2 quartzite sand surrounds the screen and riser pipe to 2 feet below grade. Bentonite fills the remaining annulus to the ground surface. A locking steel protective casing is set in a concrete collar.

<u>Toxicity Characteristic Leaching Procedure (TCLP) Extract</u> - Section 2.2.2 describes the collection of five samples of waste material for evaluation of the leachability of contaminants for the possible scenario where portions of the inactive landfill are left uncapped. These results also provide a worst case characterization of potential leachate from the landfill.

<u>2.2.4 Source Area Soils</u> - The Remedial Investigation included the collection of source area soils from the surface of the landfill, from within the waste mass, and from the soils beneath the landfill. Table 2-1 shows that all 34 source area soil samples were analyzed for VOC's, semi-volatile organics, pesticides/PCB's, and metals.

<u>Surface Samples</u> - As described in Section 2.2.2, the soil borings conducted in the "inactive" area included the collection of 14 surface soil samples. These samples were collected in a decontaminated split spoon sampler and transferred to sample containers.

Other surface soil samples were collected from the two leachate seeps that were used for leachate characterization as described in Section 2.2.3.

<u>Waste Mass Samples</u> - Soil samples were collected during the excavation of Phase 1 test pits as described in Section 2.2.1. Four of the soil samples that were collected for headspace analysis were transferred to laboratory supplied sample containers and submitted for chemical analyses. These were selected to be "worst case" samples based on discoloration, odors and/or elevated HNu headspace readings.

Soils from Beneath Landfill - Section 2.2.2 describes the collection of 12 samples from beneath the waste mass in the "inactive" area of the landfill. In addition, one soil sample was collected via split spoon sampler from the bottom of each of the MW-7 and MW-8 boreholes during the installation of those wells in Phase 1. These samples were transferred from the split spoon to appropriate sample containers and submitted for the chemical analyses described above and shown on Table 2-1.

2.3 SURFACE WATER AND SEDIMENT

2.3.1 Sampling Methodology - The purpose of the surface water and sediment sampling was to obtain samples for analysis which were representative of the water and sediment quality at each sampling location. Two locations were selected upstream of the landfill and seven downstream. Figure 2-4 shows these locations. Sampling equipment and procedures, decontamination procedures and sample handling followed the protocols established in the Field Sampling Plan (30), approved by the New York State Department of Environmental Conservation (October 1989).

Surface water samples were obtained by approaching the sampling point from a downstream direction and submerging a weighted sample collection container to the required depth. The container was then opened at that depth allowing it to fill completely. After transferring the sample to an appropriate sample container, it was stored on ice until shipment to NYTEST Environmental, Inc. for analysis.

At each surface water sampling location, field analyses of pH, temperature and specific conductance were performed. Phase 2 field analyses also included dissolved oxygen. The Quality Assurance/Project Plan (QAPP) (32) developed for Phase 1 of this investigation required the collection of an equipment blank, a daily trip blank and duplicate surface water samples at 5% of all the surface water sampling points. During Phase 2, the QAPP was modified to require a daily trip blank and a duplicate sample for every 28 water samples collected.

The hand sediment corer used for stream sediment sampling was decontaminated prior to use on-site. Samples were obtained by inserting the corer through the water column into the sediment and penetrating the stream bed with a smooth, constant pressure to the desired depth where it was rotated approximately 90° and then withdrawn in a single motion. After disassembly of the nose piece, the sample was transferred to appropriate sample containers using a decontaminated stainless steel laboratory spoon.

During the October 1991 sampling round, three bottom sediment samples were collected from different areas of Pond B using a Ponar Grab sampler. The sampler was decontaminated between sampling locations as described in the Field Sampling Plan (October 1989). The pond sediment samples were placed in appropriate sample containers and shipped to the laboratory for chemical analyses.

2.3.2 Surface Water Sampling - Phase 1 surface water sampling at locations SW-1 through SW-9 occurred during October 1989. These locations are shown on Figure 2-4. Table 2-1 shows that these samples were analyzed for VOC's, semi-volatile organics, pesticides/PCB's, metals, miscellaneous inorganics and field parameters. In accordance with the Scope of Work (31) for Phase 2 activities at the Gloversville Landfill, three additional rounds of surface water sampling were conducted in August 1991, October 1991, and February 1992.

Surface water samples were collected during August 1991 from locations SW-1 through SW-10 to satisfy NYSDEC's request for a round of samples during the summer. These samples were analyzed only for nitrogen compounds and field parameters as shown on Table 2-1.

Surface water samples collected from 10 locations during the October 1991 sampling round were analyzed by NYTEST Environmental, Inc. for VOC's, semi-volatile organics, pesticides/PCB's, metals, nitrogen compounds, and miscellaneous inorganics, as described in Table 2-1. The Tighe & Bond Environmental Laboratory analyzed the surface water samples for the treatment parameters, Biochemical Oxygen Demand (BOD_5) and Total Suspended Solids (TSS).

During the February 1992 sampling round, each surface water location was sampled and analyzed for an abbreviated list of parameters. These included the nitrogen series compounds, treatment parameters, indicator parameters and field analyses of temperature, pH, specific conductance, and dissolved oxygen. The NYSDEC approved the reduction in analytical parameters based on the fact that very few organic compounds were detected in the surface waters during the two previous sampling rounds.

2.3.3 Phase 1 Sediment Sampling - Phase 1 included the collection of 14 sediment samples from the same locations where surface water samples were collected. During October 1989, samples were collected from differing depths at three of the sampling locations as specified in the Work Plan (33). At sampling locations 3 and 7, sediment was collected from 0-6″, 6-12″ and 12-18″ below the surface of the stream bed. At sampling location 8, it was possible to collect samples from 0-6″ and 6-12″ only; attempts to collect a sample from the 12-18″ range were unsuccessful. At all other sampling locations, sediment was collected from a depth of 0-6″. Table 2-1 shows that these samples were analyzed for VOC's, semi-volatile organics, pesticides/PCB's, metals and miscellaneous inorganics. Following the requirements established in Phase 1 of the Quality Assurance Project Plan (32), duplicate samples were collected for 5% of all the stream sediment sample locations.

2.3.4 Pond B Sediment Sampling - During Phase 1 of the Remedial Investigation of the Gloversville Landfill, Pond B was identified as a groundwater discharge point and surface water leaving this pond contained concentrations of ammonia that exceed background levels. Phase 2 included the sampling and analysis of Pond B bottom sediments. Analyses performed on the three samples collected included VOC's, semi-volatile organics, pesticides/PCB's and metals as shown on Table 2-1.

2.4 MONITOR WELL SAMPLING

One of the key components of the Remedial Investigation is to identify the extent of contaminants in groundwater at the landfill site, if any, and, if present, evaluate potential migration of these contaminants from the landfill. This was accomplished through collection and analysis of 72 groundwater samples from a total of 37 groundwater monitor wells installed around the landfill. The groundwater monitor wells were installed in clusters of 2 to 4 wells, penetrating to different depths of the overburden and bedrock aquifers. The clusters are numbered MW-1 through MW-6, MW-9 through MW-11, and MW-13 through MW-15. (Wells MW-7, MW-8 and MW-12 are leachate monitor wells installed in the waste mass.)

The details of monitor well installation are provided in Section 2.7. The following subsections provide detail on groundwater sampling activities, results of which are described in Section 4.

2.4.1 Groundwater Sampling Methodology - Prior to sampling the monitor wells, the depth to groundwater was measured using either a 100-foot fiberglass tape or an electronic water level indicator. After calculating the volume of standing water in the well, a pre-cleaned, disposable bailer of known volume or a pump was used to evacuate three times the standing water volume. The same bailer was used to collect the groundwater sample once the well had sufficiently recharged. The disposable bailer was then left in the well, above the water table, to be used for purging the well during future sampling rounds. The groundwater samples were placed in appropriate sample containers and were stored on ice in coolers until and during shipment to NYTEST Environmental, Inc.

During the February 1992 sampling, purging the wells was accomplished either by use of the disposable bailer left in the well after the first round of sampling or by pumping with a Waterra pump. Several wells were screened in areas of very slow groundwater recharge. At these locations, the wells were purged dry twice and the approximate recovery rate was calculated. Sample collection occurred once the well had sufficient water to satisfy the sample volume requirements. The sample containers were filled in order of decreasing volatility (eg. VOC's first and metals last).

2.4.2 Phase 1 Sampling

__July 1990__ - Phase 1 groundwater sampling occurred from July 24 - 26, 1990. Groundwater from the 17 wells in clusters MW-1 through MW-6 was collected and analyzed for VOC's, semi-volatile organics, pesticides/PCB's, metals, miscellaneous inorganics, nitrogen compounds, and field parameters as detailed on Table 2-1.

2.4.3 Phase 2 Sampling

__October 1991__ - Phase 2 of the Remedial Investigation included two rounds of groundwater sampling. The first round of sampling began October 21, 1991 and included all the groundwater monitor wells installed during Phase 1 (MW-1 through MW-6). Samples were collected and analyzed for VOC's, semi-volatile organics, pesticides/PCB's, metals, nitrogen compounds, treatment parameters (BOD_5 and TSS), miscellaneous inorganics, and field parameters as shown on Table 2-1.

__February 1992__ - On February 20, 1992, the second round of Phase 2 groundwater sampling began and included all the existing monitor wells (MW-1 through MW-6) and the wells installed during the Phase 2 drilling program (MW-9 through MW-11 and MW-13 through MW-15). Since the data from the two previous rounds indicated that no significant concentrations of organic compounds were detected in the groundwater, the NYSDEC agreed to eliminate the semi-volatile organics and pesticide/PCB's analyses for samples from well clusters MW-1 through MW-6.

Samples of groundwater collected from wells in clusters MW-1 through MW-6 were analyzed for VOC's, metals, nitrogen compounds, treatment parameters (BOD_5 and TSS), and miscellaneous inorganics. The monitor wells installed during the Phase 2 drilling program, including the newly installed bedrock wells that were added to well clusters MW-1 through MW-4, were sampled and analyzed for the full parameter list of analytes as shown on Table 2-1. Field measurements of temperature, pH, specific conductance and dissolved oxygen were made at each well location.

2.5 PRIVATE WELLS

2.5.1 Phase 1 Summary - In the Fall of 1989, Leggette, Brashears & Graham inventoried 138 private water supplies located at residences along the roads surrounding the Gloversville Landfill. The purpose of the inventory was to select upgradient and downgradient private water wells for sampling and analysis. The inventory consisted of a Water Well Questionnaire, which asked for well specifics, such as well depth, age, type of pump, etc. The Private Well Inventory Report was included as Appendix B of the Phase 1 Report (29). Based on the scope of work established in the approved Work Plan for Phase 1 sampling (33), 16 of the 138 private wells inventoried were selected for sampling.

South of the landfill, six private wells along East Fulton Street Extension were chosen for sampling. Although these wells are located upgradient of the apparent groundwater flow direction at the landfill, they were selected because of concerns of potential reverse gradient contaminant migration and groundwater mounding beneath the landfill. Two private wells west of the landfill were chosen for sampling because they are located within the Anthony Creek watershed, but west of, and hydrologically separated, from the landfill by Anthony Creek. One private well was sampled east of the landfill. North (downgradient) of the landfill, seven private wells were sampled.

Residential well sampling began in December 1989. An inspection of the cold water system was made to determine if any in-house treatment devices, such as water softeners or filters, were installed. Water from the kitchen faucet was allowed to run for a minimum of 15 minutes to waste two volumes of the pneumatic holding tank. If any treatment device did exist, the sample was collected prior to the device. Otherwise, the samples were collected at the kitchen faucet.

Field analyses for pH, temperature and specific conductance were conducted at each private well location. A daily trip blank and duplicate samples (5%) were included with the private well sampling in accordance with the QAPP (32).

The Phase 1 results showed that the private wells in the vicinity of the landfill did not contain contaminants at levels above drinking water standards. However, one well located on Blanchard Loop Road, northeast of the landfill, contained ammonia at a concentration above groundwater standards and above levels measured in other private wells. Due to the proximity of the well to the landfill and the knowledge that ammonia is a principal contaminant migrating from the landfill, additional private well testing in the areas north and northeast of the landfill was conducted. Also, the results of the private well sampling south (upgradient) of the landfill have indicated a concern for ammonia levels in that area. The Phase 1 private well sampling locations are labelled PW-1 through PW-16 on Figure 2-5.

2.5.2 Phase 2 Sampling - Phase 2 private well sampling focused on wells screened in the bedrock aquifer in the area of Blanchard Loop Road where the top of the bedrock formation lies below 30 to 60 feet of overburden deposits. Of the private wells sampled during the Phase 2, approximately half were bedrock wells. Six were at depths of 100 feet or greater.

Phase 2 included 2 rounds of private well sampling (October 1991 and February 1992). During both rounds, the sampling protocol established for Phase 1 was maintained. A duplicate sample was taken in accordance with the QA/QC plan established for Phase 2 and a daily trip blank accompanied each shipment to the laboratory.

The private wells sampled during Phase 2 included PW-15 and PW-17 through PW-42. These locations are identified on Figure 2-5.

October 1991 - This round expanded the sampling base from 16 wells in Phase 1 to include a total of 27 private wells. Wells PW-15 and PW-17 were analyzed for the complete parameter list of analytes including VOC's, semi-volatile organics, pesticides/PCB's, metals, nitrogen compounds and miscellaneous inorganics. These were selected for complete scans because of elevated ammonia concentrations measured during the Phase 1 sampling. The remaining 25 wells were analyzed for nitrogen compounds and indicator parameters in addition to the field analytes.

<u>February 1992</u> - Since the Phase 1 study and October 1991 sampling revealed no organic compounds of concern and very few metals at levels of concern, 24 of the wells sampled during the October 1991 sampling were sampled. These included the first round private wells with the exception of two residents who had relocated out of the area and one who could not be reached to schedule an appointment. Analyses performed on these samples included an abbreviated list of analytes: nitrogen compounds and indicator parameters. The field analyses conducted at each private well location included pH, temperature, specific conductance and dissolved oxygen.

2.6 AIR SAMPLING

<u>2.6.1 Phase 1 Ambient Air Sampling</u> - In September 1990, extended duration ambient air samples were collected at the Gloversville Landfill. CC Johnson & Malhotra, PC conducted the sampling at 6 locations around the landfill each day. These locations are labelled GLP-1 through GLP-6 on Figure 2-4.

Particulate matter was collected on filters in sample cassettes and volatile organics were adsorbed in carbon tubes. Air sample pumps were used for sample collection. They were calibrated prior to use and the flow rate was adjusted to between 0.42 and 1.2 liters of air per minute. Sampling on September 27, 1990, continued for approximately 7 hours. After the pumps were shut off, the sampling cassettes and tubes were disconnected from the pumps, sealed, and stored on ice in a cooler. Calibration of pumps was conducted after sample collection to determine variation in flow rate, if any. Before and after flow rates were averaged and multiplied by the sampling duration to determine total air volumes collected.

A second round of samples was collected in an identical manner on the following day. All the samples were shipped to the laboratory on the second day via an overnight express service.

Winds were light on the sampling days. Both mornings were misty with high relative humidity (70-80%). During the morning, the sun burned off the foggy haze and the relative humidity dropped to approximately 50%. There were no landfill operations occurring on the days samples were collected.

__2.6.2 Phase 2 Ambient Air Sampling__ - Volatile organic compounds were essentially undetected during the Phase 1 sampling event and therefore the analysis was not repeated. Laboratory error in Phase 1 resulted in unusable total particulate data. Further, the Baseline Risk Assessment conducted in Phase 1 concluded that the total chromium detected in several of the Phase 1 samples presents an excessive chronic health risk. Therefore, Phase 2 ambient air sampling included analyses for total particulate matter and total chromium.

Extended duration ambient air sampling was conducted on October 29 and 30, 1991, by CCJM. Ten sampling locations were chosen by Tighe & Bond personnel, nine of which were in the immediate vicinity of the landfill. The remaining point was in downtown Gloversville, representing a background location. Sample locations for the landfill sampling points are labelled GL-1 through GL-9 on Figure 2-4.

Samples were collected on matched weight mixed cellulose ester filters (MCEF) in two-piece sample cassettes. Calibrated SKC or Gillian air pumps drew approximately two liters of air per minute through the cassettes for a six-hour period. After the pumps were turned off, the sampling cassettes were disconnected from the pumps, sealed, and placed in a clean plastic bag. Calibration of pumps occurred after sample collection to determine variation in flow rate, if any. Before and after flow rates were averaged and multiplied by the sample duration to determine total air volumes collected.

A field blank, handled identically to the sampling cassettes and opened on-site for a short time period, was included for each day of sampling. One laboratory blank provided a background check for the filters. All samples were shipped to a laboratory on October 31, 1991, via an overnight express service.

Daily weather conditions generally consisted of light winds, low relative humidity and clear, sunny skies. Air temperature, relative humidity and barometric pressure were monitored by CCJM personnel using the weather station set up near the Tighe & Bond field trailer at the south end of the landfill. Wind speed and direction were read from a hand-held indicator.

2.7 HYDROGEOLOGIC STUDIES

The Remedial Investigation (RI) hydrogeologic studies were completed in two phases. Phase 1, undertaken by WTD Environmental Drilling, was completed between May 31 and June 27, 1990. Six pilot boreholes were drilled and 19 overburden monitor wells were installed including two wells installed in the landfill to monitor leachate. These wells were installed as single wells and clusters.

Wells within a cluster were labeled in regard to the depth of the well screen settings. Shallow water-table wells were labeled "S", deep wells installed above the till or bedrock were labeled "D", and any well where the screen was installed between the deep and shallow wells was labeled "I" for intermediate.

Phase 2 drilling was completed between November 4, 1991 and February 14, 1992 by Soil & Material Testing. Twelve overburden, one leachate, and eight bedrock monitor wells were installed as either single wells or clusters. Wells were labeled in respect to the well screen settings as they were in Phase 1. The label "BR" denotes wells in which the screen was installed within the bedrock. The locations of all wells and well clusters installed during the Phase 1 and Phase 2 drilling program are shown on Figure 2-4.

2.7.1 Phase 1 Stratigraphic Logging - Stratigraphic information was obtained from the deep borehole within cluster MW-2, which was drilled using hollow stem augers. Subsequently, on June 5, 1990, the New York State Department of Environmental Conservation approved the drilling of a pilot borehole at each remaining cluster location to obtain stratigraphic profiles. Mud was allowed as a drilling fluid provided that the location of the borehole was at least 25 feet downgradient (or side gradient) from any proposed monitor well location. The purpose of the pilot borehole was to expedite the drilling program, as well as provide stratigraphic information useful in determining well screen settings.

Each pilot borehole was drilled using the mud-rotary technique. A 6-inch temporary steel casing was set approximately 5 to 10 feet in the ground to help prevent the loss of drilling fluid. After the casing was set in place, the pilot borehole proceeded using a 4-inch rotary drilling bit. With few exceptions, split-spoon sampling was continuous from grade until bedrock refusal. In very deep boreholes where the stratigraphy was very consistent (i.e., glacial till), a 5-foot sampling interval was chosen. Other deviations occurred when cobbles were encountered and split-spoon sampling was not possible. Drilling was terminated before bedrock was encountered in borehole B-1 (1990).

Once bedrock was encountered at all other boreholes, a temporary 4-inch casing was lowered down the borehole and set on the bedrock. The casing was then flushed with water and coring commenced. Five-foot core samples were collected using a 5-foot NX-core barrel at each pilot borehole and in the deep borehole of cluster MW-2. All the cores were placed into wooden boxes and labeled. When coring was complete, the temporary 4-inch casing was removed and the pilot borehole was filled with a 15-percent bentonite, 85-percent cement grout slurry. The deep borehole at cluster MW-2 was not backfilled; instead, a deep monitor well was installed. All drill cuttings, including the drilling fluid (mud), were transported in 55-gallon drums to the "active" portion of the landfill for disposal. The empty drums were steam-cleaned for reuse. The Phase 1 drill cutting disposal location is shown on Figure 2-4.

2.7.2 Phase 1 Soil Sampling - During pilot borehole drilling, split-spoon samples were collected for the purpose of geological characterization and determination of well screen settings. Samples were also collected from the boreholes of monitor wells MW-1S, MW-2D, MW-4S and MW-5S. Appendix B presents geologic logs for each of the boreholes in which split-spoon samples were collected.

All split-spoon samples were screened with a photoionization detector (PID), calibrated daily with isobutylene to detect the presence of VOC's. Tighe & Bond field personnel used benzene as a calibration gas, which has a response similar to isobutylene for PIDs. The PID used for screening was an HNU, Model No. HW-101.

The sample screening procedure consisted of transferring a representative portion of the split-spoon sample into a clean glass jar and sealing the jar with aluminum foil. After at least 15 minutes, the aluminum foil seal was punctured with the probe tip of the PID to obtain a headspace reading. All results were recorded in a field book.

2.7.3 Phase 1 Monitor Well Installation - Monitor well boreholes were drilled using the hollow-stem auger drilling method and all wells were completed in separate boreholes. The borehole for Monitor Well MW-2D was drilled with hollow-stem augers and the drive-and-wash drilling method. At each cluster location, the site geologist determined the monitor well screen depths based upon the stratigraphic information obtained in the adjacent deep pilot borehole. The monitor well boreholes were drilled with 7.5-inch o.d. (outside diameter), 4.25-inch i.d. (inside diameter) hollow-stem augers. Cluster MW-6 was drilled using 10.5-inch o.d. hollow-stem augers.

WTD advanced the monitor well borehole with a wooden knock-out plug in the lead auger to prevent sediment from entering the hollow stem. When the borehole reached the desired screen depth, the plug was knocked out with a string of drill rods and a split-spoon sample was collected for sieve and hydrometer analysis. The plug was then abandoned in place.

The shallow monitor wells consist of 10 feet of screen extending from approximately 4 feet above to 6 feet below the water table. The deep monitor wells have 10 feet of screen located immediately above the till or the bedrock, if no till was present. The intermediate wells are also screened over a 10 foot length at a depth between the shallow and deep screen zones.

The screen material for all of the wells, excluding clusters MW-4 and MW-6, consists of pre-cleaned, factory-sealed, 2-inch diameter PVC in 5-foot lengths with 0.010-inch slot openings and flush-joint threads. A 6-inch sediment trap is attached to the bottom of the screen. The PVC screen was set within 1 foot of the bottom of the borehole with sufficient pre-cleaned, factory-sealed, 2-inch diameter PVC riser pipe to extend from the top of the screen to approximately 2 feet above the ground surface. Monitor wells MW-2D and MW-7 have a bentonite and/or cement grout backfill from the bottom of the borehole to approximately 1 foot from the bottom of the well screen.

Monitor well cluster MW-6 is constructed in the same manner as above, except that the well screen and riser pipe are 4-inch PVC. This cluster also lacks a sediment trap at the bottom of the well screen.

The screen material for cluster MW-4 is pre-cleaned, factory-sealed, 2-inch diameter, stainless-steel, wire-wound screen in 10-foot lengths with 0.010-inch slot openings and flush-joint threads. The stainless-steel screen for each well in the cluster is set 1 foot from the bottom of the borehole with sufficient pre-cleaned, factory-sealed, 2-inch diameter stainless-steel riser pipe to extend from the top of the screen to approximately 2 feet above the ground surface.

Once the well screen and riser pipe of each well were placed into the borehole, the hollow-stem augers were slowly withdrawn as the annular space was backfilled with sandpack, bentonite and grout. No. 2 quartzite sandpack fills the annular space around the wells from below to at least 1 foot above the screen. A bentonite seal consisting of Volclay powder or bentonite pellets at least 1-foot thick lies immediately above the sandpack. Volclay powder was pumped down the borehole using a Tremie pipe. Where pellets were used, they were allowed to fall freely down the borehole. The remaining annular space is filled with a 15-percent bentonite, 85-percent cement grout slurry, which was pumped down the borehole through a Tremie pipe.

A 4-inch diameter, 5-foot long steel casing with a locking cap secures each well location. Wells in cluster MW-6 have 6-inch diameter steel casings. A cement pad, approximately 2.5 feet in diameter and 1-foot thick, was constructed around the security casing in such a way as to direct surface runoff away from the casing. The casings are painted bright orange for rust protection, as well as easy visibility, and labeled with lettering to identify the well, i.e., MW-2S. A summary of well construction details is presented in Table 2-2. Phase 1 wells are numbered MW-1 through MW-8. Appendix C provides well construction diagrams for each monitor well.

2.7.4 Phase 1 Piezometer Installation - On June 24, 1990, four piezometers were installed to provide additional water-level monitoring locations for the Phase 1 pumping tests that were conducted at cluster MW-6. All piezometer boreholes were drilled using 7.5-inch o.d., 4.25-inch i.d. hollow-stem augers. A wooden plug installed in the lead auger prevented sand from entering the auger. At the desired screen depth, the plug was knocked out with a string of drill rods and piezometer installation was undertaken.

The 10 foot piezometer screen extends from approximately 4 feet above to 6 feet below the water table. The screen material consists of pre-cleaned, factory-sealed, 2-inch diameter PVC, with 0.010-inch slot openings and flush-joint threads. Pre-cleaned, factory-sealed, 2-inch diameter PVC riser pipe extends from the top of the screen to approximately 2 feet above the ground surface. The annular space around the well screen is filled with graded No. 2 quartzite sandpack from below the well screen to at least the top of the well screen. A bentonite seal is emplaced on top of the sandpack in all of the piezometers, except for PZ-4, in which the sandpack ends at grade because there was insufficient distance between the top of screen and the ground surface. The remaining annular space is backfilled with the drill cuttings. No protective security casings are provided on piezometers. Table 2-3 presents a summary of piezometer construction details. Phase 1 piezometers are numbered PZ-1 through PZ-4. Piezometer construction diagrams are included in Appendix C.

2.7.5 Phase 1 Monitor Well Development - Monitor well construction was not considered complete until each well was properly developed. Between June 24 and 29, 1990, all of the wells installed during the Phase 1 drilling program were developed using one of four different methods, the most common being the nitrogen airlift. Other methods included hand-bailing, rig-mounted bailing, and use of a hand-operated Brainard Kilman pump. Any well development equipment, including any tubing that was in contact with the well water, was cleaned with Alconox (a detergent) and steam-rinsed to avoid cross-contamination. Periodic surging of each well created turbulence within the screen zone which cleared the well screen and sandpack of fine material and stabilized the formation material immediately surrounding the well screen. All evacuated water was pumped into 55-gallon drums to measure volume and to facilitate transport to a discharge pit. A discharge pit located at the base of the slope on the east side of the "active" portion of the landfill received all development water. Figure 2-4 shows the location of the discharge pit. Water color, pH, conductivity and turbidity were periodically monitored during pumping. Development time per well was limited to eight hours, or less, if the turbidity levels had dropped to 50 NTUs (nephelometric turbidity units) or below. Due to the silty nature of the sediments in which some of the wells are screened, 50 NTU turbidity levels were not achievable, even after eight hours of development. Table 2-4 summarizes well development information.

The piezometers installed during the Phase 1 drilling program were developed by hand with a bailer. Development was considered complete when all the sediment that had collected at the bottom of the piezometer had been removed. Color, pH, electrical conductivity and turbidity were not monitored.

2.7.6 Phase 2 Soil Sampling - Split-spoon sampling conducted at each cluster location provided a means to characterize the site geology. Samples were collected from the deep borehole at clusters MW-9, MW-10, MW-11 and MW-13. Samples were also collected from the boreholes of MW-9I, MW-12, MW-14S and MW-15BR. A 5-foot sampling interval was chosen for the intermediate and deep boreholes in Cluster 9. Continuous sampling in advance of the borehole occurred at all other well locations. Appendix B presents geologic logs for the boreholes.

The split-spoon samples were screened for total ionizable compounds (VOCs) with an HNU PID, Model No. HW-101 which was calibrated daily with isobutylene. All soil samples collected during the Phase 2 drilling program were screened in the same manner as described in Section 2.7.2, Phase 1 Soil Sampling.

2.7.7 Phase 2 Overburden Monitor Well Installation - Soil & Material Testing (S&MT) used the hollow-stem auger drilling method to complete all monitor well boreholes except MW-11D. Most of the wells were completed in separate boreholes that were drilled with 4.25-inch i.d., 8-inch o.d. hollow-stem augers. The boreholes for cluster MW-9 were drilled with 8-inch i.d., 12-inch o.d. hollow-stem augers. The monitor well boreholes were advanced with a drag bit attached to the rods to prevent sediment from entering the augers. The borehole for monitor well MW-11D was advanced using the drive-and-wash drilling method and 4-inch temporary steel casing. The onsite geologist determined monitor well screen depths based upon the stratigraphic information obtained from the split-spoon soil samples. When the desired borehole depth was attained, monitor well installation was undertaken.

Ten feet of screen extends from approximately 5 feet above to 5 feet below the water table in the shallow monitor wells. The deep monitor wells are installed immediately above the till, if present, or above the bedrock. Monitor well MW-12 is installed in waste material with 15 feet of screen extending approximately 7 feet above to 8 feet below the water level encountered in the landfill.

The screen material for most of the monitor wells consists of factory-sealed, 2-inch diameter, 0.020-inch slot, flush-joint PVC. Monitor wells MW-13S and MW-14S are constructed with 0.010-inch slot screen. Monitor wells MW-9S and MW-9I were constructed with 6-inch, 0.020-inch slot, PVC screen. Factory-sealed, 2-inch diameter (6-inch diameter for monitor wells MW-9S and MW-9I), flush-joint PVC riser pipe extends from the top of the screen to approximately 2 feet above grade.

As the hollow-stem augers were withdrawn, the annular space from approximately 1 foot below the well screen to approximately 1 foot above the well screen was filled with sandpack. No. 1 sandpack surrounds wells with a 0.010-inch slotted well screen and No. 2 sandpack encircles wells with a 0.020-inch slotted screen. A bentonite seal was emplaced on top of the sandpack by allowing bentonite pellets to fall freely down the borehole. The thickness of the bentonite seal varies from 1 to 3 feet. For the deep monitor wells and monitor well MW-14S, a 15-percent bentonite, 85-percent cement grout slurry fills the annulus from the top of the bentonite seal to approximately 2 feet below grade.

A locking 4-inch diameter, 5-foot long steel casing secures each well location. Monitor wells MW-9S and MW-9I required an 8-inch steel security casing. A concrete pad, approximately 2 feet thick and 2 feet in diameter, surrounds each security casing and is mounded to direct surface water away from the casing. The wells are painted for weather protection and labeled for identification. Table 2-2 presents a summary of well construction details. Construction diagrams for each monitor well are included in Appendix C.

<u>2.7.8 Bedrock Monitor Well Installation</u> - One of two different drilling techniques was used to drill through the unconsolidated material at each bedrock borehole location. The air-rotary technique was the most common. S&MT drove twenty-foot sections of 6-inch steel casing into the ground and removed the material within the casing with compressed air that was injected through a 5 7/8-inch roller bit. Additional 20-foot sections were then threaded and welded onto the section(s) already driven into the ground, and the process was repeated until bedrock was encountered. The drillers then drove the steel casing approximately 1 foot into the bedrock and advanced the borehole 3 to 5 feet below

the bottom of the casing with the 5 7/8-inch roller bit. A thick, dense till prevented the 6-inch steel casing from being advanced to bedrock in monitor well MW-1BR. It was completed by continuing to drill using the rollerbit without advancing casing.

The second technique used to drill through the unconsolidated material at the bedrock borehole locations was hollow-stem auger drilling. The borehole was advanced through the overburden aquifer and the underlying till to bedrock refusal using 4.25-inch i.d., 8-inch o.d. hollow-stem augers, and withdrawn. Twenty-foot threaded sections of 6-inch steel casing were lowered down the borehole until bedrock was encountered. The drillers drove the casing approximately 1 foot into the bedrock with an air hammer and then advanced the borehole 3 to 5 feet below the casing with a 5 7/8-inch roller bit.

Once the 6-inch steel casing was set into the bedrock, the borehole was flushed with a thick drilling mud to remove all drill cuttings. Subsequent flushing with water removed the drilling mud. When all debris had been removed from the borehole, coring commenced through a 4-inch temporary steel casing set on the bedrock. Coring was accomplished with a 5-foot NX core barrel (3 inches in diameter). Coring proceeded at least 20 feet or until sufficient water was found, but coring was limited to 100 feet. All core sections collected during the Phase 2 drilling program are stored in wooden boxes.

The entire cored section of most boreholes is screened with factory-sealed, 2-inch diameter, 0.020-inch slot PVC screen. Monitor wells MW-2BR and MW-15BR are installed with 0.010-inch slotted screen. Factory-sealed, 2-inch diameter, flush-joint PVC riser pipe extends from the top of the well screen to approximately 2 feet above grade. The annular space from the bottom of the corehole to approximately 2 feet above the top of the well screen is filled with sandpack. A No. 1 sandpack is used in Monitor Wells MW-2BR and MW-15BR; all other bedrock wells are filled with No. 2 sand. A seal was then emplaced on top of the sandpack by allowing bentonite pellets to fall freely down the borehole. The bentonite seal extends up inside of the 6-inch steel casing at all wells except MW-1BR, where a grout seal extends from the top of the bentonite seal to the surface (Table 2-5). A 15-percent bentonite, 85-percent cement grout fills the remaining annular space above the bentonite seal. A locking cap installed on the 6-inch steel casing secures the well. A cement pad approximately 2 feet in diameter and 2 feet thick constructed around the well casing directs surface water away from the casing. The well casing is painted for weather protection and labeled for identification. A summary of bedrock well-construction details is presented on Table 2-5. The bedrock monitor well construction diagrams are attached in Appendix B.

2.7.9 Phase 2 Piezometer Installation - During the Phase 2 drilling program, eight additional piezometers were installed north of the "active" portion of the landfill. The Phase 2 piezometer numbering sequence continues with that already established during Phase 1, i.e. PZ-5 through PZ-12.

Piezometers PZ-5 and PZ-6 are located within 200 feet of cluster MW-9 so changes in the groundwater elevation could be monitored during the extended pumping test. Piezometers PZ-7 through PZ-12 were installed in boreholes installed in the "inactive" area, north of the fire road in the northwest area of the property.

Piezometer boreholes were drilled using either 4.25 inch i.d. or 2.25 i.d hollow stem augers, depending on the difficulty of drilling through fill material. With the exception of PZ-6, all piezometers are constructed with screens which intersect the watertable. The screened interval of PZ-6 extends from 32 to 42 feet below grade, in a unit of medium to coarse sand and gravel. This unit corresponds in elevation and is believed to be hydraulically connected to the coarse-grained unit screened at well MW-9I.

The Phase 2 piezometers are completed with 10 feet of 1-inch diameter, flush-jointed PVC 0.20-inch screen and riser pipe. At each piezometer location, the borehole was advanced through the entire thickness of fill material. Once the bottom of the fill was encountered the borehole was backfilled with No. 2 quartzite sandpack to the desired depth for setting the bottom of the well screen.

No. 2 quartzite sandpack fills the annular space surrounding the screen and the riser pipe to the ground surface. PVC slip caps cover the top of the riser pipe. No protective steel casings were installed around piezometers. Table 2-3 summarizes peizometer construction details.

2.7.10 Phase 2 Monitor Well Development - Between December 17, 1991 and February 14, 1992 all of the overburden and bedrock monitor wells which were installed during the Phase 2 drilling program were developed using three different development methods. The primary means was the airlift method. A trailer-mounted air compressor supplied the air for development, which forced air down the well through a 1-inch rigid plastic tube into the screen zone. The air created turbulence which removed fine material from the sandpack and stabilized the formation immediately surrounding the well screen.

When recharge to the well was low, or if the depth to water was too great for airlift, a hydrolift pump and bailer were used for development. Periodic surging during bailing created turbulence which aided in stabilizing the surrounding formation.

During development, pH, electrical conductivity, temperature and turbidity were monitored. Development time was limited to 8 hours or less, if the turbidity levels had dropped below 50 NTUs and the pH, electrical conductivity and temperature had stabilized. The volume of water removed from each well was measured using a 5-gallon pail and/or a 55-gallon drum. All development water subsequently was discharged onto the ground adjacent to the well being developed. Table 2-4 summarizes Phase 1 and 2 well-development information.

During the development of monitor well 9S, a significant amount of sandpack material was present in the discharge. Examination of the well indicated that the screen section had become separated from the riser pipe allowing the sandpack to enter the well. Frost heave is believed to be the cause of the separation. To save the well, a 2-inch diameter PVC well screen and riser pipe assembly was installed inside of the 6-inch well.

2.7.11 Results of PID Screening - A majority of the sample screening results obtained during Phase 1 were below 1 ppm (part per million). The highest reading occurred in the 61 to 63-foot split-spoon sample collected from Borehole B-1(1990) with a 40 ppm reading. However, this reading appears to be erroneous due to very humid weather conditions resulting in erratic PID readings. Elevated readings were also found in the deep borehole in cluster MW-2 from 8 to 26 feet below grade and 44 to 50 feet below grade. These zones are located above less permeable silt and clay bands. Boring P-6, at cluster MW-6, also showed higher than average site readings from 2 to 12 feet below grade. Elevated PID readings were found in the headspace of soil samples collected during the Phase 2 drilling event from boreholes MW-9I and MW-12; however, both of these boreholes penetrated the landfill. Boring MW-11D also showed higher than site average readings in the 1 to 2-foot split-spoon sample. A summary of the Phase 1 and Phase 2 PID screening results is found in Table 2-6.

2.7.12 Phase 1 Pumping Tests - Three separate 24-hour pumping tests were conducted from June 28 to July 2, 1990, at cluster MW-6 to determine aquifer transmissivity and storativity values. One test was run on each of the shallow, intermediate and deep monitor wells using a stainless-steel and Teflon Grundfos submersible pump rated at 20 gpm (gallons per minute). Power was supplied by a portable gasoline generator. The pump discharge line consisted of 575 feet of 1¼" black polyethylene tubing with a gate valve installed in line to regulate discharge.

The discharge pit shown on Figure 2-6 received the discharge water from the pumping tests.

All groundwater measurements were collected using an electrical water-level indicator and recorded to the nearest one-hundredth of a foot. The water-level indicator was cleaned between monitoring wells with Alconox and rinsed with deionized water to avoid the possibility of cross-contamination. Measurements were periodically collected during each of the 24-hour pumping tests and recorded.

2.7.13 Phase 2 Pumping Test - A 48-hour pumping test was conducted from March 24 to 26, 1992 on well MW-9I at a pumping rate of 100 gpm. The test was originally scheduled to run for 72 hours; however, delayed drainage and the high transmissivity of the sediment limited the stress on the aquifer. A longer pumping period of approximately one week to 10 days would be required to reach steady state. The test was then shut down 24 hours early because no more useful information would have been obtained during the last day.

Marcor of New York supplied and monitored the pump and generator. They used a submersible Grundfos pump, powered by a trailer-mounted generator. Six hundred feet of 3″ diameter collapsible hose directed the discharge water to a bermed area on top of the landfill near monitor well MW-7 (Figure 2-6). A valve and flow meter that were installed in line regulated and measured the discharge rate.

Data loggers recorded water-level changes during the pumping test in clusters MW-3, MW-5, MW-6 and MW-9. Cluster MW-3 was monitored to establish background trends. Additional measurements were collected manually with an electric water-level indicator from Piezometers PZ-1, PZ-2, PZ-3, PZ-4, PZ-5 and PZ-6. All manual measurements were recorded to the nearest one-hundredth of a foot.

A pumping test was also proposed for MW-9S; however, the 6-inch well was damaged by frost action. The 2-inch well, which was installed inside the 6-inch well, precluded the use of a large enough pump to provide meaningful data.

2.7.14 Permeability Testing - Short duration rising head "slug tests" were performed on all of the monitor wells installed during the Phase 1 and 2 drilling programs. These tests were used to determine the hydraulic conductivity of the material immediately surrounding the well screen. These tests are useful in determining an average hydraulic conductivity values for the various types of overburden encountered at the site.

Water level changes during the "slug tests" were measured and recorded using an ORS DL-150 data logger and pressure transducer. Prior to the start of each test, a depth to water-level measurement was made using a clean, weighted steel tape. The transducer was lowered 5 to 8 feet below the static water level and the cable was secured to the well casing with duct tape. The slug was then lowered into the well below the static water level. When the water level in the well returned to static conditions, the data logger sequence was begun and the slug was quickly removed from the well. The results of these tests are discussed in Section 3.4.4.

2.7.15 Grain-Size Analysis - LBG collected a soil sample from the screen zone of each monitor well during installation for the purpose of determining grain-size distribution. Fourteen (14) samples were chosen for grain-size distribution by sieve and hydrometer analyses. Lack of sufficient material prevented a hydrometer analysis on the sample from MW-6S. Table 2-7 lists the depth at which the samples were collected.

Results of the sieve analysis, specifically the cumulative percent retained by weight, have been graphed. The resulting graphs show the variability in grain size distribution of the site soils. This is a reflection of the stratified nature of the soils. Appendix D contains all of the data sheets and graphs of the sieve and hydrometer analyses.

2.7.16 Stream Gauging - Stream gauging was conducted during Phase 1 at various points along Anthony Creek and one of its tributaries, as shown on Figure 2-7. A pygmy flow meter was used in determining stream velocities at each measurement location. At each gauging point, a section of stream was located that was free of backeddies and clear of debris, which can result in false calculated velocities.

The stream width was measured perpendicular to its bank and then divided into various segments at each selected location. The shallow nature of the stream dictated the use of the six-tenths method, which requires the flow meter to be situated 0.6 times the total depth of the water from the water's surface. This is the theoretical point at which average velocity occurs. Meter revolutions were counted, usually for one minute, and recorded. The flow meter is designed so that velocity in feet per second equals the number of revolutions divided by 60. To determine the discharge per segment, the formula $q_i = v_i d_i w_i$ was used (Fetter, 1986). Discharge per segment q_i is equal to the average velocity per segment v_i, times the depth per segment d_i, and times the width per segment w_i. The total discharge for the stream is calculated by summing the discharge in each segment as $Q = q_i + q_2 + \ldots + q_n$, where n equals the total number of segments. Section 3.4.11 presents the results of the stream gauging.

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SECTION 3
PHYSICAL CHARACTERISTICS OF STUDY AREA

The Phase 1 investigation resulted in a reasonable understanding of the hydrogeologic framework and groundwater flow patterns in the landfill vicinity. However, both the hydrogeologic data and water quality results indicated that the bedrock aquifer was also potentially being effected by landfill leachate. As a result, the Phase 2 investigation was undertaken to investigate the bedrock, further define flow patterns in the overburden and determine the interrelationship of the aquifers. The Phase 2 investigation included an extensive pumping test to further confirm preferential flow paths and to investigate the feasibility of groundwater extraction as a possible remediation technology.

The combination of Phase 1 and 2 activities has resulted in a conceptual model of groundwater movement at the site. The conceptual model was supplemented and confirmed by a computer simulation of the overburden and bedrock system. The good correlation of the interpretations of field results and the computer model further verify that the model can be used for predictive simulations with a fair degree of confidence.

The general conceptual model of groundwater movement is an attempt to focus on major pathways of flow and to simplify, to the degree possible, what is, in reality, a complex system. Groundwater in the landfill vicinity occurs under water-table (unconfined) conditions in the shallow sediments, under semiconfined conditions in the deeper sediments, and under both confined and semiconfined conditions in the bedrock.

Preferential flow paths have been identified in the unconsolidated deposits, and it is believed that much of the leachate generated by precipitation infiltrating the landfill is channeled to surface water via a buried stream channel.

There are indications, however, that some of the groundwater and associated containments are migrating into the bedrock aquifer through areas where glacial till has been eroded from the bedrock surface. Some of these eroded areas have been confirmed through the boring work while others are suspected, based on an understanding of the glaciation of the area.

Based on geophysical surveys and bedrock drilling, it is also believed that there may be preferential flow paths in the bedrock coinciding with the regional fracture pattern. The investigations and computer simulations show that, ultimately, most of the groundwater in the landfill vicinity discharges to surface water in the downgradient areas, with some bedrock and overburden underflow continuing to the northeast.

The following sections describe the results of the site investigations, and further develop the understanding of the site hydrogeology.

3.1 SURFACE FEATURES / TOPOGRAPHY

The Gloversville Municipal Landfill is located within the area covered by the United States Geological Survey (USGS) Gloversville, New York Quadrangle, about three quarters of a mile east of the Gloversville city limits, north of East Fulton Street Extension (Route 29A). Topography has been significantly altered by landfill activities since the turn of the century. The highest elevations are found near the entrance to the landfill along Route 29A, at approximately 930 ft msl (feet above mean sea level). Land surface elevations generally decrease toward the north with the lowest elevations occurring along the northern site boundary at approximately 820 ft. msl. The landfill is tiered with the highest plateau at 890 ft. msl. The southern two-thirds of the site is dry and well drained, while the northern third is generally poorly drained with a small stream system, ponds and wetland areas.

3.2 SURFICIAL GEOLOGY

The last glacial epoch, known as the Wisconsinan glaciation, ended approximately 10,000 years ago. This last period of glacial activity is responsible for the deposition of the unconsolidated sediments which are found onsite. The first material deposited, which lies directly on top of the bedrock, was a very compact, clay-rich glacial till (lodgement till) with a thickness ranging from zero to over 65 feet. This till was deposited beneath the ice sheet under pressure during one of the glacial advances. These ice advances, which were responsible for the deposition of the lodgement till, were also responsible for removing sediment from portions of the bedrock surface. The thickest till accumulation is found in the southern site area. It thins out toward the north where it becomes a discontinuous mantle. It is speculated that the till may also be absent along the bedrock ridge under parts of "active" area. The absence of till in the vicinity of cluster MW-4, which is also a bedrock high, supports the idea that the glacial ice may have scraped material off of the bedrock ridge as it moved from the northeast to the southwest. These areas, where no confining layer exists, allow a direct hydraulic connection between the bedrock and overburden aquifers.

Clay-rich till, such as that found on site, is known to fracture and have active groundwater movement to a depth of approximately 30 feet (Ruland, Cherry and Fenestra, 1991). Although no fractures were found during the drilling program, fully penetrating fractures may be present in areas where the till is less than 30 feet in thickness. This would virtually include the entire fill area from the uppermost tier of the landfill to the northern site boundary. These fractures may act as another pathway to the bedrock aquifer for the downward migration of groundwater and any associated landfill contaminants.

As the glacier retreated, its meltwater streams carried silt, sand and some gravel which was deposited across the site area on top of the compact glacial till and the exposed bedrock. The material deposited varied in thickness from 10 to 70 feet and is generally described as stratified drift. These deposits have been described as Kame Moraine, as shown on the 1987 Surficial Geologic Map of New York, Hudson Mohawk Sheet, a portion of which is shown on Figure 3-1. The sand and silt were deposited as bands and lenses with occasional coarser zones where the stream velocity was greater.

In some cases, the glacial streams did not deposit material, but eroded the material it had previously deposited. Evidence of this is found in the soil samples collected from the borehole of MW-11D. The till is absent above the bedrock at this location, but gravel was found indicating that this may have been a stream channel at one time. Areas where till is absent due to stream erosion provide another direct hydraulic connection between the overburden and bedrock aquifers.

Ponding was also common in the site area during glacial retreat as evidenced by silt and clay accumulations. This is common in the northwest part of the site area in the vicinity of cluster MW-3.

Multiple events of glacial advance and retreat result in very complex deposits of till and stratified drift. One glacial advance may completely remove material that the glacier deposited in a previous episode and redeposit material at a later time. Complex interrelationships develop between the boundaries of differing sediment types which, on a small scale, are virtually impossible to map. Resulting groundwater movement through these glacial sediments is also complex. Groundwater moves more quickly through sandy zones than it does in silty zones, thus leading to preferred paths of flow which are difficult to follow. One particular dominant path of flow is found in the coarse sand and gravel zone found at cluster MW-9.

These various glacial deposits are illustrated on four cross sections which are based upon the geologic logs recorded during the Phase 1 and 2 drilling programs. The locations of the cross sections are presented on Figure 3-2. The cross section of lines A-A′, B-B′, C-C′ and D-D′ are presented as Figure 3-3 through 3-6.

3.3 BEDROCK GEOLOGY

3.3.1 General Description - Underlying the unconsolidated stratified drift and till is the Utica Shale, a Middle Ordovician formation of the Mohawk Series. This is shown on the regional bedrock geology map, a portion of which is reproduced as Figure 3-7. The literature describes the Utica Shale as a brownish-dark gray, weathering black, platey shale with small amounts of pyrite. This formation was formerly cited in the literature pertaining to this site as the Canajoharie Shale.

A portion of the Preliminary Brittle Structures Map of New York, Hudson-Mohawk Sheet (Isachsen and McKendree, 1977), is shown on Figure 3-8. The figure indicates a general paucity of information on brittle structures in the Gloversville area due to the blanketing glacial deposits. However, a prominent set of northeast trending, high angle normal faults bounds the study area to the northwest and southeast. In general, the orientation of these high angle faults parallels the orientation of lineaments depicted in the Precambrian rocks to the north and west. Although no structural measurements of bedrock features could be made directly in the study area, the geophysical study conducted during Phase 1 of the RI identified a bedrock surface consisting of low relief ridges and troughs. The orientation of the elongated bedrock lows corresponds to the regional orientation of large-scale lineaments. The preferred orientation of the bedrock lows identified on the bedrock elevation contour map (Fig. 3-9) may reflect a structural feature formed in response to a regional, tectonic stress or a reworking of a less competent section of bedrock by glacial activity.

Core samples of bedrock collected during the Phase 1 and 2 drilling activities confirmed the rock type. The cores consisted of black, massive shale. The upper 3 to 5 feet of bedrock exhibited signs of physical weathering, such as sharp, angular rock fragments surrounded by a matrix of silt and clay. Evidence of chemical weathering such as mineralogical alterations was scarce. The degree of saturation in this upper bedrock zone probably reflects the high density of fractures. RQD analyses suggest that deeper, through-going fractures are less common. Attempts to collect bedrock cores in this upper zone resulted in the core barrel becoming seized. Therefore, subsequent coring was initiated only after drilling through the fractured zone with a roller bit.

<u>3.3.2 Rock Quality Designation</u> - A rock quality designation (RQD) interpretation was performed on the bedrock cores collected during the Phase 2 drilling program. "RQD is an index or measure of the quality of a Rock Mass," (Bowles, 1982). The RQD is equal to the total length of recovered core pieces, 4 inches or more in length divided by the total length of the core run. Competency of a rock generally increases as the RQD approaches 100 percent. The RQD results presented on Table 3-1 were typically greater than 90 percent, and in many cases 100 percent, indicating that the shale bedrock underlying the site is very competent. These results, however, do not include the upper 3 to 5 feet of bedrock where coring was not possible.

Lower RQD results were calculated from the core samples collected from borehole MW-3BR indicating that the rock at this borehole location is more fractured than elsewhere on site. The location of MW-3BR is situated within a northeast, southwest-trending bedrock low. RQD's calculated at MW-1BR also indicate a greater degree of fracturing than at most other locations on site. This cluster is also located in a northeast, southwest trending bedrock low. These results tend to support the theory that the bedrock lows correspond to areas of increased fracturing. It then follows that the bedrock lows are probably indicative of the dominant orientation of the regional fracture pattern.

3.4 HYDROGEOLOGY

3.4.1 Phase 1 Aquifer Testing Results - Aquifer characteristics, including transmissivity (T) and storativity (S) were calculated for each monitor well within cluster MW-6. For time-drawdown relationships, transmissivity was calculated using:

$$T = 264Q/\Delta S$$

Where:

264 = a constant;
Q = is the pumping rate in gpm; and
ΔS = the slope of the time drawdown graph over one long cycle.

Storativity values were obtained by:

$$S = 0.3Tt_o/r^2$$

Where:

0.3 = a constant;

T = the transmissivity in gpd/ft (gallons per day per foot);

t_o = is the intercept of the straight line at zero drawdown in days; and

r = the distance in feet from the pumping well.

Distance drawdown relationships were also used to calculate transmissivity (T) and storativity (S). Transmissivity was calculated by:

$$T = 528Q/\Delta S$$

Where:

528 = a constant;

Q = the pumping rate in gpm; and

ΔS = the slope of the distance drawdown graph over one long cycle.

Storativity values were determined by:

$$S = 0.3Tt/r_o^2$$

Where:

0.3 = a constant;

T = the transmissivity in gpd/ft; and

t = the time in days since pumping began as chosen from the time drawdown graphs; and

r_o = the intercept of an extended straight line at zero drawdown.

Monitor well MW-6D was pump tested on June 28 and 29, 1990. The rate was held constant at 10 gpm until 1,170 minutes into the test when the rate was increased to 20 gpm. The data generated after the increase were ignored for aquifer characteristic calculations because the equations used in the determination of T and S values are only valid for constant-rate pumping situations. Appendix E contains all of the data collected during the pumping test of monitor well MW-6D, including water-level measurements, pumping rates and graphs.

Time-drawdown relationships were used for all of the observation wells, except for monitor wells M-6S, MW-6I, MW-5S, MW-5I and MW-5D. The stratified nature of the sediments caused a greater horizontal-flow component and a lesser vertical-flow component in the immediate vicinity of the pumping well. This caused MW-6S and MW-6I to respond poorly to the pumping test, resulting in unreasonable storativity values. Monitor wells MW-5S and MW-5I were not included because they did not respond to the pumpage of MW-6D. Data from MW-5D yielded unrealistic S&T values. Transmissivity values calculated from the remaining data obtained at MW-6D ranged from 30,000 to 50,800 gpd/ft, with an average of 40,800 gpd/ft. Storativity values ranged from 0.0032 to 0.0057, with an average of 0.005.

A distance-drawdown relationship was also used to determine T and S values, based on a time of 1,000 minutes obtained from each of the time drawdown graphs. Using this relationship, transmissivity was calculated to be 44,000 gpd/ft and storativity to be 0.0032. Table 3-2 summarizes the aquifer characteristics derived from analysis of the data obtained from the pumping test of MW-6D.

Monitor well MW-6I was pump tested on June 30 and July 1, 1990 at a constant rate of 22.2 gpm. All data collected during the pumping test are found in Appendix E, including water-level measurements, pumping rates and graphs. Time-drawdown relationships were used to determine transmissivity and storativity values. Again, data from MW-5D yielded unrealistic T&S values. Transmissivity values calculated from the remaining data ranged from 26,700 to 83,800 gpd/ft, with an average of 48,500 gpd/ft. Storativity values ranged from 0.0002 to 0.0276, with an average of 0.0083.

A distance-drawdown relationship was also analyzed to determine T and S values. These are based on a time of 1,000 minutes obtained from each of the time drawdown graphs, except for MW-6D which was affected by the stratified nature of the sediments. As in the previous test, the horizontal flow components were greater than the vertical flow components. A longer testing time would be required to impact the deeper screen setting of MW-6D. Transmissivity was calculated to be 24,400 gpd/ft and storativity to be 0.141. Table 3-3 summarizes the aquifer characteristics of MW-6I.

Monitor well MW-6S was pump tested on July 1 and 2, 1990 at a rate of 4 gpm. The rate was increased to 6.67 gpm 1,127 minutes into the test, but the latter data were not used in determination of aquifer characteristics, because the equations used to calculate T and S values are designed for constant-rate pumping situations. Appendix E contains all of the data collected during the pumping test of Monitor Well MW-6S, including water-level measurements, pumping rates and graphs. Time-drawdown relationships were used for all of the observation wells, with the exception of MW-6D, in determination of transmissivity and storativity. As in the pumping test of Monitor Well MW-6I, stratified drift prevented any significant vertical movement of water in the short amount of time that the test was run. Transmissivity values ranged from 8,800 gpd/ft to 23,500 gpd/ft, with an average of 16,900 gpd/ft. Storativity values ranged from 0.0013 to 0.0698, with an average of 0.018.

A distance-drawdown relationship was used to determine transmissivity and storativity values. Data based on a time of 1,000 minutes was obtained from each of the time-drawdown graphs, with the exception of the graph of MW-6D data. Transmissivity was calculated to be 25,400 gpd/ft and storativity to be 0.0016. Table 3-4 summarizes the aquifer characteristics for the pumping test of MW-6S.

Transmissivity results obtained from the three pumping tests ranged from 9,000 to 83,800 gpd/ft. When these values are divided by aquifer thickness to arrive at hydraulic conductivities, the results compare favorably with published ranges for fine to coarse sand and the upper range of silt. The wide range of calculated values reflect the stratified nature of site soil which consists of sand and silt.

3.4.2 Phase 2 Aquifer Testing Results - Monitor well MW-9I was pump tested for a 48-hour period at a constant rate of 100 gpm during March 24, 25 and 26, 1992. Aquifer characteristics, including transmissivity (T) and storativity (S) were calculated using a time-drawdown relationship, using the equations presented in Section 3.4.1. Appendix E contains all of the data collected during the pumping test of MW-9I.

Only PZ-6, which is screened in the same zone as the pumping well, MW-9I, was used for determination of (T) and (S) values. The remaining 14 wells and piezometers that were monitored could not be used for the calculation of aquifer characteristics. Monitor well MW-7 and Piezometer PZ-5 were not used because they were not impacted by the pumping. The wells in cluster MW-5 and MW-6, well MW-9S, MW-9D and piezometers PZ-1 through PZ-4, were not used because they did not vary in response to the pumping due to delayed drainage. Delayed drainage is caused by a large difference between the vertical and horizontal conductivities in sediment. This is common in glacial drift where thin bands of silt and sand impede the downward movement of water (Johnson, 1986). A much longer pumping period would be required to affect these wells. Calculated transmissivity and storativity values were 59,000 gpd/ft and 0.0036, respectively. These values compare favorably with those values obtained in the 24-hour pumping tests conducted in Phase 1. Computer simulation of the pumping test yielded results which were similar to those obtained in the field. A summary of the aquifer characteristics derived from the pumping test of MW-9I is found on Table 3-5.

The pumping test demonstrated that groundwater extraction is technically feasible and that the test results can be used to complete conceptual designs of extraction scenarios for the Feasibility Study.

3.4.3 Overburden Groundwater Elevations and Flow Directions - Twelve rounds of water-level measurements have been collected from the overburden monitor wells, which were installed during the Phase 1 and 2 drilling programs. Measurements were obtained using a clean, weighted steel tape or an electrical water-level indicator. All measurements were accurate to the nearest one-hundredth of a foot. To determine groundwater elevations and flow direction, the well locations and top of casing elevations were surveyed by A.A. Major. Survey elevation results, depth to water measurements and calculated groundwater elevations are presented on Table 3-6.

Groundwater table elevation contour maps have been constructed based upon water-level measurements obtained from the shallow monitor wells. Water-level measurements collected from MW-12 were not used in the construction of the groundwater table elevation contour maps. Groundwater elevations in MW-12 are much higher than any other well in that vicinity. This water is believed to be perched due to the past landfill activities during which the permeability of the sediment and/or fill material beneath the bottom of the well screen was altered. Three water-level measuring events have been contoured which represent high, average and low groundwater table conditions. These are shown on Figures 3-10, 3-11, and 3-12, respectively. Computer modeling of static water-table conditions has resulted in similar water-table configuration and flow pattern.

Potentiometric water-level contour maps have also been constructed based on water-level measurements obtained from the deep monitor wells. Three potentiometric water-level contour maps have been constructed which also represent low, average and high water-level conditions (Figures 3-13, 3-14 and 3-15). A potentiometric contour map of the intermediate monitor wells was not constructed due to the unrelated depths at which the wells were screened.

The convergence of the contours in the vicinity of cluster MW-9 indicates that a preferential flow path may exist in this area. The boring logs shown in Appendix B confirm that a zone of coarse sand and gravel exists in this area. As described previously, this may indicate the presence of a buried stream channel.

The shallow water table and deep potentiometric contour maps (Figure 3-13 and 3-15) show that a groundwater divide may exist in the southern part of the site between monitor well clusters MW-1 and MW-14. Most groundwater north of the divide flows toward the northeast, with a small component of flow on the western side of the site which flows toward the northwest. Groundwater south of the divide flows toward the southeast, away from the landfill.

For a groundwater divide to occur as shown on Figure 3-11, there must be a source of significant recharge at the divide. This may be provided by the borrow pit located just south of the landfill. This area of sandy soil receives a great deal of runoff from the landfill mound during rain events. Since the slope of the land south of the landfill is quite flat, the majority of the runoff from the landfill percolates to the water table in the vicinity of the borrow pit. The borrow pit is located directly above the apparent groundwater divide.

3.4.4 Analysis of Permeability Tests - The data obtained from the "slug tests" were analyzed by the Hvorslev Method to calculate the hydraulic conductivity values of the site sediments. Permeability testing data, calculations and results are shown in Appendix F. Data obtained from the slug testing of monitor wells MW-9S, MW-11S, MW-11D, MW-14S and MW-15S were not useful in determining hydraulic conductivity values. Monitor well MW-9S was repaired as a 2″ well within a 6″ well. The sandpack diameter was too large to collect any useful information to determine the hydraulic conductivity. "Slug test" data collected from MW-11S and MW-11D could not be used to calculate hydraulic conductivity values because recharge to the well was almost instantaneous, causing the water levels to oscillate. The rapid recharge may indicate a high hydraulic conductivity. This is expected in the gravel materials in which MW-11D is screened and the shallow sands in which MW-11S is screened. Useful data from MW-14S could not be obtained after numerous "slug testing" attempts. Lastly, insufficient water in MW-15S prevented a "slug test" from being run.

Hydraulic conductivities calculated from the "slug test" analysis range from 1.4 to 1,133 gpd/ft^2. The wide range of values can be attributed to the stratified nature of the sediments. In general, the lowest values obtained were found in the deep wells. These wells were screened just above the till or bedrock and in finer grained material. The highest values obtained were from the shallow wells, especially in the vicinity of monitor well clusters MW-6 and MW-9. The higher values reflect the coarser sediment that is found in these zones and may represent preferred groundwater flow paths. Hydraulic conductivity values of the sediments in which the monitor wells are completed are summarized in Table 3-7.

The widely varied hydraulic conductivity values indicate that within the overburden aquifer, there are preferential flow paths. As stated, the hydraulic conductivities are generally highest in the shallow wells, where the fine sands are the least compacted (as evidenced by blow count data shown on the boring logs in Appendix B). Other important flow paths apparently occur in some very coarse sediments found at MW-9I and possibly at MW-11D. The subsequent flow net analysis illustrates the importance of the preferential flow in the vicinity of MW-9I.

3.4.5 Flow Net Analysis - Using Darcy's law, the approximate volume of groundwater which may be degraded and which is migrating toward the north can be calculated. A plan view flow net was created using the March 16, 1992 groundwater elevation data. Lines representing groundwater flow were drawn perpendicular to the groundwater elevation contours, as shown on Figure 3-16. The area between two flow lines (flow path) contains many cells, each of which theoretically contains the same volume of groundwater. Following a flow path, the cell shape may change, reflecting changes in saturated thickness and/or hydraulic conductivities. Assuming a constant saturated thickness, a cell which is narrower is an indication that the hydraulic conductivity and seepage velocity have increased, whereas widening cells indicate decreased hydraulic conductivity and velocity. To determine the volume of groundwater flow, the flow net was broken into four segments, as defined by differing gradients, hydraulic conductivity, and the saturated thickness.

Discharge (Q) from each zone is determined from $Q=KAi$, where (K) is the hydraulic conductivity, (A) is the cross-sectional area through which flow occurs, and i is the hydraulic gradient. The site average hydraulic conductivity was used to determine discharge in Segments 1 and 3. This method was chosen because there are no wells in the immediate vicinity of Segments 1 and 3 in which a representative hydraulic conductivity can be determined. Also, cell shape through the flow paths in these particular segments has not changed significantly, indicating that the hydraulic conductivity values throughout these segments are relatively constant and that use of the site average hydraulic conductivity is an appropriate approximation.

An average of the slug test results of cluster MW-3 were used for the hydraulic conductivity in Segment 2 since cluster MW-3 falls in the middle of Segment 2 and is a more accurate representation of the hydraulic conductivity of the segment. Use of this reduced conductivity is also appropriate based on the wide divergence of the flow lines in Segment 2.

The average hydraulic conductivity of cluster MW-9 was used in the calculation of groundwater discharge through Segment 4 since cluster MW-9 falls within this segment. The convergence of the flow lines in the vicinity of cluster MW-9 provides a good indication that use of a higher hydraulic conductivity in this vicinity is appropriate.

The far western part of the flow net along the 830-foot contour has been labeled Segment 1, as shown on Figure 3-16. Utilizing a calculated hydraulic gradient of 0.025, a width of 250 feet, a saturated thickness of 31 feet, and a hydraulic conductivity of 283 gpd/ft^2, the discharge for Segment 1 has been calculated at 55,000 gpd.

The western central portion of the flow net along the 830 foot contour has been labeled as Segment 2. The width of the segment is 1,225 feet, the saturated thickness is approximately 27 feet, the hydraulic gradient is 0.001, and the hydraulic conductivity is 103 gpd/ft^2. The calculated discharge for Segment 2 is 34,000 gpd.

Segment 3 is located along the 835-foot contour on the eastern central portion of the flow net. The variables include a segment width of 525 feet, a saturated thickness of 42 feet, and a hydraulic gradient of 0.01 and a hydraulic conductivity of 283 gpd/ft^2. The calculated discharge through Segment 3 is 62,000 gpd.

The far eastern portion of the flow net, labeled Segment 4, is located along the 835-foot contour. The width of the segment is 250 feet, the saturated thickness is 42 feet, the hydraulic gradient is 0.01 and the hydraulic conductivity is 643 gpd/ft^2. The calculated discharge through Segment 4 is 68,000 gpd. The large volume of water moving through this narrow segment reflects the coarse material in this vicinity resulting in a preferred pathway for groundwater flow.

The total volume of groundwater leaving the landfill area, summarizing the four flow net segments in which groundwater may be degraded, is approximately 219,000 gpd. These calculations are based upon the March 16, 1992 groundwater level measurements. Table 3-8 presents a summary of the discharge calculations.

These results are similar to those obtained during the Phase 1 investigation where the discharge through each of the segments totaled 203,000 gpd. The increase is due to the higher hydraulic conductivity results obtained from monitor well cluster MW-9. The location of MW-9 was chosen based upon the results obtained from the Phase 1 flow net. Converging flow lines in the vicinity of Pond B indicated that a large volume of groundwater was moving through this area. Cluster MW-9 was installed during Phase 2 to verify this and to determine the feasibility of groundwater recovery.

<u>3.4.6 Vertical Groundwater Flow Potential</u> - In addition to horizontal flow, there are vertical components to groundwater flow. The difference in water-level elevations between the wells within a well cluster indicate whether ground water has an upward or downward gradient in a particular location. The vertical gradients for each well cluster for each round of water-level measurements are presented on Table 3-9. In cluster MW-1, both upward and downward gradients have been calculated. These changes in gradient direction have been small, usually ±0.003 ft/ft. Since cluster MW-1 is located uphill of the site and there are no nearby groundwater discharge points, a downward gradient has been indicated on the cross-section flow net maps. Downward gradients also have been calculated in clusters MW-2, MW-3, MW-4, MW-10 and MW-15. A slight

reversal of gradient direction has occurred on occasion between the shallow and intermediate wells in cluster MW-4.

In the vicinity of well cluster MW-5, both upward and downward gradients exist. A downward gradient exists between the shallow and the intermediate wells and an upward gradient exists between the intermediate and deep wells. These gradients indicate that ground water is discharging into Pond B which is immediately west of cluster MW-5.

Upward gradients have also been calculated for well clusters MW-6, MW-9, MW-11 and MW-13. Each of these cluster locations are near Anthony Creek or near the ponds associated with Anthony Creek, indicating that the Anthony Creek stream system is an area of groundwater discharge. Temporary vertical flow direction reversals in clusters MW-6 and MW-9 have occurred, but these are attributed to delayed drainage from precipitation events. Cross-sectional flow nets have been constructed along cross-section lines C-C′ and D-D′ (Figures 3-17 and 3-18). Flow lines on the cross sections indicate the direction in which groundwater is moving.

The vertical gradients between the deep overburden and bedrock monitor wells have also been determined. A downward gradient exists at each of the following cluster locations: MW-1, MW-2, MW-3, MW-10, MW-13, MW-14 and MW-15. An upward gradient has been calculated for cluster MW-4 between the bedrock and deep overburden monitor wells. This gradient ranges from 0.002 to 0.017 ft/ft indicating that at this location, where glacial till has been eroded of the bedrock, groundwater flow tends to discharge from the bedrock to the overburden aquifer. The upward gradient at this location tends to protect the bedrock aquifer from contaminants which may be present in the overburden.

<u>**3.4.7 Bedrock Groundwater Elevations and Flow Directions**</u> - At a minimum, three rounds of water-level measurements were obtained from all of the bedrock monitor wells, which were installed during the Phase 2 drilling program. The bedrock wells were also surveyed by A.A. Major for location and top of casing elevation. Survey elevation

results, depth to water measurements and water-level elevations are presented on Table 3-6.

The way in which groundwater moves through rock can be complex depending on rock type. This is the case with the massive shale found beneath the site in which the primary means of groundwater flow is through fractures. Many fractures were encountered during the Phase 2 drilling program, especially in the upper few feet of the bedrock. Below this, fractures were occasionally found, as shown by the RQD results. Groundwater in rock will tend to flow from an area of high head to an area of low head, but its path of travel can be tortuous depending upon the orientation and intersection of fractures. A conceptual block diagram portraying fractured rock is shown on Figure 3-19.

The water level measurements obtained on February 20, 1992 are plotted on Figure 3-20. The piezometric surface of the bedrock aquifer is depicted by the contour lines shown on this figure. These contours generally coincide with the regional fracture patterns and is supported by the chemical data presented and discussed in Sections 4 and 5. Similar results were obtained using computer modeling to simulate a bedrock potentiometric-level contour map.

In general, the bedrock groundwater flow direction is toward the northeast following the northeast, southwest-trending regional fracture pattern. However, in the southern third of the site, there may be a component of the bedrock groundwater flow toward the south and southeast.

Elevation measurements made at MW-10BR indicate that the groundwater elevation is several feet higher than expected. This could represent the head in a fracture set that is not closely connected with the fractures in which the other bedrock wells are screened.

3.4.8 Potential for Contamination of the Bedrock Aquifer - Contaminated overburden groundwater may be entering the bedrock aquifer through areas where the lodgement till is absent. It may also reach the bedrock aquifer through till fractures in areas where the till thickness is less than about 30 feet. With the exception of cluster MW-4, downward gradients exist between the overburden and bedrock aquifers which would tend to force contaminated groundwater down through till fractures, if they exist. Convergence of groundwater in the bedrock aquifer in the vicinity of cluster MW-4 is causing an upward gradient.

At locations, where a downward gradient exists, contaminated overburden groundwater can enter the upper 3 to 5 feet of the bedrock aquifer where the rock is highly fractured. From this zone, continued migration into the deeper through-going fractures discussed in Section 3.3.1 and 3.3.2. These larger scale fractures, which are believed to be oriented along the bedrock lows beneath the site (Figure 3-9), probably correspond with the regional fracture pattern, providing preferential pathways for bedrock groundwater flow. The chemical data discussed in Sections 4 and 5 further support this theory.

3.4.9 Groundwater Recharge - The amount of groundwater recharge occurring within the Anthony Creek basin to Barker Road can be determined by watershed size and the amount of precipitation infiltrating the ground. The watershed to Barker Road has been delineated as shown on Figure 3-21 and has an area of 1,248 acres. The average annual precipitation in Fulton County for the years 1931 to 1990 is 42.7 inches (Table 3-10). To more accurately determine the amount of precipitation infiltrating the soil, the watershed has been broken into three zones based upon land usage. This method was also employed to develop a computer simulation of site groundwater conditions. The results of the computer model compare very favorably with this analysis.

Zone 1 of the watershed lies within the urban area of Gloversville and has an area of 195 acres. Impervious areas within the City and the diversion of runoff into adjacent basins decrease the amount of infiltration that would normally occur. An estimated 6 inches of precipitation infiltrates each year for a total of 87,000 gpd.

Zone 2 is limited in extent to a 83-acre portion of the landfill which is unvegetated. This area has sandy surface soils which allow an estimated 28 inches of water per year to infiltrate, resulting in a total of 172,000 gpd.

The remainder of the watershed, labeled as Zone 3, covers 970 acres and has an estimated recharge rate of 25 percent of the annual precipitation. This is based on the Eastern Oswego groundwater study which has similar glacially derived soils. Applying this percentage to the annual 42.7 inches of precipitation in Fulton County results in approximately 11 inches of infiltration for a total recharge of 793,000 gpd. These results are similar to those obtained by computer modeling. The total amount of recharge based upon precipitation infiltration for the Anthony Creek Basin to Barker Road is 1,052,000 gpd. Table 3-11 summarizes recharge in the various zones.

3.4.10 Dilution Potential

As determined in the previous section, a total of 172,000 gpd infiltrates the 83-acres of fill material (Zone 2) resulting in leachate generation. Direct dilution of this leachate occurs with upgradient groundwater moving through the site. An upgradient area of approximately 40 acres has been delineated with an annual precipitation of 42.7 inches. At a 25-percent recharge rate, approximately 11 inches infiltrates the soil for a total of 33,000 gpd. This results in a total of 205,000 gpd of potentially degraded water.

The calculations based on the flow net and slug test results compare favorably with those results obtained from the precipitation data in determining the volume of groundwater leaving the site. The volume of water moving through the flow net was determined to be 219,000 gpd, or 7 percent more than the volume calculated using the precipitation data.

Further dilution of the 205,000 gallons of potentially degraded groundwater takes place as the groundwater moves toward Barker Road. This dilution comes from precipitation infiltration within the remaining watershed area of 1,125 acres. This area, including a small section of the City of Gloversville, receives a total of 847,000 gpd of recharge

(195 acres at 6 inches plus 930 acres at 11 inches). The final dilution of the potentially degraded groundwater by unaffected groundwater at Barker Road is on the order of 5 to 1.

3.4.11 Stream Gauging Results - Stream gauging information obtained on July 10 and 18, 1990 is presented in Table 3-12. Stream gauging locations are shown on Figure 2-7. Stream gauging was also conducted on October 11, 1990, but is not included in the table because heavy rain produced a large amount of runoff and ponding which resulted in inaccurate data.

The data show that there is an increase in flow in the downstream direction. An exception occurs at Station 2 on July 10, but this can be attributed to a beaver dam located 10 feet away. The main outlet stream for the dam was gauged, but several smaller seeps which flow through the dams could not be measured. The increase in volume of water in the downstream direction is one indication that the stream is a groundwater discharge point. Stronger evidence comes from the elevation of the groundwater as measured in the well clusters. The groundwater has a higher elevation in the wells than in the nearby streams, indicating that groundwater flow is toward Anthony Creek. This can be seen on the shallow ground-water table contour maps and flow net.

3.4.12 Underflow at Barker Road - A cross section was created at Barker Road as shown on Figure 3-22. Depth to bedrock information as obtained form the June 1989 seismic survey and the groundwater elevation was determined based upon the water level in Anthony Creek and in cluster MW-13. Using Q = KAi, an estimation of the volume of groundwater underflow can be calculated. Using an area of 16,600 ft^2, a measured flow gradient (i) of 0.002 ft/ft and an estimated hydraulic conductivity (K) of 430 gpd/ft^2, results in a total overburden groundwater flow volume of approximately 14,300 gpd. The computer model was used to assess underflow in the bedrock, and this component of flow is estimated to be 19,000 gpd. Total flow past Barker Road, including stream flow (using the July 18, 1990 stream gauging information) and underflow, is approximately 670,000 gpd, 5 percent of which is groundwater underflow in the overburden and bedrock aquifers. Since all previous calculations of infiltration and groundwater flow have been based on average annual rainfall data, it is not possible to compare them with this number, which is based, in part, on a single day stream gauging. Long-term base flow would have to be established before a more accurate relationship can be made.

3.5 GROUND WATER FLOW MODEL

This section presents the results of a numerical simulation of ground-water flow beneath the Gloversville Landfill (the landfill) completed by LBG. The numerical simulation was completed as part of the Remedial Investigation/Feasibility Study to investigate remedial options for controlling the release of landfill leachate to ground water beneath the site.

3.5.1 Model Selection - The numerical simulation was completed using MODFLOW, a modular three-dimensional finite-difference ground-water flow model developed by the U.S. Geological Survey [USGS] (McDonald and Harbaugh, 1988). MODFLOW has been used extensively for analysis of ground-water flow problems in numerous hydrogeologic settings, and is considered fully verified. The computer code implementation of the model used in these simulations is MODFLOW™ compiled for the 80386-based DOS computer system by Scientific Software, Inc. Contour plots of head matrix output data were constructed using the SURFER software package from Golden Software, Inc.

The ground-water flow system simulated by MODFLOW is governed by the equation:

$$\frac{\delta}{\delta x}\left(K_{xx}\frac{\delta h}{\delta x}\right)+\frac{\delta}{\delta y}\left(K_{yy}\frac{\delta h}{\delta y}\right)+\frac{\delta}{\delta z}\left(K_{zz}\frac{\delta h}{\delta z}\right)-W=S_s\frac{\delta h}{\delta t}$$

where:
K_{xx} = hydraulic conductivity in x coordinate axis, etc.;
h = potentiometric head;
W = volumetric flux/unit volume (source or sink);
S_s = specific storage; and
t = time.

This equation is used to simulate a ground-water flow system, which is controlled by hydrogeologic characteristics (e.g. hydraulic conductivity and potentiometric head distribution), recharge from precipitation, the effects of surface water bodies, and recharging or discharging wells. The constructed model characterizes the aquifer system as a series of discrete blocks within a layered grid configuration. The governing equation is solved at each grid block by an iterative mathematical procedure.

3.5.2 Model Domain and Layer Definition - The model grid was constructed using a 27 by 28 block, 2 layer configuration. This represents total grid axis dimensions of 14,000 by 11,000 feet (Figure 3-23). Model layer 1 represents the unconsolidated silt, sand and gravel aquifer underlying the study area and vicinity; layer 2 represents the underlying shale bedrock aquifer. The selected model domain encompasses the landfill area, as well as the remainder of the Anthony Creek drainage basin upstream of the Barker Road culvert. The landfill area is located in the south-central portion of the model domain. The Anthony Creek drainage basin limits were determined by topographic analysis of the Gloversville, NY 7.5-minute USGS quadrangle map. The model domain extends slightly beyond the drainage basin within the bedrock aquifer (layer 2) to accommodate regional flow patterns. To the south of the landfill area, the model domain was extended beyond the drainage basin boundary in the overburden aquifer (layer 1) to simulate the effects of a drainage basin divide. The orientation of the model grid was selected so that anisotropic (i.e. directional) behavior of the bedrock aquifer could be simulated in the presumed predominant fracture orientation direction. The model domain was discretized using a variably-spaced grid, with a finer grid mesh centered about the landfill area (Figures 3-23 and 3-24). Grid block dimensions range from 1,000 by 1,000 feet at the perimeter of the model domain to 200 by 200 feet in the landfill area.

Layer 1 was defined as an unconfined aquifer and layer 2 as a confined aquifer based on the monitor well data. The vertical definition of the model domain provides an indirect representation of a glacial till confining unit which is located between the overburden and bedrock aquifers. The till unit is modeled as a component of layer 1, and is incorporated into the vertical conductance term which governs the flow of ground-water between layers 1 and 2. This approach suitably simulates the effects of the confining unit on the adjacent aquifers in the absence of specific hydrogeologic data for the confining unit.

3.5.3 Input Parameters and Modeling Process - Input parameters required by MODFLOW define the hydrogeologic properties and physical limits of the aquifer materials underlying the model domain. These parameters may include the horizontal hydraulic conductivity, transmissivity, storativity and aquifer thickness, each of which may be varied within a given model layer. Physical limitations of flow may be simulated by assigning variable-head, constant-head, or no-flow status to cells within the grid matrix. Effects of external forces on ground-water flow such as the presence of rivers, drains, wells, or the effects of recharge or evapotranspiration may be simulated using applicable MODFLOW stress packages.

The model calibration process involves adjustment of the various input parameters within expected ranges based on available field data. The values are adjusted within these ranges until the output head distribution and water budget calculations from the model are in reasonable agreement with the field data. Validation of the completed model may be demonstrated by simulating a stress on the model which can be compared to the reaction of the aquifer system to a similar stress imposed during aquifer testing in the field. This procedure typically involves simulating the effects of aquifer pumping tests.

Static conditions and remedial scenario simulations were run under steady-state conditions, which ignore the short-term head conditions which occur due to the effects of aquifer storage. Pumping test simulations, which represent short-term conditions by their nature, were simulated under transient conditions, which includes calculations to account for the effects of aquifer storage.

3.5.4 Boundary Conditions - Cells in layer 1 located within the model domain but outside of the Anthony Creek drainage basin were made inactive (i.e. no-flow) for the simulation. The locations of the no-flow cells in layer 1 are indicated in Figure 3-23.

Cells in layer 2 underlying layer 1 no-flow cells remained active to simulate the larger areal extent of the regional bedrock aquifer (Figure 3-24).

Ground-water flow direction in the model under static conditions is simulated by specifying boundary conditions representing sources and/or sinks for the respective aquifers. Because the entire Anthony Creek drainage basin south of Barker Road was included in the model domain, the principal source of water for the model area is recharge through infiltration of precipitation from the ground surface. Some regional underflow occurs along the boundaries of the model domain. This underflow was simulated by assigning constant-head cells along the edges of the grid. The head values of these cells were adjusted within estimated ranges during the model calibration process.

Sinks in the model domain are associated with the discharge of ground water to Anthony Creek and its tributary swamps. These areas were modeled as drains with conductance values ranging from 100 to 1,000 ft²/day, based on a characterization of the bed material. The conductance values are based on the equation:

$$\begin{matrix}\textit{Drain Bed}\\ \textit{Conductance}\end{matrix} = \frac{K\,L\,W}{M}$$

where:
K = hydraulic conductivity of drain bed material
L = length of drain reach
W = width of drain reach
M = thickness of bed material

Approximations of K were used in the above equation, because actual drain bed conductance values were not available. It is assumed that the K values for bed material in the swamp area are likely to be about 10 times lower than the corresponding K value for stream bed material.

Sinks for the model are also associated with ground-water flow leaving the model domain south of the landfill outside the Anthony Creek drainage basin. These effects were simulated by assigning appropriate constant-head values to cells along the southern edge of layer 1, and at the Barker Road/Anthony Creek culvert in layer 2. The layer 2 constant head cells simulate underflow leaving the basin beneath Anthony Creek in the vicinity of the Barker Road culvert, which occurs in an area of downward vertical hydraulic gradient (based on data from well cluster MW-13). These cells are considered "sinks" because they represent areas where ground water leaves the model domain through the bedrock aquifer (layer 2). The locations of the drain and constant head cells in layers 1 and 2 are indicated in Figures 3-23 and 3-24.

<u>3.5.5 Recharge</u> - Recharge to the model was simulated as infiltration of precipitation to layer 1. Previous LBG estimates of recharge were used for different areas of the model domain. Six inches of recharge were estimated for the urbanized areas of Gloversville, 11 inches for the vegetated rural areas over most of the model domain, and 28 inches over the unvegetated open landfill areas. A significant localized recharge source corresponding to the location of an open borrow pit immediately south of the landfill area was simulated by increasing recharge values in the corresponding model cells to 56 inches per year. This is based on an estimate of the runoff contribution area to the borrow pit area (roughly two times the pit area) and an infiltration rate equal to that used for the unvegetated landfill area.

<u>3.5.6 Hydraulic Conductivity and Transmissivity</u> - Hydraulic conductivity and transmissivity values are required by the equation governing MODFLOW to determine the rates of horizontal ground-water movement across each model grid layer. Hydraulic conductivity (*K*) values are required in layers defined as unconfined aquifers, and the transmissivity of the layer is then calculated based on the saturated thickness. Transmissivity is specified directly for confined aquifers (i.e. layer 2).

Hydraulic conductivity values for layer 1 were obtained from slug test and pumping test data from wells completed in the overburden aquifer in the vicinity of the landfill. These data indicate some vertical variability in hydraulic conductivity within this unit. In general, higher values of hydraulic conductivity were observed in the shallowest wells. However, the scale at which this variability occurred is not significant over the scope of the entire model domain, therefore, vertically-averaged values were used. No hydraulic conductivity values were available over much of the model area, and available data was projected into these areas. A range of K values between 15 to 100 ft/d (110 to 750 gpd/ft^2) was determined to be optimal to characterize the non-landfill portion of layer 1 and K values of 70 to 100 ft/d (525 to 750 gpd/ft^2) were optimal for the saturated portions of the landfill material. The K values were varied spatially across the model domain in accordance with the results of the hydrogeologic investigation. Transmissivity in the shale bedrock aquifer was approximated at 20 ft^2/day.

__3.5.7 Vertical Conductance__ - Vertical conductance defines the rate of vertical water movement between adjacent aquifers per unit head. The vertical conductance is controlled by the vertical hydraulic conductivity of each of the adjacent model layers and of any intervening confining layers between model layers. This is defined as:

$$Vcont_{i,j,k+1/2} = \frac{1}{\dfrac{\Delta z_{u/2}}{K_{z\,u}} + \dfrac{\Delta z_c}{K_{z\,u}} + \dfrac{\Delta z_{L/2}}{K_{z\,L}}}$$

where:
Δz_u = thickness of upper aquifer;
Δz_c = thickness of confining bed;
Δz_L = thickness of lower aquifer;
K_{zu} = vertical K of upper aquifer;
K_{zc} = vertical K of confining unit;
K_{zu} = vertical K of lower aquifer.

The vertical conductance values between layers 1 and 2 were calculated based on this equation (see figure 3-25). The calculated values ranged between 3.3x10^{-7}/day and 1x10^{-6}/day except in the landfill/borrow pit area, where removal of horizontal stratification and apparent decrease in till thickness indicated an increase to 5x10^{-5}/day. These values were based both on the thickness of the till underlying the overburden aquifer, assumed vertical hydraulic conductivities for the overburden aquifer and till, and on adjustment of values to achieve agreement between the projected head distribution and the observed water levels.

3.5.8 Calibration - During the calibration process, model parameters are adjusted within acceptable ranges so that the resultant head distribution resembles ground-water elevation contour maps derived from field data. Important hydrogeologic parameters adjusted during the calibration process are listed in Table 3-13, along with final values used in the calibrated model. A qualitative indication of the sensitivity of the model to adjustment of particular parameters is also presented in Table 3-13. This analysis indicates the parameters which have the greatest effect on the local hydrogeologic regime.

3.5.9 Calibrated Model

The steady-state head distribution matrix for model layer 1 was contoured using the SURFER graphics package, and the output from this procedure is presented in Figure 3-26. The head distribution would be considered an average condition that might be observed at the site over the period of one year, recognizing that the actual head distributions are highly variable. The head distribution from the model was compared for calibration purposes with actual head data obtained from the site on March 16, 1992 (Figure 3-27). These plots are not exactly matched, but several important characteristics have been simulated:

1) Hydraulic gradient of 5.9x10^{-3} ft/ft in overburden aquifer between landfill and Anthony Creek;
2) Hydraulic gradient of 8.0x10^{-3} ft/ft in bedrock aquifer between landfill and monitor well MW-13BR.
3) Potentiometric mounding in overburden aquifer beneath the borrow pit/landfill area well cluster MW-1; apparent reversal of gradient or

ground-water divide between landfill and monitor well MW-14S along East Fulton Street Extension.

4) Potentiometric mounding in bedrock aquifer beneath the landfill area; apparent reversal of gradient between landfill and monitor well MW-14BR along East Fulton Street Extension.

The contour plots confirm the existence of a reversal in hydraulic gradient (ground-water divide) in the vicinity of well cluster MW-1. The location and elevation of this divide are influenced to a large degree by the increased recharge associated with the presence of an open borrow pit along the southern edge of the landfill. The borrow pit forms a partial closed depression which allows runoff from the landfill and the surrounding terrain to accumulate and percolate into the underlying aquifer. This is contributing to the formation of a ground-water mound beneath the landfill in the overburden and bedrock aquifers. A printout of the MODFLOW data files, as well as the results of the calibrated model run, are presented in Appendix G.

<u>**3.5.10 Water Balance**</u> - Sources of ground water to the model domain include recharge from precipitation and underflow from adjacent drainage basins. The model output indicates 1,242,500 gallons per day (gpd) recharge from precipitation, of which 1,058,500 gpd occurs within the Anthony Creek basin. This agrees with previous LBG recharge estimates of about 1,052,000 gpd (Section 3.4.9).

An average outflow of 957,000 gpd or 663 gallons per minute (gpm) is estimated by the model to be discharged to Anthony Creek and tributary swamps upstream of the Barker Road culvert. This may be compared with a stream gauge measurement of about 300 gpm for a point somewhat downstream from the culvert obtained on July 10, 1990. This is considered a reasonable approximation, because July is typically a period of low stream flow, and the model calculation would include high flow periods as well as storm discharge minus evaporative losses. A total of 283,000 gpd is calculated to be leaving the model domain basin as underflow. This includes about 11,000 and 19,000 gpd underflow through the overburden and bedrock aquifers, respectively, in the vicinity of the Barker Road culvert.

3.5.11 Model Verification - A transient simulation was completed to assess the validity of the constructed model using data obtained during a pumping test of well MW-9I conducted by LBG on March 24-26, 1992. The pumping test was conducted by pumping the well at a constant discharge rate of 100 gpm for a period of 48 hours, while monitoring water levels in the surrounding monitor wells. The highly transmissive nature of the overburden aquifer materials near well MW-9I combined with the effects of delayed drainage on the response of overburden monitor wells limited the stress which the pumping test placed on the overburden aquifer. Under these conditions, the test could not be feasibly extended long enough for steady-state conditions to be established (a one-week to ten-day test would probably be required). Therefore, a transient simulation was completed which takes into account the short-term effects of storage in the aquifer surrounding the pumping well. A comparison between the transient pumping test simulation output and the calculated drawdown for the shallow monitor wells is presented in Figure 3-28, and indicates reasonable agreement between field data and model output.

3.5.12 Remediation Scenario Simulation - Steady-state simulations were completed using the calibrated model to test proposed remediation scenarios consisting of the installation of a low permeability ($1x10^{-7}$ cm/sec) cap on the landfill surface. The purpose of the cap would be to reduce the recharge entering the landfill surface, and thereby lowering the water table in the overburden aquifer. If sufficient lowering occurs, the water table may drop beneath the bottom of the landfill materials.

Two scenarios were considered in conjunction with this assessment. Scenario #1 involves capping the landfill surface only; Scenario #2 involves capping both the landfill area and the borrow pit area immediately south of the landfill. In both cases, the cap material was simulated by altering the recharge values for the capped area to 1 inch per year. The simulation was run to steady-state conditions, and the results are presented in Figure 3-29 and 3-30.

The results of the simulations indicate that scenario #1 reduces the ground-water mound by about 5 feet beneath the center of the landfill area. Based on limited field data (bottom elevation of fill materials encountered at MW-7), this alternative would appear to lower the water table to about one foot below the center of the landfill. However, thickness data are not available for the southwestern portion of the landfill, where the water table elevation is expected to be higher. Therefore, it cannot be determined whether the bottom of the southwest portion of the landfill will become desaturated as a result of this remedial scenario.

Scenario #2 reduces the ground-water mound by about 7 feet beneath the center of the landfill area, and provides even greater reduction (up to 9 feet) below the southwest corner of the landfill. Therefore, this scenario would provide the greater probability of reducing the ground-water mound beneath the fill materials in the southwest corner of the landfill.

3.5.13 Conclusions - A calibrated ground-water flow model of the Gloversville Sanitary Landfill area has been developed based on available hydrogeologic data. The model was validated using data obtained during a pumping test completed at the site March 24-26, 1992. Two remedial scenarios were simulated using the calibrated model to assess the feasibility of using a low-permeability landfill cap to reduce the height of a ground-water mound beneath the landfill. The second of these scenarios, which involves capping both the landfill and an adjacent borrow pit area, appears to most effectively reduce the groundwater mound beneath the landfill.

3.6 SUMMARY

The previous discussions show that a detailed evaluation of groundwater flow has been generated through the Phase 1 and 2 programs, supplemented with a computer model.

Preferred groundwater flow pathways in the overburden and bedrock have been identified, and the following sections on water quality further confirm these flow paths.

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SECTION 4.0

NATURE AND EXTENT OF CONTAMINATION

4.1 INTRODUCTION

4.1.1 General Description of Source - Section 2.2 of this Report describes the activities conducted during Phase 1 and Phase 2 of the RI which defined and characterized the source of contamination at the Gloversville Landfill. These activities included delineation of the refuse disposal boundaries through visual observations, magnetometer surveys, excavation of test pits, and drilling of boreholes.

Section 2.2.2 describes the drilling of 22 boreholes in the "inactive" area and the southern portion of the "active" area. Drilling logs record the depth of fill and a general description of the waste in these areas of the landfill. These data are presented in Table 4.1-1. As the table shows, the waste in this area is comprised largely of animal hides, leather scraps, glass and wood fragments. The thickness of these materials ranged from 2 to 31 feet, averaging 12 feet.

During both Phases, excavation of test pits provided general information on the types of solid waste present and on the extent of filled material. Descriptions of waste materials encountered are provided in Tables 4.1-2 and 4.1-3. The test pits also provided an opportunity to collect samples from within the waste mass to help characterize the source area soils and to determine the potential for leachate generation.

Visual observations of refuse types made during the installation of monitor wells MW-7, MW-8 and MW-12 indicate that approximately 70 - 80 feet of mixed household, tannery and demolition waste exists in the central area of the "active" portion of the landfill.

4.1.2 General Description of Chemical Characterization - Sections 2.5 through 2.7 detail how and when environmental samples were collected for the various phases of this RI. Media sampled include the refuse, leachate, soils, groundwater, private well water, surface waters and pond and stream sediments. Table 2-1 provides a sampling matrix which identifies sample locations, sampling dates and analytical parameters.

Most environmental samples were analyzed by NYTEST Environmental, Inc. (NEI), a New York State Contract Laboratory Program participant that was the low bidder on this project. Some surface water samples collected during wildlife toxicity studies were analyzed for COD, dissolved oxygen and ammonia by Tighe & Bond Environmental Laboratory. The analytical results generated by NEI were sent to an independent firm to validate the results. C.C. Johnson and Malhotra, P.C. (CCJM) reviewed all laboratory reports to determine if all applicable protocols were followed. In those cases where all of the protocol requirements were not followed or where certain performance standards were not met, qualifiers were added to the data. The data tables presented in Section 4 of Volume II contain qualifiers from both the laboratory and the data validation process. Where both firms qualified data for the same reason, only the laboratory's qualifiers are shown. In some cases, the validator found that the detection limits listed for undetected compounds are estimated because of certain analytical problems.

Subsections 4.2 through 4.6 discuss the specific results of chemical analyses. These subsections and the data tables have been organized by analytical method (volatile, semi-volatile, metals, etc.) rather than by environmental media (groundwater, soils, surface water) to allow the reader to "follow" a particular contaminant from the source through all the environmental media sampled. Air sampling data is discussed separately in Section 4.7. Field analyses are discussed in Section 4.8.

Tables summarizing the analytical data are provided in Volume II of this Report. Where applicable, a column of standards is included on the tables. Detected concentrations of contaminants are highlighted in the tables by shading. Concentrations exceeding the standards are denoted by a bold box surrounding the result. Where a compound was not detected, but the detection limit was greater then the standard, the result has not been boxed.

4.1.3 City Water Characterization - Two samples of water from the City of Gloversville water distribution system were collected during Phase 1. The purpose of the sampling was to determine if City water was a suitable source of water for the drilling program. The analytical results, presented in Tables 4.2-11, 4.3-11, 4.4-11, and 4.5-11, show that the water is uncontaminated. Therefore, City water was deemed to be an adequate supply for drilling water.

4.2 VOLATILE ORGANIC COMPOUNDS (VOC's)

4.2.1 Leachate Characterization -

TCLP - Five samples of refuse taken from test pits excavated in April 1992 were subjected to TCLP extraction and VOC analysis. The volatile analysis results are shown on Table 4.2-1. Acetone, methylene chloride, and 4-methyl-2-pentanone were detected in four of the five refuse samples. Xylenes, toluene and ethylbenzene were detected in TCLP-4 and TCLP-5. Generally, the VOC concentrations were quite low in TCLP-1, 2 and 3, moderate in TCLP-5 and highest in TCLP-4. No exceedences of toxicity characteristic standards were observed, although standards have not been established for some of the detected compounds.

Leachate Seeps/LW-1 and LW-2 - Results for leachate water collected from seeps on the northern and northeastern side of the landfill during May 1990 are presented in Table 4.2-2. No standards exist for leachate. However, of the nine compounds detected in these two samples only acetone (LW-1) and 2-butanone (LW-1 and LW-2) were present in levels greater than 1,000 ug/L. Other compounds detected include methylene chloride, 1,1-dichloroethane, 1,2-dichloroethene, 4-methyl-2-pentanone, and benzene derivatives: toluene, ethylbenzene and xylenes.

Leachate - MW-7, MW-8 and MW-12 - Leachate was collected from monitoring wells MW-7, MW-8 and MW-12 which are located within the refuse mound of the landfill. MW-7 and MW-8 were sampled three times during the RI whereas MW-12, which was installed during Phase 2, was sampled during the February 1992 sampling round only. The results, shown in Table 4.2-2, indicate that several VOC's are present in the leachate. Chloroethane, benzene, ethylbenzene, toluene and xylenes are the most common, present in most samples at concentrations ranging from about 10-500 ug/L.

4.2.2 Source Area Soils -

Leachate Seep Sediments LS-1 and LS-2 - Analytical results for sediments taken at the two leachate seep locations are presented in Table 4.2-3. Acetone, 2-butanone and toluene were detected in the leachate sediments in levels exceeding 1,000 ug/kg.

Test Pits - Table 4.2-4 presents the results of analyses of four soils samples collected from test pits (TP-5, TP-6, TP-8 and TP-10) excavated in the refuse mound during July 1990. The samples collected were of soil, possibly daily cover material, mixed with the refuse encountered during excavation. Three compounds, methylene chloride, acetone and xylenes were detected in TP-5, two compounds, 4-methyl-2-pentanone and xylenes were detected in TP-8, and 5 compounds, acetone, 4-methyl-2-pentanone, toluene, ethylbenzene and xylenes were detected in TP-10. No volatile organic compounds were detected in the soil sample collected from TP-6.

Borings - Section 2.2.2 described the drilling of 22 soil borings in the "inactive" areas of the landfill and the collection of 26 surface and subsurface soil samples from these borings. In addition, samples were collected from beneath the landfill during the drilling of leachate monitor wells, MW-7 and MW-8. Table 4.2-5 presents the results of the VOC analyses on these samples. Volatiles were detected in 15 of the twenty-two soil samples. Acetone and benzene derivatives were the most commonly detected compounds and they appear to be distributed throughout the "inactive" area. These compounds were detected in both surface and deep samples. Samples B-1S, B-1D and B-9S contained these compounds in much higher concentrations than any of the other samples.

Low concentrations of three chlorinated solvents, 1,2-dichloroethene, trichloroethene and tetrachloroethane were detected only in the surface sample at B-19, suggesting a localized source such as a surface spill or localized dumping.

4.2.3 Groundwater

Monitor Wells - As described in Section 2.4, Tighe & Bond collected 3 rounds of samples from groundwater monitor wells. Table 4.2-6 presents the results of VOC analyses for all samples. These data show trace quantities (less than 5 ug/L) of various VOC's in wells MW-1I, MW-2S, MW-3S, MW-4I, MW-6S, MW-1BR. Of these, only benzene, detected at 1 ug/L in wells MW-3S and MW-4I, exceeded the NYSDEC Groundwater Standard of 0.7 ug/L.

Groundwater from MW-9S and MW-9I, however, contained several compounds in excess of NYSDEC standards. These consisted of 1,1-dichloroethane, benzene, toluene, ethylbenzene and xylenes. Chloroform, 4-methyl-2-pentanone, and chlorobenzene were also detected in MW-9I at low concentrations. MW-10BR yielded a sample which contained 170 ug/L of acetone.

Table 4.2-7 presents the results of analysis of a sample collected from the discharge line of a pump test of MW-9I conducted in March 1992. The compounds found exceeding DEC standards in the MW-9I samples were chloroethane, 1,1-dichlorotehane, benzene, toluene, ethylbenzene and xylenes. Other VOC's detected in the discharge at relatively low concentrations included acetone, 1,2-dichloroethene, 2-butanone, 4-methyl-2-pentanone, and chlorobenzene.

Private Wells - Volatile organic contamination in the private wells is negligible. VOC's were detected in only three of the sixteen wells sampled during the first phase of the RI (PW-10, PW-11, and PW-12). All three of the wells contained acetone. However, acetone is a widely used solvent that frequently contaminates samples in the analytical laboratory.

PW-10 contained 2-butanone at levels above the NYS Department of Health (DOH) drinking water standard. The 2-butanone in PW-10 (110 ppb) is believed to be associated with the installation of the home owner's well which occurred a short time before the sampling event. 2-butanone is a solvent that may have been an ingredient in the glue used to fasten joints in the plastic pipe leading from the well to the house. PW-10 was resampled and a split sample was collected by the NYS DOH. All volatile organic compounds (including acetone and 2-butanone) were undetected in that sample.

In sampling of private wells conducted in October, 1991, Toluene and 1,1,1-trichloroethane were detected in PW-17 at 0.2 and 0.4 ug/L, respectively. These levels are well below the DOH drinking water standard of 5 ug/L.

VOC results for private wells are presented in Table 4.2-8.

4.2.4 Surface Water - Samples collected from the surface water of Anthony Creek showed little VOC contamination. Methylene chloride was detected in SW-8 at 4 ug/L in the sample collected in October 1989. In samples collected in October 1991, chloroethane was detected at 2 ug/L in SW-10 and acetone was detected in SW-6 at a concentration of 190 ug/L. None of the three compounds was detected during both sampling rounds. Table 4.2-9 presents the surface water VOC data.

4.2.5 Stream and Pond Sediments - Sediment samples were collected from each of the surface water sampling points in October 1989. Additionally, sediment samples were collected from 3 locations within Pond B in October 1991.

During the May 1989 sampling, acetone was detected in Pond A (Sample PS-2) and at location SW-7 (SS-7) in Anthony Creek. Samples collected at SW-3 and SW-4 (SS-3 and SS-4) contained carbon disulfide. SW-4 is located upstream of the landfill. Pond A sediments also contained low concentrations of toluene, chlorobenzene, ethylbenzene, and xylenes. These compounds were also found in the landfill leachate and in the soil samples collected from the "inactive" area.

Pond B samples collected in October 1991 contained chloromethane, ethylbenzene and xylenes at low concentrations. All sediment samples results for VOC analyses are shown in Table 4.2-10.

4.3 SEMI-VOLATILES

4.3.1 Leachate Characterization

TCLP - Sixteen semi-volatile compounds were detected in the five TCLP extracts. Of these, ten were phenolic compounds, four were phthalates and two were napthalenes. No semi-volatile compounds exceed the TCLP limits. Results are shown on Table 4.3-1. All concentrations were relatively low except for phenolic compounds in TCLP-4 and TCLP-5.

Leachate - Fifteen different compounds were detected in the leachate (LW-1, LW-2, MW-7, MW-8 and MW-12). These compounds consisted of 4 phenols, 3 phthalates, 3 napthalenes, 3 chlorobenzenes, benzoic acid and bis(2-chloroisopropyl)ether.

Concentrations of 4-methylphenol and benzoic acid were much greater in MW-12 and LW-1 and LW-2 than in MW-7 and MW-8. Naphthalene was found in a much higher concentration in MW-12 than in either MW-7 or MW-8. Analytical results for leachate are shown in Table 4.3-2.

4.3.2 Source Area Soils

Leachate Seep Sediments - Table 4.3-3 presents results of semi-volatile compounds analyses for sediment samples collected from the leachate seeps. Eight compounds were detected at concentrations ranging from 22 ug/kg to 1,500 ug/kg.

Test Pits - In July 1990, thirteen test pits were excavated to further delineate the extent of refuse. Soil samples from four of the test pits (TP-5, TP-6, TP-8 and TP-10) were chemically analyzed. Three phenolic compounds, two naphthalene compounds, two phthalate compounds and one chlorobenzene were detected in the soil collected from the four test pits. Twelve of the 16 detections were in concentrations exceeding 1,000 ug/kg. Six exceeded 10,000 ug/kg. Analytical data for semi-volatile compounds are presented on Table 4.3-4.

Borings - Twenty seven different semi-volatile compounds were detected in samples collected from borings conducted in January 1992. Semi-volatile compounds were detected in all of the boring samples except B-7D, B-11D, B-14D and B-18D. These borings are all located in the northern most portion of the site. Complete analytical results for semi-volatile compounds are presented in Table 4.3-5.

Samples B-1S, B-1D, B-14S, B-17S, and B-17D appear to be the most impacted by the landfill. Twenty two compounds were detected in B-1S and B-17S, 19 in B-14S, 16 in B-1D, and 14 in B-17D. The detected concentrations of like compounds were generally much greater in B-1S and B-1D than in the other samples. 4-methylphenol and naphthalene were detected in particularly high concentrations in B-1S at 130,000 ppb and 190,000 ppb, respectively.

4.3.3 Groundwater

Monitor Wells - Table 4.3-6 shows that semi-volatile contamination of groundwater appears to be limited to well cluster MW-9. In MW-9S and MW-9I, eleven compounds were detected. Four compounds in MW-9I (2-methylphenol, 4-methylphenol, 2,4-dimethylphenol, 4-chloro-3-methylphenol) exceeded the NYSDEC groundwater standards while two compounds in MW-9S (4 methylphenol, hexachlorobutadiene) exceeded the standards.

Phthalate esters were the only other semi-volatile compounds detected in groundwater. While the standard for di-n-butylphthalate was exceeded in MW-5S and the standard for bis(2-ethylhexyl) phthalate was exceeded in MW-6I, phthalates are generally considered to be ubiquitous. Therefore, they do not necessarily indicate an influence from the landfill, but may be from the sample containers or from the laboratory.

Pump Test Discharge - The results of analysis of a sample taken from the discharge line of the pump test conducted on MW-9I during April 1992 are shown on Table 4.3-7. Eight compounds were detected, with four (2-methylphenol, bis(2-chloroisopropyl) ether, 4-methylphenol, and 4-chloro-3-methylphenol) exceeding the DEC groundwater standards.

Private Wells - Semi-volatile contamination of private wells in the area of the Gloversville Landfill is negligible. During the December 1989 sampling, bis(2-ethyl hexyl)phthalate was detected at low levels in PW-8 and PW-12. Low levels of phthalates are frequently associated with laboratory or sampling contamination.

In the October 1991 sampling of PW-17, three compounds were detected in low levels: di-n-butylpyhthalate, bis(2-ethyl hexyl)phthalate, and n-nitrosodiphenylamine. The latter two were also detected in the laboratory blank. All three are likely due to laboratory or sampling contamination. No exceedences of DOH standards for semi-volatile compounds were observed. Private well analytical results are presented on Table 4.3-8.

4.3.4 Surface Water - Table 4.3-9 presents the analytical results for the two rounds of surface water sampling conducted in October 1989 and October 1991.

Surface water contamination by semi-volatile compounds was limited to benzoic acid at SW-5 and bis(2-ethyl hexyl)phthalate at SW-2, SW-3, SW-6 and SW-8. All were detected only during the October 1991 sampling round and in very low concentrations.

The Phase 1 RI Report submitted to DEC in May, 1991 inadvertently attributed detection of three phthalate compounds and two polyaromatic hydrocarbon compounds (PAH's), to the sample collected at SW-1 in 1989. These parameters should have been listed as detected compounds in SS-1 (stream sediment). The discussion and data tables in this report have been revised to reflect this.

4.3.5 Stream and Pond Sediments - Phthalate compounds were detected in the sediment collected from stream and pond locations. SS-9 contained the most detected compounds with thirteen. Most of these compounds were phthalates and PAH's. The PAH's are typically associated with fuels and oils and may result from runoff from the roadway adjacent to SS-9. Moreover, because the majority of these compounds were not found at any other stream sediment locations between the landfill and SW-9 (sample SS-9), it is unlikely that their source is the landfill.

Di-n-butylphthalate was found at all three pond B bottom sediment locations. Benzoic acid was detected in PS-1 at a very low concentration and bis(2-ethylhexyl)phthalate was found in PS-3. The phthalate compounds, which are ubiquitous, do not necessarily reflect contamination from the landfill.

Semi-volatile data for sediment sampling are included on Tables 4.3-10.

4.4 PESTICIDES AND PCBs

Pesticide contamination at the Gloversville Landfill site is limited to the landfill area itself. Migration of pesticides off-site appears to be very limited. Six pesticide compounds were detected in the leachate, three in the soils excavated from test pits TP-8 and TP-10 and six in five of the soil borings. Pesticide contamination in the soil borings was limited to the surface samples. Three of the detections were concentrated in the northwestern portion of the site at B-7, B-8 and B-11.

Methoxychlor was tentatively identified in MW-2D and MW-3D in the October 1991 sample but was not detected in the previous October 1989 sampling.

No pesticide compounds were detected in any of the private well, surface water or sediment samples. PCB compounds were not detected in any of the samples. Complete pesticide/PCB data are presented in Tables 4.4-1 through 4.4-11.

4.5 METALS

4.5.1 Leachate Characterization

TCLP - Of the twenty-three metals analyses performed, seven were undetected in all five samples: beryllium, cadmium, mercury, nickel, selenium, thallium and vanadium. The chromium concentration in sample TCLP-2 exceeded toxicity characteristic limits established by the U.S. Environmental Protection Agency, indicating that the refuse sampled is a characteristic hazardous waste. Sample TCLP-2 was taken from test pit number 13 (TP-13 on Figure 3.2-3) in April 1992. TP-13 was located on the west side of the "inactive" area. Table 4.1-2 indicates that the refuse material at this location is comprised of cover sand, leather, leather buffings, and animal hair.

Though no TCLP standard exists for sodium, concentrations of this parameter were notably high, exceeding 800,000 ug/L in all five extract samples.

Data are reported on Table 4.5-1.

<u>Leachate</u> - Results of metals analyses on leachate samples collected during Phases 1 and 2 are presented on Table 4.5-2. Generally, the results show that of the 23 metals analyzed for, only selenium, silver, and thallium were undetected in all the samples. While no standards are specifically applicable to leachate, the results provide a good indication of the contaminant levels at the source. Being associated with the liquid phase, as opposed to contaminants in the soil, these compounds have the greatest potential to migrate from the landfill in groundwater and surface water. The following paragraphs provide a very brief summary of the data that are presented on Table 4.5-2.

Aluminum concentrations varied widely in the leachate samples. Unfiltered samples from the leachate monitor wells ranged from 4,360 to 55,200 ug/L. The leachate seeps contained less: 933 ug/L in LW-1 and 703 ug/L in LW-2.

Antimony was detected in seven of the nine unfiltered leachate samples at concentrations ranging from 39 to 76 ug/L.

Eight of the nine unfiltered leachate samples contained arsenic at detectable levels. These levels were low (< 25 ug/L) in all the samples except that from MW-12, where 159 ug/L were detected.

Barium concentrations in all leachate samples ranged from 224 to 651 ug/L. This contaminant is one which is suspected to be migrating from the landfill. It is discussed in detail in Section 5.3.

Reported levels of beryllium in leachate ranged from less than 1 to 3.7 ug/L.

Sample LW-2 was the only leachate sample found to contain detectable amounts of cadmium (8.5 ug/L)

Calcium was present in all of the samples, ranging in concentration from 10,900 ug/L in DS-7 (filtered MW-7) to 526,000 ug/L in sample LW-2. Concentrations in MW-7 appear to drop from 1990 to 1991/92 while the concentrations in MW-8 remain fairly constant throughout the sampling program.

Chromium was also detected in all leachate samples. The highest concentration (6,020 ug/L) was measured in MW-12. To put this value in perspective, it exceeds the EPA toxicity characteristic standard of 5,000 ug/L. These values are not, however, directly comparable because the MW-12 sample was not subjected to the applicable TCLP protocol referenced in the EPA hazardous waste regulations. The remaining unfiltered samples contained concentrations ranging from 201 to 2,220 ug/L. Chromium migration from the landfill is also discussed in Section 5.3.

The leachate monitor wells all contained low concentrations of cobalt, while the element was undetected in the leachate seep samples.

Low levels of copper (8.6 to 83 ug/L) were detected in all samples except the July 1990 sample from MW-7, which contained 217 ug/L, and the MW-12 sample, which contained 353 ug/L.

The leachate monitor wells contained from 23,800 to 91,800 ug/L iron. The leachate seep samples contained higher concentrations (93,100 and 142,000 ug/L).

Lead concentrations in the leachate varied widely. Samples LW-2, MW-8 (7/90), and MW-12 contained 779, 377, and 330 ug/L, respectively. The July 1990 sample from MW-7 contained less than 3 ug/L, while the subsequent samplings in 1991 and 1992 showed 38.3 and 43.2 ug/L, respectively.

Magnesium was present in all samples at levels ranging from 37,600 ug/L in DS-7 to 123,000 ug/L in MW-8 (3/92).

Manganese concentrations in the leachate seeps were much higher than in the leachate monitor wells. LW-1 and LW-2 contained 7,700 and 9,970 ug/L, respectively while unfiltered monitor well samples contained 192 to 1,630 ug/L.

In Phase 1 sampling, mercury was detected in only MW-8 at 3.1 ug/L. Phase 2, however, yielded detectable mercury in all but one sample. Mercury was undetected in MW-8 (< 0.2 ug/L) in October 1991 but detected at 6.2 ug/L during the March 1992 sampling. Other Phase 2 mercury data ranged from 0.36 to 1.6 ug/L.

Nickel concentrations were fairly consistent in the leachate, varying from 29.1 ug/L in LW-1 to 159 ug/L in MW-7 (2/92).

Potassium concentrations in well MW-7 ranged from 21,900 to 29,300 ug/L over the three sampling rounds while levels in MW-8 ranged from 191,000 to 289,000 ug/L. The leachate seeps contained 165,000 and 258,000 ug/L and well MW-12 contained 35,200 ug/L.

The leachate contained high levels of sodium in all samples, ranging from 602,000 to 1,540,000 ug/L. MW-8 consistently yielded higher sodium values than the other leachate sampling points.

The July 1990 sampling of well MW-7 yielded a vanadium concentration of 121 ug/L. All other vanadium results were below the contract required detection limit, but above the instrument detection limit. These values ranged from 6.5 to 45.8 ug/L.

The leachate seeps contained significantly more zinc than the leachate monitor wells. LW-1 and LW-2 contained 5,320 and 7,290 ug/L, respectively. Zinc concentrations in the unfiltered monitor well samples ranged from 210 to 996 ug/L.

Cyanide results, which are also given on Table 4.5-2, show detectable concentrations in all the leachate samples except LW-2. The detected values range from 33.0 to 647 ug/L

4.5.2 Source Area Soils

Leachate Seep Sediments - Metals concentrations in leachate seep sediments (Table 4.5-3) were compared to typical background concentrations of metals in uncontaminated soils which are presented in Table 4.5-12. The concentrations of zinc in LS-1 and LS-2 were 174 and 164 mg/kg, respectively. These values are above the 9-50 mg/kg range of background levels found in New York State. No other metals detected in the sediments exceeded the typical background concentrations.

No detections of antimony, arsenic, cadmium, mercury, selenium, silver, thallium and cyanide were recorded.

Test Pits - Test pit sample results, presented in Table 4.5-4, were also compared to typical background levels. Calcium, chromium, mercury, nickel and zinc were detected in concentrations exceeding the range of typical values. Chromium concentrations in TP-5 and TP-8 were particularly high. No detections of cadmium, selenium, thallium and cyanide were recorded. Levels for all metals in TP-6 were within the range of typical values.

Borings - As with the leachate sediments and test pits, the analytical results of soil samples taken from the surface and bottom of the borings conducted in January 1992 were compared to the typical values for typical uncontaminated soils. The results, shown on Table 4.5-5 indicate that the arsenic, calcium, chromium, mercury, nickel, lead and zinc levels at several boring locations are above the typical background concentrations. Generally, the surface soil samples contained higher concentrations of contaminants than the subsurface samples.

Lead concentrations, for example, were markedly higher in the surface samples. Of the 14 subsurface samples, only three contained lead at greater than 14 mg/kg, the average eastern U.S. lead level in uncontaminated soil. Eleven of the 14 surface samples exceeded this value, with concentrations ranging as high as 641 mg/kg in sample B-1S.

A similar trend existed with the chromium results. Samples B-1S, B-7S, B-8S, B-9S, B-11S, and B-12S all contained chromium at levels between 1,000 and 26,400 mg/kg. On the other hand, the chromium concentrations in 13 of the 14 deep samples ranged from 4.7 to 65.9 mg/kg. The 14th, B-8D, contained 168 mg/kg of chromium.

Thallium and selenium were not detected in any of the samples. Silver was detected in only one sample (B1-S) within typical background range and cadmium was detected only twice (B1-S, B8-S) also within the range of typical background concentrations.

4.5.3 Groundwater

Monitor Wells - Data for the three rounds of monitoring wells sample are presented on Table 4.5-6. Results are compared to NYSDEC Class GA groundwater standards.

Iron was detected above DEC standards in all monitoring wells. Aluminum was also detected in a large percentage of the wells above the DEC discharge standard. (No ambient standard exists for aluminum.)

Barium was detected above DEC standards in only MW-3D (2/92), MW-4D (All), MW-5D (All), MW-6D (10/91, 2/92), MW4BR (2/92), and MW-9D (2/92). Throughout the RI, barium has been used as an indicator parameter, appearing at concentrations exceeding about 150 ug/L in monitor wells that appear to be impacted by landfill derived contaminants. Samples from the following monitor wells have contained barium at concentrations exceeding this 150 ug/L "site background" level: MW-1BR, MW-2I, MW-2D, MW-2BR, MW-3S, MW-3I, MW-3D, MW-3BR, MW-4I, MW-4D, MW-4BR, MW-5D, MW-6D, MW-9S, MW-9D, MW-10BR, MW-11D, MW-13BR, MW-14BR. Wells MW-2I, MW-2D and MW-4I have exceeded this threshold only, in the February 1992 sampling round. They may or may not be impacted by landfill contaminants. It is also interesting to note that the highest levels of barium were consistently found in the deep wells.

Cadmium was only detected twice in the groundwater monitor wells at the site. These detections were in the July 1990 sampling of MW-1S and MW-2S. Both wells are believed to be minimally impacted by landfill contaminants, based on groundwater flow directions and indicator parameter concentrations.

A relatively high chromium concentration (51.9 ug/L) was also measured in MW-1S during the July 1990 sampling, exceeding the groundwater standard. It was not detected in any of the subsequent samplings of that well. The only other chromium concentrations to violate the 50 ug/L groundwater standard were measured in February 1992 in wells MW-3I, MW-3D, MW-4I, MW-9I and MW-13BR. (The data validation report on the MW-3I and MW-3D results indicates that these values may be biased high.) MW-2I and MW-2D contained chromium at concentrations below the groundwater standard, but significantly higher than the levels detected in those wells during the two previous sampling rounds.

Lead levels in groundwater at the site typically were measured below about 10 ug/L. Concentrations higher than this "background" level were found in the February 1992 sampling of wells MW-2I, MW-2D, MW-3I, MW-3D and MW-13BR. The 25 ug/L groundwater standard was exceeded in MW-2I and MW-3D (2/92 only). (The data validation report on MW-3D results indicates that the result may or may not be a false positive.) As with the chromium and barium results, wells MW-2I and MW-2D contained significantly higher lead concentrations in February 1992 than in the two previous sampling rounds.

Nickel was detected in several samples at levels about 2 orders of magnitude below the groundwater standard of 2000 ug/L. Results from February 1992 samples from wells MW-3D and MW-9I were above 100 ug/L. "Typical" site concentrations appear to be near about 30 ug/L. The results from MW-2I, MW-2D, MW-3I and MW-3D were significantly higher in February 1992 than they were in the two previous sampling rounds.

All zinc concentrations were below the 300 ug/L groundwater standard except the February 1992 results from MW-3I and MW- 3D. However, these results were qualified during data validation as being potentially biased high. Again, the February 1992 results for wells MW-2I and MW-2D were significantly higher in February 1992 than in the two previous sampling rounds.

Manganese exceeded DEC standards in at least one of the wells of each well cluster except MW-5, MW-6 and MW-14.

MW-15 was the only well cluster that did not have any exceedences of sodium standards. Sodium concentrations were particularly high in many of the deep wells previously identified as being impacted by landfill contaminants (MW-3I, MW-3D, MW-3BR, MW-4D, MW-4BR, MW-5D, MW-6D, MW-7I, MW-9D, and MW-11D), with concentrations all above 200,000 ug/L. Wells MW-3BR, MW-9I and MW-9D had sodium concentrations exceeding 1,000,000 ug/L.

<u>Filtered Metals</u> - Groundwater samples from monitor wells frequently contain suspended soil particles. The particles flow into the wells with the groundwater as the well is evacuated by pumps or bailers. The naturally-occurring minerals that make-up the soil particles contain several of the metals on the Target Analyte List. When unfiltered samples are analyzed for total metals, the metals in the suspended soil particles are

measured in addition to the dissolved contaminants that are of primary interest in this study.

Filtering the suspended particles out of the samples prior to analysis provides the opportunity to evaluate the levels of dissolved metals in the sample. Samples from one well in each cluster were filtered prior to analysis during July 1990 sampling. The filtered samples generally contained significantly lower concentrations of metals than the unfiltered metals, especially aluminum, barium, iron, and manganese.

The filtered sample from MW-6S exhibits a trend opposite from the expected trend. Chromium concentrations are much higher in the filtered sample than the unfiltered sample. Other metals which were not detected in the unfiltered sample were measured at low concentrations in the filtered sample (e.g. Cd and Be).

Since it is not possible to add metals through filtration, other sources of error were considered. A detailed review of the laboratory results for these samples revealed that the filtered sample from MW-6S was run on the analytical instrument immediately following the extract from soil sample TP-8 which contained very high concentrations of chromium. It is possible that the chromium concentrations reported for the MW-6S filtered sample represent carry-over from the previous sample.

Laboratory quality control measures specifically designed to detect carryover problems did not reveal any carryover of chromium. However, the concentration of chromium used in this QC determination is approximately 10 times less than the concentration in the extract from TP-8.

Filtered metals results are also included in Table 4.5-6.

<u>Pump Test Discharge</u> - Table 4.5-7 presents results of metals analyses on the MW-9I pumping test discharge water. These compare favorably with the results of the MW-9I sample collected in February 1992.

__Private Wells__ - Private wells were sampled in December 1989, October 1991 and February 1992. These wells are virtually free of metals contamination. However, many of the wells had high concentrations of sodium. Although DOH does not have a drinking water standard for sodium, the agency does recommend that water containing more than 20,000 ug/L not be used for drinking water by people on severely restricted sodium diets. Additionally, DEC's groundwater standard for sodium is 20,000 ug/L.

The majority of the private wells with sodium concentrations above 20,000 ppb were located in Blanchard Loop Road area, northeast of the landfill. Some wells south of the landfill also exceeded this number.

Barium was detected in several of the private well samples at concentrations significantly below drinking water standards, but at levels that may indicate an influence from the landfill. These are discussed in more detail in Section 5.

Iron exceeded the DOH secondary drinking water standard in PW-20, PW-28 and PW-36. PW-20 was the only well with iron concentrations significantly above the standard.

The December, 1989 sample from PW-10 had a mercury concentration of 2.2 ppb which is above the DOH standard of 2.0. PW-10 was resampled and split with NYS DOH for a follow-up mercury analysis. Mercury was not detected during the second sampling round or in the split sample taken by NYS DOH.

Data for private well analyses are contained in Table 4.5-8.

__4.5.4 Surface Waters__ - Surface water samples for metals analyses were collected during October 1989, October 1991 and February 1992. Table 4.5-9 presents the analytical data. The NYSDEC surface water standard for iron was exceeded at each of the 10 surface water locations at least once during the sampling/analysis program. The zinc standard was exceeded at SW-1 through SW-7 and the aluminum standard was exceeded at SW-1 and SW-2. Zinc levels were generally lower during the 1991 sampling.

Calculated copper standards were exceeded at SW-2 during both October 1989 and October 1991 samplings and at SW-4 during the October 1989 sampling. Lead concentrations in surface water exceeded the surface water standard in samples SW-1 and SW-9 during the October 1989 sampling. SW-1 is upstream of the landfill and SW-9 is the point furthest downstream of the landfill. In October 1991, lead exceeded the calculated surface water standard at SW-2 and SW-4.

4.5.5 Stream and Pond Sediments - Stream and pond sediment analytical results were compared to the typical background concentrations of metals in uncontaminated soils as presented in Table 4.5-12. Arsenic, lead, zinc, chromium and magnesium were at levels above these background concentrations. Chromium and magnesium exceedences were limited to SS-5 and SS-8, respectively. Zinc was most often exceeded, with SS-1, PS-2, SS-3, SS-4, SS-5, SS-7 and SS-8 all above the typical background levels.

Arsenic, chromium, lead and zinc exceedences were found in data from Pond B sediments. The chromium concentrations in these sediments were significantly higher than at the other sediment sampling locations.

Table 4.5-10 presents the stream and pond sediment analytical data.

4.6 MISCELLANEOUS INORGANIC AND NITROGEN COMPOUNDS

4.6.1 TCLP - During the Phase 2 test pit excavations, five samples of refuse were collected and subjected to a TCLP extraction. The extraction was analyzed for the inorganic and nitrogen compounds listed on Table 4.6-1.

The concentrations of ammonia nitrogen varied widely from 0.11 mg/L in TCLP-2 to 450 mg/L in TCLP-5. TCLP-2 also contained the lowest level of chloride (4 mg/L) and TCLP-5 the highest (139 mg/L).

The highest concentration of total dissolved solids was detected in TCLP-4 (33,530 mg/L). All other TDS concentrations were similar, in the range of 3,400 to 5,000 mg/L.

4.6.2 Leachate Characterization - Results of miscellaneous inorganic and nitrogen compound data on leachate samples collected during Phase 1 and Phase 2 of the R1 are presented in Table 4.6-2. Three rounds of leachate sampling were conducted in July 1990, October 1991 and February 1992. The first two sampling rounds included two leachate wells, MW-7 and MW-8; the final sampling round included the addition of leachate well MW-12. Leachate seeps were sampled in the first round only.

Of the fifteen (15) inorganic and nitrogen compounds, three (3) were detected above NYSDEC groundwater standards in all of the samples, and in all rounds of sampling. These compounds include chloride, ammonia nitrogen and total dissolved solids.

The chloride concentrations ranged from 839 mg/L to 8,200 mg/L, with MW-8 having the highest levels of chloride in every round of sampling.

Ammonia nitrogen was detected at its highest concentration in MW-12 at 2,160 mg/L. The levels seen in MW-7 and MW-8 remained relatively constant at concentrations of approximately 600 to 800 mg/L while LW-1 and LW-2 contained 534 and 746 mg/L, respectively.

The concentration of total dissolved solids remained fairly constant in samples taken from MW-8. In MW-8, TDS ranged from 5,739 mg/L to 5,920 mg/L, while in MW-7 the low concentration was 2,250 mg/L and the high concentration was 6,799 mg/L. Monitor well MW-12 contained 2,900 mg/L of total dissolved solids. Concentrations measured in LW-1 and LW-2 were 5,500 and 7,900 mg/L.

4.6.3 Source Area Soils - In July 1990 a total of thirteen (13) test pits were excavated to delineate the edge of debris. Four (4) of the test pits were sampled. The soil encountered in the excavation was analyzed for hexavalent chromium and total cyanide. The analytical results can be found on Table 4.6-3.

In two of the test pit soil samples (TP-5 and TP-6) hexavalent chromium was not detected at the detection limit of the instrument (<0.02 mg/kg). The high concentration of hexavalent chromium was detected in TP-10 at 1.12 mg/kg.

Cyanide was detected at an elevated concentration only in TP-5 (1.24 mg/kg).

4.6.4 Groundwater - During the Phase 1 and 2 investigation a total of 71 groundwater samples were analyzed for the compounds shown on Table 4.6-4. Sixty-three of the samples were taken from the unconfined water table aquifer and the remaining eight were collected from the bedrock aquifer.

As was seen in the leachate wells; chloride, ammonia nitrogen and total dissolved solids were the commonly detected compounds.

Of the 63 groundwater samples collected from the water table aquifer, 37 had none of the miscellaneous inorganic and nitrogen compounds detected at levels exceeding groundwater standards. These samples were collected from overburden clusters MW-1 and MW-2, upgradient and laterally gradient of the landfill, respectively, and from the shallow and intermediate wells in clusters MW-5 and MW-6, downgradient of the landfill but beyond a major groundwater discharge point (Pond B).

In the overburden monitor wells, ammonia nitrogen exceeded the groundwater standard more often any other parameter. Cluster MW-3 appears to be the most impacted by ammonia nitrogen, with concentrations above the New York State DEC groundwater standard reported for each well in the cluster for every sampling event. Cluster MW-3 is located downgradient of MW-12, a leachate well with the highest levels of ammonia, nitrogen measured on site.

As stated previously, a total of eight (8) groundwater samples were taken from the bedrock aquifer. None of the miscellaneous inorganic and nitrogen compounds included on Table 4.6-4 were detected at concentrations exceeding groundwater standards in MW-

1BR, MW-2BR, MW-14BR nor MW-15BR. These bedrock wells are located upgradient and/or laterally gradient of the landfill.

In wells MW-3BR, MW-4BR, MW-10BR and MW-13BR, levels of ammonia nitrogen were above the groundwater standard, with MW-3BR having the highest concentration of 314 mg/l. This concentration is higher than those detected in the overburden wells in the cluster (for the February 1992 sampling).

4.6.5 Pump Test Discharge Sample - On March 25, 1992 during the pumping test of MW-9I, a sample of the discharge was taken and analyzed for the parameter list shown on Table 4.6-5. Ammonia nitrogen was detected in the pump test discharge at 687 mg/L. This concentration correlates directly to the concentration detected in the pumping well, MW-9I. Ammonia, chloride and total dissolved solids concentrations exceeded the groundwater standard.

4.6.6 Private Wells - Throughout the Phase 1 and Phase 2 sampling program, a total of 67 groundwater samples were collected and analyzed for the compounds shown on Table 4.6-6. Of the 67 samples, and with the exception of pH, only six (6) contained levels of an inorganic or nitrogen compound above an established standard. Ammonia nitrogen was detected at an elevated concentration (i.e. >2.0 mg/L) in the sample taken from PW-15 during each of the three sampling rounds. Elevated ammonia nitrogen was detected at PW-17 during the 1991 and 1992 sampling rounds and at PW-20 during the October 1991 sampling round only. The 2.0 mg/L standard is a groundwater standard; it is not a health related standard.

Other detections of ammonia nitrogen occur downgradient of the landfill, in private wells screened in the bedrock aquifer. These are discussed further in Section 5.3.1.

Laboratory results show that the pH in wells MW-4 (8.51) and PW-6 (8.95) are slightly above the drinking water standard (8.5). However, the field measurements of pH for these wells were 8.3 and 8.4, respectively, values within the standard range. Field measurements of pH are considered to be more accurate than laboratory measurements because pH values tend to change during sample shipment and transfer.

4.6.7 Surface Water - Surface water samples were taken during four separate sampling events. Table 4.6-7 summarizes the analytical results of all four sampling rounds.

Standards for unionized ammonia are temperature and pH dependent and are calculated individually for each sample of surface water. Of the thirty-nine (39) surface water samples taken throughout the four rounds of sampling, thirty-one (31) of the samples contained concentrations of unionized ammonia above its respective standard. The highest concentration of unionized ammonia (11.89 mg/l) was detected at SW-5 during the October 1991 sampling round.

Total dissolved solids were also detected in the surface water samples above the established NYSDEC standard of 500 mg/l in 12 of the samples taken. SW-10 had the highest concentration of TDS at 1,540 mg/l during October 1991 sampling round.

4.6.8 Stream and Pond Sediments - During the October, 1989 sampling round, stream and pond sediments were collected and analyzed for hexavalent chromium. The concentrations of hexavalent chromium in the sediment (Table 4.6-8) ranged from 1.1 mg/kg at SS-9 to 150 mg/kg at SS-7. These results were qualified by the laboratory because the sample matrix may have effected the results.

4.7 AMBIENT AIR SAMPLING

Ambient air samples collected in September 1990 were analyzed for volatile organic compounds, total particulate matter and total chromium. A laboratory error resulted in unusable particulate data. Therefore, a second round of particulate sampling was conducted in October 1991. Chromium samples were also collected in October 1991. Tables 4.7-1 through 4.7-3 summarize the results of the air sampling. Additionally, the complete Air Sampling and Analyses Report for the October 1991 sampling is included in Appendix H. The Air Sampling and Analysis Report for the first sampling round was included in the Draft RI Report dated May 1991.

Volatiles - Of the 34 volatile organic compounds analyzed, only five were detected above laboratory detection limits. These are methylene chloride, 2-propanone, chloroform, 1,1,1-trichloroethane and toluene. The other compounds analyzed were not detected.

Methylene chloride was present in all the samples at low concentrations. Two field blanks contained this chemical at concentrations exceeding those found in the air samples. Since methylene chloride is a common laboratory chemical, its ubiquitous presence in air and blank samples suggests that it is not a site-specific contaminant. For this reason, methylene chloride is not considered an air contaminant at this site.

2-propanone was detected in only three of twelve samples at concentrations ranging from 0.014 to 0.024 mg/m^3. Chloroform was detected in two of twelve samples, at concentrations of 0.0009 to 0.0015 mg/m^3 at GLV5-01 and GLV1-02, respectively.

1,1,1-trichloroethane was detected in ten of twelve samples at concentrations ranging from 0.0008 to 0.0017 mg/m^3. Toluene was detected in only three samples at concentrations of 0.00074, 0.0009 and 0.0013 mg/m^3.

Chloroform, 1,1,1-trichloroethane and toluene are classified by NYSDEC as moderate or low toxicity air contaminants. These chemicals were present in the ambient air at concentrations well below the proposed Ambient Guideline Concentrations, which for the three chemicals listed above are 0.023, 45.2, and 8.9 mg/m^3, respectively.

2-propanone, another contaminant present in air at low concentrations, is not listed in any of the three classifications (high toxicity, moderate toxicity, or low toxicity) into which the chemicals of concern are categorized. Volatile air contaminants, therefore, do not appear to pose any inhalation health hazards at the Gloversville Landfill.

Particulate Matter - All samples were below the detection limit of the analytical method except for 91-GL-018 and 91-GL-019 which had concentrations of 0.12 and 0.06 mg/m^3 respectively. The average concentration of particulate measured over the 12 hour sampling period indicates that the Federal primary and secondary, ambient air quality, 24-hour average particulate standard of 0.15 mg/m^3 would be met. (This standard applies only to particulate of less than 10 um diameter (PM10). If total particulates are below the standard level, the PM10 will also meet the criterion).

Chromium - Ambient air quality criteria for chromium have not been promulgated by the USEPA or NYSDEC. The Occupational Safety and Health Administration's (OSHA) permissible exposure limit (PEL) for total chromium is 0.5 mg/m^3 (8-hours per day, 40-hours per week). All air concentrations of total chromium measured at the Gloversville Landfill site during both sampling rounds were below the OSHA standards.

4.8 FIELD ANALYSES

Results of field analyses are presented in Tables 4.8-1 through 4.8-4. These were performed on samples from monitor wells, private wells and surface water during each sampling round and generally included temperature, pH, specific conductance and dissolved oxygen.

pH - The pH measured in leachate and groundwater typically ranged from neutral to moderately alkaline (6.5 - 9.0). The measured pH values do not appear to correlate with other contaminants. Surface water pH values were typically neutral to slightly alkaline (6.5 - 8.0).

Specific Conductance - Specific conductance values in "uncontaminated" samples typically were less than 500 umhos/cm. Values measured in samples known to contain other contaminants were generally 1000 umhos/cm and above.

Dissolved Oxygen - Dissolved oxygen is of greatest interest in surface waters, where it can affect the viability of aquatic species, especially fish. The values measured in the surface waters near the landfill ranged from 4 to 10. It is interesting that some of the higher dissolved oxygen readings were obtained at SW-5 and SW-6, where fish are known to be absent. This supports the theory that the absence of fish is most likely due to ammonia toxicity rather than oxygen deficiency.

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SECTION 5

CONTAMINANT FATE AND TRANSPORT

5.1 INTRODUCTION

The fate and transport of contaminants from the Gloversville Municipal Landfill are evaluated in this Section to determine the potential for off-site migration. This information is essential for assessing the potential for affecting public health or the environment and in determining the need for and the appropriateness of remedial actions.

Several factors contribute to the ultimate fate and transport of contaminants in the environment. These include water solubility, volatilization, adsorption/desorption, hydrolysis, photolysis and oxidation/reduction. Using contaminant specific physio-chemical characteristics and estimations of soil properties, the mobility of contaminants in the environment can be predicted. However, the accuracy of these predictions is limited to the accuracy of the assumptions made. Additionally, much of the physio-chemical information available for chemicals was developed in the laboratory in pure solutions. Applying these factors to field conditions, where many compounds are present in a solution, can lead to significant error. A more reliable means of determining environmental fate is to understand the transport conditions and to measure contaminant levels at key points in the transport network.

As described in Section 4, contaminant levels were measured at the source, in both the leachate and the surface and subsurface soils. The soils have little potential for direct migration, as most are covered with either clean sand (in the "active" area) or vegetation (in the "inactive" area). Contaminants in the soil do have the potential to leach out over time, if left exposed. If the landfill is permanently capped, the landfill soils will be unavailable for contact with percolating rainfall and unavailable for direct contact with potential receptors.

Leachate, however, can migrate readily both overland and through the groundwater. Leachate seeps discharge to the ground surface from the steep sides of the landfill mound and flow overland toward surface water bodies. Most of the leachate appears to re-percolate into the ground before reaching surface water. Leachate is expected to migrate with the groundwater that passes through and under the site. Throughout the migration in groundwater, contaminants are typically altered by physiochemical reactions including sorption, biodegradation, dissolution, chelation, etc. After the landfill is capped, leachate discharges through seeps will be eliminated and leachate discharge to the groundwater is expected to be greatly reduced.

5.2 SITE SPECIFIC TRANSPORT

At the Gloversville Landfill, the hydrogeology has been sufficiently defined to understand the flow characteristics of the groundwater. Section 3 describes that the flow is principally to the surface water bodies in the Anthony Creek basin with a smaller component of the flows entering the bedrock aquifer. Therefore, the fate of contaminants can be determined by monitoring the downgradient groundwater and surface water.

Several of the monitoring points used in this study provide the opportunity to evaluate the migration potential of contaminants. Surface water sampling points SW-1 and SW-4 are in the upper reaches of Anthony Creek and its principal tributary, respectively. It is unlikely that the water or sediments at these two locations are influenced by the landfill.

Groundwater elevation contours and contaminant data indicate that wells MW-1S, MW-1I, MW-2S and MW-14S are hydraulically upgradient of the landfill. The metals and miscellaneous inorganics data indicate that other wells, which are hydraulically downgradient of the landfill, are unimpacted by landfill derived contaminants. These include: MW-3S, MW-4S, MW-5S, MW-5I, MW-6S, MW-6I, MW-10S and MW-13S. Other wells, MW-2I, MW-2D, and MW-4I, exhibited few indications of landfill contaminants during the first two rounds of sampling. In the February 1992 sampling, however, these wells appeared to be impacted by landfill contaminants.

All of the bedrock wells installed during Phase 2 show indications of being impacted by landfill derived contaminants. The bedrock potentiometric contours presented in Section 3 indicate that all the bedrock wells are downgradient of at least a portion of the landfill.

5.3 FATE OF SPECIFIC CONTAMINANTS

Comparing the data shown in Section 4, several conclusions can be drawn regarding the fate of contaminants. Data on leachate and source area soils provide a basis for determining which contaminants are present at the source. The presence of these contaminants at downgradient monitoring points could indicate that the contaminants are migrating from the source toward potential receptors. The following subsections discuss the migration of selected contaminants.

5.3.1 Ammonia - The contaminant most prevalent downgradient of the landfill is ammonia. Figures 5-1, 5-2 and 5-3 show ammonia concentrations in all leachate monitor wells, groundwater monitor wells, private wells and surface waters around the site for the three rounds of sampling. The ammonia is believed to originate from the degradation of organic nitrogen in the anaerobic environment of the landfill. Under these conditions, ammonia is formed and, at neutral or acidic pH, remains primarily dissolved in the leachate and groundwater migrating from the site.

The concentrations of ammonia are expected to decrease through the natural system through a combination of dilution, volatilization, biological oxidation and photosynthetic uptake. While the biological oxidation of ammonia to nitrate is a common reaction in wastewater treatment, it may play a minor role in the natural environment. Nitrogen compounds including ammonia and nitrate participate in many of the oxidation-reduction reactions that occur in natural waters, including photosynthesis.

Ammonia can affect aquatic species in two ways: through oxygen depletion due to the biological oxidation referenced above and through direct toxicity. Ammonia is known to interfere with the ability of fish to uptake oxygen through their gills. Downstream of the landfill, surface water fisheries conditions are similar to those expected in areas of ammonia contamination. The wildlife studies conducted during Phase 1 (Section 2.1.5) identified the absence of any fish species in Pond C or in the stream just below the outlet of Pond B. Furthermore, in-situ bioassay work conducted at these two locations showed that the fathead minnows exhibited complete mortality after just two hours in this water, which is an indication of toxic or anoxic conditions.

Phase 1- Figure 5-1 presents the results from the Phase 1 sampling. The surface water sampling occurred in October 1989 and the groundwater and private well sampling took place in the summer of 1990. Ammonia was present in the leachate seeps and the leachate monitor wells at concentrations ranging from 534 to 865 mg/L. In the downstream surface waters, concentrations ranged from 86 mg/L at SW-5 to 3.4 mg/L at SW- 3. The concentrations shown for locations SW-6 and SW-7 show that Anthony Creek significantly dilutes the high concentrations from the tributary stream. The increase noted in concentrations from SW-7 to SW-8 may be due to the addition of ammonia from contaminated groundwater discharging to the creek.

In the groundwater, concentrations were highest in the intermediate and deep wells of cluster MW-3. Other significant concentrations were measured in MW-3S, MW-4I, MW-4D, MW-5D, and MW-6D. One of the reasons that cluster MW-3 exhibits the highest level of contamination may be that it lies immediately downgradient of the portion of the landfill where large quantities of hair, hides, and sludge-like material were found in the test pits and borings. Further, the divergence of groundwater flow lines in this area (Figure 3-16) indicate that there is essentially no dilution from unimpacted groundwater at this location.

In the private wells, very little ammonia was found. Well PW-15, however, contained ammonia at levels significantly higher than the other private wells that were sampled. This well is completed deep in the bedrock aquifer, in a direction believed to be downgradient of the site. While the presence of ammonia in this well could not be directly linked to the similar contamination around the landfill, it warranted a more substantial investigation of the private wells and the bedrock aquifer in Phase 2.

<u>Fall 1991</u> - The ammonia results of the fall 1991 sampling are presented on Figure 5-2. The leachate wells (MW-7 and MW-8) again yielded samples containing several hundred mg/L of ammonia. Concentrations in the surface waters exhibited similar trends to those seen during Phase 1, but the concentrations were generally two to three times higher.

Groundwater monitor wells exhibited results very similar to the Phase 1 data, except that MW-6D did not contain detectable ammonia. The private well sampling was expanded to include several other homes, primarily in the vicinity of Blanchard Loop Road. The table shown at the left of Figure 5-2 shows the depth of the private wells. Generally, the wells located north of the site that are greater than 60± feet deep are probably drawing from the bedrock aquifer. On the south side, bedrock elevation is somewhat deeper; wells in excess of 150' are probably bedrock wells.

The correlation between well depth and the presence of ammonia at detectable levels is good. Wells PW- 33 and PW-38 are the only wells over 100 feet deep located northeast of the landfill that did not contain ammonia. Wells PW-15, PW- 17, PW-18, PW-20, PW-23, PW-30, PW-34, and PW-37 are all located northeast of the landfill, are believed to be over 100 feet deep, and contain detectable concentrations of ammonia.

Well PW-42 (PW-4 in Phase 1) contained a small amount of ammonia. It is located south of the landfill and is believed to be completed in bedrock. PW-28 is also believed to be a bedrock well located just southwest of the landfill. This well contained no detectable ammonia.

The bedrock wells located north of the landfill that are less than 100 feet deep did not contain detectable ammonia.

<u>Winter, 1992</u> - Figure 5-3 shows that ammonia concentrations in MW-7 and MW-8 were again in the 600 mg/L range. The new leachate monitor well, MW-12, contained ammonia at 2,160 mg/L. This well is completed through wastes which are largely comprised of sludge and hair which are sources of organic nitrogen. As previously described, ammonia is a byproduct of the anaerobic degradation of organic nitrogen.

Groundwater monitor well clusters MW-1, MW-3, MW-4, MW-5, and MW-6 exhibited trends very similar to those found during the first two rounds of sampling. Cluster MW-2, however, exhibited a very low concentration of ammonia in the deep well that was not observed previously.

In the new overburden monitor wells, ammonia was detected in the MW-11D and MW-15D at 65.9 and 0.23 mg/L, respectively. The shallow wells of these clusters did not contain detectable ammonia. Well MW-11D is situated in a coarse grained material downgradient of the "inactive" area, as shown on Figures 3-14 and 3-15. MW-15D is located southeast of the landfill mound. Based on the contours shown on Figures 3-14 & 3-15, it should be uninfluenced by the landfill. However, with minor variations in the predicted potentiometric contours, contamination from the landfill could easily reach the deep well in this cluster.

Monitor well cluster MW-9, which is located in the principal flow path leaving the east side of the landfill, contained 945 mg/L of ammonia in the intermediate well, 47.4 mg/L in the shallow well, and only 1.46 mg/L in the deep well. This supports the theory presented in Section 3 that the coarse sand and gravel zone in which MW-9I is screened serves as a preferred flow path for groundwater discharging from the landfill area.

This being the case, the vertical flow net along section C - C′ shown on Figure 3-17 indicates that most of the groundwater is likely to flow upward, discharging to Pond B, while a smaller component of the flow may continue beneath Pond B in the deep portions of the overburden aquifer. The 179 mg/L of ammonia measured in MW-5D supports the theory that a component of the groundwater flow does pass beneath Pond B.

The surface water data also supports the groundwater flow hypothesis, showing the highest concentrations of ammonia in the tributary from Pond B to Anthony Creek. Sample SW-10, collected from the eastern edge of Pond B contained only 4.42 mg/L of ammonia. However, this is likely east of the principal groundwater discharge area to the pond. Little mixing is expected within the Pond B, and therefore, the concentration at

SW-10 is probably not indicative of ammonia levels in the entire pond. The ammonia concentrations in the surface waters were very similar to those measured in the fall of 1991.

Phase 2 also included an investigation of the bedrock aquifer around the landfill. Ammonia was detected in all eight bedrock monitor wells at concentrations ranging from 0.05 mg/L in MW-15BR to 314 mg/L in MW-3BR. Bedrock well ammonia concentrations were highest in wells located north and northeast of the site, supporting the theory that the majority of bedrock flow from the site is along northeast-southwest trending fracture patterns.

The private well sampling results showed the same trends as previously discussed for the fall 1991 data. Samples from MW-13BR contained ammonia at 6.96 mg/L, a concentration slightly higher, but in the same range, as many of the private wells located in the same vicinity.

5.3.2 Barium - Barium has been used as an indicator parameter in this RI because it has been found at low or non-detectable levels upstream and upgradient of the site while it has been found at significantly higher concentrations in the leachate and in groundwater immediately downgradient of the site. Based on barium concentrations in wells that are believed to be uninfluenced by landfill contaminants, the typical site background groundwater concentration is about 100 ug/L. In surface waters, background appears to be about 30 ug/L, based on concentrations found at the upstream sampling locations, SW-1 and SW-4.

Phase 1 - The leachate samples contained barium at concentrations ranging from 224 to 651 ug/L. Figure 5-4 shows the concentrations of barium at all the Phase 1 monitor wells, private wells, and surface water sampling points. Similar to the ammonia results, the highest concentrations were detected in wells MW-3I, MW-3D, MW-4D, MW-5D, and MW-6D.

Surface water data showed the only detectable concentrations of barium to be at SW-5 and SW-6, again supporting the theory that groundwater discharges to this tributary. Barium was also detected in PW-1 at 105 ug/L, PW-4 at 77.3 ug/L, and PW- 15 at 140 ug/L, concentrations which are higher than the other private wells and slightly higher than the estimated site "background" concentration.

Fall 1991 - Figure 5-5 shows the barium results for the fall 1991 sampling. The leachate samples from MW-7 and MW-8 contained barium at 290 and 524 ug/L, respectively. Wells MW-4D, MW-5D, and MW-6D all contained barium at greater than 1000 ug/L. Elevated levels of barium were also measured in cluster MW-3.

The surface water sampling results showed elevated barium levels at SW-5 and SW-6, similar to the Phase 1 data. However, barium was detected in Anthony Creek samples at concentrations above the surface water "background" level of 30 ug/L.

Elevated barium concentrations are more difficult to discern than ammonia because barium is naturally present in the groundwater. While this is true, wells PW-15, PW-17, PW-18, PW-20, PW-23 and PW-34 yield samples with greater than 70 ug/L of barium, a level higher than most other wells in the area. Well PW-42, located south of the landfill, contains barium at 76.6 ug/L, potentially indicating an influence by the landfill.

Winter 1992 - Figure 5-6 shows the barium concentrations measured in the samples collected during February 1992. The results of surface water and private well sampling closely mirror the results of fall 1991. Results from Phase 1 monitor wells were also very similar to the fall 1991 data except that barium concentrations in cluster MW-3 and in MW- 2I and MW-2D were significantly higher.

The Phase 2 overburden groundwater monitor wells showed "elevated" barium in MW-9S, MW-9I, MW-9D, MW-10S, MW-11D, and MW-13D. Surprisingly, the levels in MW-9I were more than an order of magnitude lower than those in MW-9D, suggesting that barium may preferentially flow to the deeper portions of the aquifer. The level measured in MW-10S was 120 ug/L. This shallow well, however, should not be influenced by landfill leachate, based on the flow patterns shown in Figures 3-11 and 3-12.

The results from the bedrock monitor well showed that they all appear to be somewhat influenced by the presence of barium from the landfill. As with the ammonia results, the concentrations in MW-3BR and MW-4BR were higher than the other bedrock wells.

5.3.3 Chloride - Chloride is also a good indication of the presence of contamination from the landfill. While a variety of sources exist for chloride, the high concentrations measured in the leachate monitor wells show that its presence at downgradient points may support the groundwater flow theories developed in Section 3 and supported by other data discussed in the Section.

Phase 1 - Figure 5-7 shows the chloride data collected during Phase 1. The leachate monitor wells contained 1824 and 2227 mg/L of chloride. The groundwater monitor wells that show elevated concentrations (> 100 mg/L) include MW-3I, MW-3D, MW-4D, MW-5D and MW-6D. Lower, but detectable concentrations were measured in MW-2I, MW-2D, MW-3S, MW-4I and MW-5I.

Surface water samples revealed the highest chloride concentrations at SW-3, SW-5, SW-6, SW-7 and SW-9. The high level found at SW-9 likely is due to runoff from Route 349, which is located immediately upstream from the sampling point. A low concentration at SW-8 indicates that significant dilution occurs downstream of point SW-7.

The highest chloride concentration detected in the private wells was at PW-3, a shallow well located southeast of the landfill. Topography and the groundwater flow patterns at the site indicate that this value is unrelated to contaminants at the landfill.

Fall 1991 - Chloride concentrations from samples collected during the fall of 1991 are shown on Figure 5-8. These results are quite similar to those found in Phase 1. Additional private wells sampled during this round generally contained very little chloride. Exceptions are PW-25, PW-27, PW-29 and PW-41, which are all believed to be shallow wells outside the potential influence of the landfill. Groundwater quality in shallow wells often reflects nearby land uses. Chloride frequently is found in shallow wells near roadways as a result of road salting. Well PW-28 contained 143 mg/L of sodium and is located just southwest of the landfill. However, this well has not shown other signs of contamination.

Winter 1992 - Figure 5-9 summarizes the data from the February 1992 sampling. Leachate, groundwater (from Phase 1 wells), and surface water chloride data showed similar trends to those discussed previously. The overburden groundwater monitor wells installed during Phase 2 contained very low levels of chloride, except in cluster MW-9 where levels near 2000 mg/L were measured in the intermediate and deep wells and in MW-11D where 792 mg/L was detected. MW-13S and MW-13D contain chloride at slightly elevated levels, but this is probably due to local sources, since the landfill's impact does not likely extend to the overburden aquifer at this point.

The bedrock monitor wells surrounding the site contained very low concentrations of chloride, with the notable exception of MW-3BR and MW-4BR, which both contained over 1500 mg/L of chloride.

The private wells showed concentrations very similar to those found during the fall 1991 sampling.

5.3.4 Chromium - Several of the source area soils contained high concentrations of chromium. Leachate samples and TCLP extract were also found to contain high concentrations of chromium at various points in the RI. This contaminant is of particular interest, since a large portion of the wastes buried at the landfill were associated with the tanning industry, known for its extensive use of chromium. Typically, however, the chromium used in the industry is trivalent, which is a relatively stable positively charged ion that tends to bind to soil particles. In the reducing environment which is typical of a landfill, chromium is not expected to oxidize to the less stable hexavalent form, which typically exists as a negatively charged chromate ion.

Phase 1 - Figure 5-10 presents a summary of chromium data from Phase 1 sampling. The leachate samples from MW-7 and MW-8 contained 2220 and 281 ug/L of chromium. Low concentrations were measured in downgradient monitor wells, including MW-2D, MW-3I, MW-3D, cluster MW-4, MW-5S and MW-5D. Upgradient wells MW-1S, MW-1I, and MW-2S also contained low levels of chromium. No chromium was detected in any of the surface water or private well samples.

Fall 1991 - Chromium concentrations measured in the leachate during the fall 1991 sampling ranged from 201 ug/L in MW-8 to 735 ug/L in MW-7, as shown on Figure 5-11. Detectable concentrations ranging from 6.2 to 23.9 ug/L were found in monitor wells MW-3I, MW-3D, MW-4I and MW-5I. Surface water concentrations ranged from 5.6 ug/L at SW-8 to 18.2 ug/L at SW-10. Upstream of the landfill, 6.5 ug/L were measured at SW-1. No private wells were analyzed for chromium in this round.

It is difficult to establish whether the chromium concentrations measured at the downstream points and in the downgradient wells are a result of migration from the landfill, or from natural, background conditions. The levels in MW-3I and MW-3D appear to be significantly higher than those measured in other wells, and therefore may be a result of migration from the landfill.

Winter 1992 - As shown on Figure 5-12, chromium was only measured in the monitor wells during this sampling round. Concentrations in the leachate ranged from 158 ug/L in MW-8 to 6,020 ug/L in MW-12. This latter high level is possibly attributable to the fact that MW-12 is installed through waste that appears to be primarily from industrial sources.

In the overburden groundwater monitor wells, some concentrations of chromium were significantly higher than those detected in the previous rounds. Specifically, MW-3I and MW-3D contained 112 and 243 ug/L, respectively, concentrations about ten times higher than detected in the fall of 1991. Data validation, however, qualified these results as possibly being biased high. MW-2I and MW-2D also yielded samples with elevated chromium concentrations, 39.4 and 47.2 ug/L. Chromium was measured in MW-4I at 81 ug/L, nine times the level measured in the previous round. In the Phase 2 overburden wells, chromium was only measured in MW-9I and MW-9D.

Chromium was detected in three of the eight bedrock monitor wells. Concentrations were 13.6 ug/L at MW-3BR, 36.4 ug/L at MW-1BR and 51.7 ug/L at MW-13BR. While these detections are at locations believed to be downgradient, the absence of chromium at other downgradient bedrock wells makes the connection to the landfill less evident than the apparent connection with other contaminants such as ammonia.

No chromium was detected in the private well samples.

<u>5.3.5 Organics</u> - Minimal migration of organic compounds has been noted throughout the study. Volatile and semi-volatile compounds were measured in MW-9I, indicating that these compounds are migrating from the landfill. However, there were no significant concentrations of these compounds found in surface water or groundwater further downgradient of the site.

Some volatile organic compounds (toluene, chlorobenzene, ethylbenzene, and xylene) were detected in the Pond A sediments. These same compounds were measured at elevated levels in the soil samples collected from the "inactive" area. The two may or may not be connected, since these compounds are common contaminants that are frequently associated with gasoline.

5.4 SUMMARY

The data indicate that while most contaminants are not migrating from the landfill at high concentrations, a contaminant plume is evident. Based on the hydrogeologic study, and supported by chemical data, this plume appears to primarily discharge through preferred flow channels to Anthony Creek and Pond B.

While the groundwater flow volume and velocity has been shown to be low in the vicinity of cluster MW-3, the persistent presence of contaminants in the intermediate and deep wells indicate that it is downgradient of the landfill and within the migrating plume. The divergence of groundwater flow lines in this vicinity helps explain the lack of dilution from uncontaminated groundwaters.

The high hydraulic conductivities measured in MW-9I, coupled with the presence of many landfill derived contaminants, indicates that this is a preferred flow path for groundwater and associated contaminant flow. The vertical flow nets and the contaminant data from MW-5D indicate that a portion of the groundwater flow passes beneath Pond B. Contours in the vicinity of cluster MW-10, however, indicate that some or all of the underflow from Pond B may discharge to Pond C, or to Anthony Creek. Only a small amount of groundwater leaves the Anthony Creek basin in the overburden aquifer.

The analytical data indicate that groundwater quality in the bedrock monitor wells has been degraded by landfill derived contaminants. The actual means of transport from the overburden aquifer to the bedrock aquifer can only be speculated. Most well clusters showed a downward gradient between the overburden and bedrock aquifers. However, a dense lodgement till was encountered between the two aquifers at these points. It is speculated, based on published studies of similar compact tills, that vertical fractures may exist in the till providing conduits to the bedrock aquifer.

Another potential route of transport between the overburden and bedrock aquifers is through areas where till has been removed from the bedrock either through glacial or fluvial erosion. The absence of till at cluster MW-4 is an indication of the former and the absence of till and the presence of coarse deposits at cluster MW-11 is indicative of fluvial erosion. It is speculated that till may also be eroded from the bedrock ridge that runs beneath the landfill, providing an opportunity for landfill derived contaminants to migrate directly to the highly fractured upper portion of the bedrock.

Contaminant movement in the bedrock aquifer is nearly exclusively through fractures that are believed to trend northeast-southwest along the bedrock lows. These fracture sets create a primary flow direction toward the northeast. Data also indicate that contaminants are migrating in bedrock toward the southeast. This may be occurring primarily in the fractured uppermost portion of the bedrock, where flow may follow the bedrock surface toward the southeast, to the bedrock low in the vicinity of cluster 14. Continued migration to the southeast may be through secondary fractures that exist perpendicular to the primary northeast-southwest trend.

To date, the contaminants that have migrated from the landfill have not created exceedences of drinking water standards in any of the private wells. The apparent connection between the bedrock aquifer, which supplies some of the private wells in the area, and the landfill, however, demands that remedial alternatives consider the long term effects on the private wells.

The contamination of Anthony Creek with metals and other inorganics in excess of surface water standards likely will continue until a remedial action is implemented at the landfill.

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SECTION 6
BASELINE RISK ASSESSMENT

6.1 INTRODUCTION

This baseline risk assessment was performed to evaluate the impact of chemical contamination at the Gloversville Landfill site and to assess if actual or threatened releases of hazardous substances pose potential risks to human health and/or the environment under current or possible future exposure circumstances.

The risk assessment follows U.S. EPA guidance for risk assessments in general and for Superfund sites in particular (U.S. EPA, 1986 a, b, c, and 1989), and is based primarily on data generated during the Phase 1 and Phase 2 Remedial Investigation (RI) as summarized in previous sections of this report. In keeping with U.S. EPA guidance, this risk assessment incorporates conservative assumptions about contaminant exposures, intakes, and risks. Pathway-specific models are used to characterize contaminant release and transport mechanisms, behavior patterns, and processes leading to potential contaminant exposure. Standard U.S. EPA risk characterization models and toxicological parameters were used to develop quantitative estimates of risk for each exposure pathway and potentially exposed population.

The data generated during Phase 1 and Phase 2 Remedial Investigation (RI) was utilized as follows:

Groundwater -	Phase 1 and Phase 2 data were combined.
Surface Water -	Phase 1 and Phase 2 data were combined.
Private Wells -	Phase 1 and Phase 2 data were combined.
Sediment -	Phase 1 and Phase 2 data were combined.
Leachate -	Phase 1 and Phase 2 data from leachate monitor wells were combined with Phase 1 leachate seep data. No leachate seep samples were collected in Phase 2.

"Active" Area Surface Soil -	Only Phase 1 data were used because no surface soil samples were collected during the Phase 2 investigation.
"Active" Area Subsurface Soil -	Only Phase 1 data were used because no subsurface soil samples were collected during the Phase 2 investigation.
Ambient Air -	Phase 1 and Phase 2 data were combined.
"Inactive" Area Surface Soil -	Only Phase 2 data were used because inactive surface soil was not sampled during the Phase I investigation.
"Inactive" Area Subsurface Soil -	Only Phase 2 data were used because inactive subsurface soil was not sampled during the Phase 1 investigation.

Section 6.2 describes the methodology used to evaluate the environmental sampling data. The methods used to evaluate spatial patterns of contaminant distribution and to identify the site-related pollutants are described. This section concludes with the selection of chemicals of concern for the various environmental media. These chemicals of concern are then used for the quantitative risk analysis.

In Section 6.3, exposed populations and potential present and future routes of human exposure are identified. The pathway-specific models used to calculate the intake of contaminants are also described.

Section 6.4 provides a brief discussion of the toxicologic properties of the chemicals of concern at the site and presents toxicity values used in the quantitative risk assessment.

Section 6.5 presents the results of the quantitative risk assessment. The uncertainties associated with the quantitative risk estimates are also identified.

Potential impacts on nonhuman receptors and the environment resulting from the chemicals of concern present in various environmental media are evaluated in Section 6.6.

6.2 DATA EVALUATION AND SELECTION OF CHEMICALS OF CONCERN

6.2.1 Data Evaluation - The data used for the risk assessment for the site were collected from the RI activities summarized in the previous sections. Chemical analyses from the Phase 1 and Phase 2 field investigations were used in the selection of chemicals of concern and in the quantitative risk assessment. All RI samples were analyzed using EPA's Contract Laboratory Program (CLP) and were validated in accordance with EPA protocols.

The data were reviewed and evaluated using the following criteria:

- Evaluate data qualifiers. Data qualifiers used by the laboratory may differ in identity or meaning from those used for data validation. Both laboratory and validation data qualifiers, their definitions, and the determination if data with a qualifier is acceptable for use in the quantitative risk assessment are presented in Table 6-1. If data had both laboratory and validation qualifiers, and they appeared contradictory, the laboratory qualifier was ignored and the validation qualifier was used.

- Identify sample locations to serve as estimates of local background concentrations for groundwater and soils.

- Determine which contaminants are present at concentrations greater than background concentrations.

- Identify chemicals which could be linked to site activities, either due to site-related concentration gradients or using historical data concerning the wastes disposed of at the site.

- Select site-related chemicals of concern which are most likely to pose risks to public health.

- Estimate exposure point concentrations for chemicals of concern and define chemical concentration distributions to serve as inputs for the risk assessment.

To assist in identifying the chemicals of concern and in determining their representative concentrations, the following criteria were used to evaluate the analytical data:

- The chemicals detected in upgradient groundwater and background soil samples were compared with samples from downgradient or contaminated areas.

- Duplicates were considered as one sample and the arithmetic mean of the duplicate values was used.

- For non-detected results, one-half of the sample quantitation limit (SQL) was used as a proxy concentration. This arbitrarily selected value (one-half) is commonly assigned to non-detects when averaging data for risk assessment purposes since the actual value can range from zero to a value just below the detection limit.

- The upper 95 percent confidence limit of the arithmetic mean was used as the representative concentration (C-mean). If the upper 95 percent confidence limit of the arithmetic mean was greater than the maximum detected value then the maximum detected value was used as the representative concentration.

__6.2.2 Selection of Chemicals of Concern__ - The procedure used for the selection of chemicals of concern is designed to identify the "highest risk" chemicals at the site. The following criteria were used to select a representative and conservative array of chemicals of potential concern:

- Frequency of detection: If a chemical was detected infrequently (one out of 10 samples), the chemical was not selected for further evaluation. In addition to low frequency of detection, the following criteria were also used for eliminating a chemical: 1) the chemical was detected infrequently in one or two media; 2) the chemical was not detected in any other media or not detected at high concentrations if detected, and 3) there was no reason to believe that the chemical was site related.

- Common laboratory/field contaminants: Common laboratory and/or field contaminants such as acetone, 2-butanone (methyl ethyl ketone or MEK), phthalate esters, methylene chloride, and toluene were excluded unless their concentrations exceeded the maximum detected field blank concentration by at least ten times.

- Comparison with field blanks: The concentrations of other contaminants (not common laboratory and/or field contaminants) were excluded unless their concentrations exceeded the maximum detected field blank concentration by five times.

- Comparison with appropriate background samples: Contaminants detected at levels above background levels were retained for further evaluation. Representative concentrations of chemicals were compared with the background concentrations to determine if the detected levels were higher than the background levels. Representative chemical concentrations which were less than or equal to the representative background values were considered to be present at naturally occurring levels and were not retained for further evaluation.

- Nutrients: Among the inorganics detected, many are considered essential nutrients in the human diet for which more attention has been paid to problems of deficiency rather than toxicity. Among the essential nutrients are calcium, iron, magnesium, potassium, and sodium. These chemicals were not included as chemicals of concern unless they were present at exceptionally high concentrations and might have potential adverse public health effects.

Data were summarized for each environmental medium by determining frequencies of detection, concentration ranges of detected values, and representative concentrations (C-mean). The extent of contamination in the areas sampled and the identification of site-related chemicals by medium sampled are presented in the following subsections.

<u>Downgradient Groundwater</u> - Seventy groundwater samples were collected in potentially impacted areas. A summary of the data is presented in Table 6-2. In this table, the detected chemical concentrations are compared with levels detected in upgradient wells (MW-1S, MW-1I, and MW-14S) and the maximum detected field blank concentrations to assist in the selection of chemicals of concern for risk characterization.

Cyanide, mercury, 1,2-dichloroethene, chloroform, benzene, toluene, xylene, ethylbenzene, acetone, and butyl benzyl phthalate were eliminated because of their infrequent detection. Bis (2-ethyl hexyl) phthalate was eliminated because its concentration did not exceed ten times the maximum detected concentration in the field blank. Calcium, iron, magnesium, potassium, and sodium are essential nutrients, are not considered hazardous at the levels detected, and were not retained for risk characterization.

The chemicals of concern selected for risk characterization in downgradient groundwater along with their representative concentrations are as follows:

Chemical	C-mean (ug/l)
Aluminum	11,577
Antimony	30.8
Arsenic	4.3
Barium	677
Beryllium	1.7
Chromium	54.9
Cobalt	16.0
Copper	27.2
Lead	24.1
Manganese	801
Nickel	31.2
Vanadium	54.2
Zinc	94.1
Di-n-butyl phthalate	13.7

Potentially Impacted Private Wells Groundwater - Nine of the private wells included in sampling program are considered potentially impacted based on location, well depth, and the presence of indicator parameters in the well sample. Five samples were collected from some of the potentially impacted private wells near the site and analyzed for all 24 metals, plus cyanide. Eleven additional well samples were analyzed for barium, iron, magnesium, and sodium only. Additionally, 8 of these samples were analyzed for zinc. A summary of the data is presented in Table 6-3. In Table 6-3, the detected chemical concentrations are compared with the maximum detected field blank concentrations to assist in the selection of chemicals of concern for risk characterization.

Zinc was eliminated because its concentrations did not exceed five times the maximum detected concentrations in the field blanks. Calcium, iron, magnesium, and potassium are essential nutrients, are not considered hazardous at the concentrations detected, and were not retained for risk characterization. The New York Department of Health has established drinking water guidelines for sodium for individuals on sodium restricted diets. The guidelines state that water containing more than 20 mg/l of sodium should not be used for drinking by people on severely restricted sodium diets and 270 mg/l of sodium for people on moderately restricted sodium diets (the C-mean value is 35,339 u/gL, or 35.3 mg/L). However, the toxicological data to evaluate risks associated with elevated levels of sodium is inadequate (EPA, 1987). Therefore, sodium was not retained for risk characterization.

The chemicals of concern selected for risk characterization in potentially impacted private well groundwater along with their representative concentrations are as follows:

Chemical	C-mean (ug/l)
Barium	133.3

Due to reduced list of parameters analyzed on the "impacted wells", C-mean values for copper, manganese, mercury, acetone, and 2-butanone will be assumed to be equal in the two groups of private wells.

Nearby Private Wells Groundwater - Thirteen samples were collected from private wells near the site for analysis of all 24 metals plus cyanide. Fifty-one samples were collected for barium, iron, magnesium, and sodium only. Additionally, 22 of these private well samples were analyzed for zinc. A summary of the data is presented in Table 6-4. In Table 6-4, the detected chemical concentrations are compared with the maximum detected field blank concentration to assist in the selection of chemicals of concentration risk characterization.

Lead was eliminated because of infrequent detection. Zinc was eliminated because its concentration did not exceed five times the maximum detected concentration in the field blanks. Calcium, iron, magnesium, and potassium, are essential nutrients, are not considered hazardous at the concentrations detected, and were not retained for risk characterization. Information concerning sodium is as described for the potentially impacted wells. The C-Mean for sodium in this group of wells is 17.4 mg/L.

The chemicals of concern selected for risk characterization in nearby private well groundwater along with their representative concentrations are as follows:

Chemical	C-mean (ug/l)
Barium	27.6
Copper	90.4
Manganese	16.4
Mercury	1.0
Acetone	42.5
2 -Butanone	110

Surface Water - Nineteen surface water samples were collected from Anthony Creek. In Table 6-5, the detected chemical concentrations are compared with field blanks and the levels detected in upstream sampling locations (SW1 and SW4) to assist in the selection of chemicals of concern for risk characterization.

Methylene chloride was eliminated because its concentration did not exceed ten times the maximum detected concentration in the field blank. Calcium, iron, magnesium, potassium, and sodium are essential nutrients, are not considered hazardous at the concentrations detected, and were not retained for risk characterization. Arsenic, cobalt, and vanadium were eliminated because of infrequent detection. Manganese and lead were eliminated because its concentration was less than the background concentration.

The chemicals of concern selected for risk characterization in surface water along with their representative concentrations are as follows:

Chemical	C-mean (ug/l)
Aluminum	602.77
Antimony	25.13
Barium	70.76
Chromium	10.27
Mercury	0.39
Zinc	99.32
Bis(2-ethyl hexyl) phthalate	2.0

Sediment - Seventeen sediment samples were collected from Anthony Creek. A summary of the data is presented in Table 6-6. In this table the detected chemical concentrations are combined with levels detected in upstream sampling locations (SS1 and SS4) to assist in the selection of chemicals of concern.

Nickel, toluene, chlorobenzene, ethyl benzene, xylenes, diethyl phthalate, phenanthrene, anthracene, pyrene, benzo (a) anthracene, chrysene, benzo (b) fluoranthene, indeno (1,2,3-cd) pyrene, dibenz (a,h) anthracene and benzo (g,h,i) perylene were eliminated because of infrequent detection. Calcium, iron, magnesium, potassium, and sodium are essential nutrients, are not considered hazardous at the concentrations detected, and were not retained for risk characterization. Aluminum, arsenic, cobalt, copper, manganese, mercury, vanadium, and carbon disulfide were eliminated because their concentrations were less than the background concentrations.

The chemicals of concern selected for risk characterization in sediment along with their representative concentrations are as follows:

Chemical	C-mean (mg/kg)
Antimony	59.71
Barium	219.86
Chromium	70.45
Lead	90.3
Zinc	207.94
Acetone	0.1
Xylene	0.088
Di-n-butyl phthalate	4.44
Fluoranthene	0.78
Butyl benzyl phthalate	0.18
Bis (2-ethyl hexyl) phthalate	4.27
Benzo (a) pyrene	1.60

<u>Leachate</u> - Nine leachate samples were collected during the Phase 1 and Phase 2 field investigations. In Table 6-7, the detected chemical concentrations are compared with field blanks to assist in the selection of chemicals of concern for risk characterization.

Calcium, iron, magnesium, potassium, and sodium are essential nutrients, are not considered hazardous at the concentrations detected, and were not retained for risk characterization.

The chemicals of concern selected for risk characterization in leachate along with their representative concentrations are as follows:

Chemical	C-mean (ug/l)
Aluminum	27,053
Antimony	61.6
Arsenic	64.9
Barium	534
Beryllium	2.3
Cadmium	8.5
Chromium	2,722
Cobalt	31.8
Copper	180.4
Lead	389
Manganese	5,269
Mercury	3.0
Nickel	135.0
Vanadium	65.8
Zinc	3,806.0
Cyanide	87.1
Chloroethane	43.2
Methylene chloride	131.7
Acetone	1,051
1,1-Dichloroethane	16.7
1,2-Dichloroethene	16.1
2-Butanone	4,670
Benzene	18.1
4-Methyl-2-pentanone	153.2
2-Hexanone	20
Toluene	516.8
Chlorobenzene	17.8
Ethyl benzene	80.9
Styrene	23
Xylenes	294.5
Phenol	226.6
Benzyl alcohol	9.0
2-Methyl phenol	22.3
4-Methyl phenol	1,808.9
2,4-Dimethyl phenol	60.44

Chemical	C-mean (ug/l)
Benzoic Acid	5,678.8
Naphthalene	558.9
4-Chloro-3-methyl phenol	14
Diethyl phthalate	67.15
Di-n butyl phthalate	7.0
Bis (2-ethyl hexyl) phthalate	29.2
gamma-BHC	0.05
Aldrin	0.15
Heptachlorepoxide	0.28
Endrin ketone	0.32

Active Area Surface Soil - Two surface soil samples were collected from the leachate seeps. A summary of the data is presented in Table 6-8.

The chemicals of concern selected for risk characterization in surface soil from the leachate seeps along with their representative concentrations are as follows:

Chemical	C-mean (mg/kg)
Zinc	174.0
Acetone	1.1
2-Butanone	2.1
Toluene	1.2
Xylenes	0.04
4-Methyl phenol	1.5
Benzoic acid	1.1
Diethyl phthalate	0.05
Phenanthrene	0.07
Di-n-butyl phthalate	0.2
Fluoranthene	0.05
Pyrene	0.02
Butyl benzyl phthalate	0.05
Bis (2-ethyl hexyl) phthalate	0.4

Active Area Subsurface Soil - Six subsurface soil samples (test pit soil and landfill mound samples) were collected during the Phase 1 RI activities. Table 6-9 presents the detected chemical concentrations and the background soil concentrations to assist in the selection of chemicals of concern for risk characterization.

Among the inorganics, aluminum, barium, iron, potassium, sodium, and vanadium were detected at concentrations lower than the background soil levels and were not retained for the risk characterization. Calcium, iron, magnesium, potassium, and sodium are essential nutrients, are not hazardous at the concentrations detected, and were not retained for the risk characterization.

The chemicals of concern selected for risk characterization in the active area subsurface soil along with their representative concentrations are as follows:

Chemical	C-mean (mg/kg)
Antimony	139.0
Arsenic	5.4
Beryllium	0.7
Chromium	9,982.2
Cobalt	6.2
Copper	59.4
Lead	224.9
Manganese	899.2
Mercury	0.7
Nickel	35.4
Silver	12.0
Zinc	222.2
Methylene chloride	0.03
Acetone	0.5
4-Methyl-2-pentanone	0.08
Toluene	0.1
Ethyl benzene	0.04
Styrene	0.009
Xylenes	0.3
Phenol	3.1
4-Methyl phenol	45.5
1,2,4-Trichlorobenzene	1.6
Naphthalene	37.3
4-Chloro-3-methyl phenol	0.9
2-Methyl naphthalene	6.9
Di-n-butyl phthalate	0.8
Bis (2-ethyl hexyl) phthalate	14.5
Beta-BHC	0.2
4-4'-DDE	0.07
4-4'-DDD	0.07

Inactive Area Surface Soil - Fourteen inactive soil samples were collected within the landfill area. A summary of the data is presented in Table 6-10. In Table 6-10 the detected chemical concentrations were compared with the background soil concentrations to assist in the selection of chemicals for risk characterization.

Among the inorganics, aluminum, barium, beryllium, and vanadium were detected at concentrations lower than the background soil levels and were not retained for risk characterization. Calcium, iron, magnesium, potassium, and sodium are essential nutrients, are not considered hazardous at the concentrations detected, and were not retained for risk characterization.

The chemical of concern selected for risk characterization detected in the inactive area surface soils along with their representative concentrations are as follows:

Chemical	C-mean (mg/kg)
Antimony	110.0
Arsenic	14.6
Cadmium	3.2
Chromium	10,984
Cobalt	9.9
Copper	88.2
Lead	326
Manganese	323
Nickel	20.7
Zinc	269
Mercury	0.28
Acetone	23.30
2-Butanone	6.1
Toluene	2.7
Xylene	5.0
Ethylbenzene	2.4
Chlorobenzene	0.89
4-Methylphenol	37.2
Diethyl phthalate	0.6
Benzoanthracene	0.9
Chrysene	0.10
Fluorene	1.0
Pyrene	1.4
Napthalene	5.1

Chemical	C-mean (mg/kg)
2 Methyl napthalene	5.1
Bis (2-ethyl hexyl) phthalate	3.5

Inactive Area Subsurface Soil - Thirteen subsurface soil samples were collected during the Phase 2 RI investigation. Table 6-11 presents the detected chemical concentrations and the background soil concentrations to assist in the selection of chemicals of concern for risk characterization.

Among the inorganics, aluminum, barium, cobalt, copper, manganese, nickel, and vanadium were detected at concentrations lower than the background soil concentrations and were not retained for the risk characterization.

The chemicals of concern selected for risk characterization in subsurface soil along with their representative concentrations are as follows:

Chemical	C-mean (mg/kg)
Antimony	11.00
Arsenic	117.78
Chromium	61.09
Lead	106.0
Mercury	0.28
Zinc	129
Beryllium	0.7
Acetone	0.07
Toluene	2.0
Ethyl benzene	.86
Xylenes	5.9
4-Methyl phenol	3.6
2,4,6-Trichlorophenol	.5
Fluoranthene	0.24
Octylphthalate	.22
Chrysene	.25
Bis (2-ethyl hexyl) phthalate	.5
Diethylphthalate	.30
Phenenthsene	.24

Ambient Air - Twelve ambient air samples were collected for chromium and organics analysis during the Phase 1 RI activities. Twenty ambient air samples were collected for chromium and particulate analysis only during the Phase 2 investigation. A summary of the detected chemicals is presented in Table 6-12. In Table 6-12, the detected chemical concentrations were compared with the field blanks to assist in the selection of chemicals of concern for risk characterization.

Methylene chloride, chloroform, 1,1,1-trichloroethane, and toluene were eliminated because their concentrations did not exceed five times the maximum detected concentrations in the field blanks. 2-Propanone and chromium were retained as chemicals of potential concern in ambient air.

The chemicals of concern selected for risk characterization in the ambient air along with their representative concentrations are as follows:

Chemical	C-mean (mg/kg)
2-Propanone	10.5
Chromium	11.72

6.3 EXPOSURE ASSESSMENT

This section addresses the potential pathways by which human populations could be exposed to contaminants originating from the Gloversville Landfill site. In identifying potential pathways of exposure, both current and possible future site and nearby land use conditions were considered.

The potential pathways of contaminant exposure were identified based on data concerning the distribution of contaminants at the site, physical characteristics of the site, potential routes and mechanisms of contaminant transfer, and identified patterns of land use at the site and surrounding areas.

An exposure pathway consists of the following four elements:

- a source and mechanism of chemical release (e.g., leaching)
- an environmental transport medium (e.g., groundwater)
- a point of potential human contact with the contaminated medium (e.g., residential wells)
- an exposure route (e.g., ingestion of the groundwater).

The exposure pathway is considered to be "complete" if all of the above elements are present. If one or more elements are not present, exposure does not occur.

6.3.1 Identification of Exposure Pathways - To determine potential sources of release and potential transport media, descriptions of the site presented in previous chapters of the IR and the data collected from the field investigations were utilized. Site-specific release mechanisms/sources, exposure media, potentially exposed populations, and exposure routes are identified in this section.

There are three primary routes via which individuals may be exposed to site related contaminants: ingestion, inhalation, and dermal absorption. The following subsections describe the potential exposure pathways for each environmental medium associated with the Gloversville Landfill site under present and potential future site use conditions.

The six receptor groups for which public health risks were evaluated at the Gloversville Landfill site are as follows:

1. An Adult Landfill Worker (Off-Site Resident and Non-Resident of the Area):
   The Off-Site Resident Adult Landfill Worker is an individual who works at the landfill, lives near the landfill, and uses private well groundwater. The Non-Resident Adult Landfill Worker is an individual who works at the landfill, but does not live within the potentially impacted area. Both of these receptor groups are applicable under the current and future exposure scenarios.

2. An Adult Landfill Area Resident (On-Site and Off-Site):
   The On-Site Adult Landfill Area Resident is an individual who uses contaminated groundwater from the site. This receptor group is applicable only under the future exposure scenario. The Off-Site Adult Landfill Area Resident is an individual who lives near the landfill. This receptor group is applicable under both the current and future exposure scenarios. Under the current exposure scenario, the individual uses private well groundwater. Under the future exposure scenario, the individual uses contaminated groundwater that has migrated off the site.

3. An Adult Off-Site Landfill Area Resident and Landfill Trespasser:
   The Off-Site Adult Landfill Area Resident and Landfill Trespasser is an individual who lives near the landfill. This receptor group is applicable under both the current and future exposure scenarios. Under the current exposure scenario, the individual uses private well groundwater. Under the future exposure scenario, the individual uses contaminated groundwater that has migrated off the site. Under both scenarios, the receptor group is additionally exposed to landfill contaminants as a result of trespassing.

4. An Adolescent Landfill Area Resident (On-Site and Off-Site):
   The On-Site Adolescent Landfill Area Resident is an individual who uses contaminated groundwater from the site. This receptor group is applicable only under the future exposure scenario. The Off-Site Adolescent Landfill Resident is an individual who lives near the landfill. This receptor group is applicable under both the current and future exposure scenarios. Under the current exposure scenario, the individual uses private well groundwater. Under the future exposure scenario, the individual uses contaminated groundwater that has migrated off the site.

5. An Adolescent Off-Site Landfill Area Resident and Landfill Trespasser: The Off-Site Adolescent Landfill Area Resident and Landfill Trespasser is an individual who lives near the landfill. This receptor group is applicable under both current and future exposure scenarios. Under the current exposure scenario, the individual uses private well groundwater. Under the future exposure scenario, the individual uses contaminated groundwater that has migrated off the site. Under both scenarios, the receptor group is additionally exposed to landfill contaminants as a result of trespassing.

6. A Child Landfill Area Resident (On-Site and Off-Site): The On-Site Child Landfill Area Resident is an individual who uses contaminated groundwater from the site. This receptor group is applicable only under the future exposure scenario. The Off-Site Child Landfill Area Resident is an individual who lives near the landfill. This receptor group is applicable under both current and future exposure scenarios. Under the current exposure scenario, the individual uses private well groundwater, and the future exposure scenario, the individual uses contaminated groundwater that has migrated off the site.

Potential exposure pathways are summarized in Table 6-13. An identified pathway does not imply that exposures are actually occurring, only that the potential exists for the pathway to be complete. The components of the exposure pathways identified for the Gloversville Landfill site are described in the following subsections.

6.3.1.1 Environmental Media

Air - Two contaminants were identified as chemicals of concern in the ambient air. The potential population exposed via inhalation are current and future site workers and trespassers, and future site residents. However, the reference concentration (RfC) and slope factor (SF) for chromium and propanone is not available. Thus Hazard Quotient and cancer risk for these chemicals were not determined. During the Phase 1 Risk

Assessment, risk calculations for airborne chromium were calculated based on the RfC and SF established by EPA at that time. This report showed a significant impact due to airborne chromium. Since that time, EPA has withdrawn the RfC and SF values. These values are currently under review by the EPA study group.

Active Area Surface Soil - One inorganic, four volatile organic, and nine semivolatile organic chemicals were identified as chemicals of concern in surface soil. The exposure pathways for chemicals of concern in surface soil are incidental ingestion and dermal absorption by current and future site workers and trespassers, and future on-site residents.

Active Area Subsurface Soil - Twelve inorganic, seven volatile organic, and eight semivolatile organic chemicals, and three pesticides were retained for risk characterization of subsurface soil. Since the landfill surface is vegetated and no construction activity is occurring, present exposure through subsurface soil has not been considered for risk characterization. For the future scenario, a short-term construction project involving excavation of subsurface soil was assumed. Under this future risk scenario, it has been assumed that exposures to chemicals of concern contained in all of the subsurface (2 feet and below) soil samples collected would occur.

Inactive Area Surface Soil - Eleven inorganic, six volatile organic and nine semivolatile organic chemicals detected in inactive surface soils were retained for risk characterization. The exposure pathways for chemicals of concern in surface soil are incidental ingestion and dermal absorption by current and future site workers and trespassers, and future on-site residents.

Inactive Area Subsurface Soil - Seven inorganic, four volatile organic, and eight semivolatile organic chemicals were retained for risk characterization of subsurface soil. Since the landfill surface is vegetated and no construction activity is occurring, present exposure through subsurface soil has not been considered for risk characterization. Under the future scenario of a short-term construction project involving excavation of subsurface soil, it has been assumed that exposure to chemicals of concern contained in all of the subsurface (2 feet and below) soil samples collected would occur.

Sediment - Five inorganic, two volatile organic, and five semivolatile organic chemicals were retained for risk characterization. The potential population exposed to incidental ingestion and dermal absorption of sediments includes current and future site workers and trespassers, and future on-site residents. The scenario involving a future child resident was not considered as part of the risk characterization. It was assumed that the likelihood of a child (approximately 1 year old) being exposed to sediment is very low due to the young age and thus risk from this media will be minimal. Additionally, the risk characterization for a child will be similar to that of an adolescent.

Groundwater - Thirteen inorganic and one semivolatile organic chemicals detected in groundwater from on-site monitoring wells have been retained for risk characterization. One inorganic chemical was retained for risk characterization for current exposure as a result of potentially impacted private well water usage. However, due to reduced list of parameters analyzed on "impacted wells", C-mean values for copper, manganese, mercury, acetone, and 2-butanone will be assumed to be equal in the two groups of private wells. Four inorganic and two volatile organic chemicals were retained for risk characterization for current exposure as a result of nearby private wells. Residents using groundwater from private wells were evaluated for present risk via the ingestion, inhalation (during showering), and dermal absorption routes. For the future case, it was assumed that in the future, off-site groundwater contained the same contaminants and contaminant levels as on-site groundwater and that the site would be used for residential purposes. This is a very conservative assumption that EPA typically requires for risk assessment evaluations. Future off-site residents and potential on-site residents were evaluated for future risk via the ingestion, inhalation (during showering), and dermal absorption of contaminated groundwater.

Surface Water - Six inorganic and one semivolatile organic chemical were retained for risk characterization. The risk characterization for the inorganic and organic chemicals of concern in surface water was evaluated for current and future site workers and trespassers, and future on-site residents. The scenario involving a future child resident was not considered as part of the risk characterization. It is assumed that the likelihood of a child (approximately 1 year old) being exposed to surface water is very low due to the young age, thus the risk from this media will be minimal. Additionally, the risk characterization for a child will be similar to an adolescent.

Leachate - A total of forty-five chemicals of concern were identified in the leachate. Of the forty-five chemicals, sixteen are inorganic, fourteen are volatile organic, eleven are semivolatile organic chemicals, and four are pesticides. Current and future site workers and trespassers, and future on-site residents may be exposed to leachate through accidental contacts. The potential route of exposure is through dermal contact with the leachate. Ingestion of leachate is unlikely because of the small size of the leachate seep and its shallow depth.

6.3.1.2 Exposure Routes - Potential exposure routes for the identified media of concern at the Gloversville Landfill site are as follows:

- Inhalation of ambient air (site workers, trespassers, and future site residents).
- Incidental ingestion of active area surface soil (site workers, trespassers, and future site residents).
- Dermal contact with active area surface soil (site workers, trespassers, and future site residents).
- Incidental ingestion of excavated active area subsurface soil (future site workers).
- Dermal contact with excavated active area subsurface soil (future site workers).
- Incidental ingestion of inactive area surface soil (site workers, trespassers, and future site residents).

- Dermal contact with inactive area surface soil (site workers, trespasses, and future site residents).
- Incidental ingestion of excavated inactive area subsurface soil (future site workers).
- Dermal contact with excavated inactive active area subsurface soil (future site workers).
- Incidental ingestion of sediment (site workers, trespassers, and future site residents).
- Dermal contact with sediment (site workers, trespassers, and future site residents).
- Current use of private well water (area residents via ingestion, inhalation of volatiles while showering, and dermal contact while bathing or showering).
- Future use of contaminated groundwater (future area residents via ingestion, inhalation of volatiles while showering, and dermal contact while bathing or showering).
- Ingestion of surface water (site workers, trespassers, and future site residents).
- Dermal absorption of surface water (site workers, trespassers, and future site residents).
- Dermal absorption of leachate water (site workers, trespassers, and future site residents).

<u>6.3.1.3 Exposure Populations</u> - The exposure routes identified above were combined for different groups of receptors. The receptor groups and the relevant exposure routes considered are identified below. All the receptor groups include current and future exposure routes for various environmental media, except groundwater. The private well groundwater is considered for the current use scenario and the contaminated site groundwater is considered for the future use.

Receptor Group	Relevant Exposure Routes
Adult Workers	Current and potential future use (ingestion, inhalation while showering, and dermal absorption) of groundwater (for workers who are also residents of the landfill area).
	Incidental ingestion of active surface soil, active subsurface soil, inactive surface and subsurface soil, sediment, surface water, and leachate.
	Dermal contact with active surface soil, active subsurface soil, inactive surface and subsurface soil, sediment, surface water, and leachate.
	Inhalation of ambient air.
Adult Residents	Current and potential future use (ingestion, inhalation while showering, and dermal absorption) of groundwater.
	Incidental ingestion of active surface soil, inactive surface soil, sediment, surface water, and leachate.
	Dermal contact with active surface soil, inactive surface soil, sediment, surface water, and leachate.
	Inhalation of ambient air.
Adult Trespassers	Incidental ingestion of active surface soil, inactive surface soil, sediment, surface water, and leachate.
	Dermal contact with active surface soil, inactive surface soil, sediment, surface water, and leachate.
	Inhalation of ambient air.
Adolescent Residents	Current and potential future use (ingestion, inhalation while showering, and dermal absorption) of groundwater.
	Incidental ingestion of active surface soil, inactive surface soil, sediment, surface water, and leachate.
	Dermal contact with active surface soil, subsurface soil, inactive surface and subsurface soil, sediment, surface water, and leachate.
	Inhalation of ambient air.

Adolescent Trespassers	Incidental ingestion of surface soil, inactive surface soil, sediment,surface water, and leachate.
	Dermal contact with surface soil, inactive surface soil, sediment, surface water, and leachate.
	Inhalation of ambient air.
Child Residents	Current and potential future use (ingestion, inhalation while showering, and dermal absorption) of groundwater.
	Incidental ingestion of surface soil and inactive surface soil.
	Dermal contact with surface soil and inactive surface soil.
	Inhalation of ambient air.

<u>6.3.2 Exposure Assessment Methods and Assumptions</u> - The methods/models used to assess human intakes via each of the potential exposure routes are described in this subsection.

6.3.2.1 Exposure to Ambient Air

<u>Inhalation Exposure</u> - Intakes for inhalation of ambient air were generated using the following equation:

$$\text{Intake (mg/kg/day)} = \frac{C \times IR \times ET \times EF \times ED}{6BW \times AT \times 365}$$

Where:

C = Contaminant concentration in air (mg/m^3)
IR = Inhalation rate (m^3/hr)
ET = Exposure time (hrs/day)
EF = Exposure frequency (days/yr)
ED = Exposure duration (yrs)
BW = Body weight (kg)
AT = Averaging time (yrs)
365 = Conversion factor (days/yr)

The values for the variables used in the above equation are as follows:

Parameter	Adult Worker	Adult Trespasser	Adolescent Trespasser	Adult Resident	Adolsecent/ Child Resident
C (mg/m^3)	C-mean	C-mean	C-mean	C-mean	C-mean
IR (m3/hr)	1.25	1.25	1.5	1.25	1.50/1.25
ET (hrs/day)	8	3	3	24	24/24
EF (days/yr)	60	24	60	365	365/365
ED (yrs)	40	70	*	70	*
BW (kg)	70	70	35	70	35/10
AT (yrs)	40/70**	70	*	70	**

*** For noncarcinogenic risk, ED and AT are equal and will not affect the intake calculation. For carcinogenic risk, ED and AT are not applicable.**

**** 40 years for noncarcinogenic risk and 70 years for carcinogenic risk.**

Source: "Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual" (EPA, December 1989), and "Exposure Factors Handbook" (EPA, March 1990).

The exposure time (ET) and exposure frequency (EF) are best professional estimates based on current and future land uses and activities in the area of the landfill.

6.3.2.2 Exposure to the Active Area Surface Soil

Ingestion exposure - Intakes for ingestion of surface soil were generated using the following equation:

$$\text{Intake (mg/kg/day)} = \frac{(C \times IR \times FI \times EF \times ED)}{(BW \times AT \times 365 \times 10^6)}$$

Where:

C = Representative concentration (C-mean) of the contaminant in soil (mg/kg)
IR = Soil ingestion rate (mg/event)
FI = Fraction ingested from the contamination source (unitless)
EF = Exposure frequency (events/yr)
ED = Exposure duration (yrs)
BW = Receptor body weight (kg)
AT = Averaging period (yrs)
365 = Conversion Factor (days/yr)
10^6 = Conversion factor (mg/kg)

The values for the variables used in the above equation are as follows:

Parameter	Adult Worker	Adult Trespasser	Adolescent Trespasser	Adult Resident	Adolsecent/ Child Resident
C (mg/kg)	C-mean	C-mean	C-mean	C-mean	C-mean
IR (mg/event)	100	100	200	100	200/200
EF (events/yr)	10	6	10	180	180/180
ED (yrs)	40	70	*	70	*
BW (kg)	70	70	35	70	35/10
AT (yrs)	40/70**	70	*	70	*

* For noncarcinogenic risk, ED and AT are equal and will not affect the intake calculation. For carcinogenic risk, ED and AT are not applicable.

** 40 years for noncarcinogenic risk and 70 years for carcinogenic risk.

Source: "Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual" (EPA, December 1989).

The soil ingestion rates (IR) and exposure frequencies (EF) are best professional estimates based on current and future land uses and activities in the area of the landfill. A conservative value of 1.0 was used for the fraction ingested from the contaminated source (FI).

Dermal Exposure - Dermal absorption intakes for surface soil were generated using the following equation:

$$\text{Intake (mg/kg/day)} = \frac{C \times SA \times AF \times ABS \times EF \times ED)}{(BW \times AT \times 365 \times 10^6)}$$

Where:

C = Representative concentration (C-mean) of the contaminant in soil (mg/kg)
SA = Exposed surface area of the skin (cm^2/day)
AF = Adherence factor (mg/cm^2)
ABS = Absorption fraction of the chemical
EF = Exposure frequency (days/year)
ED = Exposure duration (yrs)
BW = Receptor body weight (kg)
AT = Averaging period (yrs)
365 = Conversion factor (days/yr)
10^6 = Conversion factor (mg/kg)

The variable values for the above equation are as follows:

Parameter	Adult Worker	Adult Trespasser	Adolescent Trespasser	Adult Resident	Adolsecent/ Child Resident
C (mg/kg)	C-mean	C-mean	C-mean	C-mean	C-mean
SA (cm2/day)	3940	3940	2740	3940	2740/1580
EF (days/yr)	10	6	10	180	180/180
ED (yrs)	40	70	*	70	*
BW (kg)	70	70	35	70	35/10
AT (yrs)	40/70**	70	*	70	*

*** For noncarcinogenic risk, ED and AT are equal and will not affect the intake calculation. For carcinogenic risk, ED and AT are not applicable.**

****: 40 years for noncarcinogenic risk and 70 years for carcinogenic risk.**

Source: "Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual" (EPA, December 1989), and "Exposure Factors Handbook" (EPA, March 1990)

An adherence factor of 1.45, and absorption fractions of 0.1 for volatile organic chemicals, 0.05 for semivolatile organic chemicals, and 0.01 for inorganic chemicals were used for the intake calculations. The exposure frequency (EF) estimates represent best professional judgements based on current and future land uses and activities in the area of the landfill. Fiftieth percentile values for SA along with average body weight were used because of the strong correlation between surface area and body weight.

6.3.2.3 Exposure to the Active Area Subsurface Soil

Ingestion Exposure - Intakes for the ingestion of subsurface soil were generated using the same equation used for surface soil. However, model input values were modified as follows:

Parameter	Value for Adult Worker
C (mg/kg)	C-mean
IR (mg/event)	100
EF (events/yr)	240
ED (yrs)	0.25

Parameter	Value for Adult Worker
BW (kg)	70
AT (yrs)	0.25/70*

* 0.25 year for noncarcinogenic risk and 70 years for carcinogenic risk.

Dermal Exposure - Dermal absorption intakes for subsurface soil were generated using the same equation used for surface soil. However, the model input values were modified as follows:

Parameter	Value for Adult Worker
C (mg/kg)	C-mean
SA (cm^2/day)	3940
EF (days/yr)	240
ED (yrs)	0.25
BW (kg)	70
AT (yrs)	0.25/70*

* 0.25 year for noncarcinogenic risk and 70 years for carcinogenic risk.

An adherence factor of 1.45, and absorption fractions of 0.1 for volatile organic chemicals, 0.05 for semi-volatile organic chemicals, and 0.01 for inorganic chemicals were used for the intake calculations. It is assumed that the on-site excavation scenario would be completed in three months.

6.3.2.4 Exposure to the Inactive Area Surface Soil

Ingestion Exposure - Intake values for the ingestion of inactive surface soil were generated using the same equation used for active surface soil. Model input values were also the same.

Dermal Exposure - Dermal exposure values for inactive surface soil were generated using the same equation used for active surface soil. Same model input values were used.

6.3.2.5 Exposure to the Inactive Area Subsurface Soil

Ingestion Exposure - Intakes for ingestion of inactive subsurface soil were generated using the same equation used for active subsurface soil. The same model input values were used.

Dermal Exposure - Dermal absorption values were generated using the same equation used for active subsurface soil. The same model input values were used.

6.3.2.6 Exposure to Sediment

Ingestion Exposure - Intakes for ingestion of sediment were generated using the same equation as previously described for surface soil.

The values of the variables used in the equation were modified as follows:

Parameter	Adult Worker	Adult Trespasser	Adolescent Trespasser	Adult Resident	Adolsecent/ Child Resident
C (mg/kg)	C-mean	C-mean	C-mean	C-mean	C-mean
IR (mg/event)	100	100	200	100	200
EF (events/yr)	2	2	4	4	10
ED (yrs)	40	70	*	70	*
BW (kg)	70	70	35	70	35
AT (yrs)	40/70**	70	*	70	*

* For noncarcinogenic risk, ED and AT are equal and will not affect the intake calculation. For carcinogenic risk, ED and AT are not applicable.

** 40 years for noncarcinogenic risk and 70 years for carcinogenic risk.

Source: "Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual" (EPA, December 1989).

The values used for the soil ingestion rate (IR) and for exposure frequency (EF) are best professional estimates based on current and future land use and activities in the area of the landfill.

<u>Dermal Exposure</u> - Exposure via this route was estimated using the same equation as previously described for surface soil dermal exposure.

The variable values were modified as follows:

Parameter	Adult Worker	Adult Trespasser	Adolescent Trespasser	Adult Resident	Adolsecent/ Child Resident
C (mg/kg)	C-mean	C-mean	C-mean	C-mean	C-mean
SA (cm^2/day)	3940	3940	2740	3940	2740
EF (days/yr)	2	2	4	4	10
ED (yrs)	40	70	*	70	*
BW (kg)	70	70	35	70	35
AT (yrs)	40/70**	70	*	70	*

* For noncarcinogenic risk, ED and AT are equal and will not affect the intake calculation. For carcinogenic risk, ED and AT are not applicable.

** 40 years for noncarcinogenic risk and 70 years for carcinogenic risk.

Source: "Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual" (EPA, December 1989), and "Exposure Factors Handbook" (EPA, March 1990).

An adherence factor of 1.45 and the absorption fractions of 0.1 for volatile organic chemicals, 0.05 for semivolatile organic chemicals, and 0.01 for inorganic chemicals were used for the intake calculations. The exposure frequency (EF) estimates are best professional estimates based on current and future land uses and activities in the area of the landfill. Fiftieth percentile values for body surface area (SA) were used along with average body weight because of the strong correlation between surface area and body weight.

6.3.2.7 Exposure to Groundwater

Ingestion Exposure - Intakes from ingestion of groundwater were generated using the following equation:

$$\text{Intake (mg/kg/day)} = \frac{(C \times IR \times EF \times ED)}{(BW \times AT \times 365)}$$

Where:

C = Representative (C-mean) chemical concentration in groundwater (mg/L)
IR = Water ingestion rate (L/day)
EF = Exposure frequency (days/yr)
ED = Exposure duration (yrs)
BW = Receptor body weight (kg)
AT = Averaging period (yrs)
365 = Conversion factor (days/yr)

Values used for the variables are as follows:

Parameter	Adult Resident	Adult Worker/ Resident	Adolescent/ Child Resident
C (mg/kg)	C-mean	C-mean	C-mean
IR (L/day)	2	2	1.5/1
EF (days/yr)	365	365	365/365
ED (yrs)	70	70	*
BW (kg)	70	70	35/10
AT (yrs)	70	70	*

* For noncarcinogenic risk, ED and AT are equal and will not affect the intake calculation. For carcinogenic risk, ED and AT are not applicable.

Source: "Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual" (EPA, December 1989).

Inhalation Exposure - Inhalation exposure during showering was estimated using a mass transfer model developed for this specific exposure route and an intake estimate model. The method employed is as follows (Foster and Chrostowski, 1987, and EPA, 1989b):

$$\text{Intake (mg/kg/day)} = \frac{(S \times IR \times K \times EF \times ED)}{(BW \times AT \times R_a \times 10^6)}$$

Where:

$$K = \frac{D_s + \exp(-R_a D_t)}{R_a - \exp[R_a(D_s - D_t)]/R_a}$$

And:

S = Volatile chemical generation rate (ug/m^3/min)
IR = Inhalation rate (L/min)
EF = Exposure frequency (days^{-1})
ED = Exposure duration (yrs)
BW = Receptor body weight (kg)
AT = Averaging period (yrs)
R_a = Air exchange rate (min^{-1})
D_s = Shower duration (min)
D_t = Total duration in the bathroom (min)
10^6 = Conversion factor (mg/ug/m^3/L)

The volatile chemical generation rate was estimated using the Foster and Chrostowski mass transfer model which is based on two-phase film theory. The model employs contaminant-specific mass transfer coefficients, Henry's Law constant, droplet diameter, drop time, viscosity, and temperature to determine the chemical concentration in vapor phase.

The following supplementary calculations were also used:

$$S = (C_{wd} \times FR)/SV$$

Where:

C_{wd} = concentration leaving water droplet (ug/l)
FR = shower flow rate (L/min)
SV = shower room air volume (m^3)

$$C_{wd} = C_{wo} [1 - \exp(-K_{aL} \times ts/60xd)]$$

Where:

C_{wo} = concentration of chemical in water (ug/l)

K_{aL} = adjusted overall mass transfer coeff. (cm/hr)

t_s = shower droplet time (sec)

d = shower droplet diameter (mm)

$$K_{aL} = K_L [T_1 M_s)/(T_s M_1)]^{-0.5}$$

Where:

K_L = overall mass transfer coefficient (cm/hr)

T_1 = calibration water temperature (K) of K_L

T_s = shower water temperature (K)

M_1 = water viscosity @ T_1 (cp)

M_s = water viscosity @ T_s (cp)

$$K_L = [1/K_l + (R \times T) / (H \times K_g)]^{-1}$$

Where:

K_L = liquid phase mass transfer coefficient (cm/hr)

K_g = gas phase mass transfer coefficient (cm/hr)

T = temperature at which Henry's Law Constant is estimated (K)

R = Ideal Gas Law Constant (atm x m^3/mol/K)

H = Henry's Law Constant (atm x m^3/mol)

$$K_L = 20 \frac{cm}{hr} \left(\frac{44}{MW}\right)^{0.5} \text{ (based on } CO_2\text{)}$$

$$K_g = 3000 \frac{cm}{hr} \left(\frac{18}{MW}\right)^{0.5} \text{ (based on } H_2O\text{)}$$

The values of the variable are as follows:

Parameter	Adult Resident	Adult Worker/ Resident	Adolescent/ Child Resident
C (mg/kg)	C-mean	C-mean	C-mean
IR (L/min)	14	14	11/11
EF (days-1)	1	1	1/1
ED (yrs)	70	70	*
BW (kg)	70	70	35/10
AT (yrs)	70	70	*

* For noncarcinogenic risk, ED and AT are equal and will not affect the intake calculation. For carcinogenic risk, ED and AT are not applicable.

Source: "Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual" (EPA, December 1989).

An air exchange rate (Ra) of 0.0083 min^{-1} was used for all receptors. The shower duration (Ds) and total duration in the bathroom (Dt) were assumed to be 12 minutes and 20 minutes, respectively.

Dermal Exposure - Dermal exposure during bathing or showering was estimated as follows:

$$\text{Intake (mg/kg/day)} = \frac{(C \times PC \times AV \times ET \times EF \times ED)}{(BW \times AT \times 10^3 \times 365)}$$

Where:

C = Representative (C-mean) chemical concentration in groundwater (mg/L)
PC = Permeability constant of water (cm/hr)
AV = Skin surface area available for contact (cm^2)
ET = Exposure time (hrs/day)
EF = Exposure frequency (days/year)
ED = Exposure duration (years)
BW = Receptor body weight (kg)
AT = Averaging period (years)
10^3 = Conversion factor (cm^3/L)
365 = Conversion factor (days/year)

This approach is based on the assumption that groundwater contaminants are present in dilute form and that percutaneous penetration is controlled by the absorption of water through the skin.

The variable values are as follows:

Parameter	Adult Resident	Adult Worker/ Resident	Adolescent/ Child Resident
C (mg/kg)	C-mean	C-mean	C-mean
PC (cm/hr)	0.0008	0.0008	0.0008/0.0008
AV (cm2)	18,150	18,150	10,600/5,900
ET (hr/day)	0.20	0.20	0.20/0.20
EF (days/yr)	365	365	365/365
ED (yrs)	70	70	*
BW (kg)	70	70	35/10
AT (yrs)	70	70	*

* For noncarcinogenic risk, ED and AT are equal and will not affect the intake calculation. For carcinogenic risk, ED and AT are not applicable.

Source: "Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual" (EPA, December 1989), and "Exposure Factors Handbook" (EPA, March 1990).

6.3.2.8 Exposure to Surface Water

Ingestion Exposure - Ingestion of surface water while swimming was estimated using the following equation:

$$\text{Intake (mg/kg/day)} = \frac{(C \times IR \times ET \times EF \times ED)}{(BW \times AT \times 365)}$$

Where:

C = Representative (C-mean) surface water concentration (mg/L)
IR = Water ingestion rate (L/hrs)
ET = Exposure time (hrs/event)
EF = Exposure frequency (events/yr)
ED = Exposure duration (yrs)

And:

BW = Receptor body weight (kg)
AT = Averaging period (yrs)
365 = Conversion factor (days/yr)

Assumptions regarding the preceding variables are as follows:

Parameter	Adult Worker	Adult Trespasser	Adolescent Trespasser	Adult Resident	Adolsecent/ Child Resident
C (mg/kg)	C-mean	C-mean	C-mean	C-mean	C-mean
IR (L/hr)	0.05	0.05	0.05	0.05	0.05
ET (hrs/event)	0.50	0.50	1.0	0.50	1.0
EF (events/yr)	2	2	4	4	10
ED (yrs)	40	70	*	70	*
BW (kg)	70	70	35	70	35
AT (yrs)	40/70**	70	*	70	*

* For noncarcinogenic risk, ED and AT are equal and will not affect the intake calculation. For carcinogenic risk, ED and AT are not applicable.

** 40 years for noncarcinogenic risk and 70 years for carcinogenic risk.

Source: "Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual" (EPA, December 1989).

Ingestion rate (IR) and exposure frequency (EF) are best professional estimates based on current and future land uses and activities in the area of the landfill.

Dermal Exposure - Dermal exposure intakes for surface water were generated using the same equation as previously identified for groundwater dermal exposure. However, the variable values were modified as follows:

Parameter	Adult Worker	Adult Trespasser	Adolescent Trespasser	Adult Resident	Adolsecent/ Child Resident
C (mg/kg)	C-mean	C-mean	C-mean	C-mean	C-mean
PC (cm/hr)	0.0008	0.0008	0.0008	0.0008	0.0008
AV (cm2)	18,150	18,150	10,600	18,150	10,600
ET (hr/day)	0.75	0.75	1.25	0.75	1.25
EF (days/yr)	2	2	4	4	10
ED (yrs)	40	70	*	70	*
BW (kg)	70	70	35	70	70
AT (yrs)	40/70**	70	*	70	*

** For noncarcinogenic risk, ED and AT are equal and will not affect the intake calculation. For carcinogenic risk, ED and AT are not applicable.

* 40 years for noncarcinogenic risk and 70 years for carcinogenic risk.

Source: "Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual" (EPA, December 1989), and "Exposure Factors Handbook" (EPA, March 1990).

Exposure time (ET) and exposure frequency (EF) are best professional estimates based on current and future land uses and activities in the area of the landfill.

6.3.2.9 Exposure to Leachate

Dermal Exposure - Dermal exposure intakes for leachate were generated using the same equation as previously identified for groundwater dermal exposure. However, the variable values used in the equation were modified as follows:

Parameter	Adult Worker	Adult Trespasser	Adolescent Trespasser	Adult Resident	Adolsecent/ Child Resident
C (mg/kg)	C-mean	C-mean	C-mean	C-mean	C-mean
PC (cm/hr)	0.0008	0.0008	0.0008	0.0008	0.0008
AV (cm2)	18,150	18,150	10,600	18,150	10,600
ET (hr/day)	0.2	0.2	0.2	0.2	0.2
EF (days/yr)	1	1	2	2	4
ED (yrs)	40	70	*	70	*
BW (kg)	70	70	35	70	35
AT (yrs)	40/70**	70	*	70	*

[*] For noncarcinogenic risk, ED and AT are equal and will not affect the intake calculation. For carcinogenic risk, ED and AT are not applicable.

[**] 40 years for noncarcinogenic risk and 70 years for carcinogenic risk.

Source: "Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual" (EPA, December 1989), and "Exposure Factors Handbook" (EPA, March 1990).

Exposure time (ET) and exposure frequency (EF) are best professional estimates based on current and future land uses and activities in the area of the landfill.

6.4 TOXICITY ASSESSMENT

In this section, toxicologic properties of each selected chemical of concern are presented. In addition, the toxicologic criteria used to evaluate human health risks are discussed. The primary sources of toxicity information are U.S. EPA's Integrated Risk Information System (IRIS) data base and the Health Effects Assessment Summary Tables (HEAST) (USEPA, March 1992).

6.4.1 Classification of Health Effects - For the purposes of risk assessment, individual contaminants are separated into two categories of chemical toxicity depending on whether they exhibit the potential for carcinogenic or noncarcinogenic effects in humans.

6.4.1.1 Health Effects Criteria For Carcinogenic Effects - For chemicals exhibiting carcinogenic effects, it is recognized by U.S. EPA as well as other scientific authorities that one or more molecular events can evoke changes in a single cell or a small number of cells that can lead to tumor formation. This is the non-threshold theory of carcinogenesis which assumes that any level of exposure to a carcinogen can result in some finite possibility of causing the disease. Generally, regulatory agencies assume the non-threshold hypothesis for carcinogenic effects in the absence of information about the mechanisms of action for a particular chemical.

U.S. EPA's Carcinogen Assessment Group (CAG) has developed cancer slope factors (i.e., dose-response values) for estimating excess lifetime cancer risks associated with various levels of exposure to potential human carcinogens. The cancer slope factor in units of (mg chemical/kg body weight/day)$^{-1}$ is a number which, when multiplied by the lifetime average daily intake of a potential carcinogen (in mg chemical/kg body weight/day), yields the upper-bound lifetime excess cancer risk associated with exposure at that intake. Upper bound is a term used by U.S. EPA to reflect the conservative nature of the cancer slope factors; risks estimated using cancer slope factors are considered unlikely to underestimate actual risks but they may overestimate actual risks for a given exposure. This multiplication approach can be used for low doses corresponding to cancer risks lower than 1E-02. Excess lifetime cancer risks are generally expressed in scientific notation and are probabilities. An excess lifetime cancer risk of 1E-06, for example, represents the incremental probability of one in one million that an individual will develop cancer as a result of exposure to a carcinogenic chemical over a 70-year lifetime period under specified exposure conditions. EPA has suggested developing remedial alternatives for cleanup of Superfund sites to achieve total excess lifetime cancer risk ranging from no more than 1E-04 (one in ten thousand) to 1E-07 (one in ten million) (EPA, 1986a).

In practice, cancer slope factors are derived from the results of human epidemiology studies or chronic animal bioassays. For the latter, data from animal studies are fitted to the linearized multistage model and a dose-response curve is obtained. The 95th percentile upper confidence limit slope of the dose-response curve is subjected to various adjustments and an interspecies scaling factor is applied to conservatively derive the cancer slope factor for humans. Thus, the actual risks associated with exposure to a potential carcinogen quantitatively evaluated based on animal data are not likely to exceed the risks estimated using these cancer slope factors, but they may be much lower. Dose-response data derived from human epidemiological studies are fitted to dose-time-response curves on ad hoc basis. These models provide the upper limit estimates on lifetime cancer risk. Cancer slope factors based on human epidemiological data are also derived using very conservative assumptions and, as such, they too are considered unlikely to underestimate risks.

In addition, there are varying degrees of confidence in the weight of evidence for carcinogenicity of a given chemical. U.S. EPA (EPA, 1986a) has proposed a system for characterizing the overall weight of evidence for a chemical's carcinogenicity based on the availability of animal, human, and other supportive data. The weight-of-evidence classification is an attempt to determine the likelihood that an agent is a human carcinogen and thus qualitatively affects the estimation of potential health risks. Three primary factors are considered in characterizing the overall weight of evidence for carcinogenicity: (1) the quality of evidence from human studies, (2) the quality of evidence from animal studies which are combined into a characterization of the overall weight of evidence for human carcinogenicity, and (3) other supportive information which is assessed to determine whether the overall weight of evidence should be modified. U.S. EPA's final classification of the overall weight of evidence includes the following five categories:

Group A--Human Carcinogen: This category indicates that there is sufficient evidence from human epidemiological studies to support a causal association between an agent and cancer.

Group B--Probable Human Carcinogen: This category indicates that there is at least limited evidence from epidemiological studies of carcinogenicity to humans (Group B1) or that, in the absence of adequate data on humans, there is sufficient evidence of carcinogenicity in animals (Group B2).

Group C--Possible Human Carcinogen: This category indicates that there is limited evidence of carcinogenicity in animals in the absence of data on humans.

Group D--Not Classified: This category indicates that the evidence for carcinogenicity in animals is inadequate.

Group E--Evidence of Non-Carcinogenicity for Humans: This category indicates the evidence for noncarcinogenicity in humans (no evidence of carcinogenicity in adequate studies).

Quantitative carcinogenic risk assessments are performed for chemicals in Groups A, B, and C. Cancer slope factors are developed based on epidemiological or animal bioassay data for a specific route of exposure, either oral or inhalation. No slope factors are available for the dermal route of exposure. Carcinogenic risks associated with dermal exposure have been evaluated using oral slope factors (except in cases where evidence of carcinogenicity is highly route-specific). For example, it is inappropriate to use the oral slope factor to evaluate the risks associated with dermal exposure to carcinogens such as benzo(a) pryrene, which causes skin cancer through a direct action at the point of application. Generally, only a qualitative assessment of risks from dermal exposure to these chemicals is possible.

6.4.1.2 Health Effects Criteria For Noncarcinogenic Effects - For chemicals that exhibit noncarcinogenic effects, many authorities consider organisms to have repair and detoxification capabilities that must be exceeded by some critical concentration (threshold) before the health effect is manifested. For example, an organ can have a large number of cells performing the same or similar functions that must be significantly depleted before the effect on the organ is seen. This threshold view holds that a range of exposures from just above zero to some finite value can be tolerated by the organism without an appreciable risk of adverse effects.

Health criteria for chemicals exhibiting noncarcinogenic effects for use in risk assessment are developed using reference doses (RfDs) developed by the U.S. EPA's RfD Work Group. The RfD is expressed in units of mg chemical/kg body weight/day. The RfD is an estimate of an average daily exposure to an individual (including sensitive individuals) below which there will not be an appreciable risk of adverse health effects. The RfD is derived using conservative safety factors (e.g., to adjust from animals to humans and to protect sensitive populations) to ensure that it is unlikely to underestimate the potential for adverse noncarcinogenic effects to occur. Doses that are significantly higher than the RfD may indicate that an inadequate margin of safety could exist for exposure to that substance and that an adverse health effect could occur.

6.4.2 Potential Health Effects for Chemicals of Concern - A brief summary of the critical human health effects associated with long-term (chronic) exposure to each of the selected chemicals of concern is presented in this section. In addition, the toxicity values used in the quantitative risk assessment (cancer slope factors and RfDs for each chemical of concern) are presented.

Some chemicals exhibit different health effects depending upon the route of exposure (e.g., inhalation versus oral). For example, a chemical may be associated with an increased risk of cancer when inhaled but may have no evidence of carcinogenicity when ingested [e.g., nickel and cadmium (EPA, 1987a and b)]. For these chemicals, route-specific toxicity values, where available, are used. Table 6-14 presents the oral and inhalation toxicity values which were used to estimate quantitative risk.

A brief discussion of the toxicologic properties of chemicals of concern is presented in the following subsections. The primary source for toxicologic information is the IRIS data base.

6.4.2.1 Inorganic Chemicals

Aluminum - Aluminum in the form typically found at hazardous waste sites is not considered hazardous. Inhalation of finely divided aluminum powder has been reported to cause pulmonary fibrosis.

Antimony - Antimony occurs in nature both in the trivalent and pentavalent oxidation states. Exposure to antimony can produce electrocardiogram changes and myocardial lesions. Oral exposure of humans to therapeutic doses has been associated with nausea, vomiting, and hepatic necrosis. EPA has derived an oral RfD of 4E-4 mg/kg/day for antimony based on a chronic oral study in which rats given the metal in drinking water had altered blood glucose and blood cholesterol levels and decreased life span.

Arsenic - Arsenic is readily absorbed via the oral and inhalation routes. Acute exposure of humans to metalloid arsenic has been associated with gastrointestinal effects, hemolysis, and neuropathy. Chronic exposure of humans to arsenic can produce toxic effects on both the peripheral and central nervous systems, precancerous dermal lesions, and cardiovascular damage. Arsenic is a known human carcinogen. Epidemiological studies of workers in smelters and in plants manufacturing arsenic pesticides have shown that inhalation of arsenic is strongly associated with lung cancer. EPA has classified arsenic in Group A as a human carcinogen. EPA has reported an inhalation slope factor of 5E+1 $(mg/kg/day)^{-1}$ and oral RfD of 1E-3 mg/kg/day for arsenic.

Barium - Ingested or inhaled soluble barium compounds are highly toxic in humans. The most important effect of acute barium poisoning is a strong, prolonged stimulant action on muscle. Accidental ingestion of soluble barium salts has resulted in gastroenteritis, muscular paralysis, and extra systoles. Baritosis, a benign pneumoconiosis, is an occupational disease arising from the inhalation of barium sulfate, barium oxide, and barium carbonate dust. EPA has reported an oral RfD and an inhalation Reference Concentration (RfC) of 5E-2 and 1E-4 mg/kg/day respectively.

Beryllium - Beryllium is not readily absorbed by any route of exposure. Acute respiratory effects due to beryllium exposure include rhinitis, pharyngitis, tracheobronchitis, and acute pneumonitis. Dermal exposure to soluble beryllium compounds can cause contact dermatitis, ulceration, and granulomas. The most common clinical symptom caused by chronic beryllium exposure is granulomatous lung inflammation. Beryllium is a Group B2 carcinogen and has caused lung and/or bone tumors in experimental animals. Oral and inhalation slope factors of 4.3 and 8.4 $(mg/kg/day)^{-1}$, respectively, have been reported for beryllium. An oral RfD of 5E-3 mg/kg/day has been reported for beryllium.

Cadmium - Two oral RfDs for cadmium have been estimated: 1E-3 mg/kg/day for food ingestion and 5E-4 mg/kg/day for water ingestion. These matrix-specific values were based on studies of cadmium-induced kidney damage in humans. An inhalation slope factor of 6.1 $(mg/kg/day)^{-1}$ has been reported based on a epidemiological study of occupationally exposed cadmium smelter workers.

Chromium - Gastrointestinal absorption of chromium III is low, whereas chromium VI is more readily absorbed following oral exposure. Chromium is an essential micronutrient and is not toxic in trace quantities. The epidemiological studies of worker populations have established that inhaled chromium VI is a human carcinogen, and respiratory passages and lungs are the target organs. Inhalation of chromium III and ingestion of chromium III or VI have not been associated with carcinogenicity in humans or experimental animals. EPA has classified inhaled chromium VI in Group A as a human carcinogen. An oral RfD of 5E-3 mg/kg/day has been established for chromium (VI). An inhalation reference concentration of 6.0E-07 (mg/kg/day) and an inhalation slope factor 4.1E+1 was earlier recommended for chromium (VI). These values were used to calculate the hazard quotients and cancer risks during Phase 1 Risk Assessment, which showed a significant impact due to airborne chromium. Currently, no inhalation RfC or SF are available from EPA data base as the previous values have been withdrawn and are under review by EPA study group. Therefore, a risk calculation for airborne chromium could not be performed.

Cobalt - Little information regarding toxic effects of exposure to cobalt is available. The oral lethal dose (LD_{50}) value for cobalt is 1,500 mg/kg in the rat. The absence of positive carcinogenic responses in experimental animals and the lack of epidemiologic evidence suggest that cobalt is unlikely to pose a carcinogenic risk to humans. No chronic toxicity value for cobalt is available.

Copper - Copper has been given a weight of evidence classification of D (not classified as a human carcinogen) based on no human data and inadequate animal data. An oral RfD of 4E-2 mg/kg/day has been established by EPA.

Cyanide - Cyanide has been given a weight of evidence classification of D (not classified as a human carcinogen) based on a lack of pertinent data regarding carcinogenicity in the available literature. An oral RfD of 2E-2 mg/kg/day has been established for cyanide.

Lead - Lead causes alterations in the hematopoietic and nervous systems in humans. The toxic effects are generally related to concentration of lead in blood. Higher blood concentration of lead can cause severe, irreversible brain damage, encephalopathy, and possible death. The Center for Disease Control (CDC) has used the value of 25 ug/l as an acceptable level of lead in blood. EPA has classified lead as a Group B2 - probable human carcinogen. The Group B2 classification is based on the increased renal tumors in experimental animals. No cancer slope factor or RfD has been established for lead.

Manganese - Manganese is a normal constituent of soil and is not particularly toxic. An oral RfD of 1E-1 mg/kg/day and an inhalation RfC of 1E-4 mg/kg/day has been established for manganese.

Mercury - Occupational exposure of workers to elemental mercury vapors has been associated with mental disturbances, tremors, and gingivitis. The central nervous system is a major target for organic mercury compounds. An oral RfD of 3E-4 mg/kg/day and an inhalation RfC of 9E-5 mg/kg/day has been established for mercury.

Nickel - Epidemiological evidence indicates that inhalation of nickel refinery dust and nickel subsulfide is associated with cancers of the nasal cavity, lung, larynx, kidney, and prostate. Inhalation of nickel dust is classified in Group A (human carcinogen). An inhalation cancer slope factor of 1.7 $(mg/kg/day)^{-1}$ and an oral RfD of 2E-2 mg/kg/day has been reported for nickel.

Vanadium - Occupational exposure to airborne vanadium compounds can produce eye and skin irritation. Oral exposure may produce gastrointestinal disturbances and discoloration of the oral mucosa and tongue. The toxicity of vanadium increases with the valence number. The pentavalent chemical forms, such as vanadium pentaoxide, are the most toxic compounds. An oral RfD of 7E-3 mg/kg/day has been reported for vanadium.

Zinc - Zinc is an essential trace element that is necessary for normal health and metabolism and therefore is nontoxic in trace quantities. Chronic oral exposure of humans to zinc may cause anemia and altered hematological parameters. Reduced body weights have been observed in studies in which rats were administered zinc in the diet. There is no evidence that zinc is teratogenic or carcinogenic. An oral RfD of 2E-1 mg/kg/day has been established for zinc.

6.4.2.2 Organic Chemicals

Acetone - Acetone is a volatile chemical and is rapidly absorbed in the respiratory tract when inhaled. Prolonged inhalation of high concentrations of acetone may produce irritation of the respiratory tract, coughing, headache, drowsiness, and incoordination. The toxic concentration of acetone in human blood is 200 to 300 ug/ml, with a lethal concentration of 550 ug/ml. Acetone is not known to have any carcinogenic effects. An oral RfD of 1E-1 mg/kg/day has been established for acetone.

Aldrin - Aldrin has been given a weight of evidence classification of B2 (probable human carcinogen). An oral slope factor of 1.7E+1 $(mg/kg/day)^{-1}$, an inhalation slope factor of 1.7E+1 $(mg/kg/day)^{-1}$, and an oral RfD of 3E-5 mg/kg/day have been established for aldrin.

Benzene - A cancer slope factor of 2.9E-2 $(mg/kg/day)^{-1}$ has been established for both ingestion and inhalation exposure. This cancer potency estimate is based upon the increased risk of leukemia among individuals occupationally exposed to airborne benzene. No RfD is established for benzene.

Benzoic Acid - Benzoic acid is a mild irritant to skin, eyes, and mucous membranes. An oral RfD of 4 mg/kg/day has been established for benzoic acid.

Benzyl alcohol - An oral RfD of 3E-1 mg/kg/day for benzyl alcohol has been reported. It is not known to have any carcinogenic effects.

Beta-BHC - Beta-BHC (Beta-benzene hexachloride) is classified as a Group C chemical (possible human carcinogen). An oral and inhalation slope factor of 1.8 $(mg/kg/day)^{-1}$ has been derived for beta-BHC based on increases in benign liver tumors in mice fed with the chemical in the diet. No RfD has been established.

Bis (2-ethyl hexyl) phthalate - Bis (2-ethyl hexyl) phthalate is reported to be carcinogenic in rats and mice, causing increased incidences of hepatocellular carcinomas or neoplastic nodules following oral administration. Chronic oral exposure to high concentrations of the chemical can cause retardation of growth and increased liver and kidney weights in laboratory animals. Bis (2-ethyl hexyl) phthalate has been classified as a Group B2 chemical (probable human carcinogen). An oral RfD of 2E-2 mg/kg/day and oral slope factor of 1.4E-2 $(mg/kg/day)^{-1}$ have been established.

2-Butanone - 2-Butanone inhalation exposure caused retarded fetal development, teratogenic effects, hepatotoxicity and neurological effects in rats. An oral RfD of 5E-2 mg/kg/day and an inhalation RfC of 9E-2 mg/kg/day has been established for 2-butanone.

Butyl benzyl phthalate - Limited data exist on the absorption of butyl benzyl phthalate. It is classified as a Group C agent (possible human carcinogen). An oral RfD of 2E-1 mg/kg/day has been established.

Carbon disulfide - The principal toxic effect of carbon disulfide is on the central nervous system. There is currently no evidence that carbon disulfide is a carcinogen in either animal or human studies. An oral RfD of 1E-1 mg/kg/day and an inhalation RfC of 3E-3 mg/kg/day have been established.

Chlorobenzene - An oral RfD of 2E-2 mg/kg/day and inhalation RfC of 5E-3 mg/kg/day have been established for chlorobenzene. No cancer slope factors have been reported.

Chloroethane - No RfD and cancer slope factors are reported for chloroethane.

4-chloro-3-methyl phenol - No RfD or cancer slope factors are reported for 4-chloro-3-methyl phenol.

4-4'-DDD - 4-4'-DDD has been given a weight of evidence classification of B2 (probable human carcinogen). An oral slope factor of 2.4E-1 $(mg/kg/day)^{-1}$ has been established for 4-4'-DDD. No RfD values are available.

4-4'-DDE - 4-4'-DDE affects the central nervous system, liver, kidneys, and skin. DDE is highly persistent in living organisms, so a major concern for DDE toxicity is its chronic effects. No RfD has been established for DDE. It is classified as a Group B2 carcinogen (probable human carcinogen) and has a cancer slope factor of 3.4E-1 $(mg/kg/day)^{-1}$.

1,1-Dichloroethane - General symptoms following exposure to 1,1-dichloroethane include salivation, dizziness, nausea, vomiting, eye, and skin irritation. Human inhalation of high doses of 1,1-Dichloroethane results in central nervous system depression and may be hepatotoxic. An oral RfD of 1E-1 mg/kg/day and an inhalation RfC of 1E-1 mg/kg/day have been established. 1,1-Dichloroethane is a Group C chemical (possible human carcinogen).

1,2-Dichloroethene - Information on the health effects of trans-1,2-Dichloroethene is limited. Acute exposure to higher dose levels can cause narcosis and death in rats. High concentrations also have anaesthetic properties in humans. An oral RfD of 1E-2 mg/kg/day for cis 1,2-dichloroethene and an oral RfD of 2E-2 mg/kg/day for trans 1,2-dichloroethene have been established.

Diethyl phthalate - A chronic RfD of 8E-1 mg/kg/day for oral exposure to diethyl phthalate has been established. No inhalation route RfC has been derived for diethylphthalate. Diethyl phthalate has not been evaluated for carcinogenicity.

2,4-Dimethyl phenol - 2,4-Dimethyl phenol has been shown to be a cancer promoting agent in skin painting studies on rats, but has not been tested for its total carcinogenic potential. Dermal exposure of 2,4-dimethyl phenol was more toxic to rats than oral dosing. An oral RfD of 2E-2 mg/kg/day has been established for 2,4-dimethyl phenol.

Di-n-butyl phthalate - Di-n-butyl phthalate is readily absorbed following oral and inhalation exposure. Reduced fetal weight and dose-related musculoskeletal abnormalities have been observed in fetuses from rats and mice exposed to very high doses of di-n-butyl phthalate during gestation (Shiota and Nishimura, 1982). An oral RfD of 1E-1 mg/kg/day has been established for di-n-butyl phthalate.

Endrin ketone - A cancer slope factor or RfD have not been established for endrin ketone.

Ethyl benzene - In humans, short-term inhalation exposure to ethyl benzene can result in sleepiness, fatigue, headache, and mild eye and respiratory irritation. Liver and kidney effects were observed in rats orally exposed to ethyl benzene (Wolf et. al., 1956). An oral RfD of 1E-1 mg/kg/day has been established for ethyl benzene.

Gamma-BHC - Gamma-BHC has been given a weight of evidence classification of B2/C (probable or possible human carcinogen). An oral RfD of 3E-4 mg/kg/day and an oral slope factor of 1.3 $(mg/kg/day)^{-1}$ has been established for gamma-BHC.

Heptachlor epoxide - Heptachlor epoxide is classified in Group B2 as a probable human carcinogen. An oral RfD of 1.3E-5 and oral and inhalation slope factor of 9.1 $(mg/kg/day)^{-1}$ has been established for heptachlor epoxide based on the experiments in which mice developed hepatocellular carcinomas. Human carcinogenicity data is limited.

2-Hexanone - An oral lethal dose (LD_{50}) of 2,590 mg/kg has been established for rats. No RfD or slope factor values are reported.

Methylene chloride - Methylene chloride is absorbed following oral and inhalation exposure. Because of methylene chloride's high solubility in water and lipids, it is probably distributed throughout all body fluids and tissues. Chronic exposure to the chemical can produce renal and hepatic effects in animals. An oral RfD of 6E-2 mg/kg/day, an inhalation RfC of 1 mg/kg/day, an oral slope factor 7.5E-3 $(mg/kg/day)^{-1}$, and an inhalation slope factor of 1.4E-2 $(mg/kg/day)^{-1}$ have been established for methylene chloride.

2-Methyl naphthalene - 2-Methyl naphthalene is a solid insoluble in water. Limited animal experimental data suggest high toxicity. The RfD and slope factor values are not available.

4-Methyl-2-pentanone - 4-Methyl-2-pentanone is a clear, colorless, watery liquid with a mild pleasant odor. No RfD and slope factors are available for 4-methyl-2-pentanone.

2-Methyl phenol - 2-Methyl phenol is a colorless solid or liquid with a phenolic odor that darkens upon exposure to air. No toxicity data are available for 2-methyl phenol.

4-Methyl phenol - 4-Methyl phenol is highly irritating to the skin, mucous membranes, and eyes. It also may impair kidney and liver functioning and cause central nervous system and cardiovascular disturbances. An oral RfD of 5E-2 mg/kg/day has been established for 4-methyl phenol. It is a Group C chemical (possible human carcinogen).

Polycyclic Aromatic Hydrocarbons (PAHs) - PAHs occur in the environment as complex mixtures of many components with varying noncarcinogenic and carcinogenic potencies. PAHs found at the site are discussed below:

Benzo(a) pyrene - Benzo(a)pyrene is a Group B2 (probable human carcinogen) chemical. No toxicity data has been established for benzo(a)pyrene.

Fluoranthene - Fluoranthene has an oral RfD of 4E-2 mg/kg/day. No slope factors are available.

Naphthalene - Systemic reactions of naphthalene include nausea, headache, diaphoresis, hematuria, fever, anemia, liver damage, convulsion, and coma. An oral RfD of 4E-3 mg/kg/day has been established for naphthalene.

Phenanthrene - Phenanthrene is not considered to be carcinogenic based on the available data. No RfD values are available for phenanthrene.

Pyrene - Pyrene is a colorless solid which is insoluble in water. An oral RfD of 3E-2 mg/kg/day has been established for pyrene.

Styrene - Styrene may be irritating to eyes and mucous membranes. It is a Group B2 chemical (probable human carcinogen). Styrene has an oral RfD of 2E-1 mg/kg/day, an oral slope factor of 3E-2 $(mg/kg/day)^{-1}$, and an inhalation slope factor of 2E-3 $(mg/kg/day)^{-1}$.

Phenol - Signs of acute phenol toxicity in humans are central nervous system depression, collapse, coma, cardiac arrest, and death. Phenol has an oral RfD of 6E-1 mg/kg/day.

2-Propanone - No RfD or slope factor values for 2-propanone are reported.

Toluene - Toluene is absorbed in humans following both inhalation and dermal exposure. In humans, the primary effects of toluene vapors are central nervous system depression and narcosis. An oral RfD of 2E-1 mg/kg/day and an inhalation RfC of 6E-1 mg/kg/day have been established for toluene.

1,2,4-Trichlorobenzene - 1,2,4-Trichlorobenzene is a colorless liquid with a pleasant aromatic odor. An oral RfD of 1.31E-3 mg/kg/day and an inhalation RfC of 3E-3 mg/kg/day have been established for 1,2,4-trichlorobenzene. No slope factor values are available.

Xylenes - The three xylene isomers (o,m, and p) have similar toxicological properties. In humans, acute inhalation exposure to high concentrations of xylene adversely affects the central nervous system and lungs. An oral RfD of 2 mg/kg/day and an inhalation RfC of 1E-1 mg/kg/day have been established for xylenes.

6.5 RISK CHARACTERIZATION

Potential human health risks resulting from the exposure scenarios identified in Section 6.3 are characterized on a quantitative basis in this section. The quantitative risk estimates were calculated based on risk assessment methods presented in the U.S. EPA document "Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual" (EPA, December 1989). In this section, the toxicity and exposure assessments are integrated into quantitative expressions of risk. Noncarcinogenic risk estimates are presented in the form of hazard quotients and Hazard Indices (HI) determined through the comparison of estimated intakes with verified reference doses (RfD). Incremental cancer risk estimates are presented in the form of dimensionless probabilities based on verified cancer slope factors. The quantitative risk estimates generated do not include risks due to some chemical contaminants which have been eliminated in Section 6.2 based on U.S. EPA recommended screening methods for chemicals of concern. Also the quantitative risk estimates presented in this section do not include risks from chemical contaminants for which no toxicity values are available. Chemical contaminants with no toxicity values were identified in Section 6.4.

Estimated human intakes were developed for each of the specific exposure routes and receptor groups discussed in Section 6.3.2. Complete calculations and assessment methods for human intake values and quantitative risks are provided in Appendix A.

6.5.1 Noncarcinogenic Risk Assessment - Noncarcinogenic risk is assessed using the concept of hazard quotients and His. The hazard quotient is the ratio of the estimated intake and the reference dose (RfD) for a selected chemical of concern, as follows:

$$\text{Hazard Quotient} = \text{Intake}/\text{RfD}$$

Where:

Intake = Exposure level or intake of toxicant (mg/kg/day)
RfD = Reference dose for toxicant (mg/kg/day)

The hazard quotient for noncarcinogenic effects assumes that there is a level of exposure (i.e., reference dose) below which it is unlikely for even sensitive populations to experience adverse health effects.

To assess the overall potential for noncarcinogenic health effects posed by more than one chemical, a hazard index (HI) approach is used (EPA, 1986b). This approach assumes that simultaneous subthreshold exposures to several chemicals could result in a cumulative adverse health effect. It assumes that the magnitude of the adverse health effect will be proportional to the sum of the ratios of the subthreshold exposures to acceptable exposures (reference dose). Thus the Hazard Index is equal to the sum of the hazard quotients as described in the following equation:

$$\text{Hazard Index} = \text{Dose}_1/\text{RfD}_1 + \text{Dose}_2/\text{RfD}_2 + \ldots\ldots + \text{Dose}_i/\text{RfD}_i$$

Where:

Dose_i = Dose for the i^{th} toxicant (mg/kg/day)
RfD_i = Reference dose for the i^{th} toxicant (mg/kg/day)

The HI is not a mathematical prediction of the severity of toxic effects; it is simply a numerical indicator of the possibility of the occurrence of noncarcinogenic effects. When the HI exceeds unity, there may be concern for potential health effects. While any single chemical with an exposure level greater than its toxicity value will cause the HI to exceed unity, for multiple exposures, the HI can also exceed unity even if no single chemical exposure exceeds its RfD.

Although the HI is not expressed as a probability of an individual experiencing an adverse noncarcinogenic effect, the greater the value of the HI above unity, the greater the level of concern.

Present Risk - The estimated HIs to potential receptors through present exposure routes are summarized in Table 6-15. The associated hazard quotients for selected chemicals of concern are presented in Appendix I. The estimated HIs for impacted private well water exposure routes range from 4.5E-04 for an off-site adult resident to 1.1E+00 for an off-site child resident. The estimated HIs for nearby private well water exposure routes ranged from 3.7E-04 for an off-site adult resident to 8.9E-01 for an off-site child resident. For exposure to active area surface soil, inactive area surface soil, surface water, sediment, and leachate, the estimated HI does not exceed unity (1.0) for any of these exposure pathways. Two contaminants, chromium and 2-propanone were identified as chemicals of concern during the ambient air monitoring. However, no risk calculations for these contaminants could be conducted since reference doses (RfC) and slope factors (SF) for these chemicals are not available. During the Phase 1 Risk Assessment, risk calculations for airborne chromium were calculated based on the RfC and SF established by EPA at that time. This report showed a significant impact due to airborne chromium. Since that time, EPA has withdrawn the RfC and SF values. These values are currently under review by the EPA study group. However, typically, municipal waste does not generate large quantities of dust since the wastes usually have sufficient moisture, thereby minimizing dust generation. In addition, the wastes are being covered with clean soil. The landfill cover does have areas of exposed soil which could generate dust.

Table 6-16 presents the estimated total HIs from exposure to various environmental media. The HI for each environmental medium is derived by summing the estimated hazard indices from the ingestion, inhalation, and dermal absorption routes for that particular medium. As presented in Table 6-16, the HIs for current exposure to active area surface soil, inactive area surface soil, surface water, sediment, and leachate are less than 1. The HI for off-site child resident exposed to impacted well water exceeds unity. The total HIs for individuals exposed to impacted well water are greater than individual exposed to nearby well water. Due to the reduced list of parameters analyzed on the "impacted" wells, c-mean values for copper, manganese, mercury, acetone, and 2-butanone were assumed to be equal in the two groups of private wells.

The estimated HIs for each environmental medium listed in Table 6-16 have been summed to calculate the total HIs for potential receptors. Summing the estimated HIs for each environmental medium is overly conservative in that it assumes that the same individuals would consistently face the "reasonable maximum exposure" (RME) for each exposure pathway. The RME estimate for each exposure pathway includes many conservative and upper-bound parameter values and assumptions (e.g., upper 95th confidence limit on amount of water ingested, upper-bound duration of occupancy of a single residence). Under actual situations in which contaminant concentrations vary over time and space, the same individual may or may not experience the RME for more than one pathway over the same period of time. However, to be conservative it is assumed that the RME assumption for different pathways apply to the same individual and the RME for different pathways are combined. The HI for the inactive area surface soil is higher as many organic compounds were discovered during sampling.

Elevated levels of ammonia were detected in several of private wells and groundwater monitoring wells. Reference Doses (RfD) or slope factor (SF) for ammonia are not available, thus human health risk calculations for ammonia could not be made.

<u>Future Risk</u> - The estimated HIs for potential receptors through future exposure routes are summarized in Table 6-17. Hazard quotients for selected chemicals of concern are presented in Appendix A. Although the future health effects from exposure to surface soil, surface water, sediment, and leachate remain the same for potential receptors

identified under the present risk scenario, the following three assumptions were utilized to characterize additional future noncarcinogenic exposures under future scenarios: (1) the site is used for residential purposes in future, (2) the off-site groundwater contains the same contaminants and contaminant levels as on-site groundwater, and (3) a short term construction project involving excavation of subsurface soil occurs on the site. Assumptions 1 and 2 are very conservative, unlikely scenarios that EPA typically requires to be evaluated.

As presented in Table 6-17, the estimated HIs from ingestion of groundwater are 5.2 and 1.2E+01 for adolescent and child residents, respectively. For exposure to active area surface soil, surface water, sediment, and leachate, the HI is less than 1 for the ingestion and dermal absorption routes for on-site residents. The estimated HI for a potential exposure of an adult worker to active area subsurface soil is 2.2 for ingestion and 1.3 for dermal absorption. These values exceed unity (1.0) and represent a concern. These higher values are due to detection of many inorganic and organic compounds. The estimated HI for a potential exposure of an adult worker to inactive area subsurface soil is 1.5E-01 for ingestion and 8.7E-02 for dermal absorption. The amount of exposed skin assumed for dermal absorption calculations was the average of typical case and reasonable worst case. Typical case includes long-sleeve shirt, pants, and shoes. Reasonable worse-case includes short-sleeve shirt, shorts, and shoes.

Table 6-18 presents the estimated total HIs for exposure to various environmental media under future scenarios. The HI for each environmental medium is derived by summing the estimated hazard indices from the ingestion, inhalation, and dermal absorption routes for that particular medium. As presented in Table 6-18, the HIs are below 1 for exposure to surface soil, surface water, sediment and leachate. The HI is above 1 for groundwater intakes. Antimony contributes to most of the HI value. The HI for active area subsurface soil also exceeds unity for non-resident adult worker and off-site adult worker.

The estimated HIs for each environmental medium in Table 6-18 have been summed to calculate the total HIs for potential receptors. Groundwater exposure and active area subsurface soil dominate the total RME for workers, trespassers, and residents. Summing the estimated HIs for each environmental medium is overly conservative in that it assumes that some individuals would consistently face the RME for each exposure pathway.

<u>**6.5.2 Carcinogenic Risk Assessment**</u> - For a carcinogen, risk is estimated as the incremental probability of an individual developing cancer over a lifetime as a result of exposure to the potential carcinogen. Incremental cancer risk estimates are generated for each of the exposure pathways using the estimated intake and verified cancer slope factor (SF) of a carcinogen, as follows:

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

Where:

Risk = A unitless probability of an individual developing cancer
Intake = Chronic daily intake averaged over 70 years (mg/kg/day)
SF = Slope Factor $(\text{mg/kg/day})^{-1}$

The SF converts the estimated daily intake averaged over a lifetime of exposure directly to the incremental risk of an individual developing cancer. The above equation is valid only at low risk levels (i.e., below estimated risks of 1E-02). The risk determined by using this expression is a unitless expression of an individual's likelihood of developing cancer as a result of exposure to a carcinogenic chemical present in the exposure media. An incremental cancer risk of 1E-06 indicates that the exposed receptor has a one in one million chance of developing cancer after a lifetime exposure. Alternately, such a risk may be interpreted as representing one additional case of cancer in an exposed population of one million people. The U.S. EPA's target risk range for the aggregate lifetime excess cancer risk is 1E-06 to 1E-04. If the calculated lifetime excess cancer risk exceeds 1E-04, the contamination is of sufficient concern to consider remedial actions.

If the calculated lifetime excess cancer risk is between 1E-06 and 1E-04, the need for remedial action varies and is site-specific. For sites where the calculated lifetime excess cancer risk is below 1E-06, generally no remedial action is required.

If a chemical intake is high and results in risk levels of 1E-02 or higher, an alternate calculation equation is used. The equation used, which is consistent with the linear low-dose model, is as follows (Risk Assessment Guidance for Superfund, Volume I, December 1989):

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

Where:

Risk = A unitless probability of an individual developing cancer
exp = The exponential
Intake = Chronic daily intake averaged over 70 years (mg/kg/day)
SF = Slope factor $(\text{mg/kg/day})^{-1}$

Because the slope factor is often an upper 95th percentile confidence limit of the probability of cancer response based on experimental animal data used in the multistage model (Risk Assessment Guidance for Superfund, Volume I, December 1989), the carcinogenic risk estimate obtained by the use of the slope factor will generally be an upper-bound estimate. Actual human risk, while not identifiable, is not expected to exceed the calculated upper limit based on the slope factor, and in fact, may be much lower.

The cancer risk equation described below estimates the lifetime excess cancer risk for an individual for simultaneous exposure to several carcinogens and is based on EPA's risk assessment guidelines (1986 a, b):

$$Risk_T = Risk_1 + Risk_2 + \ldots\ldots + Risk_i$$

Where:

$Risk_T$ = the total cancer risk, expressed as a unitless probability

$Risk_i$ = the risk estimate for the i^{th} substance

This equation represents an approximation of the precise equation for combining risks which accounts for the joint probabilities of the same individual developing cancer as a consequence of exposure to two or more carcinogens. The difference between the precise equation and the approximation is negligible for total cancer risks less than 0.1. The risk summation equation described above assumes that the intakes of individual substances are small. It also assumes independence of action by the chemicals involved (i.e., there are no synergistic or antagonistic chemical interactions and that all chemicals produce the same effect, i.e., cancer).

Present Risk - The estimated lifetime excess cancer risk (LECR) to potential receptors through present exposure routes are summarized in Table 6-19. The detailed risk calculations are presented in Appendix A. No carcinogenic chemicals of concern were found in impacted private wells, as well as nearby private wells. For surface water exposure routes LECR of 4.5E-11 was estimated for an adult worker. For active area surface soil exposure, the estimated LECRs range from 1.3E-10 for ingestion to 3.6E-10 for dermal absorption. For sediment exposure, the estimated LECR range from 2.7E-10 for ingestion to 7.6E-10 for dermal absorption. Dermal contact with leachate water contributes a LECR of 1.2E-09 for an adult worker.

Chromium may exist in two oxidation states: chromium III and chromium VI. Chromium III is not carcinogenic whereas chromium VI is carcinogenic. The RI data is insufficient to make a distinction between chromium III and chromium VI. Slope factors for chromium for inhalation are not currently available. Therefore, calculations for risk characterization for chromium could not be made. During the Phase 1 Risk Assessment, risk calculations for airborne chromium were performed based on the RfC and SF established by U.S. EPA at that time. Those calculations indicated a significant health impact due to airborne chromium. Currently, no RfC and SF are available through the U.S. EPA IRIS data base or HEAST tables. These values have been withdrawn and are currently under review by the EPA study group.

Table 6-20 presents the estimated total LECR from exposure to various environmental media. The LECR for each environmental medium is derived by summing the estimated LECR from the ingestion, inhalation, and dermal absorption routes for that particular medium. The LECR for a potential adult worker or trespasser exposed to active area surface soil is 4.9E-10. For exposure to sediment, the LECR ranges from 1.0E-09 for an adult worker to 1.8E-09 for an adult trespasser.

The estimated LECR for each environmental medium presented in Table 6-20 are summed to calculate the total LECR for potential receptors.

<u>Future Risk</u> - The estimated LECR for potential receptors via future exposure routes are summarized in Table 6-21. The detailed risk calculations are provided in Appendix A. Although the future risk from exposure to surface soil, surface water, sediment, and leachate remain the same for potential receptors identified under the present risk scenario, the following three assumptions were utilized to characterize additional carcinogenic exposures under future scenarios: (1) the site is used for residential purposes in future, (2) the off-site groundwater contains the same contaminants and contaminant levels as on-site groundwater, and (3) a short-term construction project involving excavation of subsurface soil occurs at the site. Again, assumptions 1 and 2 are very unlikely scenarios.

For groundwater exposure, the estimated LECRs range from 3.6E-07 for dermal absorption to 2.5E-04 for ingestion. For exposure to active area surface soil, sediment, and leachate, the LECR is below 1E-06. For surface water, the estimated LECR for potential exposure for an adult worker is 1.4E-11 for dermal absorption. The estimated LECR for potential exposure of an adult worker to active area subsurface soil is 1.2E-08 for ingestion and 1.1E-08 for dermal absorption. The estimated LECR for potential exposure of an adult worker to inactive area subsurface soil is 2.3E-11 for ingestion and 3.4E-11 for dermal absorption.

Table 6-22 presents the estimated total LECR resulting from exposure to various environmental media. The LECR for each environmental medium is derived by summing the estimated LECR from the ingestion, inhalation, and dermal absorption routes for that particular medium. As presented in Table 6-22, the estimated LECR resulting from exposure to active area surface soil, active area subsurface soil, surface water, inactive area surface soil, inactive area subsurface soil, sediment, and leachate is below 1E-06. The exposure of an adult resident to groundwater results in a LECR of 2.5E-04. This is due to the high slope factor of beryllium.

The estimated LECR for each environmental medium presented in Table 6-22 were summed to calculate the total LECR for potential receptors.

6.5.3 Conclusions - For non-carcinogenic effects, under the present conditions, the highest estimated HI is 1.1E+0 for child resident exposed to impacted private well water. Under future scenarios, the highest estimated HI is 7.1E+00 for adult worker. This is due to high HIs for groundwater and active area subsurface soil.

For carcinogenic effects, under present conditions, the highest lifetime excess cancer risk is 7.8E-09 for an adult resident. Under future scenario, the highest lifetime excess cancer risk is 2.5E-04 for off-site adult worker, and off-site trespasser. This higher value is attributed to higher LECR for groundwater under future scenarios. All the above estimated risks are qualified by the uncertainties addressed in the following section.

6.5.4 Uncertainties Associated With the Health Risk Assessment - All phases of this Risk Assessment process have involved a factor of uncertainty. On one hand, the Risk Assessment deals with scientifically verifiable findings (i.e., chemical concentrations in media); on the other hand, judgement and assumptions (i.e., duration of exposure, number of events or site visits, cancer slope factors estimated from animal studies, etc.) are utilized.

Uncertainties which apply to the risk evaluation of all of the identified exposure scenarios include the following:

- The exposure to and concentration of contaminants at exposure points was held constant over a 70-year lifetime. The contaminants detected and their concentrations were identified based upon site and monitoring data which was obtained over a relatively short period of time. Chemical fate and transport mechanisms associated with the compounds at the point source may alter actual concentrations over time. For example, chromium concentrations in the ambient air were held constant over a 70-year period. This may be a very conservative assumption.

- The use of the upper 95 percent confidence limit of the arithmetic mean or the maximum detected value is a very conservative estimate of the representative concentration for the site-specific chemicals. Consequently, the risks calculated would be significantly higher than the actual risk associated with the site.

- The assumptions regarding body weight, average lifetime exposure, intake of contaminants, population characteristics, and lifestyle may not be representative for the actual exposed population. This Risk Assessment utilized EPA established parameters where possible (i.e., for body weight, lifetime, daily water ingestion rate, daily inhalation rate, etc.). In those instances where there was no established standard, conservative values quoted in the literature or a conservative assumption based upon knowledge of the site or professional judgement was utilized.

- Cancer slope factors are primarily derived using laboratory animal studies and when available, human studies. Extrapolation of data from high to low doses, from one species to another, and from one exposure route to another introduces uncertainty in the values of cancer slope factors. For instance, cancer slope factors are based on the upper 95 percent confidence limit of the carcinogenic potency of the chemical. As a result, human health risk characterization using the slope factors may err on the side of conservatism.

- The slope factor for a chemical is based on the upper 95th percentile estimate of cancer potency. The upper 95th percentiles of probability distributions are not strictly additive, and hence the estimated total cancer risk may become artificially more conservative as risks from a number of different carcinogens are summed.

- The total cancer risk for multiple substances sums all carcinogens equally, giving as much weight to Class B or Class C carcinogens as Class A carcinogens.

- The action of two different carcinogens might not be independent. The assumption that there are no synergistic or antagonistic effects of different organic and inorganic chemicals may not be true.

- The risk assessment is based on the present understanding of the site characteristics. Conditions at the site or understanding of the site may change over time, resulting in either an increase or decrease in the risk associated with the site.

- No RI could possibly identify every potential migration route and it is not possible to quantify and include risk from all possible exposure scenarios.

The above uncertainties apply to the risk evaluation for all of the identified potential receptors/populations.

6.6 ECOLOGICAL ASSESSMENT

This section assesses the potential impacts to nonhuman receptors potentially exposed to chemicals of concern from the Gloversville Landfill site. The approaches used in this ecological assessment are similar to those used for the human health risk assessment. Potentially exposed receptors are identified, and information on exposure and toxicity is combined to assess potential impacts. However, because there is little data available, a quantitative assessment has not been performed. Instead, contaminant concentrations are compared to standard and criteria.

6.6.1 Receptor Characterization - The area surrounding the site contains a variety of wildlife habitats. To the north, northeast, and northwest of the landfill are wooded areas intermixed with a number of types of wetland habitats including riverine (Anthony Creek), open water (three ponds), vernal pool, scrub-shrub, and forested wetlands. These wetland areas offer food, cover, and water, three essential elements for productive wildlife habitat. To the south of the landfill, private residences on small open lots predominate beyond a wooded buffer area.

Twelve mammalian species including shrews, chipmunks, moles, and white-tailed deer have been identified near the site. Forested wetlands, such as those adjacent to the landfill, are breeding areas for wood ducks and hooded mergansers which utilize cavities in dead trees as nesting sites. Anthony Creek and several ponds near the site support the production of macroinvertebrate and vegetative life upon which fish, fowl, and mammals feed. The fish species found include white sucker, chain pickerel, tesselated darter, brown bullhead, pumpkinseed, large-mouth bass, golden shiner, yellow perch, creek chub, and brook trout. A number of reptile and amphibians species including red-backed salamanders, red efts, spotted salamanders, toads, green frogs, and pickerel frogs have been identified near the site. No state or federal endangered or threatened species are known to occur on or near the site.

6.6.2 Potential Exposures and Estimates of Impact - Surface soil, subsurface soil, groundwater, surface water, sediment, leachate, and ambient air have been sampled at the Gloversville Landfill site. This ecological assessment is limited to an evaluation of potential impacts associated with chemicals of concern in surface soil, surface water, sediment, leachate and chemicals in groundwater potentially released to surface water. Potential impacts associated with contamination in subsurface soil were not evaluated because no pathways exist by which plants or wildlife can be exposed to chemicals in deep subsurface soils. Potential impacts associated with ambient air contamination will not be evaluated (even though potential exposure pathways do exist) because insufficient data is available to support plant and wildlife exposure and toxicity assessments for chemicals in air.

Potential exposures and impacts are discussed below by receptor type.

6.6.2.1 Terrestrial Plants - Terrestrial plants may be exposed to chemicals of concern in the surface soils. Limited information is available on the phytotoxicity of these chemicals, and therefore it is not possible to accurately predict the ability of natural vegetation to survive in contaminated soils at the site. However, the presence of abundant natural vegetation on and surrounding the site indicates that the chemicals at the site probably are not present at lethally toxic levels for most plant species. However, the concentrations of chemicals of concern at the site could be high enough to cause sublethal effects (e.g., reduced growth) in some plants or death in sensitive species. Based on the vegetation that currently exists at the site, such effects (if occurring) do not appear to be significantly affecting the plant community or wildlife habitats of the area.

6.6.2.2 Terrestrial Wildlife - Terrestrial wildlife may be exposed to chemicals of concern present in surface soil, surface water, sediment, and leachate via the following pathways:

- Ingestion of soil or sediment while foraging or grooming

- Ingestion of food that has accumulated chemicals from soil, surface water, sediment, or leachate
- Ingestion of surface water or leachate
- Dermal absorption from soil, surface water, sediment, or leachate

However, evaluations of receptor-specific exposures via these pathways is limited due to the lack of appropriate exposure assessment and toxicity information. For example, although it is known that small mammals (e.g., mice) and birds can ingest soil contaminants while preening or foraging, it is not known how much soil is actually ingested during these activities. Similarly, the amount of a chemical that is dermally absorbed following direct contact with contaminated environmental medium (e.g., soil) by wildlife cannot be estimated given the available data. For these reasons, the evaluation of potential wildlife exposures and impacts at the site is limited to an evaluation of potential impacts associated with ingestion of surface water, leachate, and contaminated food.

Ingestion of Surface Water - The water in Anthony Creek, ponds, and wetlands could be used as drinking water sources by some terrestrial species. These species include deer, small mammals, birds, etc.

Potential toxic effects in deer, small mammals, and birds can be evaluated by comparing surface water concentrations with recommended limits of concentrations in drinking water for livestock and poultry. The National Academy of Sciences (NAS, 1974) reviewed the toxicity of several inorganic chemicals and estimated safe drinking water limits for livestock, including dairy and beef cattle, sheep, and horses. These values were derived by examining a broad toxicity data base which included data from a variety of mammals (e.g., cattle, rats, mice, rabbits) and by considering the range of daily drinking water requirements of the livestock species of concern. Although different species of animals may react differently to a toxic substance and the young and healthy may not respond in the same way as mature or unhealthy animals, it is believed that the toxicity data base upon which the drinking water limits were based was sufficiently broad to encompass the range of potential sensitivities of terrestrial wildlife to the inorganic chemicals.

Table 6-23 compares recommended limits of toxic substances in drinking water for livestock and poultry with exposure concentrations (C-mean) in surface water. As the table shows, the concentrations for copper, lead, mercury, and zinc in surface water at the site are well below the recommended limits. Although no recommended limits have been established for aluminum, barium, and manganese, the exposure concentrations of these chemicals are not expected to be of concern. Therefore, it appears that terrestrial wildlife (e.g., deer, small mammals, and birds) is not at an increased risk of toxic effects from ingesting water near the site.

Ingestion of Leachate - The leachate is unlikely to be used as a drinking water source by terrestrial wildlife because of the small size of the leachate seeps and its shallow depths. Although it may be possible that some of the small terrestrial wildlife (e.g., small mammals, birds, etc.) may occasionally ingest leachate, the availability of an abundant water supply in the area suggests a low probability of such an occurrence. Because of the unlikelihood of terrestrial wildlife drinking the leachate on a regular basis, and inadequate toxicity information for infrequent water ingestion by small mammals, birds, etc., the health effects from the ingestion of leachate can not be assessed, but are expected to be low.

Ingestion of Food - Some inorganic chemicals and various organic chemicals were detected in the surface soil and sediment samples. No pesticide or PCBs were detected in surface soil and sediment samples. The detected chemicals in surface soil and sediment are not expected to accumulate to a significant degree in plants or animals. Only limited data was collected during the RI concerning uptake in plants and animals. Therefore, food chain exposures to the chemicals of concern at the site cannot be estimated.

Aquatic Life - The study by New England Environmental (NEE) indicted that macroinvertebrate populations at all downstream locations had been moderately impacted compared to the control station. A fisheries survey indicated that the entire eastern tributary to Anthony Creek does not contain any fish. Groundwater discharge to this tributary, discussed in Sections 3 and 5, is the probable route by which contaminants may enter the tributary. The representative concentration (C-mean) of ammonia-nitrogen in surface water samples is significantly higher than the Federal Water Quality Criteria for un-ionized ammonia as established by EPA. Un-ionized ammonia has been demonstrated to be the principal toxic form of ammonia. Ammonia consumes dissolved oxygen in receiving waters and is toxic to fish life. It has been established that ammonia in the un-ionized form interferes with oxygen transfer in the gills of fish. Concentrations of ammonia acutely toxic to fishes may cause loss of equilibrium, hyperexcitability, increased breathing, cardiac output, coma, and death. Risk characterization parameters i.e. reference dose, slope factor for ammonia are not available. Thus, no risk calculations were done in this case. Factors that have been shown to affect ammonia toxicity include dissolved oxygen, temperature, pH, carbon dioxide concentration, salinity, and the presence of toxicants. Fish populations on the western branch of Anthony Creek appear much less affected by the landfill. The bioassay study indicated that flow in Anthony Creek was sufficient to dilute any potential effect of the landfill.

Potential surface water impacts at the site were evaluated by comparing the exposure concentrations with federal Ambient Water Quality Criteria (AWQC) for the protection of aquatic life. The exposure concentrations evaluated were the representative concentrations in surface water, as well as those in on-site groundwater which could discharge to surface water. As discussed in Sections 3 and 5, the groundwater at the Gloversville Municipal Landfill site is believed to discharge to the Anthony Creek.

Table 6-24 compares the representative concentrations of chemicals of concern in surface water to federal AWQC values for aquatic life. The exposure concentrations of ammonia, lead, mercury, and zinc exceeded the federal chronic AWQC values. Except for ammonia and zinc, no chemicals exceeded the acute AWQC values. Federal AWQC values are not available for aluminum, barium, or magnesium.

Table 6-25 compares the representative concentrations of chemicals of concern in groundwater to federal toxicity criteria for aquatic life. The exposure concentrations of copper and cyanide exceeded the federal AWQC values. Federal AWQC values are not available for aluminum, barium, cobalt, manganese, vanadium, and di-n-butyl phthalate.

The above comparisons suggest that aquatic life in surface water near the site may be affected by releases of chemicals from the site. Aquatic life impacts are likely to be greatest in the wetlands adjacent to the landfill and in the areas of the creek where the dilution of chemicals is limited. Chemical concentrations in downstream waters of the creek are most likely diluted significantly.

<u>6.6.3 Conclusions</u> - The environmental risks posed by the site appear to be limited to potential impacts to aquatic life exposed to chemicals present in the surface water of the eastern tributary to Anthony Creek. Adverse impacts in plants and terrestrial wildlife are unlikely. The studies performed during the RI have indicated that macroinvertebrate populations at downstream locations had been moderately impacted. The bioassay study indicated that the contaminants present in the water of Anthony Creek were sufficiently diluted downstream of the landfill to support fish populations. Fish populations on the western branch of Anthony Creek appear unaffected by the landfill. Although the exposure concentrations of a few metals in surface water/site groundwater are above acute and/or chronic toxicity levels, the impact is not expected to be significant. The ammonia concentrations detected in surface water samples are significantly higher than the Ambient Water Quality Criteria as established by EPA and therefore may be adversely affecting the fish population.

The evaluations and conclusions made in this ecological assessment are not absolute. Many assumptions were made regarding exposure and toxicity to derive the estimates of impact, and these assumptions add uncertainty to the conclusions. In addition, toxicity data were not available for many chemicals detected at the site, and therefore the potential impacts from exposure to these chemicals could not be evaluated completely.. Because of these uncertainties and data gaps, the conclusions of the assessment should not be regarded as precise determinations of impacts, but rather only as indications of the potential for impacts and the types of impacts which may occur.

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SECTION 7
SUMMARY AND CONCLUSIONS

7.1 DESCRIPTION OF THE REMEDIAL INVESTIGATION

The Remedial Investigation of the Gloversville Landfill, which commenced in 1989, was conducted to determine whether hazardous substances have been or may be released from the landfill site and, if so, the lateral and vertical extent of such release and the environmental impacts thereof. To accomplish this objective, several types of data were collected. The studies, conducted in two phases (Phase 1 and Phase 2), included:

- Topographic Mapping
- Private Well Inventory
- Hydrogeologic Assessment
    - Geophysical Studies
    - Soil Borings
    - Bedrock Corings
    - Monitor Well Emplacement
    - Hydraulic Conductivity Determinations
    - Pumping Tests
    - Stream Gauging
- Chemical Characterization of:
    - Leachate
    - Landfill Soils
    - Groundwater
    - Private Water Supplies
    - Surface Water
    - Stream and Pond Sediments
    - Ambient Air
    - Wildlife Tissues
- Test Pits/Borings
- Wildlife Studies
- Wetlands Delineation
- Risk Assessment

7.2 RESULTS OF HYDROGEOLOGIC STUDIES - The hydrogeologic studies conducted during Phases 1 & 2 have resulted in a detailed conceptual model of groundwater movement at the landfill site providing the basis for evaluating the potential for contaminant migration from the site.

The overburden material that was deposited during the glaciation of the region consists mainly of stratified drift. This drift is comprised of horizontal beds of sands and silt with sporadic areas of gravel. Underlying the drift over much of the site is a very dense lodgement till, a highly compacted mixture of sand, silt and clay. The till is discontinuous beneath the site, occurring in a very thick band at the south end of the site, tapering down toward the north. Some of the borings and the regional geomorphology indicate that the till is absent under portions of the site due to erosion by the advancing glacier and/or glacial outwash streams.

The stratified drift provides a relatively permeable media for groundwater flow. Certain deposits within the drift have higher hydraulic conductivities than others, setting up preferred flow patterns in the overburden aquifer. Discharge of groundwater from the majority of the site is to Pond B on the northeast side of the landfill. Another component of the flow moves toward Anthony Creek to the northwest of the landfill. A third, smaller component appears to flow toward the southeast from the southern end of the landfill.

The Phase 2 hydrogeologic studies focused primarily on the bedrock aquifer underlying the site and its interrelationship with the overburden aquifer. These studies concluded that there is the potential for groundwater to migrate between the two aquifers in areas where the lodgement till has been eroded or where the till may be fractured.

The hydrogeologic study also included a computer simulation of groundwater flow at the site, providing an opportunity to model the effects of certain remedial alternatives. Two landfill capping scenarios were evaluated during the RI as described in Section 3.5.12. The results show that a landfill cap will lower the groundwater table sufficiently to desaturate most of the buried refuse that now lies beneath the water table. This, combined with the near elimination of percolating rainfall, should limit the production of contaminated leachate.

7.3 RESULTS OF CHEMICAL CHARACTERIZATION - As described in Section 2, samples were collected from a variety of environmental media to determine what contaminants are present at the source (in the leachate and in the soils at the landfill) and which contaminants are present at downgradient surface water, groundwater and sediment monitoring points.

Sections 4 and 5 describe the results of the sampling and analyses which indicate that while most contaminants are not migrating from the landfill at high concentrations, a contaminant plume is evident. Based on the hydrogeologic study, and supported by chemical data, this plume appears to primarily discharge through preferred flow channels to Anthony Creek and Pond B.

The analytical data indicate that groundwater quality in the bedrock monitor wells has been degraded by landfill derived contaminants. Most well clusters showed a downward gradient between the overburden and bedrock aquifers. However, a dense lodgement till was encountered between the two aquifers at these points. It is hypothesized, based on published studies of similar compact tills, that vertical fractures may exist in the till providing conduits to the bedrock aquifer.

Another potential route of transport between the overburden and bedrock aquifers is through areas where till has been removed from the bedrock either through glacial or fluvial erosion. The absence of till at cluster MW-4 is an indication of glacial erosion and the absence of till and the presence of coarse deposits at cluster MW-11 is indicative of fluvial erosion. It is speculated that till may also be eroded from the bedrock ridge that runs beneath the landfill, providing an opportunity for landfill derived contaminants to migrate directly to the highly fractured upper portion of the bedrock.

Contaminant movement in the bedrock aquifer is nearly exclusively through fractures that are believed to trend northeast-southwest along the bedrock lows. These fracture sets create a primary flow direction toward the northeast. Data also indicate that contaminants are migrating in bedrock toward the southeast. This may be occurring primarily in the fractured uppermost portion of the bedrock, where flow may follow the bedrock surface toward the southeast, to the bedrock low in the vicinity of cluster MW-14. Continued migration to the southeast may be through secondary fractures that exist perpendicular to the primary northeast-southwest trend.

To date, the contaminants that have migrated from the landfill have not created exceedences of drinking water standards in any of the private wells. The apparent connection between the bedrock aquifer, which supplies some of the private wells in the area, and the landfill, however, demands that remedial alternatives consider the long-term effects on the private wells.

Metals and other inorganics measured in Anthony Creek exceed certain surface water standards.

7.4 RESULTS OF WILDLIFE STUDIES

The wildlife studies conducted during Phase 1 showed that macroinvertebrate species were moderately to severely impacted at several locations along Anthony Creek. Fish populations were limited at some of the locations in Anthony Creek. Fish were non-existent in Pond C and at location SW-5. In-situ bioassays confirmed that the water in Pond C and at location SW-5 is toxic to fish, in that complete mortality was exhibited in less than 2 hours.

7.5 RESULTS OF RISK ASSESSMENT

For non-carcinogenic effects, under the present conditions, the highest estimated Hazard Index (HI) is 1.1 for a child resident exposed to impacted private well water. The calculation of the HI assumes that simultaneous subthreshold exposures to several chemicals could result in a cumulative adverse health effect and that the magnitude of the adverse health effect will be proportional to the sum of the ratios developed for each individual contaminant. This is a very conservative approach in that it assumes that the same individuals would consistently face the "reasonable maximum exposure" (RME) for each exposure pathway.

Under actual situations in which contaminant concentrations vary over time and space, the same individual may or may not experience the RME for more than one pathway over the same period of time. Therefore, the HI of 1.1 for the child resident exposed to impacted well water is an indication that landfill derived contaminants are impacting the quality of private wells downgradient of the site that may result in a health impact. It is not indicative of an immediate and imminent health threat.

Similarly, under future scenarios, where it is assumed that off-site private well water quality deteriorates to the point where contaminant concentrations equal current on-site groundwater quality and people establish residences on the landifll, calculated hazard indecies are 40 for on-site child residents and 12 for off-site child residences.

For carcinogenic effects, under present conditions, the highest lifetime excess cancer risk is 7.8E-09 for an adult resident. Under future scenario, the highest lifetime excess cancer risk is 2.5E-04 for off-site adult worker, and off-site trespasser. This higher value is attributed to higher LECR for groundwater under future scenarios. EPA typically requires that remedial alternatives be designed to achieve a total carcinogenic risk less than 1.0E-04.

7.6 CONCLUSIONS

The Remedial Investigation has resulted in several conclusions that will affect the selection of an appropriate remedial alternative in the Feasibility Study:

- Wastes which fail the Toxicity Characteristic Leaching Procedure are present in the landfill.
- Chemical contaminants (primarily inorganics) are migrating from the landfill through groundwater and surface water.
- Contaminant migration has caused NYSDEC groundwater and surface water standards to be exceeded.

- Migration of the plume (ammonia and barium) into the bedrock aquifer appears to have impacted several private wells in the vicinity, but not at levels which exceed drinking water standards.

- Contaminant migration has limited the survivability of fish species in Anthony Creek and its tributary.

- Risk assessment has shown a significant non-carcinogenic health impact to child residents consuming water from impacted private wells.

- The computer simulation of hydrogeologic conditions indicates that closure of the landfill with low permeability materials will probably reduce leachate generation by lowering the groundwater table and substantially reducing the percolation of rainwater through the refuse.

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LIST OF ABBREVIATIONS

ARAR	Applicable or Relevant and Appropriate Requirement
AT	Averaging Time
AV	Skin Surface Area Available
AWQC	Ambient Water Quality Criteria
BW	Body Weight
C	Chemical Concentration
CAG	Carcinogen Assessment Group
CCJM	C.C. Johnson & Malhotra, P.C.
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act
CGI	Combustible Gas Indicator
cm	Centimeter
DCS	Direct Contract Score
DEC	Department of Environmental Conservation
DEX	Dermal Exposure Dose
DOH	Department of Health
DOL	Department of Law
D_s	Shower Duration
D_t	Total Duration in Bathroom
E	Estimated Value Due to Interference
EA	Endangerment Assessment
ED	Exposure Duration
EF	Exposure Frequency

EM	Electromagnetic
EPA	Environmental Protection Agency
ET	Exposure Time
eV	Electron Volt
FES	Fire and Explosion Score
FID	Flame Ionization Detector
ft	Foot or Feet
g	Gram
gpd/ft_2	Gallon Per Day Per Square Foot
GW	Groundwater
H	Henry's Law Constant (Atm-m^3/mole)
HEAST	Health Effect Assessment Summary Table
HI	Hazard Index
HRS	Hazard Ranking System
IEX	Ingestion Exposure Dose or Inhalation Exposure Dose
IR	Ingestion Rate or Inhalation Rate
IRIS	Integrated Risk Information System
J	Estimated Value
K	Hydraulic Conductivity (sometimes referred to as permeability)
LBG	Leggette, Brashears & Graham, Inc.
LC	Lethal Concentration
LD	Lethal Dose
LECR	Lifetime Excess Cancer Risk
LEL	Lower Explosion Limit

M^3	Cubic Meter
mg/L	Milligram Per Liter (equal to ppm in water)
mg/m^3	Milligrams per Cubic Meter
ml	Milliliter
mm	Millimeter
MS	Migration Score
MW	Monitoring Well
NEE	New England Environmental, Inc.
NYCRR	New York Code of Rules and Regulations
NYS	New York State
NYSDEC	New York State Department of Environmental Conservation
NYSDOH	New York State Department of Health
OVA	Organic Vapor Analyzer
PAH	Polycyclic Aromatic Hydrocarbon
PC	Permeability constant
PID	Photoionization Detector
ppb	Parts per Billion
ppm	Parts per Million
PW	Private Well
QAPP	Quality Assurance Project Plan
R_a	Air Exchange Rate
RCRA	Resource Conservation and Recovery Act
RfC	Reference Concentration for Particular Noncarcinogen
RfD	Reference Dose for Particular Noncarcinogen

RI	Remedial Investigation
RI/FS	Remedial Investigation/Feasibility Study
RME	Reasonable Maximum Exposure
S	Storativity
S	Solubility or Volatile Chemical General Rate (in Section 6)
SA	Skin Surface Area Available
SARA	Superfund Amendments and Reauthorization Act (1986)
SD	Sediment
SF	Slope Factor for Particular Carcinogen
Site	Gloversville Municipal Landfill Site
SW	Surface Water
T	Transmissivity
TIC	Tentatively Identified Compound
TVGA	Tallamy, VanKuren, Gertis and Associates
ug	Micrograms
ug/kg	Micrograms of Compound per Kilogram of Soil/Sediment
ug/L	Micrograms of Compound per Liter of Water/Leachate
USEPA	United State Environmental Protection Agency
UV	Ultraviolet

ANALYTICAL PARAMETERS

Al	Aluminum
Sb	Antimony
As	Arsenic
Ba	Barium
Be	Beryllium
Cd	Cadmium
Ca	Calcium
Cr	Chromium
Co	Cobalt
Cu	Copper
Fe	Iron
Pb	lead
Mg	Magnesium
Mn	Manganese
Hg	Mercury
Ni	Nickel
K	Potassium
Se	Selenium
Ag	Silver
Na	Sodium
Tl	Thallium
V	Vanadium
Zn	Zinc
CN	Cyanide
TDS	Total Dissolved Solids
Cl	Chloride
pH	Negative Logarithm of Hydrogen Ion Concentration
TOC	Total Organic Carbons
TOX	Total Organic Halogens
Eh	Negative Logarithm of Electron Activity
TKN	Total Kjeldahl Nitrogen

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