REMEDIAL INVESTIGATION REPORT
FOR
HERTEL LANDFILL SITE
PLATTEKILL, NEW YORK
VOLUME 1
EPA WORK ASSIGNMENT NO. 003-2LH7
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UNITED STATES ENVIRONMENTAL PROTECTION AGENCY

REGION II

ALTERNATIVE REMEDIAL CONTRACTING STRATEGY (ARCS) FOR HAZARDOUS WASTE REMEDIAL SERVICES

EPA Contract No. 68-S9-2001

TAMS CONSULTANTS, Inc.

and

TRC ENVIRONMENTAL CONSULTANTS, Inc.

EXECUTIVE SUMMARY

A remedial investigation of the Hertel Landfill site, Plattekill, New York (National Priority List #810, August 1990) was performed for the U.S. Environmental Protection Agency - Region II by TAMS Consultants, Inc. and TRC Environmental Consultants, Inc. The purposes of the investigation were to: investigate the physical characteristics of the site, both natural and man-made, determine the nature and extent of contamination due to landfilling activities, and characterize environmental impact and potential health risks.

The landfill operated between the mid 1960's and the late 1970's, and changed ownership several times. The site was known to receive municipal wastes, but waste deposits visible at the surface or reportedly disposed of at the landfill included drums, engine blocks, oil wastes, printing waste, fibrous materials, paint waste, farm equipment and, possibly, rubber wastes. Eight distinct disposal areas were tentatively identified. Previous investigations at the site conducted for the State of New York revealed the presence of chlorinated solvents, toluene, phthalates and several metals in ground water, and phenols, chlorinated solvents and metals in seep samples.

The remedial investigation activities began in 1989 with the preparation of a site work plan ("Final RI/FS Work Plan for Hertel Landfill Site, Plattekill, New York", September 1989) and a detailed field operations plan ("Final RI/FS Field Operations Plan for Hertel Landfill Site, Plattekill, New York", October 1989). The following field activities were completed between September 1989 and August 1990.

- Geophysical surveys were conducted over a 14 acre area of the site using electromagnetic terrain conductivity and magnetometry methods to delineate areas of buried metal and possible plumes of contamination in the ground water. A metal detector was also utilized in an attempt to distinguish surface from buried-metal anomalies.

- A generalized soil gas survey was conducted over the landfill area to identify areas of organic vapors in the soil which may be indicative of wastes present. In addition, a specialized soil gas survey was performed in an area known to receive paint wastes in order to identify the presence of specific volatile organic compounds.

- Surface soil samples were collected from each of the tentatively identified disposal areas and other areas for a total of 22 samples.

- Test pits were excavated in seven of the eight tentatively identified disposal areas and in other areas to delineate the extent of the fill. A total of 10 soil samples and 4 water samples were collected from the test pits. Piezometers were installed in 3 test pits to provide ground water level data.

- Test borings were drilled and monitoring wells were installed to identify the geologic and hydrogeologic conditions and to permit the sampling of subsurface soils and ground water. A total of 19 monitor wells were installed at 12 locations including 7 well clusters. The wells were completed in either the shallow overburden, deep overburden or shallow bedrock beneath the site. Piezometers were installed at four additional locations to monitor water levels.

- Two rounds of ground water samples were collected from monitoring wells; one round of ground water samples from nearby private residential water supply wells; one round of surface water samples from seeps, streams and wetlands near the site; and one round of sediment samples from seep, stream and wetland locations.

- An ecological study was performed which included wetlands delineation, the examination of macroinvertebrates, mammal trapping and wildlife observation to provide baseline information on the biological resources of the site and to estimate possible impacts to local organism populations.

The investigation indicated that the fill covers approximately 13 acres of the 80-acre site, consists primarily of household refuse with some metal debris and varies in thickness to over 16.5 feet. Other distinct areas of waste disposal include surficial paint wastes, and oily wastes.

The samples collected and analyzed from the environmental media at the site provided an overview of contaminant types and distribution. Volatile organic compounds, base/neutral/acid extractable compounds, and metals (inorganics) were the primary contaminants detected at the site and were distributed as follows:

- The volatile organic compounds (VOCs) were distributed in samples of soil, ground water, sediment, seep water and surface water adjacent to the fill. The most commonly occurring VOCs identified at the site were aromatic hydrocarbons which were present in samples of ground water, subsurface soil, seep water and seep sediment, but not present in surface water samples. Other VOCs, including the chlorinated hydrocarbons, were detected in samples of subsurface or surface soils, ground water, seep water, seep sediment, surface water and sediment samples. In most cases, the VOCs were not observed in sediment or surface water samples from downgradient of the site. VOCs exceeded Federal or New York Standards for ground water and surface water. No VOCs exceeded New York Sediment Criteria Guidance.

- The base/neutral/acid extractable (BNA) compounds were identified in all the media sampled on site. Polynuclear aromatic hydrocarbons (PAHs), a subset of the BNA compounds, were present in samples from surface and subsurface soils, on-site ground water, seep water and seep sediment. PAHs were not detected in on-site surface water or sediment. Phenols and phenol compounds were detected in on-site subsurface soil, ground water and surface water samples. Phenols were not detected in off-site surface water and were detected in one downgradient sediment sample. Phthalate esters were present in samples from all sampled media including background samples of soil, sediment and ground water. Other BNA compounds are present in samples from on site soil, ground water, seep water, seep sediment and wetland sediment, but were not detected in downgradient surface water or sediment samples. BNAs exceeded Federal or New York Standards for ground water and surface water. For one seep sediment sample, three PAHs exceeded the New York Sediment Criteria Guidance.

- Metals and other inorganics are widespread in nature and their presence must be compared with natural background in order to determine if landfill impacts are present. Aluminum, arsenic, barium, calcium, copper, chloride, cadmium, chromium, iron, potassium, magnesium, manganese, lead, mercury, sodium, zinc and cyanide all appeared at elevated concentrations in one or more samples. However, only calcium, chloride, potassium, magnesium, manganese and sodium were noted at above-background concentrations in off-site surface water samples. Cyanide was present in off-site stream sediment. Dissolved arsenic, iron, magnesium, manganese and sodium (as detected in filtered ground water samples) exceeded New York ground water standards. Ten metals exceeded the New York Sediment Criteria Guidance.

A health and environmental risk assessment was conducted to quantitatively and qualitatively assess the potential impacts of the landfill on human and ecological health. For the human health component of the risk assessment, both current and future land use scenarios were considered. The primary cancer and non-cancer risks were associated with the future use of the site as a residential area. The routes of exposure of most concern included dermal contact with soil, ingestion of ground water and ingestion of soil resulting in a cumulative cancer exposure risk to children of 2E-03, and a cumulative hazard index of 100. The chemicals of primary concern include arsenic in soil and ground water, manganese in ground water and PAHs and chromium in soil. Tables ES-1 and ES-2 provide a summary for each scenario of pathway risk and the cumulative total exposure risk, for only the significant chemicals which govern the risk assessment.

There were no federal threatened or endangered species located on site. Thirteen species of plants were identified on-site which are protected by New York State. The red-shouldered hawk which was sighted is the only New York State threatened species. The benthic macroinvertebrate study conducted on site was inconclusive; the potential exists for site contaminants to produce adverse effects to aquatic organisms. Additionally, there is some indication that the potential exists for elevated inorganics (selenium, cadmium and mercury) in soil to produce adverse environmental effects.

TABLE ES-1
SUMMARY OF CANCER RISK ESTIMATES
SCENARIO 1 - TRESPASSING ON SITE

CHEMICAL	CHRONIC DAILY INTAKE(CDI) (mg/kg/day)	CDI ADJUSTED FOR ABSORPTION	SF (mg/kg/day)-1	WEIGHT OF EVIDENCE	TYPE OF CANCER	SF BASIS/ SOURCE	CHEMICAL SPECIFIC RISK	TOTAL PATHWAY RISK	TOTAL EXPOSURE RISK
EXPOSURE PATHWAY: DERMAL CONTACT WITH SURFACE WATER - ADULTS								1.3E-07	3.7E-04
INORGANICS									
Arsenic	7.4E-09	0.1	1.75	A	Bladder, Liver, Lung	Water/IRIS	1.3E-07		
EXPOSURE PATHWAY: DERMAL CONTACT WITH SOIL - CHILDREN								5E-04	5E-04
INORGANICS									
Arsenic	2.9E-05	0.1	1.75	A	Bladder, Liver, Lung		5E-04		

SUMMARY OF CANCER RISK ESTIMATES
SCENARIO 3 - RESIDENTS ON SITE

CHEMICAL	CHRONIC DAILY INTAKE(CDI) (mg/kg/day)	CDI ADJUSTED FOR ABSORPTION	SF (mg/kg/day)-1	WEIGHT OF EVIDENCE	TYPE OF CANCER	SF BASIS/ SOURCE	CHEMICAL SPECIFIC RISK	TOTAL PATHWAY RISK	TOTAL EXPOSURE RISK
EXPOSURE PATHWAY: INGESTION OF CHEMICALS IN DRINKING WATER - CHILDREN								2E-04	2E-03
INORGANICS									
Arsenic	9.8E-05	No	1.75E+00	A	Skin	Water/IRIS	2E-04		
EXPOSURE PATHWAY: DERMAL CONTACT WITH CHEMICALS IN SOIL - CHILDREN								2E-03	
INORGANICS									
Arsenic	1.1E-04	0.1	1.75E+00	A	Skin	Water/IRIS	2E-03		
EXPOSURE PATHWAY: INGESTION OF CHEMICALS IN DRINKING WATER - ADULT								5E-04	7E-03
INORGANICS									
Arsenic	3.0E-04	No	1.75E+00	A	Skin	Water/IRIS	5E-04		
EXPOSURE PATHWAY: DERMAL CONTACT WITH CHEMICALS IN SOIL - ADULT								7E-03	
INORGANICS									
Arsenic	3.8E-04	0.1	1.75E+00	A	Skin	Water/IRIS	7E-03		

TABLE ES-2
SUMMARY OF CHRONIC HAZARD INDEX ESTIMATES
SENARIO 1: TRESPASSING ON SITE

CHEMICAL	CHRONIC DAILY INTAKE(CDI) (mg/kg/day)	CDI ADJUSTED FOR ABSORPTION	RFD (mg/kg/day)	CONFIDENCE LEVEL	CRITICAL EFFECT	RFD SOURCE/ BASIS	RFD UNCERTAINTY ADJUSTMENTS	MODIFYING FACTORS	HAZARD QUOTIENT	PATHWAY HAZARD INDEX (HI)	TOTAL EXPOSURE HI
EXPOSURE PATHWAY: DERMAL CONTACT WITH CHEMICALS IN SOIL - ADULTS										4.9E-01	5.4E-01
INORGANICS											
Arsenic	4.8E-05	0.1	1E-03	NA	Keratosis and hyperpigmentation	NA/HEAST	1		4.8E-01		
EXPOSURE PATHWAY: DERMAL CONTACT WITH CHEMICALS IN SOIL - CHILDREN										2.0E+00	3.0E+00
INORGANICS											
Arsenic	2.3E-04	0.1	1E-03		Keratosis and hyperpigmentation	NA/HEAST	1		2.0E+00		

SUMMARY OF CHRONIC HAZARD INDEX ESTIMATES
SENARIO 3: RESIDENTS ON SITE

CHEMICAL	CHRONIC DAILY INTAKE(CDI) (mg/kg/day)	CDI ADJUSTED FOR ABSORPTION	RFD (mg/kg/day)	CONFIDENCE LEVEL	CRITICAL EFFECT	RFD SOURCE/ BASIS	RFD UNCERTAINTY ADJUSTMENTS	MODIFYING FACTORS	HAZARD QUOTIENT	PATHWAY HAZARD INDEX (HI)	TOTAL EXPOSURE HI
EXPOSURE PATHWAY: INGESTION OF CHEMICALS IN DRINKING WATER - CHILDREN										84	100
INORGANICS											
Arsenic	1.1E-03	No	1E-03	NA					1E+00		
Chromium VI	1.6E-02	No	5E-03	Low	No effect reported	Water/IR	500	1	3E+00		
Manganese	7.6E+00	No	1E-01	Medium	CNS effects	Diet/HEA	1	1	8E+01		
EXPOSURE PATHWAY: DERMAL CONTACT WITH CHEMICALS IN SOIL - CHILDREN										13	
INORGANICS											
Arsenic	1.3E-03	0.1	1E-03	NA	Keratosis and hyperpigmentation	NA/HEAST	1		13		
EXPOSURE PATHWAY: INGESTION OF CHEMICALS IN SOIL AND HOUSE DUST - CHILDREN										3	
INORGANICS											
Chromium VI	6.3E-03	No	5E-03	Low	No effect reported	Water/IR	500	1	1E+00		
EXPOSURE PATHWAY: INGESTION OF CHEMICALS IN DRINKING WATER - ADULT										51	60
INORGANICS											
Chromium VI	9.6E-03	No	5E-03	Low	No effect reported	Water/IR	500	1	2E+00		
Manganese	4.6E+00	No	1E-01	Medium	CNS effects	Diet/HEA	1	1	5E+01		
EXPOSURE PATHWAY: DERMAL CONTACT WITH CHEMICALS IN SOIL - ADULT										9	
INORGANICS											
Arsenic	8.9E-04	0.1	1E-03	NA	Keratosis and hyperpigmentation	NA/HEAST	1		9		

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APPENDIX

1.0 INTRODUCTION

TAMS Consultants, Inc. (TAMS) and TRC Environmental Consultants, Inc. (TRC) performed a remedial investigation at the Hertel Landfill site, Plattekill, New York (National Priority List #810, August 1990) in accordance with the requirements of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) as amended by the Superfund Amendments and Reauthorization Act (SARA). The investigation was performed for the U.S. Environmental Protection Agency (USEPA)-Region II under the Alternate Remedial Contracting Strategies (ARCS) program (Contract No. 68-S9-2001, WA No. 003-2LH7).

The purposes of the investigation were to investigate physical characteristics at the site, sources of contamination, determine the nature and extent of contamination and characterize potential health risk and environmental impact. This Remedial Investigation (RI) Report presents the findings of this investigation. Other pertinent reports prepared by TAMS and TRC and others on the Hertel Landfill site include:

- Final RI/FS Work Plan for the Hertel Landfill Site, September 1989
- Final RI/FS Field Operations Plan (FOP) for the Hertel Landfill Site, October 1989
- Site Analysis, Hertel Landfill, 1990
- Site Analyses, Hertel Landfill, 1990 (EPA/ORD/Environmental Monitoring Systems Laboratory)

The Remedial Investigation Report is presented in a format consistent with the "Guidance for Conducting RI/FS Under CERCLA" (USEPA, 1988) as follows:

Section	Description of Contents
1	Introduction/Background Information
2	Description of Site Investigations
3	Physical Characteristics of the Study Area
4	Nature and Extent of Contamination
5	Contaminant Fate and Transport
6	Baseline Risk Assessment
7	Summary and Conclusions
8	References

The text of the RI Report is presented in Volume 1. The tables, figures and plates referenced in the text are presented in Volume 2. Volume 3 contains the Appendices to the RI Report.

1.1 Site Description

The Hertel Landfill site is located in the town of Plattekill, Ulster County, New York, just south of U.S. Route 44/NY Route 55 as shown in Figure 1-1.

The property occupies approximately 80 acres and is oriented in a north-south direction; the entire 80-acre property is hereinafter considered the site. The landfill area occupies approximately 13 acres of the property.

The topography of the site is generally flat with a gentle overall slope to the east. Abundant vegetation covers most of the property with the exception of limited portions of the landfill. This landfill is located roughly at the center of the site and is covered with rocky soil and wastes with patches of grass and small shrubs. Previous investigations identified a number of waste disposal areas which are located on the site:

- Disposal Area #1 - engine block and oil waste materials
- Disposal Area #2 - trailer wreckage and scattered drums
- Disposal Area #3 - oil stain area and sanitary waste
- Disposal Area #4 - farm equipment debris
- Disposal Area #5 - printing waste
- Disposal Area #6 - fibrous material piles
- Disposal Area #7 - paint waste and municipal landfill
- Disposal area #8 - possible rubber waste

The approximate locations of these disposal areas and the landfill are shown in Figure 1-2.

Wetlands border the site to the north, south, and east. Based on the Tentative Freshwater Wetlands Map of Ulster County (NYSDEC, 1986), areas identified as potential wetlands also cover approximately 13 percent of the total area of the site. A small unnamed stream (H-128-6-2-1-2) crosses the southern and eastern area of the site and flows in a northeasterly direction, to a pond just east of Disposal Area #8. Flow from this pond (also referred to as Wetland B later in this report), continues due east to Pancake Hollow Creek (H-128-6-2-1). The creek flows northward to Routes 44/55, where just north of the roadway it forms a pond/marsh area.

A locked gate exists across the main access road near Routes 44/55, however there is no perimeter fence.

1.2 Site History

The Hertel Landfill was established in 1963 as a municipal waste landfill. It is believed that about 10 acres of the site were used when the landfill was operating. Until 1975 the landfill was operated by Carlo Hertel and later by his family (Hertel Enterprises). Around 1970, Dutchess Sanitation Services, Inc. began hauling refuse from Dutchess County to the Hertel Landfill and in 1975, Dutchess Sanitation Services, Inc. purchased the landfill.

In April 1976, the Ulster County Department of Health (UCDOH) revoked the landfill permit for a variety of violations, among which were allegations of illegal industrial dumping. This UCDOH action and a Town of Plattekill ordinance prohibiting the dumping of out-of-town garbage resulted in the permanent closing of the site in March of 1977.

Ownership of the site then passed from Dutchess Sanitation Services, Inc. through two subsequent parties [a partnership known as F.I.C.A. and then to Hudson Valley Environmental Services, Inc. (HVES)] to its current owner, Paul V. Winters and his corporation, Environmental Landfills, Inc., (ELI) based in New Windsor, New York. No landfilling operations or other activities are currently performed under the present proprietor, ELI.

During this time, the New York State Departments of Environmental Conservation (NYSDEC), Health (NYSDOH), and Law (NYSDOL) had filed suit for clean up of the landfill site. In 1981, NYSDEC directed HVES to conduct ground water monitoring. Based on the results of this monitoring, the NYSDEC placed the Hertel Landfill site on the New York State List of Hazardous Waste Disposal Sites. In 1983, the site was recommended for inclusion on the National Priority List (NPL) by the NYSDEC and in October 1984, the USEPA proposed the Hertel Landfill site for inclusion on the NPL (a "Superfund" site). In June 1986, the Hertel Landfill site was placed on the final list of federal Superfund sites. At the time, it was ranked 649 of approximately 1,200 NPL sites. As of August 1990, the site was ranked 810.

Subsequent to the conclusion of the field activities for this RI, a report was released by the USEPA (McDonald, 1990) which presented an analysis of historical aerial photography of the Hertel Landfill site. This report reviewed aerial photographs between the period 1959 and 1989 in order to determine the nature and extent of landfilling activities. The analysis and observations contained in the report are summarized below:

- Landfilling activities partially filled wetland areas. Although most landfilled material was covered with earthen fill, large amounts of scattered refuse was visible after the landfill was reportedly closed.
- From the review of aerial photographs, no landfilled materials were specifically identified. However, areas of wet ground and standing liquid were identified on the aerial photographs (dated, 1974) within the area of landfilling activities, but were not observed during the field investigation.

1.3 Previous Investigations

In 1981, five ground water monitoring wells were installed at the site under the supervision of Wehran Engineering, Inc. and at the direction of the

State of New York. These include one well on the western portion of the site and two pairs of wells, one deep and one shallow, on the expected downgradient eastern-edge of the site near the accumulation of the paint waste and brown fibrous materials (Figure 1-2).

Sampling and analysis of ground water from the two pairs of wells in 1980 and 1982 revealed measurable amounts of chloroform, methylene chloride, toluene, 1,1,1-trichloroethane, diethylphthalate, bis-(2-ethylhexyl)phthalate, and a number of metals (arsenic, barium, beryllium, chromium, copper, iron, mercury, manganese, nickel, lead, selenium and zinc). Water samples from the upgradient well revealed 1,1-dichloroethane, bis-(2-ethylhexyl)phthalate, and a number of metals (arsenic, beryllium, chromium, copper, lead, nickel, and zinc).

Three surface water samples, described as leachate, were collected in March and May 1981 by the NYSDEC. Analysis showed detections of phenols, organic carbon, methylene chloride, trans-1,2-dichloroethane, 1,1,1-trichloroethane and a number of metals (iron, arsenic, mercury, manganese, nickel and zinc).

In 1987, Dynamac Corporation, on behalf of the current owner, ELI, prepared an "RI/FS Work Plan/Scoping Document" under the guidance of the NYSDOL. The RI/FS Work Plan and FOP were modeled after the Dynamac Work Plan.

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2.0 SITE INVESTIGATION

This section provides a description of site investigation activities. A more detailed description of the field procedures is provided in Appendix A or in the final Field Operations Plan (FOP), dated October 1989.

The RI field activities were staged so as to direct and optimize subsequent field activities. Geophysical surveys and soil gas screening techniques were performed to assist in identifying desirable surficial and subsurface soil sampling locations and to detect ground water contaminant plumes, if present. Test pits were excavated next to characterize refuse and optimize the placement of monitoring wells for ground water sampling. Subsequent field activities included test borings for monitoring well/piezometer installation, ground water sampling, geological characterization, on- and off-site surface water and sediment sampling, private well sampling, hydraulic testing, ecological sampling. Generally, the activities are presented in this report in their order of occurrence. Field activities were conducted intermittently between September 1989 and October 1990.

The locations of the field activities were surveyed by C.T. Male Associates, P.C. of Latham, New York and are shown on Plate 1.

2.1 Site Mobilization

An initial site reconnaissance was conducted in September 1989 by TAMS/TRC personnel prior to field site activities to initially assess ambient conditions for worker safety and to locate any obvious areas of environmental concern.

The site reconnaissance was performed by TAMS/TRC personnel using the following instrumentation: Century 128 organic vapor analyzer (OVA); HNu model PI-101 photo ionization detector (HNu); Mine Safety Appliances (MSA) Model 361 oxygen/combustible gas indicator (O_2/LEL), and hydrogen sulfide (H_2S) detector; and a Victoreen Model 580 radiation survey meter. All instruments were calibrated prior to their use by TAMS/TRC personnel according to the manufacturer's instructions.

The initial site reconnaissance did not discern any special health and safety concerns or additional environmental issues from those previously identified by others. The instrument readings for the initial site reconnaissance are tabulated in Table 2-1; instruments were calibrated in accordance with TAMS Standard Operating Procedures.

Site activities in late September 1989 included partial clearing of the site access road and the delineation of the exclusion zone around the perimeter of the site. The site compound was constructed consisting of a site trailer, an equipment shed, a decontamination pad, a contamination reduction zone and a perimeter chain link fence with locking gate and razor wire to secure the compound.

2.2 Geophysical Surveys

Geophysical surveys were conducted to locate buried metallic wastes and identify specific areas of concern for further investigations. Three geophysical techniques were used at the Hertel Landfill site: electromagnetic conductivity (EM), magnetometry, and metal detection. The EM and magnetometry surveys were performed on a 25-foot grid over the area indicated in Figure 2-1, which is considered the area of the suspected landfilling activities. The metal detection survey was performed only in areas where results of the other techniques suggested the presence of buried metal.

The electromagnetic conductivity survey was conducted by TAMS/TRC personnel on October 25-26, 1989 using a Geonics EM-31 conductivity meter.

The magnetometer survey was conducted by Hager-Richter Geoscience, Inc. (Hager-Richter) of Salem, New Hampshire on October 26, 1989 using an EG&G Model G856 proton precession portable magnetometer with a gradiometer option. The metal detector survey was conducted by TAMS/TRC personnel using a Schonstedt magnetic cable locator in October and November 1989. The instruments were calibrated in accordance with recommended operating procedures.

The EM-31 survey indicated that large portions of the landfilled area included metallic objects within the waste. It was not possible to discriminate between shallow and deeper buried objects nor determine the size of the metallic debris. The EM-31 survey was unable to discern areas of ground water contamination because of interference from metallic objects over most of the landfill area. Some EM-31 results were useful in interpreting the area of landfilling activities. The EM-31 survey results are shown in Figure 2-2. All EM-31 survey readings are tabulated in Appendix B.

Seventeen surface/near-surface anomalies and 19 buried-object anomalies were identified by Hager-Richter in the magnetometer survey and are shown in Figure 2-3. Generally, large buried object anomalies were found in the southern area of landfilling activities. The Hager-Richter magnetometer survey report (without appendices) is incorporated in Appendix B.

Due to the heterogeneity of the subsurface materials and interference of surface metallic objects, the metal detector survey did not provide any useful data regarding the specific landfill area. As shown by the EM-31 and magnetometer surveys, metallic debris is ubiquitous in the landfilled area. Therefore, the metal detector's efficiency to discriminate specific metallic objects was greatly diminished due to interference from nearby metal debris. Since this instrument does not yield numerical readings, a summary of the instrument readings is not provided.

It is important to note that there is more uncertainty with the geophysical anomoly data beyond the limits of the actual grid sampling locations.

2.3 Soil Gas Survey

The soil gas survey was performed to assess the location, extent, and characteristics of waste materials existing in the subsurface at the site. Two separate soil gas surveys, a screening survey and a specialized soil gas survey, were conducted by TAMS/TRC personnel at the site. Soil gas survey results were utilized along with the results of the geophysical survey for final site location of the planned borings, monitoring wells, and test pits. Specific sampling methods are described in the FOP.

The soil gas screening survey was performed by TAMS/TRC personnel at fifty-foot intervals across the site from October 28 through November 23, 1989. Soil gas survey anomolies are shown on Figure 2-4; the summary data are presented in Appendix C. This survey was conducted using the following instruments: a Century 128 organic vapor analyzer, an HNu Model PI-101 equipped with a 10.2 electron-volt bulb, an MSA Model 361 oxygen (O_2)/combustible gas indicator (LEL) and hydrogen sulfide (H_2S) meter. This equipment permitted the detection of a variety of organic vapors but provided limited identification and quantification of the soil vapors. The soil gas sampling was conducted by drilling or driving a shallow hole, inserting a length of teflon tubing, purging the tubing and collecting a gas sample directly into the instrument. The procedures used are as detailed in the FOP. The only significant change from the FOP procedures was the use of a 2.5-foot deep vapor hole versus the 4.5-foot hole noted in the FOP. The soil gas hole depth was decreased in the field principally for practical considerations, as the fill material (rock, metal, plastic garbage) often made drilling difficult to advance.

The soil gas screening survey revealed that high organic vapor concentrations were present throughout the surveyed area. The OVA and LEL readings, specifically sensitive to methane, were constantly off-scale. However, the HNu readings were generally negligible, except in the area in the vicinity of Disposal Area #7 as well as in a few other select locations (Figure 2-4). The soil gas screening survey results are summarized in Appendix C.

A specialized soil gas survey, which permitted more accurate identification and quantification of specific organic vapors, was conducted in the paint waste/municipal fill disposal area (Disposal Area #7) using an HNu Model 311 portable gas chromatograph (GC). This survey involved the sampling of soil gas at fifteen points located on the basis of the soil gas screening survey results. The portable GC was calibrated to benzene, toluene, trichloroethylene (TCE) and tetrachloroethylene (a.k.a. perchloroethylene or PCE). These compounds were selected because of their prior detection at the site (toluene) or because they are common organic contaminants with a strong tendency to volatilize. Samples were collected from the teflon tubing using a glass syringe and injected into the GC; syringe and instrument blanks were collected to assure sample quality. Instrument calibration was performed daily, in accordance with standard operating procedures.

The GC soil gas survey was conducted in the area of paint waste along the eastern edge of the fill, where the largest area of elevated HNu readings were noted in the screening survey. The GC soil gas survey identified compounds within the paint waste/municipal fill disposal area contributing to the elevated organic vapor concentrations detected during the HNu survey. The four specific organic vapors that were calibrated for were identified in this area: benzene, toluene, TCE and PCE. At grid location 1825 and 50R none of these four organic vapors was detected (see Plate 1 for specific grid locations). Grid location 1925 and 00 however, had the most elevated concentrations of toluene, TCE and PCE in this disposal area. The GC soil gas results are presented in Table 2-2.

2.4 Surface Soil Sampling

Surface soil sampling was conducted on November 20-21, 1989 to determine the nature and extent of surface soil contamination in the vicinity of the disposal areas. Surface soil sampling was conducted prior to any other major field sampling activity (i.e., drilling, excavating), to minimize cross-contamination by heavy equipment traffic.

A total of 22 surface soil samples were collected for chemical analysis. Most of the samples were collected within the area of landfilling activity, focusing on the eight reported waste disposal areas. Surface soil sampling locations are shown in Figure 2-5. The rationale for the surface soil sampling locations is presented in Table 2-3. At the time of the surface soil sampling, there was no discernable physical evidence indicating the existence of Disposal Area #8 (rubber waste). Two surface soil samples were collected in this undeveloped area of the site. In addition, one background surface soil sample was taken from an apparently clean, upgradient portion of the site.

Surface soil samples were collected in accordance with the procedures detailed in the FOP. All sampling equipment (bowls, spoons and spatulas) were decontaminated according to the eight-step procedure detailed in the FOP (Appendix A). Two matrix spike/matrix spike duplicate (MS/MSD) samples (SS-8 and SS-17), two duplicate samples (SS-06 and SS-14) and one field blank (one per decontamination event) were collected for quality control purposes.

2.5 Test Pit Excavation

The test pit excavation and sampling activities were conducted at the site to visually investigate the presence and areal extent of the waste disposal areas. Historical aerial photos and the results of the geophysical and soil gas surveys were used to locate the test pits. All test pitting activities were conducted according to the procedures detailed in the FOP.

A total of 25 test pits were excavated at the site. Test pit activities were conducted by Enroserv, Inc. of Piscataway, New Jersey under the supervision of TAMS/TRC personnel. Test pit activities commenced on December 3, 1989 and ended on December 22, 1989. The rationale for the test pit locations is presented in Table 2-4 and the locations are shown in Figure 2-6. Eighteen of those test pits (designated by TP in the table and on the plate) were excavated at known areas of waste disposal in order to characterize the waste, to investigate magnetic anomalies and to determine the vertical extent of contamination. Six test pits were excavated to investigate the paint waste/municipal landfill area (Disposal Area #7 on Figure 1-2). One test pit was excavated in each of the disposal areas numbered 1 through 6. No test pits were excavated in Disposal Area #8, as no evidence of this disposal area's existence was found in initial site surveys. Another six test pits were excavated within the landfill area to investigate magnetic anomalies and to characterize the vertical extent and nature of wastes.

For exploratory purposes, seven test pits were excavated outside the apparent disposal areas throughout the central and southern portions of the site to verify the absence of waste disposal and provide additional information on site stratigraphy. These exploratory test pits were designated "EP" and numbered 1 through 7. Test pit excavation logs for all 25 test pits are submitted as Appendix D.

A soil sample was collected for chemical analysis from the bottom of four test pits in Disposal Area #7 (TP-1, TP-2, TP-7, and TP-15) to characterize the nature of the soils. One sample for chemical analysis was collected from each test pit associated with Disposal Areas #1 through #6. A total of ten soil samples were collected from test pits within the landfill area, as summarized in Table 2-5.

Samples for chemical analysis were collected from the backhoe bucket using a field-decontaminated stainless-steel spoon and bowl. The samples were collected from the contents of the backhoe bucket which were not in direct contact with the steel bucket. All sampling equipment was decontaminated in accordance with the procedures outlined in the FOP. The backhoe bucket was steam cleaned prior to use at each test pit location and at the conclusion of all test pitting activities.

Ground water was observed in contact with the landfill waste in test pits TP-2, TP-7, TP-8 and TP-15. Sampling of water for chemical quality was conducted at these four test pit locations: three test pits in Disposal Area #7 (TP-2, TP-7 and TP-15) and one test pit in Disposal Area #1 (TP-8). A sample of the accumulated water was collected from each of the four test pits as summarized in Table 2-5. All test pit soil and water samples were collected according to the procedures detailed in the FOP.

Soil and water samples collected for chemical analysis included environmental samples (10 soil samples, 4 water samples), environmental duplicates, matrix spike/matrix spike duplicates (MS/MSD), and field blanks (one per decontamination event) for QA/QC purposes. Trip blanks (VOC only) were submitted with each sample shipment. Samples were collected as summarized in Section 2.13.

Shallow piezometers were installed in selected exploratory test pits (EP-2, EP-4, and EP-5) where ground water was encountered to provide additional water table elevation data for areas removed from landfill disposal areas and likely well installation areas. The piezometers consisted of a five-foot length of two-inch diameter Schedule 40 PVC well screen (10-slot) and PVC riser extending approximately to three feet above grade. Backfill from the excavation of the exploratory test pit was placed around the piezometer. Water levels in the piezometers were periodically monitored with an electronic water level indicator.

No samples were collected for physical characteristics as proposed in the FOP. Attempts to collect Shelby tubes in the test pits were not successful due to the coarse-grained texture (sand, gravel and boulders) of the subsurface soils.

2.6 Test Borings

Test borings were drilled at the site to permit the collection of subsurface soil samples for geologic characterization and laboratory analysis. The test borings were also used for the installation of ground water monitoring wells and piezometers; these procedures are discussed in Section 2.7. Historical aerial photographs, the geophysical and soil gas surveys, and the findings of the test pit investigation were used to locate the test borings.

A total of 23 test borings were drilled at the site at 16 locations. At seven locations, shallow and deep test borings were drilled for the installation of well clusters. These well clusters were installed in order to vertically characterize ground water quality and geology. Well clusters, consisting of one shallow well and one deep well, were drilled at the following locations: MW-1, MW-2, MW-3, MW-6, MW-7, MW-10, and MW-11. At nine locations (MW-5, MW-8, MW-9, MW-12, MW-13, P-1, P-2, P-3, and P-4), a single test boring was drilled and one monitoring well or piezometer was installed. The rationale for the test boring locations is provided in Table 2-6.

Test boring activities commenced on January 10, 1990 and terminated on June 6, 1990. Drilling was performed by W.C. Services, Inc. of Woodbury, New Jersey under the supervision of TAMS/TRC personnel.

A number of drilling techniques were used to drill the test borings at the Hertel site including methods not discussed in the FOP. Alternative drilling technologies were necessitated due to the subsurface geologic materials encountered. Large boulders (up to eight feet in diameter) were encountered at many deep well locations at the site (MW-1D, MW-2D, MW-3D, MW-6D, MW-10D) and one shallow well location (MW-1S). Attempts to drill by conventional drilling techniques (e.g., hollow stem augers, tri-cone roller bit, tri-cone roller bit with casing and Nx coring) proved unsuccessful and time intensive. Therefore, the Tubex method, an air-hammer-with-casing drilling technology, was used to advance the borehole for all the deep borings.

The Tubex method drills through the subsurface materials as follows:

- compressed air circulates through the center of the drill pipe, engaging the air hammer which pounds into and advances through the subsurface material;
- the air hammer simultaneously advances a string of casing which is located approximately one foot behind the drill bit;
- the compressed air circulates through holes out the bottom of the drill bit to carry cuttings from a discrete depth interval (approximately one foot) into the annulus between the drill pipe and casing;
- the drilled materials are carried up the annulus to the surface and directed to a discharge area for observation by the geologist.

All shallow borings, except for MW-1S, were drilled by either 3.25- or 6.25-inch hollow-stem auger. An air hammer without casing was used at the MW-1S location after repeated attempts by conventional drilling methods proved fruitless.

All deep borings were drilled using the Tubex method described above to reach either: 1) the final borehole depth (for deep overburden borings) or 2) five feet into bedrock (for bedrock wells). Prior to advancing the borehole deeper into the bedrock, a four-inch diameter steel casing was installed and grouted into bedrock. The test boring was then advanced to its final depth by Nx coring and standard rotary methods, as per the FOP.

At three locations (MW-5S, MW-6D, MW-7D), steel casing was installed to isolate the fill material prior to drilling any significant depth below the fill. At MW-5S, eight-inch diameter steel casing was used, while at MW-6D and MW-7D, fourteen-inch diameter steel casing was used.

Subsurface soil samples were collected at three locations in order to assess background subsurface soils quality. These subsurface soil samples were collected at the following locations and intervals: MW-1S (15-17 feet), MW-2S (6-8 feet) and MW-3S (0-2 feet). Subsurface soil samples were collected in accordance with the procedures described in the FOP, except that three-inch inside diameter split spoon samples were used to ensure that sufficient volume was recovered for all analyses and associated QA/QC samples.

Test boring logs are attached as Appendix E.

2.7 Monitoring Well/Piezometer Installation

Nineteen ground water monitoring wells were installed to investigate the hydrogeology of the Hertel Landfill site and to permit the collection of ground water samples (see Plate 1). Four piezometers were installed within boreholes to provide additional ground water elevation data away from the landfilled areas of the site.

Monitoring wells and piezometers were installed between January 10 and June 6, 1990 by W.C. Services of Woodbury, New Jersey under the supervision of TAMS/TRC personnel.

Each of the monitoring wells were completed in one of three geologic intervals: the shallow unconsolidated overburden deposits (all wells suffixed with an "S"), the deep unconsolidated overburden deposits at approximately 45 feet below grade (MW-1D, MW-10D and MW-11D) and the shallow sedimentary bedrock (MW-2D, MW-3D, MW-6D and MW-7D). Most wells and the piezometers were completed in the shallow unconsolidated material to permit a detailed characterization of the water table beneath the site. The seven deep monitor wells installed as part of this program were paired with shallow unconsolidated monitor wells and were intended to monitor the shallow bedrock, if present at depths less than 50 feet (as per the FOP). Four of those wells were completed in the shallow bedrock, permitting an estimation of the shallow bedrock piezometric surface. Three of those wells were completed in the deep overburden, where bedrock was not encountered, enabling a comparison of the deep unconsolidated piezometric surface to the water table. In addition, the well clusters permit a comparison of vertical hydraulic head differentials at several locations.

Monitoring wells were installed according to the procedures outlined in the FOP. These procedures are summarized as follows:

- Two-inch diameter, flush-joint stainless-steel riser and screen (0.010-inch slot) were used for all wells. Wells were constructed with ten feet of screen, except for MW-2S and MW-9S, which were constructed with five and nine feet of well screen, respectively. Riser was extended from the top of the screen to approximately two feet above grade.

- A clean silica sand gravel pack (Morie #1) was placed in the annulus between the borehole wall and the well. Generally, the gravel pack was placed to one to two feet above the top of the well screen. However, physical constraints (e.g. shallow water table) sometimes necessitated completing the gravel pack nearer to the top of the well screen.

- A one- to two-foot thick bentonite-pellet seal was placed above the gravel pack. Shallow water table conditions prevented the use of a standardized two-foot seal in some cases.

- A cement/bentonite or 100% bentonite grout was used to fill the remaining annulus to the ground surface.

- A locking steel protective casing was installed and cemented into place.

Piezometers were installed according to the procedures outlined above. However, instead of two-inch diameter stainless-steel riser and casing, two-inch diameter Schedule 40 PVC was used for the piezometers' construction. All well construction materials were wrapped in individual plastic sleeves and opened immediately prior to use.

Monitoring well construction diagrams are included with the boring logs as part of Appendix E. Construction details for all monitoring wells and piezometers are presented in Table 2-7.

After installation, all monitoring wells were developed using a centrifugal pump, peristaltic pump and/or a stainless-steel bailer. During well development, pH, specific conductance, temperature and turbidity were monitored to determine when the well was sufficiently turbidity-free. Well development was concluded when either the well purge water had acquired a turbidity of less than 50 nephelometric turbidity units (NTUs) or had stabilized to a point that no further reduction in turbidity was observed. A summary of the well development activities is presented in Table 2-8.

Periodic water level measurements were conducted with an electronic water level indicator on all monitoring wells and piezometers at the site.

2.8 Ground Water Sampling

The ground water investigation consisted of the installation and sampling of monitoring wells to determine ground water quality at the site and to assess if potential contaminants were migrating from the site via the ground water.

The nineteen newly installed monitoring wells and the five existing Wehran monitoring wells were sampled during the field investigation. Two rounds of ground water sampling were performed. The first event was conducted in June 1990, the second event occurred in August 1990. Samples were analyzed in both rounds for the following parameters:

- Field measurements of pH, specific conductance, temperature, Eh
- Volatile Organic Compounds (VOC)
- Total Organic Halogen (TOX)
- Total Organic Carbon (TOC)
- Extractable Organics (Base/Neutral/Acid Extractable Compounds, Pesticides and PCBs)
- Total Metals (unfiltered)
- Dissolved Metals (filtered)
- Cyanide
- Ions (sulfate, chloride, carbonate, bicarbonate)
- Turbidity, dissolved solids, suspended solids
- Nitrogen (N) and compounds (TKN, ammonia N, inorganic N)

All ground water sampling activities were conducted in accordance with the FOP except as noted below. All ground water samples were collected with dedicated, laboratory-cleaned, stainless-steel bailers and laboratory-cleaned teflon-coated stainless-steel cables.

During the June sampling event, ground water samples for dissolved metals were field filtered using a Millipore stainless-steel filtering apparatus with a hand pump as specified in the FOP. Despite well development, formation silt was present in a number of ground water samples and, as a result, field filtration was difficult and time consuming with the hand pump.

To facilitate sampling during the second (August) ground water sampling event, the dissolved metal samples were field-filtered using a 0.45 micron high-capacity in-line sample filter (Quick Filter) manufactured by QED Environmental Systems, Inc. The body of the filter is constructed of polypropylene and the filter is made of an acrylic copolymer. This modification to the field sampling procedures was approved by the USEPA prior to its use. The ground water collected for the filtered metals analysis was poured from the bailer into a laboratory-grade 1,000 ml plastic bottle. The samples were drawn through the filters using a low-discharge (0.25 gpm) peristaltic pump (ISCO) equipped with dedicated tubing. The initial volume of filtrate (approximately 200 ml of ground water) was discarded prior to the collection of the sample for the metals analysis in the pre-preserved 1,000 ml polyethylene sample container. The filter apparatus and tubing were disposed after one use. Each extra sample volume for field duplicates, matrix spikes, and matrix spike duplicates was filtered through the same disposable filter and tubing as the corresponding sample.

The two rounds of ground water sampling each required three days to complete. A total of 65 ground water samples was collected during the two rounds of sampling from the ground water monitoring wells at the site. A total of 32 samples comprised of 24 ground water samples from monitoring wells, one field blank (one for each decontamination event), three trip blanks (VOC only), two environmental duplicate samples, and two MS/MSD sample pairs was collected during the June 11-13 sampling event. A total of 33 samples comprised of 25 ground water samples from monitoring wells, one field blank, three trip blanks (VOA only), two environmental duplicate samples and two MS/MSD sample pairs was collected during the August 20-22 sampling event. During the August sampling event, piezometer P-2 was sampled in lieu of the collection of a surface water sample at the SW-26 location (June 1990 sampling event) because the leachate seep was dry. The ground water sample taken from the P-2 location was submitted for chemical analyses.

2.9 Private Water-Supply Wells

Eleven private water-supply wells were sampled and analyzed during June 19-20, 1990 ground water sampling event, at the homes shown in Figure 2-7. Most of these private wells had been previously sampled by NYSDOH. These locations were selected based on their prior NYSDOH sampling, proximity to the Hertel Landfill site and ability to obtain permission from the homeowners. A summary of the private well sampling information is presented in Table 2-9.

The private water-supply wells were sampled according to the procedures described in the FOP. A total of 14 samples was collected comprised of 11 private well ground water samples, one trip blank (VOC only), one environmental duplicate and one MS/MSD sample.

2.10 Surface Water/Sediment Sampling

The surface water and sediment sampling was performed in order to identify if the disposal activities at the site are impacting sediments and/or surface water through surface runoff and ground water discharge.

A total of 33 surface water samples was collected consisting of 25 environmental samples, two environmental duplicates (SW-40 and SW-41), two MS/MSD samples and four trip blanks.

Surface water grab samples, including leachate seep samples, were collected from the 25 sampling stations shown in Figure 2-8. (Surface water and sediment sample locations in the streams and wetland are designated both on Figure 2-8 and on Plate 1 as "SW". The leachate seep sample locations are designated "LS". General siting of surface water sampling locations was based upon a comprehensive site inspection, review of available information and the protocols of the FOP. The rationale for the selection of the twenty-five surface water sample locations during the site field investigation program is presented in Table 2-10. Figure 2-8 and Plate 1 reference location SW-26. At the time of the surface water sampling, no water was present in the leachate seep previously observed at location SW-26.

Sediment samples were collected at each surface water sampling location, with the addition of sample location SW-26. Only a sediment sample was collected at location SW-26. As noted above, sample locations included leachate seeps, adjacent wetlands, the creek and a tributary, background and downgradient locations. To the extent possible, samples were collected from the point of greater sediment accumulation, usually depositional areas of low stream flow velocity. The sediment sampling locations were determined based on the same criteria as the surface water samples. The rationale for the sediment samples collected during the site field investigation program is discussed in Table 2-11.

A total of 38 sediment samples was collected. Twenty-six grab samples were collected from zero to six inches in depth. At six of the twenty-six sampling areas, two additional sediment samples were collected from six to twelve inches below grade and twelve to eighteen inches below grade. The rationale for the collection of the deeper samples was to provide a stratigraphic column in order to determine if there is a downward migration of contaminants. In addition, two MS/MSD samples (SED-1 and SED-20) were collected, and three field blanks were taken off sampling equipment used when collecting samples SED-29, SED-24, SED-10. Associated trip blanks are discussed with the surface water samples.

Surface water and stream sediment samples were collected according to the procedures detailed in the FOP. Field measurements were performed at each sampling location and were obtained from a separate sample aliquot not used for chemical analysis. For sediment samples this included field measurements of sediment oxidation/reduction potential which were made at each location. Field measurements for surface water samples consisted of conductivity, pH, Eh, and dissolved oxygen.

All equipment was field decontaminated employing the eight-step procedure outlined in Appendix A. While collecting the sediment samples, care was taken to minimize disturbance and sample washing as the sample was retrieved through the water column above. If the water above was flowing or deep, finer-grained sediment could be carried out of the sample during collection.

Analytical parameters are summarized in Section 2.13.

Surface water monitoring also included the measurement of water levels at staff gages at the site. Nine staff gages (SWL-2 through SWL-10) were installed at the site at various locations to collect surface water level information. Periodic water level measurements were made at these staff gages with an electronic water level indicator.

2.11 Ecological Investigation

The purpose of the ecological investigation was to provide baseline information on the biological resources of the Hertel Landfill site. The study was designed to document existing floral and faunal species through population surveys, with particular emphasis placed on any threatened, rare, endangered, or protected species.

The field investigation entailed the following sampling/surveying components:

- wetlands identification,
- macroinvertebrates,
- birds,
- fish,
- mammals,
- herpetofauna,
- vegetation.

Field activities were conducted in two parts. In October 1989, the following surveys were conducted: wetlands, macroinvertebrates, mammal trapping, and incidental mammal, bird and herpetofaunal observations. In May 1990, the investigation included intensive avian, herpetofauna, and vegetation surveys as well as incidental mammal observations.

Prior to field sampling, pertinent background information was obtained which included the Clintondale USGS quadrangle, New York State Department of Environmental Conservation (NYSDEC) wetland mapping (no National Wetland Inventory mapping is currently available), the Ulster County Soil Survey, and documentation from the NYSDEC Wildlife Resource Center regarding the existence of endangered species and/or significant habitats on-site.

To determine the extent of jurisdictional wetlands subject to Section 404 of the Clean Water Act which is regulated by the Army Corps of Engineers (COE), wetlands are delineated pursuant to the manual entitled: Federal Manual for Identifying and Delineating Jurisdictional Wetlands (January, 1989) (or the Manual). The Manual presents technical guidelines to identify wetlands and distinguish them from non-wetlands. In order to apply the guidelines, the Manual provides a set of scientific methods and supporting information. A positive indication of wetlands must be present for all three parameters - vegetation, soils and hydrology. (Chapter 3 and Appendix H also report on the NYSDEC methodology for wetland delineation).

The U.S. Army Corps of Engineers (1989) provides step-by-step methods for both routine and comprehensive delineations of wetlands, guidelines on handling atypical situations (e.g., man-induced wetlands or natural events such as beaver dams), recommendations for determining if normal environmental conditions are present, and forms for recording data.

Field investigations for wetland assessment took place in mid-October 1989 by a TAMS/TRC wetlands ecologist. During the field investigation, eighteen observation points from three wetland areas were selected in order to accurately represent the characteristics of the site. The location of each observation point was marked in the field and identified by a three-digit reference number. The first digit designated an observation point (0), distinguishing it from upland/wetland boundary markers. The second digit corresponded to the sequential observation point numbers which include both an upland and wetland component. The location of each component was selected in order to quantify the characteristics of the upland and wetland communities. If the observation point was located in an upland adjacent to the wetland, a /U' followed the number. A /W' in this position indicated that the point was within the boundary of the wetland. For example, 0-3-U indicates the third observation point encountered (0-3) characterizing the upland community (U). In addition to the above borings, many additional borings were made to identify the limit of hydric soils and facilitate the determination of the wetland/upland boundary.

Benthic macroinvertebrate samples were collected throughout the site, in order to characterize resident macroinvertebrate communities both upstream, adjacent and downstream of the landfill as well as in the on-site pond.

A total of twenty macroinvertebrate samples was collected in October 1989 at the Hertel site. At the on-site stream, six sampling stations were established - two upstream (US-A and US-B), two contiguous with the landfill (IS-A and IS-B) and two downstream from the landfill (DS-A and DS-B). At each station, three replicate samples were collected, for a total of eighteen samples (6 stations x 3 replicates). Additionally, two stations were established within the ponded area on-site, from which, one sample each was collected.

In addition to documenting resident macroinvertebrate populations on-site, TAMS also wanted to compare differences in community composition at different points in the stream respective to the landfill. To accomplish this, habitats were sampled at two sampling stations established in each of three areas (upstream, adjacent landfill and downstream). In general, one station exhibited no flow (ponded area) and had a mucky, organic substrate, and the other station exhibited slow to moderate flow and a sandy/silty substrate with leaf matter, rocks and tree branches. Listed in Appendix H is a more detailed description of each sampling station.

An intensive avian survey was conducted in May 1990. The survey was conducted by establishing two ecological inventory transects per vegetative cover type (i.e., old field, forested wetland and forested upland) and performing a strip census of birds. Resident and transient birds on the project site were identified using sight and sound observations, along with other available evidence including feathers, eggs and nests. Wide-angle binoculars and avian taxonomic guides were utilized and only positive identifications were recorded in the field notes.

Ecologists conducted ten-minute bird counts at 500-foot intervals along each transect. Additionally, daily observations (30-minute surveying periods) were conducted at the ponded area to document water-dependent avian utilization (i.e. waterfowl or wading birds) of the site. All bird surveys were performed in the morning (commencing just before dawn) and/or in the early evening, for three sample periods.

In addition to the structured transect survey discussed above, less rigid bird observations were conducted on site in October 1989, when other ecological work was being performed. The autumn observations were conducted to document those species which might be seasonal migrants to the area.

Small and medium-sized mammals on-site were inventoried in October 1989 using paired, baited live traps set along ecological transect lines. Two 300-foot transect lines per cover type (old field, forested upland and forested wetland) were established. Paired and baited HAV-A-HART and Sherman Box Traps were set at 100-foot intervals along each line.

The traps were set for three consecutive nights during each of the two weeks of sampling. All traps were checked in the early morning in order to identify and release all mammals contained within as soon as possible.

In addition to trapping, mammals were surveyed through direct (actual sightings) and indirect (tracks, burrows, dens, tree rubs and scat) observations.

Herpetofauna were surveyed in May 1990 by searching suitable habitats including stream banks, ponded areas, underneath logs and rocks, in leaf litter and on sun-exposed rocks and outcroppings. Searches were conducted in various types of weather, but certain groups were sought most intensively when conditions were optional for them (i.e., warm, sunny days for turtles and snakes; warm evenings for frogs and toads). Additionally, eleven, two-gallon pit traps, with logs set in place to serve as barriers, were set in appropriate habitats on warm, damp nights, for a period of three nights. The pit traps were checked in the early morning in order to identify and release any organisms contained within as soon as possible.

Three dominant vegetation communities present at the Hertel site were identified as: old field, upland forest and wetland forest. These communities were sampled along transect lines randomly picked thorough the communities without any bias to particular individual species. Two transects were established varying in length from 250 to 1000 feet, depending on the extent of the community. Each transect, regardless of length, had at least two observation points at which specific species were logged (see Figure 1 in Appendix H).

At each observation point the overstory trees were identified within a 30-foot radius. Each tree at or over 4.0 inches diameter at breast height (DBH) was identified by species and its relative basal area was estimated. All tree saplings (under 4.0 inches DBH and over 4.5 feet tall) and shrubs (less than 20 feet tall with several stems) within a 30-foot radius were identified. All woody and non-woody plants under 4.5 feet in height were included in the herb layer. Herbs were sampled within a 5-foot radius plot. Woody vines were counted within a 30-foot radius of the observation point. Visual estimates of percent areal cover were calculated for each observed species in the sapling/shrub, herb, and woody vine vegetation layers.

Each vegetation layer was separately evaluated as to its dominant species. Criteria used were basal areas for trees, and percent areal cover for saplings/shrubs, herbs, and vines, in each respective sampling plot. The dominant species were determined, listed and assigned a wetland indicator status. The indicator status was made according to National List of Plant Species that Occur in Wetlands (Northeast) (USFWS, 1988). Special attention was given to identifying and documenting plants identified in the NYSDEC list of Protected Native Plants.

2.12 Hydraulic Testing

Hydraulic testing was performed on the ground water monitoring wells at the site in order to define the hydraulic properties of the water-bearing units.

Hydraulic tests were performed on most of the newly installed wells and all of the previously installed Wehran wells. Some wells were not hydraulically tested due to oily contamination noted in the ground water sampling (MW-6S) or due to difficulties resulting from insufficient saturated thickness (MW-7S), or due to the depth to water coupled with higher formation conductivity (MW-1D). At MW-1D, the depth to water precluded the use of a pump (beyond suction limits), while the well recovery was too rapid to gather useful data using the slug removal test.

Slug removal tests were performed in most instances, in accordance with the procedures specified in the FOP. Slug removal tests were analyzed using a computer program (Thompson, 1987) based on the Cedergren method (1977).

At five locations (MW-10D, MW-11D, MW-13S, MW-W1D, MW-W2D), the water level recovery was nearly instantaneous using a slug removal test, therefore insufficient data were collected to adequately compute the hydraulic conductivity of the formation. However, the nearly instantaneous water level recovery at these locations suggest that the hydraulic conductivity is high. At these five locations, pumping and recovery tests were conducted. These testing procedures are outlined as follows:

- Install pressure transducer and data logger; monitor water level until it stabilizes.
- Start pumping test and periodically monitor discharge rate to maintain a constant rate.
- Monitor water level to ascertain when conditions stabilize.

- Shut off pump and monitor water levels in well during recovery period.

These pumping tests were evaluated as single well pumping tests using the Theis non-equilibrium method as modified by Cooper and Jacob (Driscoll, 1986). The recovery tests were evaluated using the Theis recovery method presented by Driscoll (1986). Hydraulic testing was conducted from August 27 through September 12, 1990.

All hydraulic testing calculations are presented in Appendix G.

2.13 Field Investigation Sample Summary

A total of 154 environmental samples were collected during the environmental field investigation conducted at the Hertel Landfill site. In addition, sixty-two (62) QA/QC samples were collected consisting of duplicate samples, matrix spike and matrix spike duplicate samples, field blanks, and trip blanks. Two (2) water blanks were also collected during the course of the field investigation and submitted for laboratory analysis.

The sample collection information is summarized in Table 2-12.

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3.0 PHYSICAL CHARACTERISTICS OF THE STUDY AREA

Summarized below are the physical characteristics of the Hertel Landfill site and surrounding area. The physical characteristics are based on observations during the site investigation and published literature. Plate 1 shows the locations of all field activities performed at the site.

3.1 Surface Features and Land Use

The Hertel Landfill site is located in the Valley and Ridge physiographic province of Ulster County, New York, between the Shawangunk Mountains and the Marlboro Mountains. The Shawangunk Mountains are located approximately eight miles to the northwest of the site while the Marlboro Mountains are located approximately two miles to the east of the site. With the exception of the two mountain areas, most of the land within this province has been cleared for farms and orchards, and is low and gently rolling in relief. The more rugged areas remain forested.

The land surface at the site gently slopes to the east from hills to the west, south and southeast. The hills surrounding the site range in elevation from approximately 680 to over 700 feet National Geodetic Vertical Datum (NGVD). Land surface slopes to the east at approximately 7 feet per 100 feet across the site, reaching a low of about 614 feet NGVD at the stream and wetland exiting the site to the east. Topography is flatter in or near the wetland areas and steeper near the ridges and hills.

There are at least three wetland areas on the site which comprise 10.3 acres or 13% of the total 80-acre site area. The wetland elevations vary depending on the location: the wetland on the southern portion of the site is at an elevation of about 645 feet NGVD and provides the head waters for a small stream which flows to the northeast; the wetland areas in the northern and eastern areas of the site are at an approximate elevation of 614 feet NGVD. All three wetlands drain off-site to the east via a small stream.

Access to the site is provided via an unpaved road which leads south from Route 55 directly toward the center of the site. Three other cleared paths, trails or former roads lead from the site to the south, southwest and northwest. Based on the surveyed site map and exclusive of on-site wetlands, approximately 64 acres of the site are wooded or lightly wooded with the remaining 6 acres covered with low vegetative growth. The access road leads directly into the major cleared area at the center of the site.

The former landfill activities appear to be limited to approximately 13.2 acres in the central portion of the site (including a one-acre area of surface debris) as shown in Figure 3-1. Delineation of the landfilled area was based on background information, geophysical results and general site observations. Generally, the surficial deposits in the former landfill area consist of household refuse (plastic, glass, paper, etc.). Metal debris (empty 55-gallon drums and scrap metal) was observed along the northern edge of the former landfill (adjacent to the pond/wetland area). Paint waste was observed at the surface along the southeastern portion of the former landfill area, in the vicinity of monitor well MW-10. An area of oily-stained soils and vegetative stress was observed at the surface in the vicinity of MW-6. Fill thickness varies up to 16.5 feet (minimum) with an average thickness of about 12.0 feet. Based on an areal extent of 12.2 acres and the average fill thickness of 12 feet, the total fill volume is estimated to be about 236,000 cubic yards.

The area surrounding the site, as well as the site itself, is zoned residential, although a small industry is located to the west, adjacent to the site. A town permit is required to operate a business in this area. The site is currently inactive. The site is accessible for recreation such as walking or hunting.

3.2 Surface Water Hydrology

The Hertel Landfill site is approximately 80 acres in area with approximately 10.3 acres (COE criteria) occupied by surface water bodies and associated wetlands. These surface water bodies consist of a 5.8-acre pond and wetland (wetland C) located west of the entrance road to the site; a 2.2-acre wetland area in the southwestern portion of the site (wetland A); and a stream and associated 2.3 acres of on-site wetlands (wetland B) which leads from wetlands A and extends off-site to the east. During periods of high water, the pond has been observed to drain to the east over the access road. All three water bodies and wetland areas ultimately drain off-site to the east via a small stream and wetland.

The area which contains the landfill and which serves as the watershed for the stream draining the site is estimated to be 180 acres in area. It is defined by the ridges to the west of the landfill; hills to the northwest, south and southeast; and minor topographical divides to the north and northeast.

A simple water budget for the watershed is presented below, where flow out of the watershed or basin (Q) is equal to the difference between precipitation (P) and evapotransporation (E), and assumes no net change in basin storage.

$$Q = (P - E) * B * C$$

where

Q = Basin outflow in million gallons per year
P = 40.16 inches per year (mean annual of Poughkeepsie, NY, 1951-1980)
E = 50-75% of precipitation (Frimpter, 1972)
B = Basin area (180 acres estimated)
C = 37,000 conversion factor to standardize units

Over the area of the basin this amounts to an estimated average of 50-100 million gallons per year or 0.14-0.28 million gallons per day of combined ground water and surface water outflow from the basin (Q).

The basin drains via a series of unnamed and named streams. The applicable NYSDEC stream classification number is parenthetically noted after a stream is referenced. The first stream (H-128-6-2-1-2) originates on-site and exits the east side of the site, flows eastward and discharges into Pancake Hollow Stream (H-128-6-2-1) approximately 1,200 feet east of the site. Pancake Hollow Stream flows generally northeast across Route 44 and Crescent Avenue east of Clintondale. Pancake Hollow Stream discharges to a wetland area which is then drained by a third stream (H-128-6-2) which ultimately discharges into Black Creek (H-128-6), 3.75 miles north-northeast of the site. Black Creek meanders through a series of lakes, ponds and wetlands before discharging into the Hudson River 10 miles northeast of the site near Esopus, New York.

3.3 Geology

Geologic information for the site was gathered from published sources and from on-site field activities, including test pit excavation and test borings. Location of the geologic cross-sections discussed herein are shown in Figure 3-2.

Bedrock in the area of the site is the Austin Glen formation, described as a graywacke and shale. This is consistent with the rock cores obtained at various locations at the Hertel landfill site (MW-2D, MW-3D, MW-6D, MW-7D and MW-11D). At these well locations, an occasional variegated light blue to blue-grey fine to medium grained sandstone (graywacke) with occasional seams of shale was observed. The rock has well defined bedding planes and the upper few feet are slightly weathered. Based on a published bedrock map of the site area (Fisher et al., 1970), the bedrock unit strikes northeast-southwest and dips to the east towards the Hudson River.

The measured depth to bedrock at the site varies from 28 feet below grade in the north-central portion of the site (at the MW-6D location) to greater than 50 feet below grade (at the MW-10D location) along the eastern edge of the site. Measured bedrock elevations vary from less than 569 feet NGVD (MW-10D) to 637 feet NGVD (Wehran boring 3D). The form of the bedrock surface below the site (based on six borings which encountered bedrock and three other deep borings) is generally consistent with the form of the land surface: downward slopes from west to east in the central portion of the site and from north to south along the access road. Three generalized geologic cross sections of the site (Figures 3-3 through 3-5) present the stratigraphic sequence of deposits present across the study area.

The overburden material directly overlying bedrock is mapped in the area as glacial till, composed of an unsorted mixture of boulders, gravel, sand, silt and clay (Frimpter, 1972). Till typically blankets the bedrock surface in glaciated terrains. In the site investigation, a possible till deposit consisting of silt, fine-to-coarse sand, and gravel was encountered overlying bedrock at most locations. Boulders were encountered at several locations (MW-1D, MW-2D, MW-6D and MW-9S) at various depths, and were visible at the surface in the western portion of the site. The presence of boulders distributed in the matrix further indicates a till.

Overlying the till deposit in most areas of the site is a layer of light brown fine sand or fine sand and silt present at depths ranging from grade, at borings MW-9S and MW-3D, to 13.5 feet below grade at MW-7D. This may represent till reworked and redeposited by glacial meltwaters. The till and overlying sand and silt deposits vary in color across the site depending on their location. To the north, south and west, the sediments are light brown to brown in color suggesting an oxidizing depositional environment. Along the eastern edge of the site and in areas adjacent to wetlands, the sediments are grey in color, consistent with a reducing environment.

Overlying the silt, sand and gravel deposits along the eastern edge of the landfill near the wetlands is a layer of brown silt (MW-13S, MW-12S, MW-W1D, MW-11D, MW-W2D and MW-10D). A peat deposit is present over the silt layer at locations MW-11, MW-W2, TP-1, TP-2, TP-5, TP-6, TP-7, TP-8, TP-10, and TP-15. These deposits represent recent alluvium and wetland deposits.

Surficial deposits at boring and test pit locations outside of the boundary of the landfill consist of clean natural fill comprised of red-brown fine to medium sand, some silt, some rock fragments and organic debris (leaves, roots, etc.).

Surficial deposits at boring and test pit locations within the former landfill boundary generally consist of household refuse with occasional lenses of green-grey silt and fine sand. Data obtained from test pit excavations and boring logs indicated refuse at the site consisted generally of paper, plastic, glass and household waste. Metal was encountered at some boring and some test pit locations. The landfill surficial deposits vary in thickness from 2 feet along the eastern edge of the site (at boring location MW-W2D) to greater than 16.5 feet towards the western-central portion of the landfill (at test pit TP-12).

3.4 Hydrogeology

The two water-bearing geologic units encountered during drilling at the site are the unconsolidated overburden deposits and the consolidated sedimentary bedrock. All monitoring wells were completed in one of the three following horizons: shallow overburden deposits, deep overburden deposits, and shallow sedimentary bedrock.

The water table, deep overburden, and bedrock piezometric surface contour maps are provided as Figures 3-6 through 3-11. The water level elevations on those dates as well as a summary of all water level measurements (monitoring wells, piezometers, test pit piezometers and surface water staff gauges) during the program are included in Appendix F.

The overburden deposits are composed of materials ranging in size from peat to silt to sand and gravel. These grain size heterogeneities occur vertically, as seen by the general coarsening of materials proceeding stratigraphically downward at a location. Grain size also varies laterally across the site, but to a lesser degree. Although limited occurrences of fine-grained materials were noted, no geologic unit within the overburden deposits appears to serve as a confining or semi-confining layer over the entire site. In fact, at the MW-11 cluster, the pumping test of the deep well (MW-11D) at 4 gpm for approximately one hour induced a lowering of the water level of the adjacent shallow well (MW-11S) by 0.21 feet. This indicates a hydraulic connection between the shallow and deep portions of the overburden deposits.

Ground water flow direction in the overburden deposits closely corresponds with the general topography at the landfill area of the site. As shown in Figures 3-6 and 3-7, ground water in the upper portion of the overburden deposits flows eastward toward the landfill base and the wetland, which borders the landfilled area to the east. The hydraulic gradient decreases closer to the landfill base, where the shallow ground water discharges into the adjacent wetland. Using the water level data from the three deep unconsolidated wells installed as part of this program and the Wehran-installed wells MW-W1D and MW-W2D, ground water flow in the deeper portion of the unconsolidated unit also follows the general topographic trend to the east (Figures 3-8 and 3-9). Seasonal ground water level changes did not induce changes in ground water flow direction in the water table aquifer. Overburden saturated thickness varies from 23 feet at MW-2D and MW-1D to greater than 46 feet at MW-10D.

The bedrock is composed principally of sandstone. It is generally competent and consistent across the site. The rocks of the Valley and Ridge physiographic province are well-cemented (Frimpter, 1972), therefore ground water flow occurs principally along zones of secondary permeability such as fractures and joints. No geologic units were observed to isolate the bedrock from the overlying unconsolidated geologic materials. The direction of the ground water gradient in the bedrock aquifer is generally toward the northeast or east (Figures 3-10 and 3-11). It should be noted that the number of bedrock water level monitoring points is limited. As a result, the direction of the bedrock piezometric surface gradient is not well defined. Seasonal fluctuations in ground water levels observed during the course of this investigation did not appear to significantly alter the flow direction in the bedrock aquifer.

Downward vertical gradients were observed at most well cluster locations across the site (Table 3-1). This is especially true for clusters involving bedrock wells (e.g., MW-2S/2D). At two overburden clusters near the eastern wetland and stream, MW-10S/10D and MW-11S/11D, an upward vertical gradient was observed. The geological characteristic unique to this lower area of the landfill near the wetland was the presence of a gray silt layer. The largest downward gradients were observed at locations MW-6S/6D and MW-7S/7D located in the middle of the landfilled area.

Based on visual observations (e.g., orange-colored surface water emanating from landfill base) at its contact with the wetland and water level data from monitoring wells and nearby staff gages (Table 3-2), the shallow ground water at the site is discharging to the surface water bodies. An exception to this was noted along the access road, where the pond is recharging the ground water to the east (toward P-1 and MW-13). Here, the water level in the pond is higher than the water table level observed in the monitoring well MW-13 and piezometer P-1.

Hydraulic conductivities generally ranging from 0.4 to 64.6 ft/day were calculated from slug and pumping tests of the wells screened in the shallow overburden. Most calculated values were less than 10 ft/day. The wide range in calculated values is attributed to geologic and well construction/development variations, as well as the short-term nature of the tests which affect only a limited volume of aquifer near the wells. Generally, lower hydraulic conductivities (0.4 to 1.6 ft/day) were found in the shallow gray silt unit along the base of the landfill (MW-9S, MW-10S, MW-11S). Calculated hydraulic conductivities at other well locations at the base of the fill ranged up to 13 ft/day. Hydraulic conductivities of 0.7-5.1 ft/day were calculated for the deeper overburden deposits, except for MW-W2D which had an average value (pumping and recovery tests) of 123 ft/day. This may be attributed to a localized coarser-grained sand and gravel deposit at this location.

Bedrock hydraulic conductivities ranged from 0.1 to 3.3 ft/day. Generally, the hydraulic conductivity measured in bedrock wells increased in a northward direction across the site.

A summary of the testing method and hydraulic conductivity for each well is presented in Table 3-3.

Utilizing the calculated values for formation hydraulic conductivity, estimates of formation porosity and hydraulic gradient across the site, an estimate of ground water flow velocity can be made using the following equation:

$$\bar{V} = K\overset{1}{L}/n$$

where:

$\bar{V}$ = average linear ground water flow velocity in ft/day

K = calculated formation hydraulic conductivity in ft/day

$\overset{1}{L}$ = hydraulic gradient, based on ground water contour maps in ft/ft

n = estimated formation porosity in ft^3/ft^3

Utilizing an average overburden hydraulic conductivity range from the slug and pumping tests of 1.6 to 4.7 ft/day, a hydraulic gradient across the center of the landfill of 0.047 ft/ft (10/29/90 data), and an average porosity for sand and silt of 0.30 (Walton, 1983), the average linear ground water velocity across the site in the upper overburden deposits is 0.3-0.7 ft/day. Because of the wide range in calculated hydraulic conductivities, this is considered an order-of-magnitude estimate at best. Long-term, constant-rate pumping tests would need to be conducted to more precisely determine the aquifer parameters.

3.5 Flora and Fauna

3.5.1 Flora

A total of five ecological community types were identified on-site. They are old field, forested upland, forested wetland, stream and open water (pond). Particular vegetation, topography and hydrologic regimes attract species that are specialized or adapted for each particular ecological niche. Below is a brief description of each community type.

- Old Field

The old field is limited to the extent of the landfill located on the Hertel site. Its boundaries are the pond to the north and naturally occurring vegetation surrounding the field. The old field is dominated by opportunistic species which are indicative of disturbed and road side areas. Within the field, there are a few scattered trees and an area directly in the center where a dense cluster of shrubs are growing. The remainder of the field varies from barren to densely vegetated with herbaceous perennials.

- Forested Upland

The forested upland is diverse in ecotones or forest subtypes. It is the dominant community on-site. Most areas have a dense canopy, a mixed shrub layer and a greatly varied herbaceous layer or none at all. The topography varies greatly here from steep areas with rock outcroppings to flat areas.

- Forested Wetland

The forested wetland is located in a depressional basin in the south west of the site. The vegetation species in this community are more uniformly distributed than the previously mentioned communities, with the exception of the herbaceous layer, which is dictated by the topography. For example, in depressional pocket areas of standing water, tussock sedge were present, but in flatter soggy areas, sphagnum moss and various hydrophytic perennials and annuals were found.

- Stream

An intermittent stream runs though the site draining from the forested wetland north eastward across and off the site. Along its course, the stream varies from a very narrow ravine-like corridor, to a broad open area where the center course slows down and meanders around exposed bedrock and trees. It is in the quiescent area or where water overflows, that hydrophytic shrubs and herbaceous species were found.

- Pond

The ponded wetland area is in the northern section of the site. It contains floating, submergent and emergent vegetation. It also contains numerous tree stumps and snags. It is the only extensive open water area on-site.

Within this geographic region a large diversity of forest communities exists. In particular, the two principal communities which occur on site are beech-maple mesic forest with inclusions of Hemlock-northern hardwood forest as a subtype and red maple-hardwood swamp. These forested uplands exist in "rich, moist, well drained soils, usually acid soils and mid-elevation slopes on moist, well-drained sites at the margins of swamps (Rescke, 1990). The forested wetlands or red maple-hardwood swamp "occurs in poorly drained depressions throughout NY State - usually on inorganic soils".

In addition to the two natural communities classified on site, there is an additional community which is classified as a cultural community by the DEC. "The cultural subsystem includes communities that are either created and maintained by human activities, or are modified by human influences to such a degree that the physical information of the substrate, or the biological composition of the resident community is substantially different from the character of the substrate or community as it existed prior to human influence" (Rescke, 1990). Specifically this area is classified as landfill/dump. This classification is described as: "A man-made site that has been cleared or excavated, where garbage is disposed. The bulk of the material in the landfill or dump is organic and biodegradable, although some inorganic material (plastic, glass, metal, etc.) is usually present" (Rescke, 1990) and in the case of this landfill possibly even a larger diversity of materials exists.

An overall list of plant species identified on site is provided in Appendix H. The site is comprised of several plant communities including: beech-maple mesic forest with inclusions of hemlock-northern pine forest, old field and three areas containing hydrophytic plant communities. This also provides a list of dominant plant species found at each observation point and identifies each wetland area (A through C).

Thirteen plant species identified on the Hertel site are listed in the DEC 193.3, Protected Native Plants list (Table 3-5), pursuant to section 9-1503 of the Environmental Conservation Law. Eleven (11) are listed under clause "(d) exploitably vulnerable native plants likely to become threatened in the near future throughout all or a significant portion of their ranges within the state if casual factors continue unchecked." One (1) is listed under clause "(e) rare native plants that have from 20 to 35 extant sites or 3,000 to 5,000 individuals state." And one (1) is listed under clause "(b) endangered native plants in danger of extinction throughout all or a significant portion of their ranges within the state and requiring remedial action to prevent such extinction." No federally listed threatened or endangered species were found during field studies on the site. The dominant species list associated with the following communities, forested upland, forested wetland, and old field are summarized in Appendix H.

3.5.2 Fauna

The organisms collected were compared with published lists (USEPA, 1973 and Illinois Environmental Protection Agency, 1989) to determine their pollution tolerance classifications. The U.S. Environmental Protection Agency (1973) defines these classifications as follows:

-- Tolerant: Organisms frequently associated with gross organic contamination and are generally capable of thriving under anaerobic conditions.

-- Facultative: Organisms having a wide range of tolerance and frequently are associated with moderate levels of organic contamination.

-- Intolerant: Organisms that are not found associated with even moderate levels of organic contaminants and are generally intolerant of even moderate reductions in dissolved oxygen.

Macroinvertebrates

Appendix H presents summary data for the ecological surveys including taxa presence/absence, percent relative abundance, complete lists of species, etc.

The benthic macroinvertebrate survey resulted in the collection of 25 taxa at the Hertel site (Appendix H). The dominant organisms were insect larvae, molluscs, and oligochaetes. Of the 25 taxa, 22 are considered facultative and three are tolerant (Lumbriculidae-Oligochaete worm, Chironomous and Physella). No intolerant organisms were identified.

Inspection of Table 3-4 reveals that the number of macroinvertebrate taxa collected at each area of the stream (upstream, adjacent and downstream of the landfill) is equivalent (12, 13 and 13, respectively). The total number of individuals collected at each area (305, 186 and 250) is similar, particularly when it is taken into account that of the 305 individuals collected upstream, 179 were the gregarious midge larvae (Chironomous). As demonstrated in Section 6.7, species diversity and equitability is also similar at each area.

Birds

A total of 45 bird species have been identified on the Hertel site (Appendix H). None of these species has federally protected status; however, the red-shouldered hawk (Buteo lineatus) is listed under the New York State Environmental Conservation Law as a threatened species.

The observed species can be categorized in six major taxonomic groups; (Appendix H). The most common group observed was the passerines (perching birds). Thirty-two passerine species were identified which account for 71 percent of the total species recorded. Five of the observed species belong to the water-dependent taxonomic groups (i.e. wading birds, waterfowl and shorebirds). These species were only observed at the large ponded area at the site's northern boundary.

The number of species observed at each community type is similar (Appendix H). It is noted that five species were only observed at the pond (great blue heron, Canada goose, mallard, spotted sandpiper and common bobwhite), the mourning dove and Northern Oriole were only observed at the old field, the veery bluejay and black-capped chickadee were only observed at the forested wetland, and the rose-breasted grosbeak and field sparrow were only observed at the forested upland.

The most numerous type of bird observed on the Hertel site was the red-winged blackbird (n=87) followed by the American crow (53), American robin (31), tree swallow (23) and bluejay (20). These five passerines account for approximately 68 percent of the total individuals recorded. This data is summarized in Appendix H.

Fish

The scope of work also called for the surveying of fish in the on-site stream. However, due to its narrow, shallow and very intermittent nature, it was determined by field scientists that no fish existed in the stream. Due to the numerous snags and emergent vegetation present, seining for fish in the ponded wetland area on-site was impossible. In lieu of this, small fish traps were set in the pond. No fish were collected, nor were any observed, although numerous aquatic newts were collected in the traps.

Mammals

A total of ten mammal species have been identified on the Hertel site (Appendix H). None of these species has a federal or state protected status. The species observed are characteristically found in Northeastern old fields and deciduous forests.

Investigation of the food requirements of each of the ten species identified reveals that there is an equitable representation of trophic levels. There are three primary consumers/herbivores (eastern cottontail, gray squirrel, and white-tailed deer), four omnivores (eastern chipmunk white-footed mouse, raccoon and striped skunk) and three carnivores (mink, short-tailed shrew and cat).

The distribution of captured mammal species among each of the three community types is shown in Appendix H. Three species were captured in the old-field (eastern cottontail, striped skunk, and domestic cat), two species were captured in the forested wetland (white-footed mouse and raccoon) and only the raccoon was trapped in the forested upland. The most numerous mammal collected during the trapping survey was the white-footed mouse, with ten individuals collected at the forested wetland.

Herpetofauna

A total of nine herpetilian species have been identified on the Hertel site, comprised of five amphibian and four reptilian species (Appendix H). The amphibians include one salamander (Eastern newt), one toad (American toad) and three frog species (green frog, wood frog and spring peeper). The reptiles observed include two turtle species (snapping turtle and painted turtle) and two snake species (racer and common garter snake). None of the species observed have a federal or state protected status. No herpetilian species were captured in the pit traps. Only insects and crayfish were collected in those traps. All herpetofaunal data collected were based on intensive searches and subsequent observations of herpetiles on-site.

3.6 Wetlands

3.6.1 Soils

The parcel is identified on atlas sheet 123 of the Ulster County Soil Survey. Five soil mapping units are listed by the Soil Conservation Service on this site. These units are Canandaigua silt loam, tilt substrate (Cd);

Bath-Nassau complex, 8-25% slope (BnC); Bath-Nassau Rock outcrop complex, hilly (BOD); Lyons-Atherton complex, very stony (LY); and Volusia, very stony soils, gently sloping (VSB). Canandaigua loam, Lyons loam and Volusia loam are considered hydric soils (USDA, 1988).

Because the scale of the Ulster County soil survey map is so large, it does not provide the detail needed in a wetlands delineation. The precise location of hydric and non-hydric soils, and the location of the wetland/upland boundary were determined in the field in order to provide the level of precision necessary for a site specific wetlands delineation.

In most instances, the field investigation supported the soil mapping units identified on the Ulster County Soil Survey. Three major areas of hydric soils were identified on this parcel. In most instances the hydric/non-hydric zones were clearly defined.

3.6.2 Hydrology

Wetland hydrology was observed on the parcel during the field investigations. Table 3-6 summarizes the hydrological findings. On the parcel, three areas possessed hydrologic characteristics typical of wetlands (see Figure 3-12). Wetlands "A" and "B" are associated with the on-site stream while wetland "C" is a result of the access road impeding surface waters and creating a large ponded area. Matted leaves, hummocky soil, inundation and saturated soils within a depth of 12 inches provided clear evidence of wetland conditions. When either inundation or shallow depth to saturated soils was observed, it was interpreted as direct evidence of wetland hydrology. Generally this occurred in most of the wetland areas.

3.6.3 Vegetation

Vegetative species were identified at each observation point (see Table F.3 in Appendix H) and were then determined to be wetland or upland plants according to the Fish and Wildlife Natural List of Plant Species that occur in wetlands: Northeast (Region 1) (May 1988).

Wetlands "A" and "B" are plant communities dominated by red maple (Acer rubrum) swamp and forested wetlands. The forested wetland are dominated by wetland tree species such as slippery elm (Ulmus rubra), pin oak (Quercus palustris) and swamp white oak (Quercus bicolor) in the over story. Shrub species include red-osier dogwood (Cornus stolonifera), spice bush (Lindera benzoin) and highbash blueberry (Vaccinium corymbosum) in the understory and the herbaceous layer was dominated by a variety of species including sensitive fern (Onoclea sensibilis), tussock sedge (Carex stricta), purple loosestrife (Lythrum salicaria) and bog clubmoss (Lycopodium inundatum).

Wetland "C" is an emergent wetland along the perimeter of an open pond dominated by herbaceous species including cattail (Typha sp.), purple loosestrife (Lythrum salicaria) and various hydrophytic sedges (Carex sp.).

3.6.4 Conclusions

Using the COE (three parameter) approach and the DEC technical statement, wetlands were identified and mapped on the parcel. The parcel contains a total of approximately 10.3 acres of freshwater (COE) wetlands. Wetland area "A" is approximately 2.2 acres. Wetland area "B" extends beyond the property boundary; 2.3 acres are located within the site boundary. Both wetlands are Palustrine Broad-leaved Deciduous Forested (PFO1) as classified by the U.S. Fish and Wildlife Service (Cowardin, et. al.; 1979). Wetland area "C" is the largest wetland on site containing 5.8 acres of Palustrine Emergent (PEM) wetland. Wetland and observation point locations are shown in Figure 3-12; data forms for wetland delineations are presented in Appendix H. Figure 4 in Appendix H also shows the location of NYS delineated wetlands at the Hertel site.

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4.0 NATURE AND EXTENT OF CONTAMINATION

Presented in the following sections are the sample analytical results and discussions of their significance relative to site background sample concentrations and published ranges of background or typical concentration, where applicable. The compounds detected are summarized in tables associated with each of the subsections which follow. These tables present "hits only"; compounds and analytes which were not detected are not included in these tables. Following the presentation of all the sample analytical results, is a discussion on comparisons of analyte concentrations observed in the sampled media to available standards and guidance values. The sampling locations are shown on the Figures and on Plate 1.

The samples were analyzed in accordance with USEPA Contract Laboratory Program (CLP) protocol for TCL organic and TAL inorganic analytes. The organic data from samples collected 11/89 through 2/90 were validated in accordance with USEPA Region II Standard Operating Procedures (SOP) No. HW-6, Revision 6 (March 1989). Organic data from samples collected 6/90 and 8/90 were validated in accordance with SOP No. HW-6, Revision 7 (March 1990). The Inorganic data for samples analyzed under CLP Statement of Work for Inorganic Analysis 787 were validated in accordance with USEPA Region II SOP HW-2, Revision 9 (December 1987). Inorganic data for samples analyzed under the 788 SOW were validated in accordance with SOP HW-2, Revision 10.

Non TCL/TAL analyses were analyzed in accordance with EPA approved methods. Laboratories were solicited through CLP Special Analytical Services (SAS) request. Data were evaluated for adherance to the specified methods and whenever possible, organic and/or inorganic data validation protocols were adapted for validation of SAS data.

It should be noted that there was one significant change in the data validation protocol from the organic revision 6 to revision 7 which has affected some of the data. Revision 6 called for rejection of all data in which the percent moisture of a soil sample exceeded 50%. However, revision 7 changed this action to qualification of all analytes as estimated. Therefore, some data from 11/89 through 2/90 rounds of sampling has been rejected for this criterion, while data from 6/90 and 8/90 has been qualified as estimated.

Data qualifiers used on the summary tables (Tables 4-2 to 4-18) and corresponding definitions are as follows:

"U"	The analyte was analyzed for but not detected.
"UJ"	The analyte was analyzed for but not detected at an estimated detection limit due to quality control deficiencies.
"J"	The reported concentration is considered estimated due to quality control deficiencies. This qualification is applied in many cases, such as calibration deficiencies, holding time exceedances, non-compliant internal standard area counts, non-compliant surrogate recoveries, etc.
"R"	The data point is rejected due to serious quality control problems. Frequently data gets rejected for multiple quality control exceedances, such as two or more non-compliant surrogates recoveries in one sample, gross exceedance of holding time, poor instrument response to a particular analyte, etc.

Infrequently used qualifiers are explained on the specific tables in which they appear.

4.1 Fill and Adjacent Soils

As noted in Section 3-1, the landfilled area covers about 13 acres. Seven disposal areas were identified in the main fill area prior to the field investigation, and an eighth disposal area was tentatively identified south of the main fill area. Surface soil samples and/or subsurface soil samples were collected from the waste disposal areas and from other areas of the fill to characterize contaminants in the fill and to determine if the wastes are RCRA characteristic wastes (hazardous by RCRA definition). Subsurface soil samples were collected at select boring locations to provide additional background data for subsurface soils. The concentrations reported for the surface soil samples and test pit samples from the waste disposal areas were compared to samples collected from test borings drilled in on-site areas believed to be uncontaminated and to published ranges of values for uncontaminated soils (Table 4-1). (Published values were used for comparison when on-site background was not well established due to sample rejections.) The positive analytical results for the chemical analyses and the RCRA characterization are summarized on Tables 4-2 through 4-6 (surface soil and subsurface soil samples from test pits). The analytical results for soil boring background samples are presented on Table 4-7 for comparison. The analytical results and observations are discussed below by disposal area.

Disposal Area #1: Engine Block and Oil Waste Materials

Surface soil sample SS-01 was collected from Disposal Area #1 in an area of metal debris. Sample SS-02 was collected across the access road to the west of Disposal Area #1. Pesticides, PCBs and volatile organic compounds (VOCs) were not detected. The only base/neutral/acid extractable (BNA) compound detected was bis(2-ethylhexyl)phthalate (SS-01), which was also detected in one background sample. A subsurface soil sample was collected from test pit TP-8 near the SS-01 location. Pesticides, PCBs, and BNAs were not detected. A trace concentration (5 ppb) of toluene was detected.

In contrast to the organic compounds discussed above, soils normally contain numerous and varied concentrations of inorganic analytes. Copper, chromium, mercury and sodium were present in sample SS-01 at concentrations four to eighteen times higher than the highest on-site background values. However, for three of these elements (copper, mercury, and sodium) the on-site background range was not well established due to rejection of some analytical results in the QA/QC data validation. Copper, mercury and cadmium were present at concentrations above a range of typical soil concentrations published by the EPA (EPA, 1983) in one or both samples. Inorganic concentrations in sample TP-8 were within two times the background values reported (or detection limits reported for the background samples).

A sample from TP-8 was also submitted for EP toxicity testing which is used to determine if a waste is hazardous by characteristic. All results were less than the EP toxicity limit for determination of hazardous waste.

Disposal Area #2: Trailer Wreckage and Scattered Drums

Surface soil sample SS-03 was collected in Disposal Area #2 in an area of dense undergrowth where metal debris and assorted trash were evident. Sample SS-04 was collected on the approximate southern perimeter of this disposal area.

Volatile organic compounds were not detected in either sample. One pesticide (4,4'-DDT) and several polynuclear aromatic hydrocarbons (a subgroup of the BNAs) were identified in sample SS-03, all at estimated concentrations. No BNAs were detected in the SS-04 sample; however, it should be noted that detection limits reported by the laboratory were elevated due to sample dilution. Pesticide/PCB results for sample SS-04 were not usable (rejected during data validation).

Several inorganic analytes were detected at concentrations greater than two times the observed natural background: cadmium, chromium, copper, iron, lead, mercury, and nickel. Cadmium, chromium, copper, lead and mercury were all present at concentrations above the published typical element concentrations in soils (EPA, 1983). In SS-04, cadmium, chromium and copper were detected at 7 to 100 times the observed on-site background concentrations.

Disposal Area #3: Oil-Stained Area and Possible Sanitary Waste

Five soil samples were collected in the vicinity of Disposal Area #3: surface soil samples SS-05, SS-07 and SS-09 and test pit soil samples TP-3 and TP-10.

The surface soil samples were collected in an oil-stained area (SS-05) and at the perimeter of this disposal area (SS-07 and SS-09). Volatile organic compounds, pesticides and PCBs were not detected in these three surface soil samples. Some BNA compounds, primarily polynuclear aromatic hydrocarbons (PAH), were detected in sample SS-07. It is important to note that the laboratory detection limits for the BNA compounds in sample SS-05 were significantly raised and may account for the absence of BNAs in sample results for SS-05. PAH compounds are commonly present in an oily sample.

Several inorganic analytes were present in one or more surface soil samples at concentrations greater than two times the observed background concentrations. They include: cadmium, chromium, copper, iron, lead, mercury, nickel and sodium. The most significantly elevated elements were cadmium, chromium and copper present in sample SS-05 at concentrations 12 to 180 times those reported in the background samples.

Test pit TP-3, located near surface soil sample SS-05, and test pit TP-10, located at the northern fringe of Disposal Area #3, showed assorted fill/waste materials to depths of greater than 12 feet and 5 feet, respectively. The sample TP-3 showed the presence of several aromatic hydrocarbons and polynuclear aromatic hydrocarbons which could be associated with petroleum-derived materials. Also reported was the presence of one PCB Arochlor (1248). PCBs are sometimes associated with waste oils. Pesticides were not detected. The sample of waste from TP-10 did not indicate the presence of pesticides, volatile organics or BNAs. However, one PCB, Aroclor 1260, was reported at an estimated concentration. No inorganic analytes were reported at concentrations greater than two times the observed background concentrations. However, cyanide was reported at near the detection limit in sample TP-10.

One sample was collected for EP toxicity analysis from TP-10. The extract contained several RCRA metals, all at or below the EP toxicity limit for determination of hazardous characteristics.

Disposal Area #4: Farm Equipment Debris

The area of farm equipment debris is located along the northwestern edge of the fill area. Surficial debris in this area consisted of assorted metal and landfill waste. Two surface soil samples (SS-23 and SS-24) and two subsurface soil samples from test pits (TP-11 and TP-13) were collected.

Pesticides, PCBs and volatile organics were not detected in the surface soil samples. The only BNA detected was bis(2-ethylhexyl)phthalate, which was also detected in one of the background samples. The inorganic analytes were at concentrations less than two times the observed background concentrations.

Three test pits were excavated in the vicinity of Disposal Area #4: TP-11, TP-13 and TP-24. Samples were collected from soils beneath the fill in test pits TP-11 and TP-13.

Pesticides and PCBs were not detected in either subsurface soil sample. Aromatic hydrocarbons, typical of petroleum products, were detected in both samples. Some polynuclear aromatic hydrocarbons were also detected in sample TP-11. The inorganic analytes were detected at concentrations less than two times the observed background concentrations.

Subsurface samples were collected for EP toxicity testing at sample locations TP-11 and TP-13. The results did not exceed the EP toxicity limits for determining hazardous by characteristic. The extract also showed the presence of the herbicide 2,4-D, although at a concentration less than the EP toxicity limit.

Disposal Area #5: Printing Waste

The area of printing waste was not easily discernable. Two surface soil samples (SS-20 and SS-21) were collected in the location of Disposal Area #5. Sample SS-20 was collected near what appeared to be carbon paper shreds. Neither of the samples contained detectable quantities of volatile organics, pesticides or PCBs. A trace concentration (43 ppb) of diethylphthalate was detected in sample SS-20. Other than this one phthalate finding, no BNAs were detected. All inorganic elements analyzed for were present at concentrations less than two times observed background except copper and sodium. In both cases, background was not well established due to rejected results, and concentrations reported for both elements were within published background ranges (EPA, 1983; USGS, 1984).

A representative sample of printing waste was obtained from the location shown on Figure 2-5.

Disposal Area #6: Fibrous Material Piles

Three surface and one subsurface soil samples were collected from Disposal Area #6. Surface samples SS-10 and SS-12, collected from the center and along the western perimeter of area 6, contained no detectable concentrations of volatile organics, pesticides, PCBs or BNAs. Organic analyses for sample SS-11, collected at the northern perimeter of area 6, were rejected. Inorganic analyses indicated that cadmium, chromium, copper, mercury, nickel and sodium were present at concentrations greater than two times the observed background at one or more locations. The most notable of these are chromium and mercury which were present at 13 and 14 times observed background concentrations, respectively.

Three test pits were excavated in Disposal Area #6. All three test pits (TP-4, TP-5, TP-6) encountered landfill material consisting of assorted garbage. The fill depth in this area ranged from 10 to 14.5 feet in thickness. The one sample collected from TP-4 for analysis was collected from below the fill. Pesticides and PCBs were not detected. Low concentrations of aromatic hydrocarbons were detected along with several BNA compounds (PAHs and phthalate esters). Inorganics, with the exception of lead, were within two times the observed site background concentrations. The lead concentration was less than three times the observed site background and was within the published range of concentration in natural soils (EPA, 1983).

One sample was collected from waste encountered in test pit TP-4 for RCRA waste characterization. The sample was determined to be not hazardous by EP toxicity methods.

Disposal Area #7: Paint Waste/Municipal Landfill

Disposal Area #7 extends approximately north-south and forms the approximate eastern edge of the main body of the landfill. Four surface soil samples (SS-13, SS-16, SS-18 and SS-19) and three subsurface soil samples (TP-1, TP-2 and TP-7) were collected. Additional test pits were excavated in this area (TP-5, TP-6, TP-9, TP-15) and in adjacent Disposal Area #6 to characterize the wastes.

The surface soil samples were collected near the apparent center of Disposal Area #7 near the paint waste (SS-13) and at the perimeter of this area (SS-16, SS-18 and SS-19). Tetrachloroethene was identified in sample SS-13 at a trace concentration (8 ppb). No volatile organic compounds or PCBs were detected in samples SS-16, SS-18 or SS-19. The pesticide 4,4'-DDT was present in three of the four samples and was reported by the laboratory at estimated concentrations. Polynuclear aromatic hydrocarbons were also reported in three of the four samples. In both cases, sample SS-16 was the exception which showed no detectable compounds.

Inorganic analyses of surface soil samples showed the presence of several elements (barium, cadmium, selenium, chromium, copper, lead, magnesium, mercury and zinc) at greater than two times the observed background concentrations at one or more locations. Most notable was the lead concentration in sample SS-13 (and SS-14, the duplicate of SS-13) which was almost 20 times those observed in the on-site background.

The subsurface soil samples were collected along the length of Disposal Area #7 from beneath the paint waste (sample TP-1) and from assorted waste material (samples TP-2 and TP-7). All three samples showed the presence of aromatic hydrocarbons, most apparent in samples TP-1 and TP-7. No pesticides were detected, but one PCB (Aroclor 1248) was detected in one sample (TP-2). BNA compounds were detected in all three samples. Ignoring the results for the phthalate esters, naphthalenes were detected in the TP-1 and TP-7 samples. Several polynuclear aromatic hydrocarbons (PAHs) were detected in sample TP-2. PAHs are present in heavy petroleum products and PCBs are sometimes associated with waste oils, so appearance of the PAHs and the PCB in sample TP-2 may be associated. The inorganic analytes were not reported present at concentrations greater than two times background.

One sample was collected from test pit TP-1 for waste characterization. The waste sample analytical results did not exceed the EP toxicity limits for determination of hazardous by characteristic.

Disposal Area #8: Possible Rubber Waste

Two surface soil samples, SS-26 and SS-27, were collected in the vicinity of the area tentatively identified as receiving rubber waste in the southern portion of the site. No surficial evidence of waste disposal in this area was observed. Sample SS-27 showed the presence of polynuclear aromatic hydrocarbons and pesticides. No volatile organics or PCBs were detected. (The organic analytical results for sample SS-26 were rejected.) None of the inorganic analytes exceeded two times the observed background concentrations.

Other Soil and Fill Samples

Two additional surface soil samples and two additional subsurface soil samples, not specifically associated with previously identified waste disposal areas, were collected.

Surface soil sample SS-22 was collected on the west side of the site beyond the apparent fill area. Sample SS-22 was intended to provide surface soil background data. Sample SS-22 showed the presence of pesticides, trace concentrations of two PAHs, and the highest arsenic concentration reported from an on-site soil sample. The presence of PAHs and pesticides in samples SS-22 and SS-27, both removed from obvious areas of filling, may be indicative of other site use unrelated to waste disposal.

Surface sample SS-25 was collected in an area east of an unpaved road across the site where runoff from the southern area of the site (and possibly Disposal Area #8) might have accumulated. Organic analyses for this sample were rejected. Inorganic analyses indicated the presence of lead at less than three times the reported on-site background concentrations.

Test pits TP-12 and TP-16 were excavated in the southwestern portion of the main body of fill. Both test pits encountered assorted garbage, including metal. No pesticides or PCBs were detected. Both samples indicated the presence of aromatic hydrocarbons, although the TP-16 samples showed only a trace concentration. Base/neutral/acid extractable compounds were not detected in either sample, although detection limits were high for both samples due to sample dilution at the laboratory. Inorganic analytes were within two times the observed on-site background concentrations. Cyanide was reported in the TP-12 sample at 10.4 ppb.

EP toxicity analysis of wastes from test pits TP-12 and TP-16 were within EP toxicity limits.

4.2 Ground Water

Two rounds of ground water sampling of on-site monitoring wells were conducted to assess the impact to ground water from the landfilling activities and to assess if contamination, if present, is migrating or potentially migrating off-site through any of the monitored ground water flow regimes. Positive ground water sample analytical results (hit tables) are summarized in Tables 4-8 through 4-11. Ground water at the site was also evaluated by collecting samples from test pits at selected areas of the landfill. These data are summarized in Tables 4-12 and 4-13. In addition, one round of ground water sampling was conducted from local private water-supply wells to assess if any effect has occurred to these nearby water-supply wells. Private well sampling results are summarized in Table 4-14.

The principal volatile organic compounds (VOCs) detected in the ground water sampling at monitoring wells and test pits were aromatic VOCs; namely benzene, toluene, chlorobenzene, ethylbenzene and xylenes. The distribution of these compounds is primarily in the central to eastern landfill area with the highest concentrations in the central landfill area around MW-6S and MW-7S, at the edges of Disposal Areas #3 and #6. Observed aromatic VOC concentrations decreased to the east, downgradient of MW-6S and MW-7S, as detected in test pits TP-2, TP-7 and TP-15. These aromatic VOCs were also consistently detected at MW-13S, removed from the landfill proper area but in the reported Disposal Area #1. Low concentrations (less than 10 ppb total) of aromatic VOCs were detected only during the June ground water sampling event at monitoring wells MW-2S and MW-8S.

Various halogenated and other VOCs were detected at low concentrations across the site in the June ground water sampling event. There was no apparent pattern or consistency in the observed concentrations or their distribution. Most of the reported concentrations were estimated values at or below the detection limits. Acetone was detected at low concentrations in five wells in the August ground water sampling event. Again, no pattern was discernable in the distribution of the acetone concentrations, nor was there any consistency observed between the two ground water sampling events. Acetone was used in the eight-step decontamination of field sampling equipment and is a common laboratory contaminant. Therefore, its detection here is not considered a positive indication of ground water contamination.

The distribution of detected base/neutral/acid extractable (BNA) compounds closely corresponded to the distribution of VOCs discussed above. Monitoring wells MW-6S and MW-7S and test pits TP-2, TP-7 and TP-15 were the primary locations where BNAs were detected. In addition, BNAs were detected in the August ground water sampling at monitoring wells MW-6D, MW-8S and MW-13S. The presence of these BNA compounds at MW-6D is significant because it is the only bedrock well location where there was a detection of any Target Compound List (TCL) organic compound.

During the August ground water sampling, an oily layer was noted at monitoring well MW-6S. Subsequent water level monitoring of this well in October determined an approximate floating product thickness of two feet in the well. The true thickness of free-phase floating product does not correspond with the measured apparent thickness in a well (Hall et al, 1984). Apparent thickness is generally greater than the true thickness due to the effect of the capillary fringe. The ephemeral nature of the product at MW-6S may be caused by water level fluctuations, as described by Kemblowski and Chiang, 1990. However, it is likely that the adjacent oil-stained surface soils and the floating product in MW-6S are interrelated.

The only pesticide or PCB detected in ground water was Arochlor 1248 detected in a ground water sample in contact with waste in test pit TP-2. PCB Arochlor 1248 was also detected in the soil sample collected from test pit TP-2.

Numerous metals were detected in the ground water. As shown in Tables 4-9 and 4-11, more total metals (unfiltered samples) were detected than dissolved metals (filtered samples). In addition, the total metals concentrations were generally significantly higher (two to ten times) the dissolved metals concentrations. Given the siltiness of many of the ground water samples, it is expected that the unfiltered samples would contain more metals and at higher concentrations. The filtered samples are believed to provide a more accurate representation of the ground water quality.

The refuse material at a landfill is subject to leaching by percolating waters derived from precipitation or by contact with ground water. The resulting liquid is called leachate. Leachate generation and quality are influenced by a number of processes, including: pH reduction by dissolution of carbon dioxide and organic acids, mineral dissolution (which is enhanced by a lowered pH), depletion of dissolved oxygen concentrations by aerobic organisms, depletion of biodegradable organic compounds and creation of anaerobic conditions, and reduction/oxidation potential, which controls the solubility of iron and manganese oxides. Concentrations of leachate parameters will evolve over time, as the chemical reactions and processes in contaminated ground water do not remain constant over time.

Constituents such as chloride, bicarbonate, sodium, calcium, ammonia, manganese, iron, magnesium and aluminum are typically characterized as leachate indicator parameters (Lu et al, 1985). These compounds are also mobile. Tracking these compounds in ground water can provide an understanding of the path of ground water migration at and away from the site. Comparing the range and distribution of concentrations for the leachate indicator parameters at the site, it can be determined that wells, MW-1S, MW-2S, MW-3S and MW-W3S, which are removed from the landfilled area, represent background levels for most of these compounds. Chlorides, bicarbonate, calcium, magnesium, manganese, sodium, iron, total kjeldahl nitrogen (TKN) and ammonia are generally present at concentrations much lower than the other shallow wells. Wells located in the landfill proper, MW-6S and MW-7S, consistently have higher, or the highest concentrations for these compounds. Shallow wells at the toe of the landfill (MW-10S, MW-W2S, MW-11S, MW-W1S and MW-12S) generally have the next highest concentrations. Wells adjacent to the landfilling activities (MW-8S, MW-9S, MW-13S and P-2) had levels characteristic of ground water influenced by landfill leachate, but less than the wells closer to the landfill.

Significant are the values for the leachate indicator parameters observed in the deeper wells, both overburden and bedrock. Background values for the deeper overburden were observed in well MW-1D and MW-W3S. At all four deep overburden wells (MW-W1D, MW-W2D, MW-10D and MW-11D) at the toe of the landfill, concentrations of some leachate indicator parameters were similar to those observed in the shallow wells in the landfill. Three of the four bedrock wells (MW-2D, MW-3D and MW-7D) generally had background concentrations for these leachate indicator parameters. However at MW-6D, concentrations for these parameters were consistently above background, and in fact, were similar to those values observed in the shallow landfill wells.

No TCL organic compounds were detected in any of the private wells sampled. The range of inorganic compounds and leachate parameters detected in the off-site private wells was generally comparable to the values detected in the background bedrock wells at the site.

4.3 Surface Water and Sediment

Surface water and sediment samples were collected to determine if the site is impacting surface water or sediment quality and if components of the on-site wastes are being transported off site by movement of surface water and erosion and redeposition of sediment. Surface water sampling analytical results are summarized in Tables 4-16 and 4-17. Sediment sampling analytical results are summarized in Tables 4-18 and 4-19.

4.3.1 Surface Water

Surface water could be impacted by the landfill from direct discharge of landfill runoff or seepage from the landfill or ground water discharge to the surface water. For purposes of comparison, several surface water samples were collected to establish and document background surface water quality. These samples include SW-1 and SW-3, collected from a wetland area southwest of the site and located just west of a topographic divide which separates them from the site; and samples SW-4 and SW-5, collected from the southern most on-site wetland which is the source area for the stream which flows on the east side of the landfill. Although sample locations SW-4 and SW-5 are background to the main body of fill, they may be affected by Disposal Area #8, if it is present. The background samples indicate the absence of detectable concentrations of pesticides, PCBs and base/neutral/acid extractable compounds. Trace concentrations of carbon disulfide and acetone were detected, each in one sample. Detectable concentrations of several elements and leachate indicator parameters were also reported.

The sample results and comparisons to observed on-site background concentrations, are presented below in the following groups: seep samples from the eastern toe of the fill; surface water samples from the stream along the eastern end of the site; surface water from the wetlands north of the main body of fill, and; surface water downgradient of the site.

Leachate Seep Samples

Leachate seep samples LS-21 through LS-25 were collected along the toe of the landfill, generally where staining and/or a sheen was present. Several of the seep samples showed trace concentrations of aromatic hydrocarbons, and one location (LS-21) showed a trace of 1,1-dichloroethane and chloroethane. The quality of the BNA compound scan was limited due to several QA/QC rejections. Trace concentrations of phenols, naphthalene and/or PAH compounds were detected in three of the five samples.

Many of the inorganic analytes and landfill leachate indicator parameters were present at elevated concentrations in seep samples. Aluminum, calcium, iron, magnesium, potassium, sodium, zinc, chlorides, barium, cadmium, lead, manganese and cyanide were elevated in some samples. The seep samples showed significant increases (8 to 648 times) in concentrations of these analytes over background surface water samples. In addition, biological oxygen demand (BOD), chemical oxygen demand (COD), total kjeldahl nitrogen (TKN), total dissolved solids (TDS), bicarbonate (as $CaCO_3$), nitrite/nitrate nitrogen, ammonia nitrogen, phosphate (as phosphorous), total organic carbon (TOC) and total organic halogens (TOX) were also present in several seep samples at concentrations well above background (4 to 240 times).

On-Site Stream Samples

Samples SW-6 through SW-9 were collected from the stream which flows south to north along the toe of the eastern edge of the fill. The sample numbers increase in the downstream direction, as shown on the plate.

Pesticides, PCBs and BNA compounds were not detected. Although a trace concentration (8 ppb) of acetone was detected in sample SW-6, it was not significantly above the concentration detected at upstream location SW-4. No other VOCs were detected.

It is apparent from the data that many of the inorganic analytes and other components found in the seep samples are affecting the surface water quality in the vicinity of the landfill. Barium, calcium, iron, magnesium, manganese, potassium, sodium, chlorides, bicarbonates, ammonia nitrogen and dissolved solids were present at concentrations from approximately 2 to 35 times above background in samples SW-7 through SW-9.

Samples from Wetland C - North of the Fill

Samples SW-10 through SW-15 were collected from the wetlands north of the main body of fill. Sample SW-2 was collected from a stained area between the fill and the wetland.

Sample SW-2, similar to the seep samples along the east side of the fill, showed a trace concentration of one aromatic hydrocarbon and several BNA compounds (a large portion of the BNA results were rejected). No pesticides or PCBs were detected. The sample contained several inorganic analytes and typical landfill leachate constituents at concentrations well above background.

Samples SW-10 through SW-14 were collected from the pond/wetland area north of the fill and west of the road. The southern portion of this pond/wetland abuts the fill area. No BNA compounds, pesticides or PCBs were detected. Volatile organic compounds (acetone and methylene chloride) were each detected at one location. Magnesium, manganese and sodium were elevated from 2 to 100 times background. In addition, aluminum, arsenic, barium, calcium, iron and potassium were elevated from 2 to 6 times background in sample SW-10. Several leachate indicator parameters, primarily chlorides and bicarbonates, were elevated in all five samples. BOD and TDS were elevated in several samples.

Sample SW-15 from east of the road showed no detectable concentrations of VOCs, BNA compounds, PCBs or pesticides. Several inorganic analytes and leachate indicator parameters including aluminum, calcium, iron, manganese, sodium, zinc, chlorides, bicarbonate, COD, TDS and phosphate were detected at concentrations elevated above background.

Surface Water Downgradient of the Site

Samples SW-16 through SW-20 were collected from the stream which drains the site to the east. No volatile organic compounds, pesticides, PCBs or BNA compounds were detected. Several inorganic analytes and leachate indicator parameters (calcium, magnesium, manganese, potassium, sodium, chlorides, TDS or bicarbonate) were elevated above background in most samples, although the apparent effects generally decreased with distance from the landfill.

4.3.2 Sediment

Sediment samples were collected from the wetlands, stream and seep location on-site and stream locations downgradient of the site. A total of 38 sediment samples was collected. Background sediment samples were obtained from locations SED-1 and SED-3 (0-6 inches), SED-29 (6-12 inches), and SED-30 (12-18 inches). The results are discussed below and grouped as follows: sediment samples from seep locations along the eastern toe of the fill; sediment samples from the stream along the eastern side of the site; sediment samples from the wetlands north of the main body of fill; and sediment from stream locations downgradient of the site. Sediment sample locations correlate to the surface water sample locations. At selected locations additional sediment samples were collected below the surficial samples, as noted.

Sediment from Seep Locations

Samples SED-21 through SED-25 were collected along the eastern toe of the landfill. Sample SED-21 was collected adjacent to the southern end of the landfill. No volatile organic compounds, pesticides or PCBs were detected. Low concentrations of base/neutral/acid extractable (BNA) compounds were detected. With the exception of cadmium, the inorganic analytes were within two times the range observed in the background sediment samples. The cadmium concentration was ten times the sediment background concentration and four to six times the background concentration in surface soil and subsurface soil.

Samples SED-22, SED-24 and SED-25 are located adjacent to the main body of the fill. No pesticides or PCBs were detected. Aromatic hydrocarbons were detected in all three samples. Methylene chloride was also reported in a sample collected from near the paint waste area (SED-25). Trace concentrations of aromatic hydrocarbons were also detected in the surface water samples from these locations. Base/neutral/acid extractable compounds were present in all three samples, with the greatest variety of compounds and highest concentrations in sample SED-24. Observed concentrations of cadmium and calcium were four to ten times the observed background sediment concentrations. Beryllium was present at concentrations of two times observed sediment background. Chromium and copper were both slightly elevated (greater than two times background) in sample SED-24.

Sediment sample SED-23 was collected adjacent to Disposal Area #1. The results were similar to the other seep locations, showing apparently elevated beryllium, copper, cadmium and calcium as compared with background. Positive base/neutral/acid extractable results included bis(2-ethyhexyl) phthalates and naphthalene. No aromatic hydrocarbons were detected, although one volatile compound, carbon disulfide, was detected. No pesticides or PCBs were detected.

Sediment sample SED-26 was collected from a small ravine along the southern end of the main body of fill where staining was evident. No VOCs, pesticides or PCBs were detected. The only BNA compound detected was bis(2-ethylhexyl)phthalate. The only inorganic analyte elevated significantly over background was cadmium (3 times background).

On-Site Stream Sediment

Samples SED-5 through SED-9, SED-37 and SED-38 were collected from the stream which flows from the southern wetland area north along the eastern boundary of the site. SED-4 was collected from the wetlands in the southern portion of the site. The two locations from which samples SED-4, SED-5, SED-37 and SED-38 were collected are possibly located downgradient of Disposal Area #8, the area tentatively identified as receiving rubber waste. These locations are upgradient of the main disposal area and thereby provide additional background data to the stream locations near the main disposal area.

Samples SED-4 and SED-5 showed no detectable concentrations of BNAs, pesticides or PCBs. It should be noted that detection limits for all three analytes were elevated. Trace concentrations of two VOCs (carbon disulfide and toluene) were detected in SED-5. Carbon disulfide was also detected in samples SED-37 and SED-38. The reported concentrations were at or near the detection limits reported for the background samples. The only inorganic analyte reported at a concentration greater than two times sediment background concentration was calcium. The calcium concentration was also as much as three times the background soil sample results.

Samples SED-6 through SED-9 were collected from the stream along the eastern side of the main fill area. The data indicate only limited affects from the landfill. Pesticides, PCBs and BNAs (other than phthalates) were not detected. Two VOCs were detected at trace concentrations: chloroform and 2-butanone. (The analyte 2-butanone is a common laboratory contaminant.) Cadmium, beryllium, lead and calcium were reported at concentrations from two to nine times the background samples when compared with off-site background sediment samples. (The lead concentration was within two times the observed background concentration for soil samples.) However, when compared with upstream samples SED-4 and SED-5, only cadmium appeared elevated (lead results were rejected). Mercury was also reported at a concentration greater than the off-site background (SED-1 and SED-3), but less than the detection limits for the on-site background (SED-4 and SED-5).

Samples SED-33 and SED-34 were collected at depths of 6-12 inches and 12-18 inches, respectively, at location SED-8. No detectable concentrations of VOCs, PCBs, BNA compounds or pesticides were present. The inorganic analytes are similar in occurrence to sample SED-8, although mercury, lead and calcium generally decrease with depth.

Sediment from Wetland C - North of the Fill

Samples SED-10 through SED-15 were collected from the wetland north of the main body of the fill. Sample SED-2 was collected from an undrained ditch area between the northern edge of the fill and the wetland.

Sample SED-2 showed no detectable pesticides, PCBs or BNAs. Trace concentrations of two VOCs were detected, at or below the detection limits reported for the background samples. None of the inorganic analytes were detected at concentrations above background except cadmium (approximately three times background).

Samples SED-10 through SED-15 showed no detectable volatile organic compounds or PCBs. One BNA compound was reported in sample SED-14 at a concentration below the detection limit reported for the background samples. A detectable, but trace, concentration of one pesticide was reported in one sample. Several inorganic analytes were present at concentrations between two and four times the site sediment background concentrations: calcium, lead, mercury and sodium. Cyanide was also reported at a low concentrations (near detection limits) in one sample.

Samples SED-31, SED-32, SED-35 and SED-36 were also collected from the wetland areas north of the landfill but at depths of 6-12 and 12-18 inches below grade (below samples SED-11 and SED-15). No detectable concentrations of pesticides, PCBs, BNAs or VOCs were reported. Inorganic analytes calcium and copper were elevated above two times background at location SED-31, but all reported concentrations were within three times background.

Sediment Downgradient of the Site

Sediment was sampled downgradient of the site at locations SED-16 through SED-20. Volatile organic compounds, with the exception of 2-butanone, a common laboratory contaminant, were not detected. PCBs were not detected at the downgradient locations. Pesticides were detected at three of the five downgradient sediment locations, although none of those pesticides detected downgradient were detected in the on-site sediment samples. BNA compounds were detected at two locations. The most notable was the presence of several PAHs at the most downgradient sample. This may be attributable to roadway runoff, as this sample (SED-20) was collected near the bridge on Rt. 44/55 over this stream.

Metals concentrations were significantly above sediment background concentrations in some sediment samples, primarily sample SED-17. However, when compared with soil sample background, all concentrations were within three times background except barium. Cyanide was also detected in sample SED-17.

Samples SED-27 and SED-28 were collected at 6-12 inches and 12-18 inches, respectively, at location SED-20. No pesticides or PCBs were detected and inorganic analytes were within three times the observed soil and sediment background concentrations. BNA compounds detected in sample SED-27, were consistent with those found in sample SED-20, which were attributed to roadway runoff. Volatile organic compounds (chloroform and 2-butanone) were also detected in both samples, but 2-butanone may be attributable to laboratory contamination. Toluene was also identified in SED-27, which could be attributed to laboratory contamination, roadway runoff or the landfill.

4.4 Comparison to Applicable or Relevant and Appropriate Requirements (ARARs)

Applicable, relevant, and appropriate requirements (ARARs) are presented in Tables 4-20 through 4-23. These tables present maximum concentration of the compounds detected in the RI investigation for various matrices (ground water, surface water, soil and sediment, respectively), and compares them to available Federal and State of New York standards and guidance values established for the protection of human health and the environment.

The primary standard used for ground water is the Maximum Contaminant Level (MCL). The Maximum Contaminant Level Goal (MCLG) is also used. These numbers have been developed under the Safe Drinking Water Act and MCLs are federally enforceable. In the event an MCL does not exist for a particular compound, comparisons are made with the federally established Ambient Drinking Water Quality Criteria (ADWQC). New York State standards and guidelines for drinking water and ground water quality have been issued by the NYSDOH (10NYCRR) and NYSDEC (6NYCRR), respectively.

In ground water samples, a number of compounds, including VOCs, BNAs and metals (total and dissolved) exceed Federal or New York standards or guidance values. Metals were the most common compound group exceeding these values; for some metals (barium, cadmium, chromium, lead, silver), the total concentration exceeded one of the standards or guidance while the dissolved concentrations did not exceed any standard or guidance. Generally, New York standards or guidance were exceeded more frequently than the Federal standards, because New York standards and guidance exist for a larger number of compounds and are generally more stringent than the Federal standards.

For surface water sample results, the same MCL and MCLG referenced above were utilized as federal standards for comparison. As discussed, the MCLs and MCLGs were promulgated for the protection of drinking water supplies. New York State Ambient Water Quality Standards and Guidance Values (promulgated by the NYSDEC) comprised the values for comparison. These NYSDEC regulations are divided into subclasses for drinking water supply, fishing and fish propagation, and fishing and fish survival. Each of these subclasses was shown in the ARARs table.

Federal or New York standards or guidance are exceeded in surface water samples for a number of compounds, including VOCs, BNAs, and metals. Metals were the predominant analyte group exceeding standards or guidance. The leachate seep samples, collected at the toe of the landfill, exceeded standards or guidance more frequently than the remaining surface water samples.

Because there are no known Federal standards or guidelines for assessing sediment quality, sediment ARARs were compiled using guidance from the NYSDEC; this guidance is used by the NYSDEC Bureau of Environmental Protection, Division of Fish and Wildlife. These numbers comprise neither a standard nor a policy of the department. No VOCs, one BNA (penzopyrene), and several metals (8) exceed the ARARs provided by the sediment criteria guidance.

No Federal or State of New York standards or guidance are available for assessing contamination in surface or subsurface soils. To compensate for this, this RI report is using the New Jersey Department of Environmental Protection (NJDEP) Soil Action Levels to provide an assessment on the soils at the Hertel Landfill site. The NJDEP Soil Action Levels include VOCs, BNAs, and metals. None of the soil concentrations on-site exceed any of the soil clean-up levels.

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5.0 CONTAMINANT FATE AND TRANSPORT

The previous section outlines the occurrence of contamination across the site in surface soil, subsurface soil, ground water, surface water and sediment. Observed contamination consists mainly of: selected metals and base/neutral/acid extractable (BNA) compounds in the surface soils; aromatic volatile organic compounds (VOCs), BNA compounds and selected metals in the subsurface soils; aromatic VOCs, BNA compounds, and metals in the ground water; metals and trace concentrations of organic analytes in the sediment; and inorganic analytes and some organics in the surface water.

5.1 Potential Routes of Migration

Several of the environmental media studied have the potential for off-site migration, primarily surface soils, ground water, sediment and surface water (see Plate 1 for sampling locations). Subsurface soils would not be at risk of transport off-site unless exposed by excavation. Although containing several chemicals of concern, the mode of transport of these chemicals associated with subsurface soils would be through leaching and ground water transport.

Surface soils can migrate or be carried from the site by overland runoff/entrainment (resulting from precipitation), in the form of airborne dust, and by users of the site via vehicle tires, shoes, etc. In addition, contaminants can move from the surface soils (leaving the soils in place) through leaching by infiltration of precipitation and transport by ground water, and volatilization to ambient air.

The sampling results have demonstrated that ground water has been impacted by the landfill. The overburden and shallow bedrock ground water level data show that the ground water generally flows toward the surface water body which exits the site to the east. Some discharge of shallow ground water to the surface water is expected (especially in the wetland areas at the landfill toe) where shallow ground water levels and surface water levels show the potential for ground water discharge. In addition, ground water is present in the saturated sediments beneath the stream and wetlands which drain the site to the east.

Ground water from the overburden and bedrock aquifer is used as a drinking water source in the vicinity of the site. The ground water samples collected from the nearby private water-supply wells as part of this investigation do not show any apparent impacts from the landfill.

Contamination of sediment at seep locations and in the stream and wetlands probably results from deposition of eroded landfill soils or accumulation/adsorbtion of waterborne contaminants upon discharge of contaminated ground water to the surface. Further sediment transport could occur through erosion and subsequent redeposition in response to seasonal or precipitation-related changes in surface water velocity.

Surface water flow is comprised of ground water discharge in the drainage basin and surface runoff from precipitation. Surface water quality appears impacted only in the immediate vicinity of the landfill, namely, at seep locations and at stream locations near the landfill. Dilution and/or natural renovation occurs downstream of the site so that only limited landfill impacts were discernable at a distance of 1,500 feet downgradient of the site (SW-18 and SW-20). However, surface water does represent a potential route of migration from the site.

The initial site reconnaissance indicated that vapor transport of contaminants from the site was not detectable. Vapor transport would possibly effect those aromatic and chlorinated VOCs observed in the study. In addition, landfill-generated compounds such as methane or hydrogen sulfide could also migrate from the site. Volatile organic compounds, primarily methane, are present in the landfill, as demonstrated by the soil gas survey and monitoring during test pit excavation. These are most likely being released from the landfill at non-detectable concentrations under current conditions. This rate of release could be increased if wastes are exposed. The surface soils sampled showed no significant concentrations of volatile compounds.

5.2 Contaminant Distribution and Observed Migration

The following section examines the contaminant presence across the site, as outlined in Section 4, in combination with the migration pathways presented in Section 5.1 to provide an understanding of contaminant persistence and migration at the site. The discussions below are presented with respect to contaminant or contaminant group. Contaminants observed in the environmental samples collected from the site include volatile organic compounds, base/neutral/acid extractable compounds, PCBs, pesticides, metals, and cyanide.

5.2.1 Volatile Organic Compounds

Although widespread at the site, volatile organic compounds (VOCs) were present primarily at low concentrations.

The most commonly occurring VOCs on site were the aromatic hydrocarbons, specifically benzene, toluene, xylenes, chlorobenzene, and ethylbenzene. The principal mechanism for the natural removal of aromatic VOCs is through volatilization (EPA, 1979). Vapor pressures (@ approximately 20°C) of the aromatic hydrocarbons range from 7 to 95.2 mm Hg and Henry's Law Constant range from 5.27×10^{-3} to 6.43×10^{-3} atm-m^3/mol. Biodegradation's role in the natural attenuation of these compounds is compound specific. Similarly the role of adsorption is compound specific; e.g. ethylbenzene has a strong tendency to be retained by soils; the amount adsorbed is highly related to the amount of organic carbon in the soil and is represented numerically by the organic carbon/water partition coefficient (Koc). The compounds with higher Koc would be preferably partitioned to organic matter in soils and so would be less likely to be leached from the soils and transported to the ground water. Some aromatic hydrocarbons are highly mobile. Benzene, for example, has a moderate solubility (greater than 1,000 ppm) and low (83 ml/g) Koc. Therefore, benzene because of its tendency to volatilize and biodegrade, would be mobile but would not be expected to be very persistent in the environment. Conversely, xylenes, with their lower solubilities (175 ppm) and higher Koc (240 ml/g), would not be as mobile as benzene, but would be more persistent in the environment as they would tend to sorb to soil particles. Probably owing to their volatility and to biodegradation, the aromatic compounds were not identified in the surface soil samples.

Subsurface soils from test pits did contain aromatic hydrocarbons. Concentrations generally increase with increasing boiling point and decreasing vapor pressure - xylenes and ethylbenzene appear to be the most persistent. This apparent persistence could also be attributed to the higher Koc of ethylbenzene, as compared with benzene and toluene. Aromatic hydrocarbons were identified in 10 of the 11 soil samples from test pits at concentrations ranging from 5 ppb to 353 ppb, indicating their widespread nature in subsurface soils/waste at the site. The highest concentrations observed were associated with Disposal Area #7 (TP-1 and TP-7) and in the west-central area of the landfill (TP-11 and TP-12).

Aromatic hydrocarbons were also present in some ground water samples. Aromatic hydrocarbons were noted at above trace concentrations (>10 ppb) in ground water samples from monitor wells MW-6S (117 ppb total aromatic hydrocarbons), MW-7S (228 ppb) and MW-13S (31 ppb). Three of the four test pit ground water samples, all associated with Disposal Area #7, contained aromatic hydrocarbons (test pits TP-2 (54 ppb), TP-7 (143 ppb) and TP-15 (93 ppb)). It is important to note that ground water samples from shallow and deep overburden monitor wells directly downgradient of Disposal Area #7 (wells MW-W2S/W2D, MW-10S/10D and MW-11S/11D) did not show detectable concentrations of aromatic hydrocarbons.

Shallow ground water discharges to the ground surface at seep locations and directly into the stream and wetlands. Aromatic hydrocarbons were present in trace concentrations in several seep samples along the eastern toe of the main body of fill. As noted above, ground water samples from shallow monitor wells along the eastern toe of the fill did not show detectable concentrations of aromatic hydrocarbons. The detection of aromatics in seep samples but not in nearby monitoring wells is likely due to the difference in sampling location. The seep samples represent the uppermost portion of the water column, while the monitoring wells represent an average of a 10-foot thick water column. Trace concentrations of aromatic hydrocarbons were detected in ground water from one seep location (4 ppb in SW-2) and one monitoring well (6 ppb in MW-8S) along the northern end of the fill.

Ground water from the overburden beneath the landfill eventually exits the site to the east as ground water or discharges to the surface water. With the exception of monitor well MW-13S, in which toluene was detected at 31 ppb, aromatics were not present in ground water samples from downgradient monitor wells. A review of the surface water sample results show that no aromatics were reported in surface water samples along the eastern or northern end of the fill or downgradient of the landfill.

The occurrence of aromatic hydrocarbons in sediment samples generally agrees with the surface water results. Aromatics were present in sediment samples collected at seep locations but not present at surface water sample locations from east or north of the fill. Downgradient of the landfill, the only finding of aromatics in sediment was 5 ppb toluene at the SED-20 location at 6-12 inches (SED-27). Sample SED-27 was collected near Route 44/55, and findings of contamination at that location may be attributed to roadway runoff.

Other volatile organic compounds (methylene chloride; acetone; carbon disulfide; chloroform; 2-butanone; 1,1-dichloroethane; chloroethane; 1,2-dichloroethene; 1,2-dichloropropane; trichloroethene and tetrachloroethene) although present, do not show a discernable pattern of occurrence.

Tetrachloroethene is the only VOC (other than toluene) identified in the surface soil samples. It was identified in Disposal Area #7 at a trace concentration (8 ppb in SS-13). Chlorinated alkanes or alkenes were not detected in the test pit soil samples. Chlorinated alkenes or alkanes were also identified at trace concentrations (1 to 2 ppb) in ground water samples from wells along the eastern toe of the fill (MW-W2S, MW-13S and MW-9S) and elsewhere in the fill (MW-5S and MW-6S). The compounds 1,1-dichloroethane and chloroethane were detected in one seep water sample (3 ppb and 5 ppb respectively in SW-21), but were not detected in any other surface water or sediment samples.

As noted above, a variety of VOCs were detected at the site in various sampled media. Acetone, carbon disulfide and 2-butanone were detected in some subsurface soil samples from test pits. [Acetone and 2-butanone are common laboratory contaminants and their presence in low concentrations in any sample should be considered with some reservation.] Acetone was detected in soil samples from test pits TP-3, TP-4 and TP-8. Acetone has a low Koc and is very soluble in water and would leach easily. Acetone was also detected in ground water samples from monitor wells MW-5S, MW-7S and MW-12S and piezometer P-2. Findings of acetone in these samples demonstrates the widespread occurrence of acetone. Because acetone was not detected in all the monitor wells in the center of the fill and downgradient of the center of the fill, a distinct plume is not evident. The acetone pattern may be as a result of several small occurrences of acetone in the fill.

Acetone was also detected in trace concentrations in several surface water sample locations including seep location SW-21 (17 ppb) and surface water location SW-6 (8 ppb) along the southeastern edge of the fill. The highest concentration of acetone detected in a surface water or ground water sample was reported at location SW-10 along the northern end of the fill. Surface water samples downstream from SW-10 (110 ppb) and other areas of the site do not show the presence of acetone, indicating the probable effects of dilution, volatilization or degradation.

Carbon disulfide was present in trace concentrations (<5 ppb) in subsurface soil samples from test pits TP-11, TP-12 and TP-13 in the western end of the landfill. Carbon disulfide is soluble in water but was not detected in any ground water samples from monitoring wells. The highest concentrations were observed in seep sediment samples along the eastern edge of the fill. It was also detected in some of the seep water samples. Carbon disulfide was detected at trace concentrations in sediment samples from the stream along the eastern edge of the site but was not detected in surface water or sediment samples downgradient of the site.

The compound 2-butanone was identified in one subsurface soil sample (most results were rejected). This compound was also detected in two ground water samples from the center area of the fill, but was not detected in the seep or surface water samples downgradient of the fill. The compound 2-butanone was also detected in two sediment samples, one in the stream east of the fill and one downgradient of the site.

The remainder of the VOCs (methylene chloride, chloroform and 1,2-dichloropropane) were detected only in the ground water, surface water or sediment samples. Methylene chloride was detected in sediment samples from seep locations along the eastern edge of the site and from a sediment sample upgradient of the main body of fill. Methylene chloride was detected in one on-site surface water sample, but was not detected in downgradient sediment or surface water samples. Chloroform and 1,2-dichloropropane were detected in one ground water sample from monitor MW-2S. They were present at trace concentrations (1 ppb) and so are not considered to be significant.

Previous ground water sampling data collected by Wehran Engineering (Ecological Analysts, Inc., 1983) indicated the presence of methylene chloride; toluene; 1,1,1-trichloroethane; 1,1-dichloroethane; and chloroform. The results of this Remedial Investigation (RI) confirm the presence of most of those compounds and identified several additional compounds. The RI results, however, also suggest that these compounds are largely confined to the site and they do not appear in significant concentrations in downgradient sediment or surface water. The potential risks posed to human health and the environment are examined in Section 6.

5.2.2 Base/Neutral/Acid Extractable Compounds

The base/neutral/acid extractable (BNA) compounds were identified in all the media sampled on site. The BNA compounds, particularly the PAHs, are persistent in the environment due to their complex chemical nature. Some of the lighter PAHs (fewer aromatic rings) would be subject to biodegradation or volatilization, but the chemical persistence generally increases with increasing number of aromatic rings. Base/neutral/acid extractable (BNA) compounds are generally characterized by high boiling point, low vapor pressure, and low solubility (except phenols).

The BNA compounds will be divided into the following groups for discussion: polynuclear aromatic hydrocarbons (PAHs) and naphthalene, phenols, phthalates, and other compounds (benzoic acid, benzyl alcohol, 1,2-dichlorobenzene and 1,4-dichlorobenzene).

Polynuclear aromatic hydrocarbons (PAHs) were detected at concentrations up to 15,000 ppb (SS-19) in surface soils and subsurface soils primarily in the vicinity of Disposal Areas #6 and #7, but also in other areas of the site. PAHs generally have a very low solubility (<4.0 mg/l), whereas the solubility of naphthalene is slightly greater (30 mg/l). The Koc of PAHs is generally greater than 2,500 ml/g with many greater than 100,000 ml/g. This indicates that PAHs readily absorb to organic carbon in soils. This accounts for their relative absence from the ground water samples, with the exception of phenanthrene which was detected in one ground water sample from the center of the landfill. Naphthalene was detected in four first-round ground water sample locations and one second-round sample. The highest concentration of naphthalene (39 ppb) was detected in samples from monitor well MW-7S located near the center of the fill. Both naphthalene (4 ppb) and phenanthrene (11 ppb) were detected in one bedrock ground water sample (MW-6D, first round). PAHs and naphthalene were not detected in ground water samples from the downgradient monitor wells along the eastern end of the site, but were detected along with fluoranthene and pyrene in seep water samples. PAHs were not detected in surface water samples collected from the on-site streams or wetlands or downgradient of the site.

PAHs are readily adsorbed onto particulates (Callahan, et al 1979). Biodegradation is expected to play a significant role in the fate of PAHs which accumulate onto sediments (EPA, 1979). PAHs were present in the sediment samples collected from the seep locations. PAHs and naphthalene were not detected in the sediment samples from the on-site streams or wetlands. The only other location where PAHs were detected was in sediment collected from near Route 44/55 (18,130 ppb in SED-20). Since PAHs are present in asphalt and tars, as well as particulates from vehicle exhaust, traces of PAHs at this location may be due to roadway runoff.

Phenols and phenol compounds are generally more soluble in water than other BNA compounds and display a relatively low volatility (the vapor pressure of phenol is less than the aromatic hydrocarbons but slightly greater than naphthalene; the Henry's Law Constant for phenol is much less than that of naphthalene). Based on the relatively low Koc and high solubility of phenols, they would not tend to adsorb to soils and sediment organic matter; but would tend to leach from soil into ground water. Phenol compounds of various types (phenol; 4-methylphenol; 2,4-dimethylphenol) were detected in subsurface soil, ground water, surface water and sediment. The apparent absence of phenol compounds from surface soil may be due to their solubility (leaching potential) or to their biodegradability (Callahan, et al 1979).

Phenol compounds were detected in one test pit soil sample in the center of the fill (TP-3) but were also detected in ground water samples from the center of the fill (MW-6S, MW-7S) from the northern limit of the fill (MW-13S) and from the bedrock (MW-6D). One phenol compound was detected in the seep water sample at the north end of the fill (SW-2), but phenol compounds were not detected in any other on- or off-site surface water samples. One phenol compound was detected in one sediment sample downgradient of the site.

Phthalate compounds were reported in samples from all environmental media collected at the site. It should be noted that phthalates are considered to be common laboratory contaminants and are widespread in the environment (ATSDR, 1987; ATSDR, 1989). This may explain their detection in background samples. The phthalate esters are distributed in the environmental samples from the site as follows:

- Surface soil samples distributed across the site.
- Subsurface soil samples from test pits TP-2, TP-3 and TP-4 along the east-central portion of the fill and in a background subsurface soil sample.
- Ground water samples from monitor wells in the center of the site, but also upgradient of the main body of fill (MW-3S).
- Water samples from seep locations on the east and north side of the fill.
- Sediment from seep locations on the east and south side of the fill.
- Sediment from one on-site stream sample location east of the fill and the background sediment sample location.
- Trace concentrations in one on-site and one off-site downgradient surface water samples.

The phthalate esters, although not common in all media, appear to be widespread across the site including some background sample locations. They generally occur in association with other BNA compounds. They generally exhibit low solubility and high Koc, and so would not be particularly amenable to water transport. This is consistent with the site data which show the phthalates primarily in sediment and soil samples. However, control of phthalate migration from the site would require control of sediment and soil migration as well as discharges from seeps.

Other BNA compounds detected in the environmental samples include benzyl alcohol, benzoic acid, 1,4-dichlorobenzene, and 1,2-dichlorobenzene. These compounds appear in isolated occurrences, showing no apparent pattern. Although present in on-site subsurface soils, ground water, seep water, seep sediment, and on-site wetland sediment, they do not appear in surface water or sediment samples from downgradient of the site.

5.2.3 Pesticides and PCBs

Pesticides were detected both within the fill area and in samples collected from beyond the fill area. Pesticides also have an affinity for organics in soils (e.g., Koc of DDT is 243,000 ml/g), which tends to render them immobile. Many pesticides, such as DDT, are very persistent.

Pesticides and PCBs at the site appear generally confined to the solid media: sediment and soils. Pesticides were noted in surface soil samples from the east-central area of the fill, west of the fill (background sample SS-22) and south of the main body of fill near disposal area 8. Pesticides were not detected in the subsurface soil samples, ground water samples, seep sediment samples or surface water samples. The few observed occurrences of pesticides downgradient were scattered and were not always pesticide compounds detected in the fill area, but are sometimes metabolites or degradation products of 4,4-DDT which was found on site.

PCBs are generally regarded to be a significant environmental problem because of their persistence and adverse health effects linked to PCBs. However, because of the strong tendency of PCBs to adsorb to organic matter in soils, PCBs do not tend to migrate unless solvents or oils are present (Callahan et al, 1979).

PCBs were not identified in surface soil samples in the fill area but were identified in subsurface soil samples from on-site test pits (1500 ug/kg at TP-2). PCBs were not identified in the ground water, surface water, seep water, or sediment, and so do not appear to be migrating from the site.

5.2.4 Inorganic Analytes

The presence of numerous inorganic analytes was examined in the Remedial Investigation. Many metals have an affinity for soils (particularly clay particles and organic matter in soils) which reduce their mobility. Under low pH conditions typical of landfill leachate, some metals can be rendered mobile, as exemplified by the EP Toxicity results. The presence of the inorganic analytes, particularly the naturally occurring elements, must be examined in the context of natural background concentration for the site, as discussed in Section 4. The analytes which appeared elevated above background levels in one or more samples are: aluminum, arsenic, barium, calcium, copper, chloride, cadmium, chromium, iron, potassium, magnesium, manganese, lead, mercury, sodium, zinc and cyanide. The distribution of these analytes is discussed below.

The inorganic analytes can be grouped generally based on the observed above-background distribution at the site.

- Arsenic was present in one on-site surface water sample and several ground water samples at concentrations above background.
- Copper, chromium and mercury appear to be elevated above background concentrations in on-site soils or sediment and not in filtered ground water or surface water.
- Aluminum, barium, iron and zinc appear to be elevated above background concentrations in on-site filtered ground water, seep (water) and/or surface water. The affects of these elements on surface water and sediment appear to be confined to the site.

- Cadmium and lead were detected above background concentrations in on-site soils, seeps (water) and seep sediment and/or stream sediment. Impacts were not observed in concentrations significantly above background in on-site filtered ground water, off-site surface water or sediment samples.

- Calcium, chloride, potassium, magnesium, manganese and sodium showed a wide distribution in the surface and ground water at the site. All six elements were present at above background concentrations in seep (water) samples and on- and off-site surface water. With the exception of chlorides, these elements were also present in on-site ground water at above-background concentrations.

- Cyanide was detected in on-site subsurface soils, but only sporadically in on- and off-site stream sediment, seep sediment and seep (water). Cyanide was not detected in wetland or stream surface water or ground water.

6.0 BASELINE HEALTH RISK ASSESSMENT

6.1 Objectives

This section provides a quantitative Health and Environmental Risk Assessment (Human Health Evaluation) for the former Hertel Landfill site (the site) in Plattekill, New York. Its primary objectives are to examine exposure pathways and concentrations, and to estimate the potential for adverse effects associated with the contaminants of concern at the site under current and future land use conditions (baseline risk assessment). The risk assessment follows guidelines established by the EPA (EPA, 1989).

For the baseline assessment, three site-specific exposure scenarios have been considered and developed to represent potential situations in which humans may be exposed to contaminants originating from the site. Efficacy of specific remedial programs are not included as part of this analysis.

Human health risks associated with the site are presented with regard to potential effects from the contaminants of concern. These effects may include potential risks of cancer or non-cancerous (systemic) effects. A quantitative risk assessment for carcinogens involves calculations of the lifetime incremental probabilities of cancer that take into account exposure concentrations and the carcinogenic potencies of the chemicals. Health effects associated with exposures to noncarcinogenic chemicals are evaluated primarily with regard to reference dose (RfD) values. This approach for non-cancer effects is most useful when exposure doses of the chemical are below the RfD thresholds. However, there is often no quantitative way to measure the degree of risk created when concentrations exceed the standard thresholds.

Ultimately, the risk assessment presented in this report is expected to be used within a risk management framework. In making decisions concerning what, if anything, should be done at a site (including, for example, the collection of additional data or implementation of a remedial program), the results of the risk assessment should be used in concert with other information on the site. The risk assessment should also identify site or land use conditions that present unacceptable risks. The results of the risk assessment identify contaminants and exposure pathways contributing the greatest risk to the receptor population. From this information, recommendations for future activities at the site can be made such that public health and the environment are protected.

This Human Health Evaluation focuses most strongly on the baseline conditions at the site. However, the results of this study will help decision makers focus on the areas, contaminants, media, pathways and people of greatest concern at the site, thereby helping to identify future remedial alternatives for the site.

6.2 Methodology

The methodology is structured utilizing the most current methods accepted by the EPA in the Interim Final Risk Assessment Guidance for Superfund, Volume I (Human Health Evaluation Manual - Part A) (1989). Where assumptions are made, they are realistic but conservative, i.e., protective of public health. In keeping with accepted practices for conducting such assessments, all assumptions are carefully discussed and an assessment made of the uncertainty associated with the overall health and environmental risk estimates.

Following the guidelines accepted by the EPA, the basic components of the public health risk assessment will be organized and presented as follows:

- Data Collection;
- Data Evaluation;

- Exposure Assessment;
- Toxicity Assessment; and
- Risk Characterization.

Each of these components are discussed in detail with relation to the site.

6.3 Identification of Chemicals of Potential Concern

6.3.1 Data Collection

A soil gas survey was used to direct the investigation of on site soils and ground water. Soil gas sampling points were located using a grid system. This systematic method was used to better understand the location and extent of contamination so that sampling locations for environmental media could be effectively located. Sampling points were located based on known or suspected areas of contamination and used in conjunction with the results of the systematic soil gas survey to locate the sampling points for environmental media on the Hertel site.

The surface water and sediment sample locations were chosen to identify potential contaminants that may have entered the stream sediments and the surface water through surface runoff and ground water discharge. Sampling of known or suspected areas of contamination was used to maximize the detection of chemicals released into surface waters. Potential background surface water and sediment sampling locations were identified by sampling in areas where landfilling did not occur. Background sampling points were also located upgradient of potentially contaminated surface and ground water.

The test pit soil sampling locations were identified starting from the areas suspected of having the heaviest contamination (identified disposal areas) as compared to other areas. Background subsurface soil sampling locations were drilled in areas where no landfilling had occurred. Also an estimation of ground water flow direction was deduced in order to locate background test boring locations upgradient of any past landfill activities.

Monitoring wells were installed to determine the direction of shallow overburden, deep overburden and shallow bedrock ground water flow and the extent and amount of contamination. Background ground water sampling points were located in three regions of the site where ground water was suspected to enter the site. The background wells were also located in areas where no landfill disturbances took place.

Surface soil samples were collected in areas where there were obvious landfill activities so that the most contaminated spots were quantified. Background surface soil sampling locations were identified by sampling in areas where it was believed that landfilling had not occurred. In addition, numerous background soil samples were collected from test borings to establish a probable range of background soil concentrations.

6.3.2 Data Evaluation

As detailed in Section 4.0 of this report, the site contains residues from the on site disposal of wastes between 1963 and 1976. Field studies have revealed the presence of numerous organic and inorganic contaminants in the soils, sediments, surface water and ground water.

In order to organize the data into a form manageable and appropriate for the baseline risk assessment the following nine steps were followed during the data evaluation process as described by EPA (1989):

1) Gather and sort all data by medium (e.g., soil, ground water, etc.);

2) Evaluate methods of analysis;

3) Evaluate the sample quantitation units;

4) Evaluate the data qualifiers and codes;

5) Evaluate blank data;

6) Evaluate tentatively identified compounds (TIC's);

7) Evaluate background data;

8) Develop data sets by medium; and

9) Develop a set of chemicals of potential concern from the entire data set.

Briefly, the specific methods used in this report include the following, which correlate respectively with the previously described steps (1-9).

1) All analytical data was initially sorted by media (surface soil, subsurface soil, surface water, ground water and sediments);

2) An evaluation of analytical methods was not considered to be necessary as all data used was analyzed by EPA's Superfund Contract Laboratory Program (CLP) procedures or other standard USEPA methods;

3) Unusually high sample quantitation limits (SQL's) were commonly reported, due primarily to matrix interferences. While these unusually high SQL's could not be reduced by re-analyzing the sample, the data was excluded from the data set when the high SQL's caused the 95% upper confidence level (UCL) to exceed the maximum detected concentration for a particular sample set. In these instances, the maximum detected concentration was used to represent site concentration. Consideration as to the use of the SQL or one-half the SQL included extent and degree of contamination or concentration within each media and potential for migration between media. These factors were used to evaluate the potential of chemicals to be present at concentrations below the SQL. Similarly, if a chemical was not detected in a single media, transport and fate information was used to determine the likelihood of low chemical contamination (below the SQL) in that particular media;

4) Data validation qualifiers were assessed during the data evaluation process. As indicated in EPA guidance (EPA, 1989), data qualified with U, J or UJ qualifiers were used in the quantitative risk assessment when appropriate. Chemical data characterized with unusually high SQL's (and qualified with a UJ) in combination with a lack of detection across a medium was not addressed in the quantitative risk assessment, but discussed in a qualitative sense;

5) Field and laboratory blanks were used to segregate actual site contamination from cross contamination from field or laboratory procedures. As indicated in EPA (1989) sample results were considered positive only if concentrations exceeded ten times the concentration of a common laboratory contaminant in a blank, or five times the concentration of a chemical that is not considered a common laboratory contaminant;

6) Tentatively identified compounds (TICs) were reported in surface soil samples across the site. TICs ranged from a few unknowns at low concentrations (<1 ppm) to many TICs (>10) each at elevated concentrations (>250 ppm). Due to the uncertainty associated with the quantitative and qualitative nature of these TICs, only a qualitative assessment of risk associated with exposure was included in this assessment;

7) As discussed previously in this section, background sampling locations were identified for the site. Data resulting from sampling at these locations was used in combination with national background levels as a screening method to evaluate non-site related chemicals or commonly encountered naturally occurring chemicals; and

8) Tables 6-1 through 6-5 provide the chemicals and concentrations sampled in surface soils, subsurface soils, surface water, sediments, and ground water, respectively. Table 6-6 provides a summary of chemicals of potential concern in each media. This list was not reduced based on guidance provided in Chapter 5 of EPA, 1989, with the exception of evaluation of certain compounds as essential nutrients (i.e. iron, magnesium, calcium, potassium and sodium).

6.3.3 Summary of Surface Soil Data

Table 6-1 presents a summary of the analytical data associated with chemicals detected in surface soil, organized by class including semivolatile organics, inorganics and pesticides/PCBs. Volatile organic compounds were not detected in surface soils. Each class of chemicals is discussed in detail below.

- Semi-Volatile Organics

Of the twenty-six semivolatile organics in surface soil listed in Table 6-1, nine were not detected at any of the twenty-two sampling locations, and an additional five compounds were detected at a frequency of less than 5% (1/22) (see Table 6-1). Polynuclear aromatic hydrocarbons (PAHs) detected (greater than 5%) include benzo(a)anthracene (5/22), benzo(a)pyrene (4/22), benzo(b)fluoranthene (6/22), benzo(g,h,i)perylene (3/22), chrysene (6/22), fluoranthene (7/22), indeno(1,2,3 cd)pyrene (4/22), phenanthrene (5/22) and pyrene (7/22). In general, the concentrations of these PAHs were low (close to detection limits), with a range of detection of 58-280 ppb for the group.

Phthalate esters detected in surface soils at a frequency greater than 5% include only bis(2-ethylhexyl)phthalate (10/22), which was detected at a range of 37 to 240 ppb, which is close to the detection limit. Table 6-1 presents the range of sample quantitation limits (SQL) for surface soil. Unusually high SQL occurred frequently in surface soil samples, particularly in soil sample number 5 (SS-5) (SQL = 24 ppm) which was visually contaminated with an oily substance. As indicated previously, unusually high SQL were not included in the calculation of the 95% UCL if inclusion of SQL forced the 95% UCL to be greater than the maximum detected concentration.

On site background concentrations (570 ppb for all semivolatiles except benzoic acid) were not actual hits but rather were SQL ("UJ" data). More than half of the semivolatile compounds detected on site were detected at concentrations greater than the SQL of 570 ppb.

- Inorganics

Of the inorganics analyzed, only three (antimony, selenium and thallium) were not detected at any location on site. All other elements were detected at a frequency greater than 5%. SQL for inorganics were not unusually high, and 95% UCL were not adjusted based on the exclusion of "UJ" data. Comparisons to on-site background and U.S. background levels (see Table 6-1) indicated a general trend of elevated concentrations across the site for arsenic, cadmium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, nickel and zinc.

- Pesticides/PCBs

Only 4,4'-DDE and 4,4'-DDT were detected on site, at a frequency of 2/22 and 6/22, respectively. The ninety-five percent UCL for both compounds was below on site background levels.

6.3.4 Summary of Subsurface Soil Data

Table 6-2 presents a summary of the analytical data associated with chemicals detected in subsurface soil, organized by class including semivolatile organics, volatile organics, inorganics and pesticides/PCBs. Each class is discussed in detail as follows.

- Semi-Volatile Organics

Of the twenty-six semi-volatile organics listed in Table 6-2 for subsurface soil, eight were not detected at any location on site. Furthermore, seven compounds were detected only once (out of eleven possible locations) and six were detected only twice. Of the thirteen compounds, all were detected at low concentrations (i.e., close to or less than the detection limit). Fluoranthene, naphthalene and pyrene were the most frequently detected compounds (4/11, 6/11 and 4/11, respectively) with a range of detection from 50-1,200 ppb. Bis(2-ethylhexyl)phthalate was the most frequently detected phthalate ester (5/11), with a range of concentration from 87-4,500 ppb.

Unusually high SQL occurred occasionally in subsurface soil samples, particularly in test pit sample number 12 (TP-12) with an SQL equal to 24,000 ppb. No visual contamination was noted in this sample. As indicated previously, unusually high SQL were not included in 95% UCL calculations if the inclusion of the SQL forced the UCL to exceed the maximum detected concentration. In some instances, all SQL were eliminated and the maximum detected concentration was used in place of a 95% UCL.

No background levels were available for comparison with on site concentrations.

- Volatile Organics

Of the volatile organics (VOCs) detected, only benzene was detected as infrequently as two of eleven locations. Both hits for benzene were close to or less than the detection limit. Of the remaining VOCs, ethylbenzene, toluene and total xylene concentrations exceeded the detection limits by at least two fold.

No unusually high SQL were reported for VOCs.

No background levels were available for comparison with on site concentrations of VOCs in subsurface soils.

- Inorganics

Of the inorganics analyzed, only selenium was not detected at any of the eleven sampling locations. Most inorganics were detected at a frequency of 100% (see Table 6-2). SQL for inorganics were not unusually high, and 95% UCL were not adjusted based on the exclusion of "UJ" data. Comparisons to U.S. background levels (see Table 6-2) indicated a general trend of elevated concentrations across the site for antimony, barium, cadmium, cobalt, copper, lead, manganese, nickel and zinc.

- Pesticides/PCBs

No pesticides or PCBs were detected in subsurface soil at any location on site. There was no indication of unusually high SQL in the analytical data.

6.3.5 Summary of Surface Water Data

Table 6-3 presents a summary of the analytical data for compounds detected in surface water including leachate samples. Each class of chemicals is discussed in detail below, with the exception of Pesticides/PCBs which were not detected.

- Semivolatiles Organics

Of the semivolatile organics listed in Table 6-3, only 4-methylphenol, bis(2-ethylhexyl)phthalate and phenanthrene were detected at a frequency greater than 5% (2/20 each). Of these, only one hit of 4-methylphenol exceeded the detection limit (110 ug/l vs a detection limit of 10 ug/l). This sample was located at surface water sample number 25 (SW-25) which is a leachate sampling location.

No semivolatile organic compounds were detected in on site background sampling locations (values shown in Table 6-3 are SQL).

- Volatile Organics

Carbon disulfide, chlorobenzene, ethylbenzene, toluene and total xylenes were all detected at a frequency greater than 5% across surface water sampling locations. In general, the ranges of detection were low, that is, less than or close to detection limits.

- Inorganics

Beryllium, silver and thallium were not detected at any surface water sampling location. All other elements were detected at a frequency greater than 5%. SQL for inorganics were not unusually high, and 95% UCL were not adjusted based on the exclusion of "UJ" data. Comparisons to on-site background levels (see Table 6-3) indicated a general trend of elevated concentrations across the site for all elements except antimony, selenium and vanadium.

6.3.6 Summary of Sediment Data

Table 6-4 presents a summary of the analytical data for compounds detected in sediment samples. Each class of chemicals is discussed in detail below.

- Semivolatile Organics

Bis(2-ethylhexyl)phthalate, chrysene, di-n-butylphthalate, fluoranthene, fluorene, naphthalene, phenanthrene and pyrene were the only semivolatile organics detected at a frequency greater than 5% (greater than or equal to 2 hits/21 locations). Because of the wet nature of the sediment samples, many of the SQL were unusually high, particulary sediment sample number 4 (SQL equal to 9,400 ppb). As indicated previously, unusually high SQL were not included in the calculation of the 95% UCL if such inclusion caused the UCL to exceed the maximum detected concentration. In some cases, the maximum detected value was used in place of the 95% UCL.

No semivolatiles were detected in on-site background samples. Furthermore, on-site background sampling data was qualified as "UJ" data and contained unusually high SQL. Based on this information no meaningful comparison between site related contamination and background could be made.

- Volatile Organics

Of the volatile organics listed in Table 6-4, only carbon disulfide, chlorobenzene, toluene and total xylenes were detected at a frequency greater than 5%. Of these, detected levels of carbon disulfide and toluene were less than or close to the reported detection limits. No volatile organics were detected at background sampling locations (Table 6-4 contains a range of SQL at background sampling locations). Thus, only chlorobenzene and xylene exceeded these detection limits.

- Inorganics

Antimony was the only inorganic not detected in sediment samples. All other inorganics were detected at a frequency greater than 5%. SQL for inorganics were not unusually high, and 95% UCL were not adjusted based on the exclusion of "UJ" data. Comparisons to on-site background and U.S. background soil levels (Table 6-4) indicated a general trend of elevated concentrations across the site for arsenic, cadmium, chromium, cobalt, copper, iron, lead, manganese, nickel and zinc.

- Pesticides/PCBs

No pesticides/PCBs were detected at a frequency greater than 5%. 4,4'-DDE and 4,4'-DDT were detected once (out of 21 sampling locations) at levels close to detection limits.

6.3.6 Summary of Ground Water Data

Table 6-5 presents a summary of the analytical data associated with compounds detected in two rounds of ground water sampling. Each class of chemicals is discussed in detail below, with the exception of pesticides/PCBs, which were not detected in either sampling round.

- Semivolatile Organics

In two separate rounds of ground water monitoring, only 4-methylphenol was detected at a frequency of greater than 5% and in both rounds. In round I, 2,4-dimethylphenol, benzoic acid, naphthalene and phenol were detected at a frequency greater than 5%. In round II, diethylphthalate was the only other compound besides 4-methylphenol that was detected more than once in twenty five locations. No semivolatile compounds were detected in background well locations. Values for on site background levels are presented as SQL. SQL were not unusually high, and 95% UCL were not adjusted based on the exclusion of "UJ" data. However, maximum detected concentrations were used in place of 95% UCL in some instances.

- Volatile Organics

Chlorobenzene, ethylbenzene, toluene and total xylenes were detected in both sampling rounds at a frequency of detection greater than 5%. No other VOCs were detected in either round at a frequency greater than 5%. No VOCs were detected in background well locations. Values for on site background levels are presented as SQL.

- Inorganics

Analytical data for all inorganics was based on unfiltered analyses. Selenium was the only compound not detected at a frequency greater than 5% in both rounds of ground water monitoring. Chromium, copper, mercury, thallium and vanadium were not detected or not detected at a frequency greater than 5% in round II only. All other elements listed in Table 6-5 were detected in both rounds of ground water monitoring. SQL for inorganics were not unusually high, and 95% UCL were not adjusted based on the exclusion of "UJ" data.

Comparisons to on-site background data (Table 6-5) indicated a general trend of elevated concentrations across the site for all detected inorganics.

6.3.7 Chemicals Used in the Assessment

Table 6-6 presents a summary of chemicals of potential concern in all media sampled (as a range of detection). Chemicals carried through the quantitative risk assessment are marked with a single asterik (*) to the left of the chemical name. Chemicals discussed in the qualitative risk assessment are marked with two asteriks (**) to the left of the chemical name. Those chemicals addressed both quantitatively and qualitatively are marked with three asteriks (***).

6.4 Exposure Assessment

6.4.1 Development of Exposure Scenarios

The most critical aspect of a technically sound exposure assessment is the identification of exposure routes, together with the identification of human receptors. Based on findings presented in Section 1.0 of this report and discussions with field personnel, the following potential current human exposure scenarios were identified:

Scenario 1

- Persons having access to the site (i.e., residents of Plattekill) may be potential receptors (especially children playing on the site or adults trespassing on the site for recreational purposes).
- The ground water gradient on site is towards the east or northeast so any contaminants picked up underneath the site may flow towards the east or northeast residences off site.

Several potential future exposure pathways exist at the site, including:

Scenario 2

- Construction of buildings on the site (i.e., development of the site into house lots), presenting a potential for exposure of construction workers to site contaminants.

Scenario 3

- Residential use of the site, presenting a potential for exposure of adults and children to site contaminants.

Each scenario includes a particular potential "receptor population", and a consideration of the pathways by which those receptors may encounter contaminants of concern. The values and assumptions used for each exposure scenario were prepared in keeping with generally accepted values in the discipline of risk assessment; the values are not based on a detailed time-activity studies. Specific assumptions and details for each exposure scenario are presented in Appendix I.

6.4.2 Exposure Scenarios Addressed in the Health Assessment

Recreational Use/Trespassing Scenario

Part 1 - Adults: Table I-1 of Appendix I of this report presents the model inputs for the exposure routes that recreational use/trespassing on the site could potentially occur. It is assumed that adults may make 60 hiking, hunting, etc. trips during a 4-month season, with 3 weeks of vacation and all weekend days spent on the site. Adults may enter the site while hunting or tracking game or hiking/camping, and it is assumed that 25% of the time is spent on site.

For exposure to surface water on site, it is assumed that adults would occasionally wade into or accidentally fall into a swamp on site (5 exposures per 60 trips). This would involve dermal exposure to water (resulting in exposure of lower body parts) which would last 4 hours due to clothes staying wet. This mechanism for exposure would not involve a substantial amount of water ingestion, and this route was not addressed quantitatively.

For dermal exposure to surface soil contaminants, it is assumed that adults would have exposed arms and hands. Of the total amount of organic contaminants present in soil and adhered to skin, it is expected that 10% would be absorbed (Brown, 1984). For inorganic contaminants this dermal absorption factor is much lower (EPA, 1986; EPA, 1984; EPA, 1982), and 1% absorption has been used as an approximation. However, the dermal penetration of arsenic may be greater than that for other inorganic chemicals (ATSDR, 1989), and thus, the absorption value for arsenic was set at 10%. Dermal contact with sediments was not addressed quantitatively in this report as exposure to sediments is expected to present a risk less than or equal to that associated with dermal exposure to soils. In addition to dermal exposure, adults are also expected to receive exposure to soil contaminants by ingestion. It is assumed that adults will ingest 100 mg soil/day, with 100% of that occurring on site, and that all contaminants in soil will be completely absorbed from the gastrointestinal tract.

Again, based on potential activities, exposures would not involve a substantial amount of sediment ingestion. Therefore, this route was not addressed quantitatively. Furthermore, ingestion exposure of sediments is expected to present a risk less than or equal to that associated with ingestion exposure to soils, based on the 95% UCL across contaminants.

Part 2 - Children: Table I-2 of Appendix I presents the model inputs for the exposure routes associated with children trespassing and playing on-site. It is assumed that children living within the immediate vicinity of the site may trespass onto it up to 90 days per year, which is approximately one-half of the non-school days. Additionally, on days in which children trespass/play on site, it is assumed that all soil ingestion (100 mg) for that day occurs on site. Children are not likely to enter the site on a regular basis and without adult supervision before the age of 9 years due to the distance of the site from residences. Regular exposures of this nature are not expected beyond the age of 18 years because of changes in the use of recreational time. Play activities are expected to involve contact with soil and surface water. For surface water, the exposure period is assumed to be 4 hours/day, due mostly to clothing staying wet after water activities have ended. Ingestion of surface water will be unintentional and very low. As an upper bound, the amount of water ingestion that could be incurred while swimming (50 ml/hour) was assumed to occur for receptors at this site for a 1 hour period. Dermal and ingestion exposures are modeled for only soils and not sediments, as exposure to sediments is expected to present a risk less than or equal to that associated with exposure to soils, based on the 95% UCL across contaminants. For dermal exposures, children are assumed to have exposed arms, hands and legs, and dermal penetration of contaminants in soil was modeled as described above for adults. Absorption of soil contaminants after ingestion is assumed to be 100%.

Construction Scenario

Table I-3 presents the model inputs for the exposure routes that construction workers involved in site development could potentially encounter. Excavation and site preparation activities could cause workers to receive inhalation exposure to contaminants in dust, as well as dermal and ingestion exposures to contaminants in soil. It is assumed that workers are engaged in the construction of 5 homes, with excavation and site preparation activities lasting for a 6-month period. It is also assumed that remediation of contaminants would not occur prior to construction or prior to the occupation of homes (Residential Scenario - below). The inhalation rate is based upon men undergoing moderate exertion, and dermal exposure is based upon the wearing of long pants and sleeveless, tank-top shirts. The soil ingestion rate is set at 100 mg/day.

Residential Scenario: Children and Adults

A scenario relating to current residential exposures resulting from migration of contaminants in ground water to off site wells which are east/northeast of the site, was not constructed because sampling results suggest a lack of contamination of currently existing wells (see Section 4.2). However, a future use residential scenario was constructed to evaluate the possible risks associated with residing on the site as it currently exists.

Table I-4 (Appendix I) presents the model inputs for the exposure routes that children and adults who live on site might receive. Children, aged 0-6 years, and adults are modeled to receive exposures through soil/house dust ingestion, dermal contact with soil based upon exposed arms, hands and legs, dermal contact with contaminants in water during showering, inhalation of contaminants in dust outdoors from wind erosion, inhalation of volatile organic compounds released into bathroom air during showering, and ingestion of contaminants in drinking water. These exposures are assumed to occur on 365 days/year for 6 years for children and 30 years for adults. The time period for outdoor exposure to fugitive dusts is 4 hours/day, and for showering, is 12 minutes/day. Children are assumed to ingest 756 ml water and 200 mg of soil/house dust per day, while for adults, these values of 2 liters of water and 100 mg soil/day.

6.4.3 Estimating Environmental Concentrations

All exposure point concentrations used in assessing receptor dose were calculated as specified in Chapter 6 of the Risk Assessment Guidance for Superfund (EPA December, 1989). This statistical method uses a confidence interval to calculate a theoretical concentration from actual on site samples. The confidence interval to be used, as specified by EPA (EPA December, 1989), is the 95% upper confidence limit. The results of this method represent an "upper bound" on the average concentration; the probability that the actual average concentration on the site exceeds this value is estimated to be less than 5%. The confidence intervals for this application were calculated for a log-normal distribution. This distribution was chosen based on an examination of the measured data. Most measured concentrations are relatively low, with a few values higher than the majority by at least one order of magnitude. For this type of data set, the log-normal distribution is more suitable than the standard normal distribution. The 95th % upper confidence limit was calculated for each compound in each environmental media based on actual compound concentrations found on-site. The upper confidence limits used in this assessment were calculated using the following formula:

$$UL_{95} = \exp \left(Yave + 0.5\ Sy^2 + \frac{Sy\ H_{95})}{\sqrt{n-1}} \right)$$

where

UL_{95} = the 95th percentile upper confidence limit average concentration value

y = lnC

Yave = the average of the natural logarithms of all concentrations

S_y = the standard deviation of the logarithms of the concentrations

n = the number of samples

H_{95} = a statistical parameter which depends upon n and sy, obtained from a look-up table (Gilbert, 1987).

As indicated in section 6.4 (Data Evaluation) non-detected values were included in the calculation of exposure point concentrations (i.e., soil concentrations). These non-detected values included both detection limits (indicated by a "U" qualifier) and estimated sample quantitation limits (SQL) (indicated by a "UJ" qualifier). In general, detection limits were used as reported, while SQL were evaluated in light of detection limits and quantifiable ("hits") concentrations of each contaminant. Each SQL was independently analyzed and used either as the estimated SQL or one half (1/2) of the SQL.

6.4.4 Evaluating Uncertainty

Tables 6-1 through 6-5 summarize contaminant concentrations in soil, sediment, and surface water both as a range of detection across the site and as the value used (either the 95% UCL or the maximum detected concentration) in the risk assessment. Table 6-6 provides a summary of ranges of detected contaminants across all media.

Table 6-7 summarizes the assumptions used to estimate exposure (i.e., soil ingestion rate, exposure frequency, etc.).

The exposure estimates produced for each receptor in each scenario are based on numerous variables with varying degrees of uncertainty. This discussion will focus on these parameters, and the associated range of uncertainty. Table 6-7 summarizes the parameters and values used to estimate exposure. The table is separated into those parameters which apply to all three scenarios (i.e., global variables), and those which apply specifically to an individual scenario.

- Global Variables (Scenarios 1-3)

Table 6-7 lists the parameters and associated values which are used in each of the three scenarios. Body weight ranges for children (age 9-18 years) were derived from EPA (1990b). The actual value used (49 kg) represents an average body weight for this group. Similarly, for children ages 0-6 and adults (18-65 years), a range of body weights is presented, along with the average body weight (16 kg and 70 kg, respectively) for the group. While there is a range of body weights for each age group, these ranges are not large, and are not expected to contribute a significant degree of uncertainty to this assessment.

The exposure durations (ED) used for Scenarios 1 and 3 were separated into categories for children and adults. For Scenario 1, children were assumed to spend a duration of nine years at the site, based upon the age range of children likely to trespass onto the site. In theory, this duration might range from 1 to 18 years, however, it is unlikely that children younger than 6 years of age would visit the site. For Scenario 3, children ages 0-6 were expected to spend the entire six year time frame on site. The value used is the high end of the proposed range (1-6 years). For Scenarios 1 and 3, adults were assumed to have an ED equal to 30 years, which is the national upper-bound (90th percentile) time at one residence. The ED range is 1-70 years, which spans the expected lifetime. Finally, construction workers (Scenario 2) were expected to have an ED of 1 year, based on the amount of time spent building new homes at the site.

The ranges associated with ED are only large when considering adults. However, the values used are expected to provide conservative estimates and overstate the potential risk.

Averaging time (AT) which is a pathway specific period of exposure for non-carcinogenic effects, calculated as a product of exposure duration and the number of days/year, is dependent on exposure duration, which was discussed above. AT is not expected to lend a large degree of uncertainty to the exposure estimates.

The ranges of absorption factors (AF) for organic and inorganic compounds vary from no absorption (0) to complete absorption (1). This range is likely to contribute a large degree of uncertainty to the exposure estimates. The values chosen for AF are representative for classes of compounds, with the exception of arsenic for which a specific value was used.

The permeability constant (PC) for each chemical was assumed to be equal to the penetration rate of water, rather than on a compound specific basis. Thus, PC may lend a degree of uncertainty in that some compounds will not readily penetrate skin, while others will penetrate at a rapid rate.

The range of adherence factor of soil to skin is small (0-2.77 mg/cm^2). Based upon the adherence of potting soil to skin, a value of 1.45 mg/cm^2 was used.

The fraction of soil ingested (FI) from the site ranges from 0-1. As a highly conservative estimate, and based on an event-based approach, it was assumed that all soil ingested came from the site.

Finally, concentrations of contaminants in all media were presented as a 95% UCL or as a maximum detected concentration (see section 6.4.3). For some chemicals the range of potential concentrations across the site is very large, introducing a high degree of uncertainty to the exposure estimates. However, the exposure estimates are expected to over predict rather than under predict, and therefore are protective of human health.

- Scenario 1 - Recreational Exposure: Current Use

The exposure frequency (EF; days/year) may range from 1 to 365, which may introduce the greatest degree of uncertainty. The values used (90 days for children and 60 days for adults) were based on available free time (away from work, school, etc.). Skin surface area, exposure time and soil ingestion rate also present a large range of values but the values used are not expected to introduce a large degree of uncertainty into the exposure estimates.

- Scenario 2 - Construction Exposure: Future Use

Of the parameters presented in Table 6-7, the modeled ambient dust concentration is expected to present the largest degree of uncertainty to the exposure estimates. Exposure point concentrations available at the Hertel site include concentrations in soils, sediments and water. However, airborne concentrations of contaminants (i.e., volatilization, fugitive dusts) were not sampled during the field program and thus exposure point concentrations must be modeled. Names and citations for the transport models used to estimate exposure point concentrations from laboratory measurements of field samples are given in Appendix I. As a caveat, it is always more accurate to have data for exposure point concentrations in the medium of concern at the exposure point of concern, and the use of transport models represents a good faith attempt to estimate unknown values from known values. However, the use of the models does introduce uncertainty into the results. Of the remaining parameters, the ranges of skin surface area are quite large, and may contribute a large degree or uncertainty to the exposure estimates.

- Scenario 3 - Residential Scenario: Future Use

Of the parameters presented in Table 6-7, the modeling of ambient dust concentrations and indoor airborne vapor phase chemical concentrations are expected to present the largest degree of uncertainty. Exposure point concentrations available at the Hertel site include concentrations in soils, sediments and water. However, airborne concentrations of contaminants (i.e., volatilization, fugitive dusts) were not sampled during the field program and thus exposure point concentrations must be modeled. Names and citations for the transport models used to estimate exposure point concentrations from laboratory measurements of field samples are given in Appendix I. As a caveat, it is always more accurate to have data for exposure point concentrations in the medium of concern at the exposure point of concern, and the use of transport models represents a good faith attempt to estimate unknown values from known values. However, the use of the models does introduce uncertainty into the results.

6.5 Toxicity Assessment

This section of the report presents a short description of the toxic effects of each chemical of concern including effects associated with exposure and concentrations at which such effects may be expected to occur, when available. Furthermore, Tables 6-8 through 6-11 present a summary of toxicity values associated with chronic and subchronic noncarcinogenic effects, for the oral and inhalation routes, respectively. Tables 6-12 and 6-13 summarize the slope factors associated with potential carcinogenic effects of chemicals of concern by the oral and inhalation routes, respectively.

6.5.1 Inorganics

Aluminum

There is inadequate data for the quantitative risk assessment of aluminum. Aluminum is an ubiquitous metal and can be found in the air, water and soil. Ingestion of aluminum containing compounds can alter intestinal function as well as change the absorption of other elements in the gastrointestinal tract. Chronic inhalation of aluminum leads to pulmonary fibrosis which results in both restrictive and obstructive pulmonary disease (Goyer, 1986).

Antimony

The best characterized human health effect associated with the inhalation of antimony is myocardial damage. The suggested no-observed-adverse-effect-level (NOAEL) for antimony induced myocardial damage is 0.003 mg antimony/kg body weight (bw)/day (mg/kg/day). The chronic oral Reference Dose (RfD) for antimony is 4E-04 mg/kg/day (EPA, 1991), and is based on a chronic rat bioassay. Rats were administered 5 ppm (0.35 mg/kg bw/day) potassium antimony tartrate in drinking water for two years. The critical effects associated with this study are a decrease in longevity, a decrease in fasting blood glucose levels and an alteration in cholesterol levels. An uncertainty factor of 1000 was applied to the lowest observed adverse effect level (LOAEL) of 0.35 mg/kg bw/day to obtain the RfD. The confidence level in this Rfd is low since there was only 1 dose level of antimony used and no observed adverse effect level (NOAEL) was established.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Arsenic

Symptoms of arsenic intoxication consist of fever, anorexia, hepatomegaly, melanosis, and cardiac arrythmia. Other features include upper respiratory tract symptoms, peripheral neuropathy, and gastrointestinal, cardiovascular and hematopoietic effects. Liver injury is characteristic of longer term or chronic exposure (Goyer, 1986).

The chronic oral RfD is 1E-03 mg/kg/day (EPA, 1991a). The critical effects associated with arsenic ingestion are keratosis and hyperpigmentation at a dose of 1 ug/kg/day in humans.

The EPA weight of evidence classification for the carcinogenicity of this compound is "A" - a human carcinogen. Exposure to arsenic by the oral route is known to produce skin cancer, while inhalation will cause lung cancer. The slope factors for these carcinogenic effects are 5E-05 ug/l and 4.3E-03 ug/m^3, respectively (EPA, 1991).

Barium

The chronic oral RfD for barium is 7E-02 mg/kg/day (EPA, 1991). Increasing doses of barium were administered to human volunteers in drinking water for up to ten weeks. An NOAEL of 10 mg/l (.21 mg/kg/day) was established. The critical effect of chronic human exposure to barium is increased blood pressure. An uncertainty factor of 3 was applied to the NOAEL to obtain the RfD. The confidence level for this RfD is medium.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Beryllium

The chronic oral RfD for beryllium is 5E-03 mg/kg/day (EPA, 1991). Beryllium was administered to rats over their lifetime in their drinking water at a concentration of 5 ppm (0.54 mg/kg/day). There were no observed adverse effects. An uncertainty factor of 100 was applied to the NOAEL to obtain the RfD. Chronic occupational exposure to beryllium results in chronic granulomatous pulmonary disease (berylliosis).

The EPA weight of evidence classification for the carcinogenicity of this compound is "B2" - a probable human carcinogen (sufficient animal evidence, inadequate/no human evidence). The oral slope factor for beryllium is 4.3 mg/kg/day^{-1} (EPA, 1991).

Cadmium

Ingestion of cadmium results in nausea, vomiting and abdominal pain. Inhalation of cadmium fumes may result in an acute chemical pneumonitis and pulmonary edema (Goyer, 1986).

The chronic oral RfDs for cadmium are 5E-04 mg/kg/day (water) and 1E-03 mg/kg/day (food) (EPA, 1991). The critical effects associated with chronic ingestion of cadmium are proteinuria and renal damage in humans. An uncertainty factor of 10 was applied in order to determine the RfD. The confidence level for this RfD is high.

The EPA weight of evidence classification for the carcinogenicity of this compound is "B1" - a probable human carcinogen (limited human evidence). The inhalation of cadmium has been shown to produce respiratory tract cancers and an inhalation slope factor of 6.1E+0 mg/kg/day^{-1} has been established (EPA, 1991).

Chromium VI

Note: Total chromium only was measured on site. All detected chromium in all media was assumed to be chromium VI. This is a highly conservative assumption.

The chronic oral Rfd for chromium VI is 5E-03 mg/kg/day (EPA, 1991). No adverse effects were observed in rats which received 0-11 mg/l or 25 mg/l chromium in drinking water for 1 year. No adverse effects were seen in humans drinking well water contaminated with 1 mg/l chromium VI for 3 years. An uncertainty factor of 500 was applied to the LOAEL to obtain the RfD. The confidence level in this RfD is low.

The EPA weight of evidence classification for carcinogenicity of this compound by the inhalation route is "A" - a human carcinogen. Chromium VI produces lung tumors and an inhalation slope factor of 4.1E+01 mg/kg/day^{-1} has been established (EPA, 1991). There is insufficient evidence for carcinogenicity of this compound by the oral route.

Cobalt

Cobalt is essential as a component of vitamin B12 which is required for the production of red blood cells. Cobalt is well absorbed orally, probably in the small intestine. Excessive cobalt intake is known to result in cardiomyopathy. One ppm cobalt was added to beer to enhance its foaming properties and the resultant signs and symptoms were those of congestive heart failure. Autopsy findings revealed a ten-fold increase in the cardiac levels of cobalt. Occupational exposure may result in respiratory symptoms (Goyer, 1986). No RfDs were found in either Integrated Risk Information Service (IRIS) or Health Effects Assessment Summary Tables (HEAST).

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Copper

A single dose of 5.3 mg copper resulted in local gastrointestinal tract irritation in humans. A chronic oral RfD is reported as 1.3 mg/l, which is the current drinking water standard for copper (EPA, 1991a). The Drinking Water Criteria Document concluded toxicity data were inadequate for calculating an actual RfD for copper.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Lead

The health effects of lead have been well characterized through decades of medical and scientific observation. Some of these effects include cognitive and motor defects in children, lead induced anemias, increased susceptibility to viral infections and in chronic adult lead poisoning, peripheral neuropathies. It appears that some of these effects particularly the changes in the levels of certain blood enzymes and in aspects of children's neurobehavioral development, may occur at blood lead levels so low as to be essentially without a threshold. Therefore the EPA has considered it inappropriate to develop an RfD for inorganic lead (Goyer, 1986; EPA, 1991).

The EPA weight of evidence classification for the carcinogenicity of this compound is "B2" - a probable human carcinogen (sufficient animal evidence, inadequate/no human evidence). Lead has been shown to produce renal tumors, however due to the many uncertainties associated with quantifying lead's cancer risk, it has been recommended that a numerical estimate not be used (EPA, 1991). Although not directly applicable to this risk assessment, the federal Maximum Contaminant Level for lead in drinking water is 50 ppb.

Manganese

Exposure to manganese results in two types of toxicities. The first, the result of acute inhalation exposure, results in manganese pneumonitis. The second, and more serious of the two, results from chronic exposure to manganese either by the oral or inhalation routes. Chronic manganese poisoning results in a psychiatric disorder characterized by psychological and motor difficulties (Goyer, 1986). The chronic oral RfD has been set at 1E-01 mg/kg/day (EPA, 1991) in order to prevent the central nervous system effects. The confidence level in this RfD is medium.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Mercury

Exposure to mercury vapor may produce an acute, corrosive bronchitis and interstitial pneumonitis resulting in either death or symptoms of central nervous system effects such as tremor or increased excitability. Ingestion of mercuric salts results in corrosive ulceration, bleeding and necrosis of the gastrointestinal tract usually accompanied by shock and circulatory collapse. Renal failure occurs within 24 hours. Chronic mercury poisoning mainly affects the central nervous system. Characteristic symptoms include increased excitability, tremors, gingivitis, and increased salivation. There have been some instances of proteinuria and renal damage in persons chronically exposed to mercury vapors (Goyer, 1986). The chronic oral RfD for mercury is 3E-04 mg/kg/day (EPA, 1991a), in order to prevent the critical effect of renal damage. An uncertainty factor of 1000 was applied in order to determine the RfD.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Nickel

Nickel is a common allergen which results in allergic contact dermatitis (Goyer, 1986). The chronic oral RfD of nickel is 2E-02 mg/kg/day (EPA, 1991), and is based on a chronic rat bioassay. Rats were administered 0, 5, 50, 125 mg/kg bw/day in their drinking water over their lifetimes. The critical effects associated with this study are a decrease in body and organ weights. Clinical signs in this study included lethargy, ataxia, irregular breathing, decreased body temperature and salivation. An uncertainty factor of 100 and an additional modifying factor of 3 was applied to obtain the RfD. The confidence level in this Rfd is medium.

The EPA weight of evidence classification for carcinogenicity of this compound by the inhalation route is "A" - a human carcinogen. Nickel produces lung tumors and an inhalation slope factor of 8.4E- 01 $mg/kg/day^{-1}$ has been established (EPA, 1991). There is insufficient evidence for carcinogenicity of this compound by the oral route.

Selenium

The availability as well as toxic potential of selenium is related to its chemical form. Selenates are readily absorbed from the gastrointestinal tract whereas elemental selenium is probably not absorbed. Acute selenium poisoning produces central nervous system effects including nervousness, drowsiness and sometimes convulsions. Eye and nasal irritation may occur from exposure to vapors. Signs of chronic selenium intoxication in humans may include discolored or decaying teeth, skin eruptions, gastrointestinal distress, lassitude and partial loss of hair and nails (Goyer, 1986). No RfDs were found in either IRIS or HEAST.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Vanadium

Vanadium is an ubiquitous element. The toxic action of vanadium is mostly confined to the respiratory tract. Industrial exposure results in bronchitis and bronchopneumonia. Ingestion of vanadium produces gastrointestinal disturbances, slight renal dysfunction and nervous system effects (Goyer, 1986). The chronic oral RfD for vanadium is 7E-03 mg/kg/day (EPA, 1991a) and is based on a chronic rat bioassay. Rats received 0.7 mg/kg/day vanadium in their drinking water over their lifetimes and no toxic effects were observed. An uncertainty factor of 100 was applied to the NOAEL to obtain the RfD.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Zinc

Zinc is ubiquitous in the environment so that it is present in most food stuffs, water and air. About 20 to 30 percent of ingested zinc is absorbed. Acute toxicity from the ingestion of excessive zinc is uncommon (Goyer, 1986). The chronic oral RfD for zinc is 2E-01 mg/kg/day (EPA, 1991a). This value is based on a therapeutic dosage of 2.14 mg/kg/day in man which resulted in anemia. An uncertainty factor of 10 was applied to obtain the RfD.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Cyanide

Cyanide is readily absorbed from all routes including the skin, mucous membranes, and by inhalation. Death can occur with ingestion of even small amounts of sodium or potassium cyanide. Inhalation of toxic fumes represents a potentially rapid fatal type of exposure. Symptoms of poisoning include salivation, nausea and vomiting, confusion, vertigo, stiffness, convulsions, paralysis, coma, respiratory stimulation followed by respiratory failure and death (Rumack and Lovejoy, Jr., 1986). The chronic oral RfD is 2E-02 mg/kg/day (EPA, 1991) based on a 2 year dietary study in rats. Rats received approximately 4.3 and 10.8 mg/kg/day and no treatment effects were observed. An uncertainty factor of 100 as well as a modifying factor of 5 was applied to the NOAEL of 10.8 mg/kg/day to obtain the RfD. The critical effects associated with chronic cyanide ingestion were obtained in a chronic rat study in which rats received 30 mg/kg/day cyanide. The toxic effects observed were decreased weight gain, decreased thyroxin levels and myelin degeneration. There is inadequate human data since chronic cyanide ingestion by humans has not been documented.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

6.5.2 Volatiles

Carbon Disulfide

Exposure to high concentrations of carbon disulfide results in organic brain damage, peripheral neuropathies, neurobehavioral dysfunction, and ocular and auditory effects. Occupational exposure to carbon disulfide has also been called a major contributing factor to coronary heart disease (Andrews and Snyder, 1986). Carbon disulfide may be absorbed through the skin as vapor or liquid, inhaled or ingested. The chronic oral RfD is 1E-01 mg/kg/day (EPA, 1991) based on a chronic rabbit inhalation teratogenic study. Rabbits and rats were exposed to 20 and 40 ppm carbon disulfide for 34 weeks prior to pregnancy as well as throughout gestation. There were no effects seen on fetal development. An uncertainty factor of 100 was applied to the NOAEL of 11.0 mg/kg/day to obtain the RfD. The confidence level in this RfD is medium.

This compound has not been evaluated by the EPA for evidence of human carcinogenicity.

Chlorobenzene

The chronic oral RfD for chlorobenzene is 2E-02 mg/kg/day (EPA, 1991) and is based on a 13 week dog study. Beagle dogs received chlorobenzene orally by capsule at doses of 27.25, 54.5, or 272.5 mg/kg/day for 5 days/wk for 13 weeks. The LOAEL was 54.5 mg/kg/day and the critical effects observed were histopathological changes in the liver as well as changes in the blood chemistry. An uncertainty factor of 1000 was applied to the NOAEL of 19 mg/kg/day (adjusted from 27.25 mg/kg/day to take into account X exposure) to obtain the RfD. The confidence level in this RfD is medium.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Ethylbenzene

The chronic oral RfD for ethylbenzene is 1E-01 mg/kg/day (EPA, 1991) and is based on a oral subchronic rat bioassay. Rats received oral doses of 13.6, 136, 408, or 680 mg/kg/day in olive oil for 26 weeks. The LOAEL was 408 mg/kg/day and the critical effects observed were liver and kidney toxicity. An uncertainty factor of 1000 was applied to the NOAEL of 97.1 mg/kg/day (adjusted from 136 mg/kg/day to take into account 5/7 day exposure) to obtain the RfD. The confidence level in this RfD is low. There were no adverse effects seen in human volunteers exposed to 100 ppm (435 mg/cu.m) for eight hours.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Toluene

The chronic oral RfD for toluene is 2E-01 mg/kg/day (EPA, 1991) and is based on a subchronic oral gavage study in rats. F344 rats received oral doses of 0, 312, 625, 1250, 2500, or 5000 mg/kg/day for 5 days/wk for 13 weeks. The LOAEL was 625 mg/kg/day and the critical effects observed were changes in liver and kidney weights. An uncertainty factor of 1000 was applied to the NOAEL of 223 mg/kg/day (adjusted from 312 mg/kg/day to take into account 5/7 day exposure) to obtain the RfD. The confidence level in this RfD is medium. There were no adverse effects seen in human volunteers exposed to 100 ppm for twenty minutes. When exposed to 200 ppm for twenty minutes they exhibited incoordination, exhilaration, and prolonged reaction times.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Xylenes

The chronic oral RfD for toluene is 2E+00 mg/kg/day (EPA, 1991) and is based on a chronic oral gavage study in rats and mice. Rats and mice were given oral gavage doses of 0, 250 or 500 mg/kg/d (rats) and 0, 500 or 1000 mg/kg/d (mice) for 5 days/wk for 105 weeks. There was a dose-related increase in the mortality levels seen in male rats, as well as hyperactivity and decreased body weights. An uncertainty factor of 100 was applied to the NOAEL of 179 mg/kg/day (adjusted from 250 mg/kg/day to take into account 5/7 day exposure) to obtain the RfD. The confidence level in this RfD is medium. The

EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

6.5.3 Semivolatiles

Bis(2-ethylhexyl)phthalate

The chronic oral RfD for bis(2-ethylhexyl)phthalate (BEHP) is 2E-02 mg/kg/day (EPA, 1991) and is based on a subchronic feeding study in guinea pigs. Guinea pigs received 19 or 64 mg/kg/day BEHP in their food for 1 year. There were no treatment related toxic effects, however both dose groups had increased liver weights. An uncertainty factor of 1000 was applied to the LOAEL of 19 mg/kg/day to obtain the RfD. The confidence level in this RfD is medium.

The EPA weight of evidence classification for the carcinogenicity of this compound is "B2" - a probable human carcinogen (sufficient animal evidence, inadequate/no human evidence). The oral slope factor for BEHP is 1.4E-02 mg/kg/day^{-1} (EPA, 1991), and has been shown to produce liver tumors in an animal model.

Butylbenzylphthalate

The chronic oral RfD for butylbenzylphthalate is 2E-01 mg/kg/day (EPA, 1991) and is based on a subchronic feeding study in rats. Rats received 0, 17, 51, 159, 470, 1417 mg/kg/day butylbenzylphthalate in their diet for 26 weeks. The LOAEL was 470 mg/kg/day and the critical effects observed were a decrease in body weight, decreased testes' size, decreased organ weights and hematological effects. An uncertainty factor of 1000 was applied to the NOAEL of 159 mg/kg/day to obtain the RfD. The confidence level in this RfD is medium.

The EPA weight of evidence classification for the carcinogenicity of this compound is "C" - a possible human carcinogen (EPA, 1991).

Benzo(a)anthracene

No RfDs were found in either IRIS or HEAST.

The EPA weight of evidence classification for the carcinogenicity of this compound is "B2" - a probable human carcinogen (sufficient animal evidence, inadequate/no human evidence). No oral slope factor for benzo(a)anthracene has been established, however this compound has been shown to produce liver, lung and skin cancer in animal models (EPA, 1991). Current EPA guidance suggests the use of an oral slope factor of 11.5 $mg/kg/day^{-1}$ and an inhalation slope factor of 6.1 $mg/kg/day^{-1}$. These values are derived from experimental data utilizing benzo(a)pyrene as the test compound.

Benzo(a)pyrene

No RfDs were found in either IRIS or HEAST.

The EPA weight of evidence classification for the carcinogenicity of this compound is "B2" - a probable human carcinogen (sufficient animal evidence, inadequate/no human evidence). Benzo(a)pyrene has been shown to produce lung and stomach cancer in animal models (EPA, 1991). The oral and inhalation slope factors for benzo(a)pyrene have been withdrawn by EPA. As an interim measure, the withdrawn values have been recommended for use by EPA. These values are 11.5 and 6.1 $mg/kg/day^{-1}$ for the oral and inhalation routes, respectively.

Chrysene

Data has been determined to be inadequate for quantitative risk assessment (EPA, 1991a).

The EPA weight of evidence classification for the carcinogenicity of this compound is "B2" - a probable human carcinogen (sufficient animal evidence, inadequate/no human evidence). No oral slope factor for chrysene has been established, however this compound has been shown to produce carcinomas, and malignant lymphomas in mice (EPA, 1991). Current EPA guidance suggests the use of an oral slope factor of 11.5 mg/kg/day^{-1} and an inhalation slope factor of 6.1 mg/kg/day^{-1}. These values are derived from experimental data utilizing benzo(a)pyrene as the test compound.

Benzo(b)fluoranthene

No RfDs were found in either IRIS or HEAST.

The EPA weight of evidence classification for the carcinogenicity of this compound is "B2" - a probable human carcinogen (sufficient animal evidence, inadequate/no human evidence). No oral slope factor for benzo(b)fluoranthene has been established, however this compound has been shown to produce lung and thorax carcinomas, lung adenomas and skin tumors in animal models (EPA, 1991). Current EPA guidance suggests the use of an oral slope factor of 11.5 mg/kg/day^{-1} and an inhalation slope factor of 6.1 mg/kg/day^{-1}. These values are derived from experimental data utilizing benzo(a)pyrene as the test compound.

Benzo(k)fluoranthene

No RfDs were found in either IRIS or HEAST.

The EPA weight of evidence classification for the carcinogenicity of this compound is "B2" - a probable human carcinogen (sufficient animal evidence, inadequate/no human evidence). No oral slope factor for benzo(k)fluoranthene has been established, however this compound has been shown to produce lung and thorax carcinomas, lung adenomas and skin tumors in animal models (EPA, 1991). Current EPA guidance suggests the use of an oral slope factor of 11.5 mg/kg/day^{-1} and an inhalation slope factor of 6.1 mg/kg/day^{-1}. These values are derived from experimental data utilizing benzo(a)pyrene as the test compound.

Dibenzo(a,h)anthracene

No RfDs were found in either IRIS or HEAST.

The EPA weight of evidence classification for the carcinogenicity of this compound is "B2" - a probable human carcinogen (sufficient animal evidence, inadequate/no human evidence). No oral slope factor for dibenzo(a,h)-anthracene has been established, however this compound has been shown to produce lung and mammary tumors after oral administration as well as fibrosarcomas after subcutaneous injection in animal models (EPA, 1991). Current EPA guidance suggests the use of an oral slope factor of 11.5 mg/kg/day^{-1} and an inhalation slope factor of 6.1 mg/kg/day^{-1}. These values are derived from experimental data utilizing benzo(a)pyrene as the test compound.

Indeno(1,2,3-cd)pyrene

No RfDs were found in either IRIS or HEAST.

The EPA weight of evidence classification for the carcinogenicity of this compound is "B2" - a probable human carcinogen (sufficient animal evidence, inadequate/no human evidence). No oral slope factor for indeno(1,2,3-cd)pyrene has been established, however this compound has been shown to produce lung and skin tumors in animal models (EPA, 1991). Current EPA guidance suggests the use of an oral slope factor of 11.5 mg/kg/day^{-1} and an inhalation slope factor of 6.1 mg/kg/day^{-1}. These values are derived from experimental data utilizing benzo(a)pyrene as the test compound.

1,4-Dichlorobenzene

No RfDs were found in either IRIS or HEAST.

The EPA weight of evidence classification for the carcinogenicity of this compound is "B2" - a probable human carcinogen (sufficient animal evidence, inadequate/no human evidence). The oral slope factor for 1,4-dichlorobenzene is 2.4E-02 mg/kg/day^{-1} (EPA, 1991a). In a 103 week oral gavage study in mice, 1,4-dichlorobenzene produced liver tumors.

Diethylphthalate

The chronic oral RfD for diethylphthalate is 8E-01 mg/kg/day (EPA, 1991) and is based on a subchronic feeding study in rats. Rats received 0, 150, 770, and 3160 mg/kg/day diethylphthalate in their diet for 16 weeks. The LOAEL was 3160 mg/kg/day and the critical effects observed were a decrease in body weight, decreased food consumption and altered organ weights. No changes in behavior or other clinical signs of toxicity were observed. An uncertainty factor of 1000 was applied to the NOAEL of 770 mg/kg/day to obtain the RfD. The confidence level in this RfD is low.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Di-n-butylphthalate

The chronic oral RfD for di-n-butylphthalate is 1E-01 mg/kg/day (EPA, 1991) and is based on a subchronic feeding study in rats. Rats received 0, 0.01, 0.05, 0.25 and 1.25 percent di-n-butylphthalate in their diet for 1 year. The LOAEL was 600 mg/kg/day (1.25%) and the critical effect observed was an increase in mortality. No changes in behavior or other clinical signs of toxicity were observed. An uncertainty factor of 1000 was applied to the NOAEL of 125 mg/kg/day (0.25%) to obtain the RfD. The confidence level in this RfD is low.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Di-n-octylphthalate

The chronic oral RfD for di-n-octylphthalate is 2E-02 mg/kg/day (EPA, 1991a) and is based on a subchronic feeding study in rats. An uncertainty factor of 1000 was applied to the LOAEL of 175 mg/kg/day to obtain the RfD. The critical effects observed at the LOAEL were elevated kidney and liver weights and increased SGOT and SGPT levels.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Acenaphthene

The chronic oral RfD for acenaphthene is 6E-02 mg/kg/day (EPA, 1991) and is based on a subchronic gavage study in mice. Mice received 0, 175, 350, or 700 mg/kg/day acenaphthene by oral gavage for 90 days. The LOAEL was 350 mg/kg/day and the critical effects observed were liver weight changes accompanied by microscopic alterations. No treatment related effects on survival, clinical signs or body weight changes were observed. An uncertainty factor of 3000 was applied to the NOAEL of 175 mg/kg/day to obtain the RfD. The confidence level in this RfD is low.

This compound has not yet been evaluated by the EPA for evidence of human carcinogenic potential (EPA, 1991).

Anthracene

The chronic oral RfD for anthracene is 3E-01 mg/kg/day (EPA, 1991) and is based on a subchronic gavage study in mice. Mice received 0, 250, 500, or 1000 mg/kg/day anthracene by oral gavage for 90 days. No treatment related effects on survival, clinical signs or body weight changes were observed. An uncertainty factor of 3000 was applied to the NOAEL of 1000 mg/kg/day to obtain the RfD. The confidence level in this RfD is low.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Benzo(ghi)perylene

No RfDs were found in either IRIS or HEAST.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Fluoranthene

The chronic oral RfD for fluoranthene is 4E-02 mg/kg/day (EPA, 1991) and is based on a subchronic gavage study in mice. Mice received 0, 125, 250, or 500 mg/kg/day fluoranthene by oral gavage for 13 weeks. The LOAEL was 250 mg/kg/day and the critical effects seen were neuropathy, increased salivation and increased liver enzymes. An uncertainty factor of 3000 was applied to the NOAEL of 125 mg/kg/day to obtain the RfD. The confidence level in this RfD is low.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Fluorene

The chronic oral RfD for fluorene is 4E-02 mg/kg/day (EPA, 1991) and is based on a subchronic gavage study in mice. Mice received 0, 125, 250, or 500 mg/kg/day fluorene by oral gavage for 13 weeks. The LOAEL was 250 mg/kg/day and the critical effects seen were neuropathy, increased salivation and increased liver enzymes. An uncertainty factor of 3000 was applied to the NOAEL of 125 mg/kg/day to obtain the RfD. The confidence level in this RfD is low.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

4-Methylphenol

The chronic oral RfD for 4-methylphenol is 5E-02 mg/kg/day (EPA, 1991) and is based on a subchronic gavage study in mice. Mice received 0, 50, 175, or 600 mg/kg/day 4-methylphenol by oral gavage for 13 weeks. The LOAEL was 175 mg/kg/day and the critical effects seen were CNS effects such as lethargy, increased salivation and tremors, as well as diarrhea. An uncertainty factor of 1000 was applied to the NOAEL of 50 mg/kg/day to obtain the RfD. The confidence level in this RfD is low.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Naphthalene

The chronic oral RfD for naphthalene is 4E-02 mg/kg/day[c] and is based on a subchronic gavage study in rats. An uncertainty factor of 1000 was applied to the LOAEL of 35.7 mg/kg/day to obtain the RfD. The critical effect observed in this study was decreased body weight gain.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Pyrene

The chronic oral RfD for pyrene is 3E-02 mg/kg/day (EPA, 1991) and is based on a subchronic gavage study in mice. Mice received 0, 75, 125, or 250 mg/kg/day pyrene by oral gavage for 13 weeks. The LOAEL was 125 mg/kg/day and the critical effects seen were toxic effects to the kidney including changes to the renal tubular pathology and decreased kidney weight. An uncertainty factor of 3000 was applied to the NOAEL of 75 mg/kg/day to obtain the RfD. The confidence level in this RfD is low.

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

Phenanthrene

Data has been determined to be inadequate for quantitative risk assessment (EPA, 1991).

The EPA weight of evidence classification for the carcinogenicity of this compound is "D" - not classifiable as to human carcinogenicity (EPA, 1991).

6.5.4 Pesticides/PCBs

4,4'- DDD

No RfDs were found in either IRIS or HEAST.

The EPA weight of evidence classification for the carcinogenicity of this compound is "B2" - a probable human carcinogen (sufficient animal evidence, inadequate/no human evidence). The oral slope factor for 4,4'- DDD is 2.4E-01 mg/kg/d^{-1} and has been shown to produce liver tumors in animal models (EPA, 1991).

4,4'- DDE

No RfDs were found in either IRIS or HEAST.

The EPA weight of evidence classification for the carcinogenicity of this compound is "B2" - a probable human carcinogen (sufficient animal evidence, inadequate/no human evidence). The oral slope factor for 4,4'- DDE is 3.4E-01 mg/kg/d^{-1} and has been shown to produce liver tumors in animal models (EPA, 1991).

4,4'- DDT

The chronic oral RfD for 4,4'- DDT is 5E-04 mg/kg/day (EPA, 1991) and is based on a subchronic feeding study in rats. Rats received 0, 1, 5, 10, or 50 ppm 4,4' - DDT in their food for 15 - 27 weeks. The LOAEL was 0.25 mg/kg/day (5 ppm diet) and the critical effects seen were histopathological effects to the liver. An uncertainty factor of 100 was applied to the NOAEL of 0.05 mg/kg/day (1 ppm diet) to obtain the RfD. The confidence level in this RfD is medium.

The EPA weight of evidence classification for the carcinogenicity of this compound is "B2" - a probable human carcinogen (sufficient animal evidence, inadequate/no human evidence). The oral slope factor for 4,4'-DDT is 3.4E-01 mg/kg/d^{-1} and has been shown to produce liver tumors in animal models (EPA, 1991).

6.6 Risk Characterization

6.6.1 Quantitative Risk Assessment

For potential carcinogens, risks are estimated as probabilities. The compound-specific potency factors for carcinogens are generally estimated through the use of mathematical extrapolation models (e.g., the linearized multistage model). These models estimate the largest possible linear slope, within a 95% confidence interval, at low extrapolated doses. Thus, the potency factor is characterized as a 95% upperbound estimate, such that the true risk is not likely to exceed the upperbound estimate and may be lower.

The evaluation of risk from noncarcinogenic health hazards is based on the use of RfDs (EPA, 1990; EPA, 1989a). RfDs are estimates of daily exposure to the population (including sensitive subpopulations) that are likely to be without appreciable risk of deleterious effects for the defined exposure period. The RfD is calculated by dividing the NOAEL or LOAEL derived from animal or human studies by an uncertainty factor, which is multiplied by a modifying factor. RfDs incorporate uncertainty factors which serve as a conservative downward adjustment of the numerical value and reflect scientific judgement regarding the data used to estimate the RfD. For example, a factor of 10 is used to account for variations in human sensitivity (i.e., to protect sensitive subpopulations) when the data stems from human studies involving average, healthy subjects. An additional factor of 10 may also be used for each of the following:

- extrapolation from chronic animal studies to humans,
- extrapolation from a lowest observable adverse effect level (LOAEL) to a no observed adverse effect level (NOAEL), and
- extrapolation from subchronic to chronic studies.

Finally, based on the level of certainty of the study and database, an additional modifying factor (between zero and ten) may be used.

The results of the quantitative risk analysis are presented in two basic forms. In the case of human health effects associated with exposure to potential carcinogens, risk estimates are expressed as the lifetime probability of additional cancer risk associated with the given exposure. In numerical terms, these are presented in scientific notation in this report. Thus, a lifetime risk of 1E-04 means a lifetime incremental risk of one in ten thousand; a lifetime risk of 1E-06 means an incremental lifetime risk of one in one million and so on.

In the cases of exposure to non-carcinogens, the Hazard Index Ratio is used. As noted in previous sections, the fundamental principles used to construct the RfD utilized in calculating the Hazard Index Ratio are predicated on long term or chronic (usually measured in years) exposures and health effects.

Cancer and non-cancer health risks are discussed below for trespasser/recreational (Scenario 1 - current use), construction (Scenario 2 - future use), and residential (Scenario 3 - future use) scenarios. Within the trespasser/recreational and residential scenarios, the risks to children (9-18 years old, trespasser/ recreational scenario; 0-6 years old, residential scenario and adults are presented separately. In each case, daily doses of the compounds of concern have been calculated for each exposure pathway modeled, and these doses were then used to calculate cancer risk levels and hazard index ratios. Cancer risk levels are the lifetime probability of excess cancer due to the exposure pathways emanating from use of the site. Cancer risk levels are derived by multiplying exposure dose by the appropriate cancer slope factor for each compound and exposure route. Non-cancer health risk is quantitated by the hazard index ratio which is the ratio of the exposure dose to the RfD (both in mg/kg/day). The calculated level of cancer risk can be compared to the acceptable total site risk range ((1E-04 to 1E-06) for evaluating the need for remediation, as stated in the "National Oil and Hazardous Substances Pollution Contingency Plan, Final Rule" (EPA, 40 CFR Part 300, March 8, and in the Superfund Human Health Evaluation Manual (1E-04 to 1E-07) (EPA, 1989). Regarding non-carcinogenic health hazards the Superfund Human Health Evaluation Manual (EPA, 1989) states that:

> "When the total hazard index for an exposed individual or group of individuals exceeds unity, there may be concern for potential non-cancer health effects."

Thus, the cancer risk and hazard index ratios that constitute a concern are those that are greater than 1E-04 and 1E+00, respectively. Tables 6-14 through 6-23 summarize cancer risk levels and hazard index ratios for all scenarios. Dose estimates, hazard index ratios and cancer risk estimates for all compounds and exposure pathways are presented in Appendix J.

<u>Current (Scenario 1)/Recreational Use: Cancer Risks and Hazard Index Ratios</u>

- Children

Tables 6-14 and 6-16 summarize the cancer risks and hazard index ratios for all exposure pathways considered. These tables present these risks on a chemical-by-chemical basis so that the major factors which drive the risk can be readily ascertained.

Exposure of children to contaminants while playing on-site is associated with a total cancer risk of 5E-04 which is above the acceptable risk range. The predominant factor contributing to this risk is dermal contact with arsenic in soil, which is responsible for nearly all of this risk. Arsenic and PAH compounds in soil makes an additional contribution to cancer risk

(9E-06) via the ingestion pathway, while dermal exposure to surface water contributes 1E-06.

Playing on site is associated with a total hazard index ratio of 3E+00, which is above the target HI value. Dermal exposure to arsenic is responsible for most of this risk. Ingestion of surface water causes exposure to a variety of chemicals which each make small contributions to the HI (pathway total = 2E-01).

- Adults

Tables 6-15 and 6-17 summarize the cancer risks and hazard index ratios for all exposure pathways considered.

Adults using the site for recreational purposes (hunting, hiking, etc.) would experience a total cancer risk of 4E-04, which is above the acceptable risk range (<1E-04). As for recreational use by children, the vast majority of the risk for adults is derived from dermal contact with arsenic in soil. Ingestion of arsenic in soil is responsible for a cancer risk of 8E-06 while ingestion of carcinogenic PAHs in soil contributes a total risk of 5E-06.

Use of the site by adults for recreational purposes is associated with a total hazard index ratio of 6E-01 which is below the upper limit of acceptable HI. The HI for adults is considerably below that of children because children are modeled to have more extensive contact with surface water than adults. Children have a similar amount of exposed skin surface area for dermal contact as for adults, but for a lower body weight, and children are expected to use the site more often than adults.

Construction (Scenario 2)/Future Use: Cancer Risks and Hazard Index Ratios

Table 6-18 summarizes the cancer risks associated with all chemicals and exposure pathways included in this scenario. The total cancer risk level is

1E-05, which is within the acceptable risk range (<1E-04). Dermal contact with arsenic in soil causes the vast majority of this risk, while ingestion of arsenic and PAH compounds make only a minor contribution. Inhalation of dust-borne contaminants does not appreciably contribute to the cancer risk.

Table 6-19 presents the hazard index ratios for all chemicals and exposure pathways. The total HI is 1E+00, which is just at the level of concern for non-carcinogenic effects. Again, dermal contact with arsenic in soil creates most of the HI, while soil ingestion and inhalation of dust exposure pathways makes a considerably lower contribution.

Cancer risk and HI levels are lower for construction workers than for receptors in other scenarios because construction workers will be on the site for considerably less time than other receptors, and because construction workers are exposed primarily to subsurface soils, which have lower contaminant levels than do surface soils.

Future (Scenario 3)/Residential Use: Cancer Risks and Hazard Index Ratios

- Children

Table 6-20 presents the cancer risks for each compound and each exposure pathway associated with future residential use of the site. The total cancer risk for children residing on site is 2E-03, which is above the acceptable risk range (<1E-04). The pathways of most importance are ingestion of arsenic in drinking water (risk = 2E-04), and dermal contact and ingestion of arsenic in soil (risks = 2E-03 and 8.5E-05, respectively). Additionally, ingestion of carcinogenic PAH compounds present in soil cause a combined risk of 5E-05, while other compounds and pathways do not make substantial contributions.

Table 6-22 presents the hazard index ratios for each compound and exposure pathway. The total HI for children is 100, which is considerably above that which may constitute a concern (>1E+00). The most important component of the HI is ingestion of metals in drinking water including manganese (HI = 8E+01), and arsenic (1E+00). This pathway accounts for 87% of the total HI. Other pathways of concern are ingestion of and dermal contact with soil (HI = 2E+00 and 1E+01, respectively). For soil ingestion, chromium is the major contributor while for dermal contact with soil, arsenic is most important.

- Adults

Table 6-21 presents the cancer risks for each compound and each exposure pathway. The total cancer risk for adults residing on site is 7E-03, which is well above the acceptable level (<1E-04). The major contributor to this risk is dermal contact with arsenic in soil (92% of total risk). Other pathways which contribute substantially to the cancer risk are ingestion of arsenic in drinking water (risk = 5E-04), ingestion of arsenic in soil (risk = 5E-05), and ingestion of carcinogenic PAH compounds in soil (risk = 3E-05). No cancer risk was caused by the inhalation of VOCs emanating from tap water since the VOCs found in ground water are not known to be carcinogenic.

Table 6-23 presents the hazard index ratios for each compound and exposure pathway. The total HI for all pathways is 6E+01, which is considerably greater than the target value of 1E+00 for HI. Ingestion of chemicals in tap water, most importantly manganese, chromium and other metals, accounted for the vast majority of the HI. The only other exposure route that had an elevated HI was dermal exposure to chemicals in soil (predominantly arsenic), which contributed 1E+01 to the HI.

Summary of Cancer and Non-Cancer Health Risks

This site currently contains elevated levels of certain key toxicants, which are responsible for driving the risk assessment. The residential scenario was associated with the greatest cancer risk and HI levels, due largely to the ingestion of ground water (as tap water) which was absent from the other scenarios. Additionally, the continuous exposure to surface soils (dermal and ingestion) in the residential scenario (365 days/year) caused risks to be higher in this scenario than in the others. Inhalation and dermal contact with surface or tap water were not major exposure pathways.

The chemical in ground water causing the greatest cancer risk is arsenic. Exposure to arsenic in soil is also of primary importance in each scenario. Arsenic is a group "A" carcinogen, whose carcinogenic effects are most notable in the skin after oral absorption. While the arsenic oral slope factor for carcinogenic effects is based upon the evidence of human skin cancer, arsenic exposure by the oral route has also been associated with elevated cancer incidences in bladder, lung, liver, kidney and colon (EPA, 1991 - IRIS File). The carcinogenic potency of arsenic upon dermal exposure has not been quantitatively evaluated. Arsenic also makes substantial contributions to hazard index ratios due to its potency in causing changes in skin (hyperpigmentation, keratosis) (EPA, 1990 - HEAST).

Manganese exposure in tap water produced a large elevation of the HI for both adults and children. The effect of concern for manganese is CNS damage, which has been demonstrated in humans occupationally exposed via the inhalation route. In these studies, internal organ effects for manganese, most notably liver cirrhosis, have also been seen upon chronic inhalation occupational exposure. Manganese is a trace element for which the typical dietary dose is 2-9 mg/day. A dose of 10 mg/day is considered safe (EPA, 1991 - IRIS file). Substantial exposure in drinking water could elevate the dose to a level of concern, particularly because manganese absorption from drinking water is better than that from the diet.

Other contributors to cancer risk at this site are PAH compounds present in soil. Seven carcinogenic PAH compounds, including benzo(a)pyrene, were detected on-site and included in the quantitative risk assessment. All were assigned the cancer slope factor derived for benzo(a)pyrene, which is among the most potent members of this chemical class. Most carcinogenic members of this class have been shown to induce skin cancer upon topical administration, while the more heavily studied agent, benzo(a)pyrene, has also been shown to cause lung and stomach tumors (ATSDR, 1990). The cumulative cancer risk associated with this group of chemicals was small in relation to arsenic; the highest risk for a particular receptor was 5E-05 (residential scenario - children: soil ingestion). Dermal cancer risk was not calculated because of uncertainty regarding the carcinogenic potency of the agents by the dermal route. However, given the preponderance of evidence in rodents that these agents are carcinogenic by dermal exposure, it is likely that this analysis underestimates the cancer risk due to PAH compounds present in soil. The increase in cancer risk that could be associated with dermal exposure to PAHs may be substantial since the dermal dosage to these agents was generally greater than that received via oral exposure, and oral exposure was associated with substantial risk. Further, this dermal dose represents the absorbed dose, which is only 10% of the exposure dose. Thus, the exposure dose of PAH compounds to the skin from dermal contact with soil is considerably greater than the oral dose, and thus might be associated with cancer risks greater than the highest found for oral exposure (5E-05).

6.6.2 Qualitative Analysis of Risks

Selected compounds (see Table 6-24) were addressed qualitatively rather than quantitatively because of the following reasons:

1) Highly qualified data precluded clearly confirming or denying, the presence of a chemical in a matrix;

2) Compounds were lacking cancer slope factors or RfD values.

6.6.2.1 Compounds Not Detected But Which Were Associated With Qualified Data

Although there was no clear evidence for the presence of these selected compounds in the listed matrix, there were at least some data qualifiers (UJ data) indicating that the compound might have been present. Sediment data are not included in this discussion because, as discussed previously, the profile of compounds in sediments was very similar to that of compounds in soils, with much greater exposure occurring in soils. There was much less qualified data for ground water, and in this matrix there were no compounds for which decisions could not be made regarding their presence or absence.

Semivolatile Compounds

In certain soil samples, heavy staining and resulting matrix effects caused SQLs for semivolatiles to be high. This precluded a clear determination of whether individual semivolatiles were present in such samples. However, these samples could be expected to contain at least some members of this class. The major uncertainty is whether these compounds, if they are present on site, could make a substantial contribution to risk. Since these compounds could not be addressed quantitatively, they are discussed below in the qualitative analysis.

Dibenzo(a)anthracene

This polynuclear aromatic hydrocarbon (PAH) has been estimated to have a high carcinogenic potency, similar to that for benzo(a)pyrene, (Clement, 1987;

EPA, 1985), and could therefore be expected to contribute significantly to cancer risk. However, all carcinogenic PAH compounds which have been assessed quantitatively have been assigned the benzo(a)pyrene cancer slope factor as a conservative, worst case assumption. Since data is available for six other carcinogenic PAH compounds in soil, the addition of dibenzo(a)anthracene is expected to make only a small contribution to the quantitative estimate of risk, unless the concentrations of dibenzo(a)anthracene far outweighed those of other PAHs. However, for the other six carcinogenic PAH compounds, the upper 95% confidence limits are elevated by inclusion of the qualified (UJ) data. Incorporation of this uncertain and conservative data in the analysis diminishes the chance that exclusion of the dibenzo(a)anthracene data, on the grounds that there were no clear hits, would cause an underestimate of actual risk. Therefore, the possibility that dibenzo(a)anthracene does exist on this site does not add a large degree of uncertainty to the magnitude of cancer risk.

Butylbenzylphthalate and Di-n-octylphthalate

These phthalates have RfD values typical of other semi-volatile compounds which are included in the quantitative analysis for soil (0.2 for butylbenzlpthalate; 0.02 for di-n-octylphthalate). Further, di-n-octylphthalate has not been shown to be carcinogenic; butylbenzylphthalate is regarded as a group "C" carcinogen, but a cancer slope factor has not been derived. A related carcinogen, bis(2-ethylhexyl)phthalate is not highly potent (oral slope factor = 0.014/mg•kg•day). Therefore, exclusion of these compounds from the risk analysis from soil on the grounds that they were not clearly identified in this matrix, should not greatly affect the result of the analysis.

Miscellaneous Semivolatiles: Phenol, p-Cresol, 2,4-Dimethylphenol, 1,4-Dichlorobenzene, 1,2-Dichlorobenzene, Benzoic Acid, Acenaphthene

The exclusion of the compounds in this group of miscellaneous semivolatiles is not expected to substantially alter the quantitative estimates of risk. The RfD values for these agents, where available, range from 0.02 to 0.6, which is within the range of the semivolatiles used in the quantitative risk analysis. Of these compounds, cancer slope factors are available only for 1,4-dichlorobenzene and 1,2-dichlorobenzene and these values are relatively low (0.02-0.09 $mg/kg/day^{-1}$). p-Cresol is classified as a "C" carcinogen, but no slope factor has been derived.

Based upon the toxic and carcinogenic potencies of these compounds, they do not alone, or taken in combination, appear to constitute a major factor in the risk assessment (cancer and non-cancer). Additionally, there is no reason to believe that these compounds are present on this site at levels substantially greater than the other semi-volatiles included in the analysis.

Inorganic Compounds: Antimony, Selenium, Thallium

These inorganic compounds have relatively low RfD values (0.004 to 0.00008) and could make substantial contributions to the risk assessment. Of these metals, selenium has been classified as a "B2" carcinogen, although a slope factor has yet to be derived. These agents are excluded from the quantitative assessment of soil on the basis that they are not clearly identified in this matrix. The SQLs in these soils are not unusually high for thallium compared to U.S. background soil levels (2.6 ppm SQL vs background range of 2.2-31 ppm). However, the SQL for selenium is high, leaving open the possibility that selenium levels at this site are elevated. Of note is the fact that a similar SQL has been obtained for background and test samples at this site. Selenium would not, in any case, make a quantitative contribution to cancer risk or hazard index ratio because RfDs and slope factor values have not been derived for this metal. For antimony, the moderately low RfD value (0.004) would not be expected to drive the risk assessment unless exposures are high. The antimony SQLs are approximately one order of magnitude above the background levels of this metal in U.S. soils indicating that high exposure cannot be ruled out. Therefore, it is possible that the cumulative hazard index ratio from contact with soil is somewhat underestimated due to exposure to antimony.

In summary for these three excluded metals:

1) Risks associated with the possibility that thallium may be in soil should not be any greater at this site than at other background locations;

2) Risks associated with selenium in soil could not be quantified even if it were clearly detected; and

3) Non-cancer chronic health risks associated with the possibility that antimony may be in soil are potentially significant, due to its low RfD and due to the high SQLs obtained.

Tentatively Identified Compounds (TICs)

TICs are not quantitatively addressed because their chemical identities were poorly characterized. In the vast majority of samples, the TICs are listed as "unknown hydrocarbons" or simply "unknown". In the few isolated cases where a specific chemical is listed as a TIC, the levels are generally low (<1 ppm). Total TIC levels per soil sample range up to 3,506 ppm, but without a better indication of the contaminants which comprise the TIC listing, no qualitative or quantitative assessment can be made. However, the extensive contamination in soils in which TIC levels are high caused the semi-volatile SQL levels to also be high. Since these data were classified as UJ in many such cases, they were included in the semivolatile soil data base.

Thus, in a sense, the presence of unknown contaminants appears to have increased the concentrations of semivolatile compounds used in the analysis, and thus they were indirectly added into the quantitative risk assessment.

There was one case in which a large amount of a TIC was present in soil and for which a chemical identity was listed. Sample No. SS-13 contained 7.9 ppm of total "tetramethylbenzene isomers". The tetramethylbenzene isomers durene and isodurene were researched for toxicological properties, utilizing a computerized bibliographic retrieval system (Toxline). Only one pertinent study was found, and this study was only concerned with acute toxicity (Lynch, 1978). Therefore, there is insufficient data to qualitatively or quantitatively analyze this TIC.

6.6.3 Uncertainty Assessment

- Site-Specific Uncertainty Factors

The scenarios developed for the site include exposures resulting from probable current use as a recreational area and potential future use of the site as a residential area. The risks associated with these scenarios are conditional on these land uses occurring. Observations made during field investigations indicate that activities such as hunting/hiking have occurred on the site (i.e., spent shotgun shells, campfire remains). However, contrary to this evidence was the lack of observation of any individuals other than field personnel on site. Thus, the uncertainty associated with the exposure duration for Scenario 1 may be large, and may contribute significantly to an overestimation of risk. Current zoning for the site is residential. However, uncertainty in future use of the site adds a degree of uncertainty to the risks associated with Scenarios 2 and 3.

Additional uncertainty in the risk characterization may stem from exclusion of chemicals in the quantitative risk assessment. Chemicals which were not included in the quantitative risk assessment were excluded due to either lack of quantitation in the chemical analysis or as a consequence of missing toxicity data. Chemicals for which a 95% UCL could not be estimated were evaluated for adverse health effects.

Any chemicals expected to contribute a significant uncertainty to the assessment of risk were addressed qualitatively in section 6.6.2. Briefly, the exclusion of antimony in soil may underestimate the cumulative hazard index ratio due to its low RfD and high SQL's obtained. Exclusion of other chemicals from the quantitative analysis is not expected to significantly alter the risk.

Table 6-25 summarizes the exposure pathways considered for the risk assessment, and reasons for exclusion or inclusion. Ingestion of ground water from currently used wells downgradient from the site was not addressed as no impacts were found in any well monitored (see section 4 of this report). Ingestion of and dermal contact with sediments for current and future land use scenarios was not addressed as these pathways for soil are expected to characterize an equivalent or greater risk.

Two models were used to characterize exposure point concentrations. The first, a model used to estimate concentrations of chemicals in fugitive dust, was taken from AP-42 (EPA, 1985) (see Appendix I). The key model assumptions include the time frame during which the construction on site is likely to take place and the use of a yearly average wind speed. The potential impact of these assumptions will be to underestimate risk if construction occurs for a longer period of time than originally estimated, or, if daily wind speeds exceed the annual average wind speed. The second model, volatilization of chemicals during home use of ground water (i.e., showering) (see Appendix I) was taken from Andelman (1985). A key assumption for this model is likely to include the fraction of contaminant volatilized, which is assumed to be 0.9 (90%). This assumption is likely to overpredict, rather than underpredict, risk.

As indicated in section 6.6.1, the primary routes of exposure for Scenarios 1 and 2 are dermal contact with soil, while ingestion of ground water is the primary route of exposure for Scenario 3. Site data gaps which resulted in the use of conservative assumptions for Scenario 1 include the frequency with which residents use the site for recreational purposes. Similarly, the exposure duration for construction workers was based on a conservative assumption, such that the risk estimate may be overestimated. Finally, risks associated with ingestion of ground water rely on the 90th percentile ingestion rate (2 l/day), and this may drive the risk estimate for this pathway.

Some significant uncertainties exist in the data used for this site. In all cases these uncertainties are likely to overestimate, rather than underestimate, the risk.

A few examples of data uncertainties include:

- Chemicals detected infrequently in all media were assumed to occur across the site at a 95% UCL or maximum detected concentration.
- "UJ" data (i.e., resulting from matrix effects) were generally included in calculations of 95% UCL and considered as potential locations of contamination.
- Uncertainties in background sampling locations, particularly with regard to inorganic compounds, disallowed exclusion of compounds which may occur naturally at the site.

- Uncertainties Surrounding Toxicity Values and the Calculation of Risk

Uncertainties Associated with Summation of Risks Across Multiple Exposures.

For the risk estimation of cancer and of chronic non-cancer health effects, risks from all exposure pathways and for all chemicals have been summated to yield the total site risk for a given receptor. This is a conservative approach, since, in general, different chemicals do not have the same target organ or mechanism of action. Thus, their toxic effects may be, at least in some cases, independent and not additive. Further, chemicals may antagonize one another through competition for enzymes and binding sites, and by inhibition of pathways needed for chemical transport (absorption, cellular uptake, etc.) or metabolic activation. However, it is also possible that certain chemicals can be synergistic such as is the case when a promotor-type carcinogen greatly enhances the expression of genetic damage induced by a low dose of an initiator. The uncertainties surrounding these possibilities are discussed below for the chemicals found on-site.

Cancer Risks

Interactions between carcinogens present at this site may both lead to enhanced and diminished carcinogenic responses. Arsenic, which is responsible for the greatest elevations in cancer risk on-site, is at most only weakly mutagenic, but its carcinogenic effects appear to be mediated through clastogenic effects (ATSDR, 1989). Arsenic-induced chromosomal damage may be due to an impairment of DNA replication or repair, and this effect could facilitate the genotoxic effects of other agents (ATSDR, 1989). Arsenic has been shown to greatly increase the mutagenic effects of direct-acting agents such as UV radiation, and alkylating agents. Further, arsenic appears to increase the production of lung tumors caused by benzo(a)pyrene, and is generally considered to have promotional activity (ATSDR, 1989). The target organ for arsenic's effects after oral ingestion (inhalation of arsenic is not a major concern at this site) is primarily the skin, but elevations in bladder, liver and lung cancer in humans exposed orally to arsenic have also been reported (EPA, 1991 - IRIS File; ATSDR, 1989). Therefore, it appears that arsenic might be able to enhance the carcinogenic action of other genotoxic agents at a variety of target sites.

Of the other carcinogens found on-site, only the group of PAH compounds can be classified as being genotoxic. Like arsenic, the PAH compounds exert genotoxic and carcinogenic effects in skin and at internal organs (ATSDR, 1990). The finding that arsenic can enhance lung tumor production by benzo(a)pyrene (ATSDR, 1989) supports the concept that a synergistic action is possible, particularly since arsenic and PAH compounds are found together in soil. Since the skin is an important target site for both the PAH compounds and arsenic, the synergistic effect might be most probable in the skin. Exposure to the skin may occur both directly by dermal contact, and after ingestion of soil or drinking water.

It is of note the beryllium, another carcinogenic metal found in soil, also can produce skin tumors upon oral exposure.

There is evidence that arsenic's toxic, cytogenetic, and carcinogenic effects can be antagonized by selenium, possibly through an interaction at the level of biliary excretion (ATSDR, 1989). However, the selenium content of soils at this site were below detectable levels and so the quantitative importance of this antagonistic effect is impossible to determine.

The carcinogenic PAH compounds are considered to, in general, act similarly with respect to mechanism of action and target organ. However, as a mixture their effects may not be strictly additive due to the potential for co-carcinogenic and antagonistic effects (ATSDR, 1990). These effects appear to be mediated primarily through interference with each other's metabolism -

either activation or detoxification, and by inducing, activating or detoxifying enzymes. The difference between antagonism, synergism and additivity of carcinogenic effects appears to depend upon the timing of the dosage of the different PAHs, the ratio of the different agents administered, and the exact agents involved (Baird, 1984; Slaga, 1979; Van Duren, 1976). These factors are too complex to allow prediction of the likely outcome from the interaction of PAH compounds at this site. However, this factor does introduce uncertainty in the calculation of cancer risks.

For other carcinogens detected on site, bis(2-ethylhexyl)phthalate (BEHP), butylbenzylphthalate, 1,4-dichlorobenzene, and DDT and related metabolites (DDE and DDD), the data suggest a promotional mechanism of action. Since the liver is the primary target organ for several of these agents, and since they may act via distinct mechanisms (e.g., DDT via inhibition of gap junctions; BEHP via peroxisome profileration) it is possible that additive and even synergistic interactions are possible. Further, these agents may promote the low dose effects of the genotoxic carcinogens (PAH compounds) to enhance their potency. However, the PAH compounds are not known to cause liver tumors unless the replicative state of the liver is grossly affected, as in partially hepatectomized animals (Marquardt, 1970). Thus, it appears unlikely that the expected weak promotional effects of these agents at this site would substantially increase the carcinogenic potential of PAH compounds in the liver.

Non-Cancer Effects

A variety of potential toxicant interactions affecting non-cancer health effects are possible for the chemicals found on-site. Arsenic may enhance the inhibitory effects of lead on porphyrin metabolism and thus increase the risk of lead-induced porphyria. However, this effect is not of greatest concern regarding lead toxicity.

The effect of greatest concern for lead, as well as for manganese, is neurotoxicity. Manganese in drinking water was responsible for the largest elevation in hazard index ratio on-site, due to its potential for causing CNS damage (substantia nigra neuron degeneration) and behavioral changes (EPA 1991 - IRIS File). Lead causes a wide variety of electrophysiological and neurochemical changes in the brain (EPA, 1986) and so it is impossible to predict how these two metals will interact. However, lead has been shown to decrease the synthesis of dopamine in the substantia nigra, which appears to be a primary target site for manganese. Therefore, it is possible that the deleterious effects of lead and manganese on CNS function are additive, or possibly even synergistic. However, this hypothesis has apparently not been tested.

The bioavailability of lead can be impaired by copper and zinc (EPA, 1986), suggesting that the presence of these components in soil may decrease lead absorption and toxicity. However, the quantitative importance of such an antagonism has not been clarified.

Elevations in the hazard index ratio (above 1E+00) at this site were generally not caused by adding individual HIs for different compounds. Compounds such as manganese and arsenic were capable of elevating the HI on their own. Therefore, considerations of whether it is appropriate to summate HIs stemming from non-cancer effects that occur in different tissues for different chemicals do not increase the uncertainty in this analysis.

Uncertainties In The Derivation of Toxicity Values

In numerous cases in which a toxicity value was available for one exposure route but not another, a dose route extrapolation was performed. These extrapolations were utilized to go between the oral and inhalation routes of exposure if the toxic/carcinogenic effects were systemic rather than local. The compounds for which this was done are noted in Tables 6-9 through 6-13. The oral to inhalation dose route extrapolation can underestimate potency from inhalation exposure if the chemical is irritating, insoluble, slowly absorbed or highly reactive. Under these conditions, the dose to specific lung regions may be greater than that to the g.i. tract or internal organs, creating the possibility that the lung would be at greater risk. At this site, this possibility is greatest for the oral-to-inhalation extrapolation of RfD values for the metals arsenic, beryllium, nickel and zinc. However, inhalation of these metals was due to the dust inhalation pathway which was a minor exposure route. Therefore, underestimation of toxicity values for inhalation exposure should not have a large effect on the outcome of this risk assessment.

Another form of dose route extrapolation was the use of oral toxicity values for dermal exposure. This extrapolation was utilized for all compounds except PAH compounds, whose potential for dermal effects was discussed. Similar to the case for PAH compounds, the toxicologic effects of arsenic may be greater by the dermal route of exposure. Arsenic produces primarily dermal toxicity and carcinogenesis after oral absorption. Since arsenic is readily excreted, it is likely that the amount of arsenic reaching the skin and accumulating there is considerably lower after oral compared to dermal exposure (ATSDR, 1989). Thus, the effectiveness of a dermal dose of arsenic may be considerably greater than an oral dose. The correction factor used for dermal RfDs and slope factors (0.10 for arsenic) takes into account the difference between absorbed vs exposure doses in oral vs dermal data. In general, the oral toxicity values are based upon an exposure dose, while the dermal doses for the modeled pathways are in terms of absorbed dose. The correction factor allows the dermal absorbed dose to be used with the oral toxicity value, but it does not take into account dose to the target organ. Since the dose to the target organ (skin) may be considerably greater from dermal as compared to oral exposure, the dose route extrapolation may underestimate risk.

Assignment of the benzo(a)pyrene cancer slope factors to other carcinogenic PAH compounds likely creates a considerable overestimate of risk. Benzo(a)pyrene is one of the most potent PAH compounds, and of the others on-site, only dibenz(a)anthracene has a similar carcinogenic potency (Rugen, 1989; Clement, 1987; EPA, 1985). Chrysene's potency appears to be ~200 fold below that for benzo(a)pyrene. The data upon which these relative potency estimates are based are taken from primarily dermal studies in which the development of skin tumors was studied. The degree of uncertainty in extrapolating these results to the oral route of exposure in order to adjust the oral slope factor is not known. However, these data are applicable to considerations of the cancer risk from dermal exposure. The overestimation created by using the benzo(a)pyrene slope factor as a surrogate for the other PAH compounds partially offsets the possible underestimation of risk from dermal exposure caused by not adequately characterizing the dermal exposure dose to arsenic and PAH compounds, as described above.

6.7 Environmental Assessment

The purpose of the environmental assessment is to better understand how selected biota might be impacted by elevated concentrations of organic and inorganic compounds on the former Hertel Landfill site. Endangered and threatened plant species will also be discussed in reference to the legal implications of such species on a privately owned parcel of land.

Little species specific information exists on the effects of chemicals on the organisms present at this site. However, the effect of the landfill on local organisms was evaluated through comparison of an upstream population of aquatic invertebrates found in areas adjacent to and down stream from the suspected landfill area.

Contaminants have been sampled from five different environmental media. Samples were collected from surface water, ground water, surface and subsurface soils and sediments. The analytical results were compiled and 95% UCL contaminant concentrations were tabulated for each compound at each sampling point for each of the five sampled media (see Tables 6-1 through 6-6). Contaminant concentration ranges were compared to Federal and State Guidelines and Criteria, to gain perspective on how compounds on site could potentially effect local organisms.

6.7.1 Threatened or Endangered Species

No federally listed threatened or endangered species were identified on the Hertel Landfill site. Two separate attempts were made to identify threatened or endangered species. The first involved asking the New York State Department of Environmental Conservation (NYSDEC) if there was any evidence of endangered or threatened species existing in the Hertel Landfill area (Buffington, 1989). Correspondence (10/3/89) from the DEC indicate that no protected species or habitats had been identified on the Hertel site to-date. However, the correspondence does state that "the absence of data does not necessarily mean that rare or endangered elements, natural communities or other significant habitats do not exist on or adjacent to the [Hertel] site, but rather that our files currently do not contain any information which indicates the presence of these." The second was to identify any federally threatened or endangered species through an extensive ecological survey. A complete list of plant and animal species was compiled for the Hertel Landfill site (see Appendix H).

Thirteen plant species exist on the site which are on the NYSDEC protected status list (NYSDEC, Protected Native Plants). A list of these plants is summarized in Appendix H. Currently the site is privately owned. New York State (NYS) regulations for protected plants on private property state that if any person wishes to harvest or destroy any of these thirteen species on the site the owners permission must be acquired. None of these plant species are on the federal threatened or endangered species list.

No bird species observed at the site is on the federal threatened or endangered species list, although the red-shouldered hawk (Buteo lineatus) is the only animal species on the Hertel site that is a threatened species protected under NYS Environmental Conservation Law. The NYSDEC, Division of Fish and Wildlife requires notification if a red-shouldered hawk's (Buteo lineatus) life is endangered by any activities that occur on the site.

6.7.2 Macroinvertebrates

To maximize the information generated from the raw data, several approaches were used for data analysis. Community structure was evaluated by examining species richness (number of taxa), the number of individuals, species diversity, and equitability (all reviewed in USEPA, 1973), at each station. The number of taxa and number of individuals were tabulated from the raw data. Species diversity ($\bar{d}$) was determined using the Shannon-Weaver function, which was defined in the 1973 USEPA document as follows:

$$\bar{d} = \frac{\underline{C}}{N} (N \log N - \sum ni \log_{10} Ni)$$

where C = 3.32 (converts base 10 log to base 2); N = total number of individuals and ni = the number of individuals in a given taxon. To evaluate the relative importance of certain organisms at a station, the ni ln ratio was used by itself and termed "percent relative abundance". Equitability was calculated using the formula e = S/S', where S is the total number of taxa and S' is the maximum theoretical number of taxa expected from a community with the given diversity. Equitability is a measure of eveness; the higher the equitability the more even the distribution of individuals among the component taxa. Equitability (e) as calculated, may range from 0 to 1 except in the unusual situations where the distribution in the sample is more equitable than the theoretical maximum distribution (which is based upon MacArthur's "broken stick" model which is described in USEPA, 1973). Such an eventuality will result in values of e greater than 1.0, and this occasionally occurs in samples containing only a few specimens with several taxa represented.

Appendix H presents the species richness (number of taxa), number of individuals, species diversity and equitability values at each station. At the six stream stations, species diversity ($\bar{d}$) ranged from a low of 1.5 at upstream station (US-A) to a high of 2.8 at downstream station (OS-A). The highest species diversity value however occurred at the pond, with a value of 3.1. The lowest equitability measurement (e) was also at upstream station, US-A, while the highest equitability value was at the other upstream station (US-B). The 1973 USEPA manual points out that estimates of $\bar{d}$ and e improve with increased sample size, and that samples containing less than 100 specimens should be evaluated with caution, if at all. But because of sampling limitations, samples of less than 100 were used. Upstream station US-B, intermediate stream station IS-B, downstream station DS-A, and the pond had less than 100 specimens. The remaining stations had greater than 100 specimens.

The 1973 USEPA Manual discusses a study in which the author evaluated values calculated from data collected from a variety of polluted and unpolluted waters. The report indicated that in unpolluted waters, ($\bar{d}$) was generally between 3 and 4, whereas in polluted water, ($\bar{d}$) was generally less than 1. For comparison, the species diversity values at the former landfill site ranged between 1.5 and 3.1.

The manual (USEPA, 1973) states that where environmental degradation is at slight to moderate levels, $\bar{d}$ lacks the sensitivity to demonstrate differences. Equitability (e) on the contrary, has been found to be very sensitive to even slight levels of degradation. Equitability levels below 0.5 have not been encountered in streams unaffected by oxygen-demanding wastes, and in such streams, e generally ranges between 0.6 and 0.8. Even slight levels of degradation have been found to reduce equitability below 0.5 and generally to a range of 0.0 to 0.3. The equitability values at the Hertel stream range from 0.4 to 1.1. Of these values, 4 were calculated based on less than 100 specimens. Three of these four values are 1.0 or greater, and the fourth value is 0.77. Therefore, these 4 high values for e should be viewed with caution. The other values, 0.4 at upstream station US-A, 0.4 at intermediate station IS-A and 0.6 at downstream station DS-B are more reliable because of larger sample size and indicate that there is slight degradation, although this occurred both at upstream and downstream locations. Thus, species diversity and equitability values in general, were similar at each of the stations, upstream, adjacent to, and downstream from the landfill.

6.7.3 Chemical Profiles for Ecotoxicity

The fate of each chemical of interest is discussed below in order to develop an understanding of compound mobility in various environmental matrices, of interactions with biota, and of other zones of the site which have the potential to become contaminated. These factors are important in predicting points of contamination at which substantive exposure could occur for organisms that exist on the site.

Inorganics - Essential Nutrients

Some inorganic elements found on the Hertel site are not discussed as a chemicals of interest because they are essential nutrients for plant growth; see Table 6-25. Some of these essential nutrients are in relatively high concentrations as compared with other inorganics found on site, but these particular inorganics (essential nutrients) are what comprise the materials from which they are formed (Brady, 1984). Any essential elements that are in exceedance of national average background levels will be discussed in more detail to better understand the potential effects such levels might cause. The concentration of inorganic chemicals at each sampling point can be found in Table 4-1. Table 6-1 illustrates the concentrations of inorganics in the soil at this site as compared to the average background level for the United States. The list of inorganic compounds in Table 6-25 are essential nutrients for plant growth, however toxic effects could still occur if concentrations are high enough. Concentrations of iron, calcium, sodium, potassium and magnesium were found above national average background concentrations in the surface water. However, there is no ambient water quality comparison criteria for these elements. For some of the inorganic compounds in Table 6-25, guidelines for allowable concentrations will be based upon human effects because of the lack of information and guidelines relating to other organisms.

- Aluminum

Aluminum might cause toxic effects in plants if the pH of the soils becomes extremely acidic. At pH levels in the 6 - 7 range the soil would bind most of the plant available aluminum (Brady, 1984) and there would be very little chance that vegetation would show symptoms of aluminum toxicity.

- Cobalt

Even though cobalt is found on site at concentrations above the U.S. average concentrations (30 ppm at SS-05 versus 9.1 as a national average), the toxic effects are expected to be limited. This is because plants do not readily take up cobalt from the soil. Plant tissue cobalt concentrations average 1/10th of soil concentrations (NTIS, 1980). Cobalt is toxic to animals but there does not appear to be a pathway for high concentrations to be ingested by local grazers.

- Copper

Copper toxicity has been observed in plants when soil concentrations have been in the range of 150 to 400 mg/kg (TRC, 1989). In some locations on site, copper concentrations in the soil are over 300 mg/kg. This evidence shows that there is a potential for copper toxicity in plants. However, the ability of plants to absorb copper from the soil is poor. This suggests there is little potential for animals to ingest toxic levels of copper from the edible portions of plants (NTIS, 1980). Copper concentrations in the surface soil are high enough to create potential problems to animals that are grazing near the contamination because grazers will consume soil along with their forage (Fries, 1986).

- Manganese

The Federal Ambient Water Quality Criteria for manganese in drinking water and for fish propagation is 50 ppb. Surface water concentrations on site range from 32.6-25300 ppb. Manganese can be found at concentrations as high as 1320 ppb in U.S. public drinking water supplies (Doull, 1980). The potential for toxic effects from consuming water with elevated manganese levels is unknown. The potential of manganese toxicity occurring on the Hertel site is expected to be minor.

- Zinc

Concentrations of zinc in the soil on site ranged from 48,600 - 469,000 ppb. The national average ranges from <5000 to 2,900,000 ppb. Zinc causes a toxic effect in plant tissue when concentrations reach 3,000,000 ppb. Plant tissue, on the average, bioaccumulates zinc at a level that is 1/2 of the concentration present in the soil. Thus, the zinc concentration in the plant tissue (234,500 ppb) at the site of greatest concentration in soil, is unlikely to cause a toxic effect. Levels of zinc in plant tissue that might cause toxic effects when consumed by animals range from 500,000 to 1,000,000 ppb. Using plant tissue bioconcentration factors from (NTIS, 1980) concentration in plants on site would not be expected to affect wildlife if consumption were to occur. The NYS Maximum Contaminant level for zinc for surface water and the Federal Ambient Water Quality Criteria for drinking water and fish propagation is 5000 ppb. The Hertel site had a surface water sample with a zinc concentration of 11,200 ppb.

Inorganics - Non-Essential Nutrients

The remainder of inorganics found on site are not essential nutrients for plant growth but have the potential to adversely effect local organisms. This is evaluated by comparing on site concentrations with federal and state guidelines and criteria. Additional toxicity information for each chemical is used to evaluate the acceptability of the levels found on site. For certain inorganic compounds listed in Table 6-27, guidelines based upon human effects were used because of the lack of information or guidelines relating to specific organisms. These inorganics were compared to average U.S. soil concentrations (USGS, 1984). The uptake of these non-essential elements by plants greatly varies between plants. Information regarding bioaccumulation of inorganics by specific plant species is sparse and so plant uptake will be discussed for individual species.

- Arsenic

The concentrations of arsenic in the surface soils are much greater than the surface water concentrations at the site. Once arsenic reaches the aquatic environment its mobility is greatly enhanced. This increased mobility is facilitated by arsenic's affinity to clays and iron oxides which reduce the concentrations of arsenic in the surface waters. Both iron oxides and clays are in abundance in the surface waters and soils of the landfill. Iron oxides were identified in the leachate by the presence of a layer of orange sedimentation. The particle size distribution of the soils on site indicate that soils contain 15% to 30% clay. The abundance of both iron oxides and clay suggests that arsenic can be readily transported once it enters an aqueous environment. Arsenic residence time within the aquatic organism is relatively short (i.e., it is readily excreted by organisms) therfore arsenic does not show a tendency to bioaccumulate through trophic levels. High concentrations of arsenic in the soil does not seem to effect plant growth because it is readily bound to iron oxides and clays, as soil colloids, in which state it becomes unavailable to plants. Since arsenic is not readily available for plant uptake there is little potential for transfer to animals by consumption of edible plant parts. Extremely high concentrations of arsenic (100 kg/ha) applied to the soil were not reported to cause plant toxicity (Brady, 1984). The New Jersey Department of Environmental Protection has guideline concentrations for inorganics in soil (NJDEP, ECRA). The standard set for the remediation of arsenic in the soil is 20 ppm which is greater than the highest concentration detected on the Hertel site (12.5 ppm). Based on the above discussion, arsenic at 12.5 ppm probably has little untoward effect on the local flora and fauna.

- Antimony

Antimony is very soluble and readily transported in a natural water system because of its affinity to sediments and other particulates that are present in the water column. This high solubiltiy and affinity to particulates makes water the dominant off site transport mechanism. Antimony was found in surface waters on site at a concentration of approximately 15 ppb. This is well below the Federal Ambient Water Quality Criteria for Human Health and Fish Propogation for antimony, 146 ppb. No information was located concerning antimony's ability to accumulate in the edible parts of plants and the effect on animals following consumption.

- Cadmium

The behavior of cadmium in the environment is not dominated by any single process. Cadmium is bound in an aqueous solution with organics (naturally occurring and pollutants) and inorganic components such as clays. Cadmium is readily released from sediments such that there is an equilibrium established between cadmium concentrations in the sediments and in solution. Thus, the cadmium concentration in wetlands and streams can be sustained after the source of cadmium is removed. Cadmium is readily taken up by aquatic organisms because if its similarity to zinc which is used in certain biological enzymes. This replacement of cadmium for zinc can disrupt metabolic processes and lead to a high ability to bioconcentrate factor. The Federal Ambient Water Quality Criteria for drinking water and fish propogation is 10 ppb. The highest concentration on site was 37.1 ppm at surface water sample SW-2. Elevated concentrations of cadmium could potentially create a risk to aquatic organisms on site. Cadmium is not an essential element for plant growth, and plant tissue can accumulate cadmium when elevated concentrations are found in soil. Plants on site could potentially accumulate a level of cadmium that would be toxic to animals if consumed, yet the plant would not be adversely affected (NTIS, 1980).

- Chromium

Chromium can be present in the environment in multiple valience states. The two most common forms in the environment are Chromium +3 and +6, although only total chromium was measured in this program. The Federal Ambient Water Quality Criteria for drinking water and fish propagation is 50 ppb. The highest concentration on site was 316 ppm. Concentrations of chromium (in the tissue of aquatic organism) have been shown to be much higher than in the surrounding water, indicating that chromium has a high bioaccumulation factor. High concentrations in the water and chromium's tendancy to bioaccumulate could potentially affect the aquatic life that surrounds the Hertel site. Chromium is not essential for plant growth nor is it readily absorbed into plant tissue from the soil. Because plants contain a very low concentration of chromium even when the concentration is high in the soil, animals are not likely to be affected adversely. High, acute doses of chromium do not seem to be harmful to domestic animals (Chaney, 1982).

- Lead

Lead is not an essential element for either plants or animals. Lead is bound tightly to soil particles such as iron oxides and organic matter which greatly decreases its mobility. Lead bound tightly to the soil becomes insoluble and is therefore mostly unavailable to plants. Concentrations of lead in soil above 500 ppm is considered to be unacceptable for growing crops because of the potential for human ingestion of lead and resulting toxicity (Brady, 1984). The highest concentration of lead found in the soil on site was 0.185 ppm, which is considerably less than the 500 ppm threshold. EPA has set guidelines for lead concentrations in drinking water at 50 ppb (USEPA, 1990). One surface water sample in site contains 54.9 ppb lead (at Surface Water Sample 2; SW-2), which is also somewhat elevated relative to background levels of lead in water. Once reaching surface water, lead readily binds to the sediments by forming complexes with organic and inorganic sediment materials, such that surface water concentrations diminish rapidly as the distance from the source increases. Lead seems to be moderately bioconcentrated but does not bioaccumulate through trophic feeding levels.

- Mercury

Mercury forms insoluble compounds with organic matter and clay minerals in soil, rendering the mercury unavailable to plants. Mercury can then be acted upon by microorganisms in soil and sediments to form methylmercury. Mercury in the methylmercury form moves freely in the environment by volatilizing into the air and dissolving into water. The following illustrates possible pathways mercury could take to cause toxic effects in plants and animals:

- Methylmercury is readily taken up by plants.
- Methylmercury is highly bioaccumulated by plants and animals.
- Biomagnification through trophic feeding level occurred in a study done on an aquatic ecosystem.
- These pathways are of concern at this site becuase high concentrations of mercury are present. On site concentrations of mercury are three times higher than on site background concentrations.

- Nickel

The heavy metal with the most mobility is nickel due to its great affinity for organic matter and hydrous metal oxides in the soil. This mobility is greatly dependent on the pH of the soil (e.g., if the soil has a pH of 7 or greater, nickel is mostly unavailable to plants). Similarly, as the soil becomes more acid, nickels availability to plants, like lead, is increased.

- Selenium

Selenium bioaccumulates in plants in high soil concentrations. Plants ingested by animals with high concentrations of selenium could potentially cause toxicity, as shown by the development of toxicity in livestock grazing on soils containing high levels of selenium (USEPA, 1979).

- Thallium

Thallium is readily sorbed onto clays in the soil but is also highly soluble, thus presenting a potential for migration in surface and ground water. Thallium has a high bioaccumulation factor in aquatic systems. Thallium concentrations in the surface water and the soils of the Hertel site were close to background concentrations. Soil concentrations are below the national average which is 9.4 ppm (USGS). The Federal Ambient Water Quality Criteria for drinking water and fish propagation is 13 ppb. The highest concentration of thallium in surface water on site was 4 ppb.

- Vanadium

Concentrations of vanadium in surface and subsurface soils are below the average concentration found in United States (Table 6-1 and 6-2). (Concentrations of vandium in U.S. soils range from less than 5 to 700 ppm, with an average of 80 ppm). Also, plants do not readily take up vanadium (NTIS, 1980). Based on these factors, vanadium does not appear to be a concern for ecotoxicological effects.

- Polynuclear Aromatic Hydrocarbons (PAH)

For the purpose of this ecological assessment the PAHs will be treated as a group because they behave similarly in the environment. Since concentrations of PAHs were very low to undetectable in all of the surface water samples it is unlikely that PAHs will effect the aquatic ecology of the Hertel Site. PAHs are abundant throughout the soils and the sediments of the Hertel site. Surface soils show elevated concentrations of PAHs as compared with on site background concentrations. The highest concentraion of PAHs found in the surface soils and sub-surface soil are 2,000 ppb and 710 ppb respectivly. There was no data that showed background concentrations of PAHs in the soil above detection limits. Movement of PAH's in soils and sediments is greatly inhibited because of the affinity that PAH's have for organic matter and soil colloids. Since PAH's are readily sorbed to soil particles, microorganisms have a greater opportunity to metabolize them.

Little is known about the structure of PAH metabolites. Once PAH's are exposed to sunlight, photolysis is the dominant process in converting PAHs to metabolites. The more soluble PAHs are more readily degraded by photolysis in reducing concentrations in an aquatic environment. PAH's also have an affinity for particulate matter and to a much greater extent particulate organic matter. Because of PAH's tendency to be sorbed to particulates, transport in water is greatly inhibited.

- Volatiles

The fate of volatile compounds on the Hertel site seem to be dominated by three different processes listed in decreasing order of significance: 1) Volatilization to the ambient air is dominant when a compound is transported to areas exposed to air such as the surface of the water and the soil. Volatiles are also transported through soil gases and may eventually reach the soil surface; 2) Sorption onto particulate and organic matter also codominate the fate of volatiles. Sorption onto particulates also provides a route for volatiles to migrate off site; 3) Depending on the solubility of the volatile, water transport can be route of transport; and 4) This fourth process only applies to the volatiles that are chlorinated. Volatiles with chlorine within their structure seem to bioacumulate much more readily than non-chlorinated volatiles. Bioaccumulation of chlorinated volatiles could present a risk to aquatic organisms if contact were to occur.

6.7.4 <u>Comparison to Background Soils</u>

Three inorganics (iron, lead, selenium) were identified on site that exceed the upper ranges of background concentrations in U.S. soils based on a U.S.G.S. survey and an EPA survey (see Table 4-1). These three inorganics also exceed background concentrations found on site. Iron is usually found in elevated concentrations and is not likely to adversely affect the ecology of the Hertel site. With the exception of one sampling point (surface soil sample SS-13), lead concentrations did not exceed U.S.G.S. or EPA values. Based on this information, it is not likely that adverse ecological affects will occur, with the exception of this elevated value. Elevated concentrations of selenium are located across the site, however there is little information detailing its environmental toxicity. Possible adverse ecological effects of these compounds are provided in 6.7.3.

Four inorganics (cadmium, copper, magnesium, mercury) exceeded the upper ranges of on site background concentrations and upper range concentrations documented by the EPA (see Table 4-1). Elevated concentrations of cadmium and mercury are located primarily in surface soils across the site. Cadmium and mercury are not essential for plant growth and have the potential of causing toxic effects in the ecology of the Hertel site. Details on the potential of toxic effects of cadmium and mercury along with the essential plant nutrients copper and magnesium are discussed below. Copper and magnesium are essential elements for plant growth and are not likely to cause adverse ecological effects.

Seven compounds (aluminum, calcium, chromium, barium, potassium, sodium, zinc) were identified on site at concentrations that are higher than on site background concentrations, but not USGS or EPA values (see Table 4-1). Because the concentrations of these seven inorganics are below the upper ranges of background concentrations in U.S. soils and EPA upper ranges used in land treatment of hazardous waste, the potential for adverse effects to occur is small. Chromium is the only compound that has readily avaliable toxicity information and the possible toxic effects are discussed below. Table 6-27 summarizes the elements which exceed U.S. background and on-site background concentrations.

Of all the identified inorganics of concern in soil, selenium, cadmium and mercury present a potential for adverse ecological impacts.

Similar comparisons between organic contaminant concentrations on site and background concentrations were not made based on the anthropogenic nature of these contaminants. Little information exists on the ecotoxic affects of PAHs and volatile organics.

6.7.5 Environmental Assessment Summary

Extensive data collection, analysis and evaluation indicates a general trend of elevated concentrations of organic and inorganic contaminants in one or more environmental media at the site. However, studies indicate that there are no federally listed threatened or endangered species identified at the site, and a macroinvertebrate study indicated that species diversity and equitability values in general, were similar upstream, adjacent to, and downstream from the landfill.

Of the identified inorganics of concern in soils, selenium, cadmium and mercury present a potential for adverse ecological effects. Similar conclusions were not drawn for organic compounds due to a paucity of ecotoxicological data on these compounds.

7.0 SUMMARY AND CONCLUSIONS

This section presents a brief summary of the findings of the RI investigation conducted at the Hertel Landfill site, focusing on the nature and extent of contamination, contaminant fate and transport, risk assessment, data limitations and recommended remedial action objectives.

7.1 Summary

The Hertel Landfill was operated as a municipal waste landfill from 1963 until its permit was revoked in 1976. Preliminary environmental investigations determined that ground and surface water at the site was contaminated with volatile organic compounds (VOCs), base neutral/acid extractables (BNAs) and metals. The site was codified onto the National Priority List (NPL) in 1986.

The RI investigation was comprised of the following field activities: geophysical surveys, soil gas surveys, test pit excavations, test boring drilling, monitoring well installation, private well sampling, as well as the sampling of surface and subsurface soils, ground water, surface water, and sediment. The site investigation was focused on the eight disposal areas identified through historical research and site visits.

In addition, an ecological risk assessment was performed on the site to provide baseline data. The field investigation included the following sampling/surveying components: wetlands identification, macroinvertebrates, birds, fish, mammals, herpetofauna and vegetation.

7.1.1 Nature and Extent of Contamination

The investigation determined that the landfilling activities were concentrated in a 13.2 acre area located near the center of the site. Fill thickness ranged to greater than 16 feet. Numerous metallic anomalies were identified in the landfill area by geophysics and most were investigated by excavation, but no areas of suspicious buried waste (e.g. buried drums) were observed.

The two locations with obvious surficial contamination, Disposal Areas #3 and #7, appear to be the focal points of subsurface contamination. Aromatic VOCs, at the highest concentrations observed on the site, were detected in the ground water at these locations (monitoring wells MW-6S and MW-7S). Similarly, BNAs were detected in the ground water at these locations. The presence of an ephemeral oily layer observed on the ground water at monitoring well MW-6S may be related to the oily surface-stained area nearby. Lesser concentrations of aromatic VOCs were detected in test pit leachate and leachate seeps downgradient of these areas.

Landfill leachate indicator parameters were detected in all of the shallow and deep overburden wells in the landfill area, indicative of a hydraulic connection between the shallow and deep overburden deposits. One shallow bedrock well (MW-6D) had landfill leachate parameter values similar to the landfill wells. BNA compounds were also detected in this deep well.

Subsurface contamination of soils and ground water were also observed at Disposal Area #1 where only surficial wastes were observed. Again, aromatic VOCs are the principal contaminants of concern.

Aromatic VOCs were also detected at relatively low concentrations in most of the subsurface soil sampling locations in the landfilled area. BNAs and numerous metals were also identified in the surface and subsurface soils. Although a wide range of compounds were detected over the landfill area, there was no predictive pattern to the distribution of the compounds.

Contamination was detected in the surface water; the highest concentrations and the greatest number of compounds, VOCs, BNAs, and inorganics, on-site were detected in the seep samples collected at the toe of the landfill, where ground water is discharging to the surface into the adjacent wetland. Some metals were detected downgradient. A number of PAH compounds were detected off site in the most downstream surface sample (SW-20). It is likely that the detection of PAHs at this location is attributable to the nearby road. Arsenic was detected in all sampled media at the site. The highest concentration of arsenic detected (109 mg/kg) at the site was located away from the historical landfill area at a background location.

The distribution of contamination in sediment samples was largely confined to the site and principally at the toe of the landfill at the leachate seep sampling locations.

7.1.2 Fate and Transport

Aromatic VOCs were detected in subsurface soils, ground water, surface water and sediment. The primary mechanism by which these VOCs naturally attenuate is through volatilization. Some of the VOCs detected at the site, such as benzene and halogenated VOCs, are more mobile, due to their higher solubilities and low organic carbon-water partition coefficient (Koc). Other VOCs detected on site, e.g. ethylbenzene and xylenes, are more persistent in the environment because they have lower solubilities and higher Koc values. Biodegradation also affects the fate of VOCs as some of these compounds have a strong tendency to be altered by organisms.

PAHs were the primary BNA compounds detected at the site; they were detected in surface and subsurface soils, ground water, surface water and sediment. In the environment, PAHs tend to adsorb onto soils and particles. They are persistent in the environment, due to their low solubilities and high Koc values.

Numerous inorganic compounds were detected in all sampled media at the site. Inorganics have a wide range of solubilities and Koc values. Some compounds, such as lead, tend to adsorb to organic matter or soils and this is their predominant fate process. The environmental fate of arsenic is very complex. In general, inorganic arsenic compounds are not likely to volatilize and are not soluble in water, suggesting little environmental mobility. Some evidence suggests arsenic may leach to ground water from soils with low sorptive capacity. Other inorganics, like cadmium, are mobile and can leach into the ground water. Inorganics also occur naturally in the environment.

7.1.3 Risk Assessment Summary and Conclusions

The risk assessment quantitatively analyzed the contribution of 43 different chemicals (12 inorganic, 5 volatile organics, 23 base/neutral organics, 3 pesticides) to lifetime incremental cancer risk and non-cancer health effects (Hazard Index Ratio). Other chemicals were discarded from the analysis because they were not positively identified in any matrix on-site. The assessment quantitated exposure and risk to potential receptors in the recreational/current use scenario (adults and children), in the construction/ future use scenario, and in the residential/future use scenario (adults and children). Exposure was modeled for the most likely pathways: ingestion of tap water (residential scenario), ingestion of soil (all scenarios), dermal contact with soil (all scenarios), inhalation of compounds volatilized from tap water (residential scenario), inhalation of dust-borne compounds (construction and residential scenarios), dermal contact with tap water during showering (residential scenario), and dermal and ingestion exposures to surface water (recreation scenario).

Risk levels at the site were elevated above the range of acceptable cancer risk (1E-04 to 1E-07) and above the acceptable hazard ratio (1E+00) in several cases. The greatest cancer risk levels and hazard index ratios were for children and adults in the residential/future use scenario. The total cancer risk for all chemicals and all pathways in this scenario is 7E-03 for adults and 2E-03 for children. The total hazard index ratio is 9E+01 for children and 6E+01 for adults. Cancer risk levels and hazard index ratios in the recreation/current use scenario were generally 1-2 orders of magnitude below those derived in the residential scenario, but were still of concern (total cancer risk levels: 4-5E-04; total hazard index ratios: 0.5-2E+00). The construction scenario is associated with the lowest cancer risk (1E-05) and hazard index ratio (1E+00).

The most important exposure pathways are dermal exposure to chemicals in soil, ingestion of chemicals in drinking water (derived from ground water), and ingestion of chemicals in soil. The major contaminant of concern in soil is arsenic, which contributes much more to cancer risk than does other carcinogens found in soil. Carcinogenic polycyclic aromatic hydrocarbons, as a group, add up to 5E-05 to the cancer risk through ingestion of soil. Arsenic was found in ground water (with potential future use as drinking water) at levels that substantially increase cancer risk, while other drinking water contaminants are not a major factor for cancer risk.

The primary contributor to hazard index ratio is ingestion exposure of residents to manganese in drinking water. Elevated hazard index ratios are also caused by ingestion of arsenic in drinking water, dermal exposure to arsenic in soil and ingestion of arsenic and other metals, particularly chromium, in soil.

Dermal and ingestion exposures to surface water, inhalation of dust-borne chemicals, dermal exposure to tap water, and inhalation of volatile organic chemicals in indoor air were pathways which were responsible for only a minor portion of the total cancer risk or hazard index ratio.

In conclusion, arsenic concentrations in soil and ground water, and manganese concentrations in ground water were the major driving forces in producing elevated cancer risk and hazard index ratios in the residential and recreation/current use scenarios. Other contaminants which are involved in producing elevated risks are polycyclic aromatic hydrocarbons and chromium in soil.

7.2 Conclusions

The Hertel Landfill site was operated as a municipal waste landfill from 1963 to 1976. The landfill is composed nearly exclusively of typical household waste and municipal landfill materials. Surficial evidence of deposition of other material were limited to Disposal Areas #3 and #7. A layer of oil was observed in the ground water at one location. Subsurface investigations indicate that, of 9 of 19 anomalies identified, there are no extensive buried drum deposited in the explored areas.

The analytical results demonstrate that contamination is present at the site in many media surface and subsurface soils, ground water, surface water and sediment. This contamination, primarily in the form of VOCs, BNAs, and inorganics, does exceed Federal and New York standards and guidance.

7.2.1 Data Limitations and Recommendations for Future Work

As a result of the QA/QC review of the sample analytical packages, some of the data for a particular matrix or sample were rejected. Due to interference from oily samples, some of the detection limits for BNA compounds, especially in soil samples, were extremely elevated. Because of this elevated detection limits, confidence in reporting the affected compounds as not detected is not high. Similarly, the elevated detection limits influence the risk assessment.

The Hertel Landfill was investigated extensively, using a number of surface and subsurface techniques, and included the sampling of various media. The sampling program was targeted to the eight reported disposal areas and to assess the nature and extent of contamination emanating from these disposal areas. Although flexibility was employed in the RI investigation, some data gaps do exist. Although the RI investigation determined that there are some immediate environmental concerns, contamination is largely limited to the area of landfilling and its immediate surroundings. Therefore, any data gaps would be addressed during the conceptual design phase. These data gaps include the following items:

- assessment of the remaining magnetic anomalies to confirm the presence/absence of drums at these locations;
- the installation of additional monitor wells to determine the extent of the floating oily layer; and
- the installation of a limited number of additional overburden and bedrock wells downgradient of the site to assess the extent of off-site ground water contamination and to act as an early warning system for off-site contaminant migration.

In addition to the above, additional efforts will be made during design to further delineate the wetlands at Hertel to determine necessary actions prior to construction.

Although it is unlikely that contamination from the landfill would reach Tuckers Corner Road, especially in the absence of any observed significant ground water contamination in bedrock at the site, the residential wells along Tuckers Corner Road should be sampled prior to remedial design of the selected remedy.

7.2.2 Recommended Remedial Action Objectives

General remedial response objectives for the Hertel Landfill site were originally presented in the Final RI/FS Work Plan (TAMS, September 1989). They included the following:

- Ground water - clean up ground water contamination such that available ARARs and/or risk-based levels are attained at the end of the remedy.
- Soils - prevent exposures to contaminated soil that exceed risk-based levels developed in the risk assessment.
- Surface Water/Sediment - prevent exposure to contaminated surface water and sediment such that recreational uses can be restored.

Based on the results of the RI, these objectives can be refined to reflect specific contaminants of interest and allowable exposures based on risk assessment and ARARs. The following is a general discussion on general remedial response objectives. A comprehensive discussion on this topic is presented as part of the feasibility study.

For ground water, the contaminants of interest include VOCs (benzene, chlorobenzene, ethylbenzene, xylenes and others), BNAs (phthalate esters and phenols) and metals (arsenic, iron and manganese). Potential exposure pathways include human ingestion via nearby private potable wells and exposure due to discharge of shallow ground water to surface water bodies. Based on this information, the remedial action objective for ground water is as follows:

- Prevent exposure to the VOCs, BNAs and inorganics at levels exceeding acceptable risk-based cleanup levels or ARARs/TBCs due to ground water ingestion.

For soils, the contaminants of interest include arsenic, chromium and other metals. The potential exposure pathway is dermal contact. Based on this information, the remedial action objective for soils is as follows:

- Prevent exposures to inorganics at levels exceeding acceptable risk-based cleanup levels or ARARs/TBCs, and prevent migration of contaminants that could result in ground water contamination in excess of acceptable risk-based levels or ARARs.

For surface water and sediments, the contaminants of interest are most of the inorganic analytes. Potential exposure pathways include dermal and ingestion exposures of surface water. Based on this information, the remedial action objective for surface water and sediments is as follows:

- Prevent exposure to inorganics in sediments and surface water at levels exceeding acceptable risk-based cleanup levels or ARARs/TBCs, and prevent releases of contaminants from sediments into surface water that could result in excessive contaminant levels in surface water.

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