

**The Impact of a Piedmont Sanitary Landfill on
Surface and Ground Water Quality**

by

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ABSTRACT

The primary objective of this investigation was to evaluate the chemical quality of ground and surface water at a sanitary landfill located in the Piedmont region of North Carolina. This landfill is "typical" in many respects of municipal waste disposal facilities in this region. The facility was constructed without an engineered liner or leachate control system.

Ground water immediately down gradient from the refuse disposal areas is contaminated with a wide variety of pollutants. At the downgradient boundary of the landfill, pollutant concentrations are much lower although ground water released from the landfill may violate N.C. ground water quality standards for iron, manganese and pH. Synthetic Organic Chemicals (SOC's) may also be present in concentrations exceeding ground water standards. The contaminated ground water is believed to discharge to a large stream which forms the downgradient boundary of the landfill. Adjoining water supply wells were not impacted by the landfill.

Elevated concentrations of organic and inorganic pollutants are released from the solid waste disposal areas during dry weather. The contaminated water passes through a sedimentation pond-wetland and is mixed with urban runoff before discharging from the landfill. The concentrations of most pollutants in the landfill discharge were higher than in the receiving water. Low but detectable levels of SOC's were observed in the landfill surface water discharge during dry weather. With the exception of total suspended solids, concentrations of most pollutants were significantly lower during storm events. The overall chemical quality of stormwater runoff released from the Piedmont landfill is comparable to runoff from other urban watersheds which do not receive landfill runoff. Mass loads for most pollutants released from the landfill are significantly lower than those from a wastewater treatment plant serving a comparable population.

(Keywords: sanitary landfills, ground water, surface water)

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SUMMARY AND CONCLUSIONS

1. Ground and surface water were monitored at a typical piedmont sanitary landfill located in Wake County, North Carolina. A series of monitoring wells were installed and sampled for conventional wastewater parameters, heavy metals and synthetic organic chemicals. Surface water was also sampled at several locations around the landfill to examine the variation in water quality during dry and wet weather.
2. Shallow ground water immediately downgradient from the refuse disposal areas within the boundaries of the Piedmont landfill is contaminated with a wide variety of pollutants. Elevated concentrations of chemical oxygen demand (COD), total organic carbon (TOC), iron, manganese and synthetic organic chemicals (SOC's) were observed in shallow ground water immediately downgradient from the refuse. At the downgradient edge of the working area, approximately 1,000 ft upgradient from the property boundary, concentrations of most pollutants are much lower. Ground water discharging from the landfill probably exceeds concentration limits established for Class GA ground waters for iron, manganese, and pH. Ground water discharging from the landfill may contain vinyl chloride in concentrations exceeding water quality standards; however, concentrations were below the analytical detection limit. We believe the contaminated ground water discharges to Crabtree Creek immediately downgradient from the landfill. No water supply sources have been adversely impacted by this landfill. Elevated concentrations of toxic heavy metals were not found at any location within the landfill.
3. Elevated concentrations of chloride, fluoride, ammonia nitrogen ($\text{NH}_3\text{-N}$), COD, TOC, dissolved solids, suspended solids, iron, manganese, barium, phenol, diethyl-phthalate, 4-methylphenol, acetone and 2-butanone were measured in surface waters discharging from the waste disposal areas during periods of dry weather. After passage through a sedimentation pond-wetland system located within the landfill boundary, the concentrations COD, total suspended solids, $\text{NH}_3\text{-N}$, $\text{NO}_3\text{-N}$, dissolved solids, total iron, total manganese, were reduced substantially, but remained higher than the average concentrations in the receiving water (Crabtree Creek). Low but detectable levels of phenol, diethyl-phthalate, 4-methylphenol, 2-butanone, and 2-hexanone were observed in the landfill surface water discharge during dry weather. With the exception of total suspended solids, concentrations of most pollutants were significantly lower during storm events. Very high concentrations of suspended solids were observed in runoff immediately downstream from the waste disposal areas during storm events. It was not possible to determine the effect of the sedimentation pond-wetland system on suspended solids removal.

4. Essentially all surface and ground water released from the Piedmont landfill eventually enters Crabtree Creek, immediately downgradient from the landfill. There was no obvious difference in pollutant concentrations between samples collected immediately upstream and downstream of the landfill on Crabtree Creek.
5. Concentrations of conventional pollutants (COD, TOC, TDS, iron, and manganese) and SOC's at the Piedmont landfill were much higher in shallow ground water immediately downgradient from recently deposited refuse (less than 5 years old) than in shallow ground water downgradient from older refuse (over ten years old). This decline in pollutant concentrations probably occurs as a result of decomposition within the buried refuse.
6. Significant attenuation of many pollutants (COD, TOC, TDS, iron, manganese, phenols, ketones and alkylbenzenes) is occurring during transport through the saprolite and transition zone aquifers at the Piedmont landfill. The majority of these pollutants have been removed before they reach the bedrock aquifer. Little additional removal appears to occur during transport through the bedrock fractures. Vinyl chloride and possibly 1,1-dichloroethene appear to be produced by reductive dehalogenation of more highly chlorinated solvents in the buried refuse and/or shallow ground water. These pollutants are probably not removed to any significant extent during transport through the anaerobic bedrock aquifer.
7. The overall chemical quality of stormwater runoff released from the Piedmont landfill is comparable to runoff from other urban watersheds. The concentration of ammonia nitrogen may be higher in landfill runoff, while concentrations of heavy metals may be higher in urban runoff.
8. The total mass of COD, NH₃-N, NO₃-N, lead, and zinc released to surface water from the Piedmont landfill is much lower than the total mass released from a waste water treatment plant serving the same city. The amount of suspended solids released by the landfill due to soil erosion could be much higher if proper erosion control measures are not implemented. The total mass of COD, NO₃-N, total nitrogen and lead released per acre of landfill is comparable to other urban land uses.

RECOMMENDATIONS

1. The State of North Carolina has encouraged landfill operators to locate sanitary landfills away from water supply wells and near larger streams. At the landfill examined in this study, this practice was very effective in minimizing adverse impacts on water resources. The State of North Carolina should continue to encourage the siting of landfills away from water supply wells and near larger rivers and streams.
2. At most landfills, essentially continuous earth moving operations are underway for many years. Soil erosion at these landfills may have a significant adverse impact on surface water quality. Every effort should be made to control soil erosion at sanitary landfills.
3. Methods are needed to enhance refuse decomposition during the early life of a landfill when operator control and maintenance is highest. If decomposition rates can be enhanced, leachate released as the landfill ages will contain few hazardous compounds, and the potential impact on surface and ground waters will be minimized.
4. Significant attenuation of a variety of pollutants was observed during transport through the subsurface at the Piedmont landfill. Additional research is needed to determine if extensive pollutant degradation occurs at all landfills, and if not, how the degradation of these pollutants can be enhanced. Special attention should be focused on the degradation of alkylbenzenes in the subsurface. At present, there are no well documented cases where anaerobic degradation of benzene, toluene, or xylene has been conclusively shown to occur in the subsurface.

INTRODUCTION

Environmentally safe and economically efficient solid waste disposal is becoming increasingly difficult. The number of sanitary landfills is expected to grow as our population increases resulting in increased potential for contamination of surface and ground water by landfill leachate. At the same time, recent publicity over health risks associated with trace levels of organic and inorganic contaminants has resulted in greater pressure to protect our ground and surface water resources. This combination is resulting in new challenges for the safe and efficient management of water resources and solid waste disposal facilities.

The most widely used method for disposal of municipal wastes is the sanitary landfill. Prior to the 1970's, open dumps were common. However, increased awareness of the environmental problems associated with dumps including air pollution from burning refuse, water pollution from leachate, health risks posed by vermin, and odor problems led to stricter regulations governing solid waste disposal. The sanitary landfill offered a relatively safe and economical alternative to open dumps. Waste in a sanitary landfill is disposed of by burial in trenches or cells and is covered at the end of each day. By limiting the amount of waste exposed at any given time, problems associated with open waste are greatly reduced. Unfortunately, burial of solid waste has the potential to increase surface and ground water contamination. After burial, solid waste gradually decomposes generating leachate, a noxious liquid high in dissolved organics, metals and frequently toxic organic chemicals. If the generation and movement of leachate is not properly managed, there is a potential for contamination of surface and ground water.

Leachate Characteristics

Leachate composition is highly variable and dependent on the refuse composition and waste processing practices, landfill age, rate of water infiltration, depth of leached bed and landfill temperature (Lu et al., 1985). Typical landfill leachate contains high concentrations of ammonia nitrogen, phosphorus, dissolved solids and other ionic compounds. Biochemical oxygen demand (BOD) requirements 40 to 85 times higher than most raw domestic sewage sludges have been reported (Qasim and Burchinal, 1970) (Table 1). The mean pH of landfill

Table 1. Observed Ranges of Constituent Concentrations in Leachate from Municipal Landfills (in mg/l unless noted)

Constituent	Concentration Concentration Range	Constituent	Range
Chemical Oxygen Demand (COD)	50-90,000	Biochemical Oxygen Demand (BOD)	5-75,000
Hardness (as CaCO ₃)	0.1-36,000	Acidity	2,700-6,000
Total Organic Carbon (TOC)	50-45,000	Alkalinity	0.1-20,350
Total Solids	1-75,000	Total Phosphorous	0.1-150
Total Dissolved Solids (TDS)	725-55,000	Organic Phosphorous	0.4-100
Total Suspended Solids (TSS)	10-45,000	Phosphate	0.4-150
Volatile Suspended Solids (VSS)	20-750	Nitrate Nitrogen	0.1-45
Total Volatile Solids (TVS)	90-50,000	Ammonia Nitrogen	0.1-2,000
Fixed Solids (FS)	800-50,000	Organic Nitrogen	0.1-1,000
Total Coliforms (CFU/100 ml)	0-10 ⁵	Total Kjeldahl Nitrogen	7-1,970
Fecal Coliform (CFU/1,000 ml)	0-10 ⁵	Total Volatile Acids (TVA)	70-27,700
pH (standard unit)	3.5-8.5	Turbidity (NTU)	30-450
Cd	0-0.375	Chloride	30-5,000
Cu	0.1-9	Specific Conductance (µmho/cm)	960-16,300
Hg	0-0.16	Sulfate	25-500
Na	20-7,600	Cr	0.2-18
Mn	0.6-41	Fe	200-5,500
Pb	0.001-1.44	K	35-2,300
		Mg	3-15,600
		Ni	0.2-79
		Se	0-2.7
		Zn	0.6-220

Source: U.S. Environmental Protection Agency 1986

leachate is typically around 5.5 (Fungaroli, 1971). Bacteria are found in most leachates but the populations appear to decrease with time (Qasim and Burchinal, 1970). Field measurements of coliform bacteria in leachate suggest that pathogenic organisms may also be present (Lu et al., 1985). Organic compounds found in leachate include acetone, alkanes, ethers, short chain alcohols and short chain acids, among others (Lu et al., 1985) (Table 2).

Research Objectives

The objective of this research was to evaluate the impact of a "typical" North Carolina sanitary landfill on ground and surface water quality within the landfill watershed. A secondary objective was to compare the pollutant load associated with a typical landfill with the pollutant load from urban runoff or a wastewater treatment plant.

LANDFILL DESCRIPTION

The sanitary landfill examined in this study is located in the Piedmont physiographic province of North Carolina in eastern Wake County (Figure 1). The facility has been in operation since 1972 and receives approximately 1,500 tons of municipal refuse per day including domestic, commercial and industrial solid wastes (Richards, 1989). Disposal of hazardous waste within the landfill is not permitted, although prior to 1989 an active program was not in place to exclude hazardous materials. This landfill appears to be "typical" of many landfills permitted in the 1970's in that it has no design features intended to prevent movement of leachate into the ground water. The landfill does not have an engineered liner, leachate collection system, or engineered cover system. In most areas, the surface soil was removed to within 5 feet of bedrock. In operating the landfill, a general nonsegregated solid waste is placed in lifts with 6 to 12 inches of daily cover. After the site is filled to the approved elevation, an additional 24 inch final cover of compacted soil is applied. The landfill appears to operate within its permit guidelines. Daily and final cover and soil erosion control requirements appear to be generally met. An area-type fill procedure is followed.

The landfill's watershed covers approximately 184 acres, with about 130 acres used for solid waste disposal. Runoff from the landfill is detained in a sedimentation basin at the southern end of the site, before discharging into Crabtree Creek. The original basin has gradually filled with sediment to form an extensive pond-wetland system of approximately four acres. Drainage from the watershed north of the disposal areas has been diverted eastward around the waste to the same sedimentation basin.

Figure 1 shows pertinent site features including the locations of waste deposits, diversion channel and sedimentation basin. Stream channels indicate the site

Table 2. Organic Compounds Detected in Municipal Solid Waste Landfill Leachate.

CONSTITUENT	MINIMUM	MAXIMUM	MEDIAN
Acetone	140	11,000	7,500
Benzene	2	410	17
Bromoethane	10	170	55
1-Butanol	50	360	220
Carbon Tetrachloride	2	398	10
Chlorobenzene	2	237	10
Chloroethane	5	170	7.5
bis (2-Chloroethoxyl) methane	2	14	10
Chloroform	2	1,300	10
Chloromethane	10	170	55
Delta BHC	0	5	0
Dibromomethane	5	25	10
1,4-Dichlorobenzene	2	20	7.7
Dichlorodifluoromethane	10	369	95
1,1-Dichloroethane	2	6,300	65.5
1,2-Dichloroethane	0	11,000	7.5
cis 1,2-Dichloroethene	4	190	97
Trans 1,2-Dichloroethene	4	1,300	10
Dichloromethane	2	3,300	230
1,2-Dichloropropane	2	100	10
Diethyl phthalate	2	45	31.5
Dimethyl phthalate	4	55	15
Di-n-butyl phthalate	4	12	10
Endrin	0	1	0.1
Ethyl acetate	5	50	42
Ethyl Benzene	5	580	38
bis (2-Ethylhexyl) phthalate	6	110	22
Isophorene	10	85	10
Methyl ethyl ketone	110	28,800	8,300
Methyl isobutyl ketone	10	660	270
Naphthalene	4	19	8
Nitrobenzene	2	40	15
4-Nitrophenol	17	40	25
Pentachlorophenol	3	25	3
Phenol	10	28,800	257
2-Propanol	94	10,000	6,900
1,1,2,2-Tetrachloroethane	7	210	20
Tetrachloroethene	2	100	40
Tetrahydrofuran	5	260	18
Toluene	2	1,600	166
Toxaphane	0	5	1
1,1,1-Trichloroethane	0	2,400	10
1,1,2-Trichloroethane	2	500	10
Trichloroethane	1	43	3.5
Trichlorofluoromethane	4	100	12.5
Vinyl chloride	0	100	10
m-Xylene	21	79	26
p-Xylene = o-Xylene	12	50	18

Source: U.S. Environmental Protection Agency, 1986 (units in µg/l)

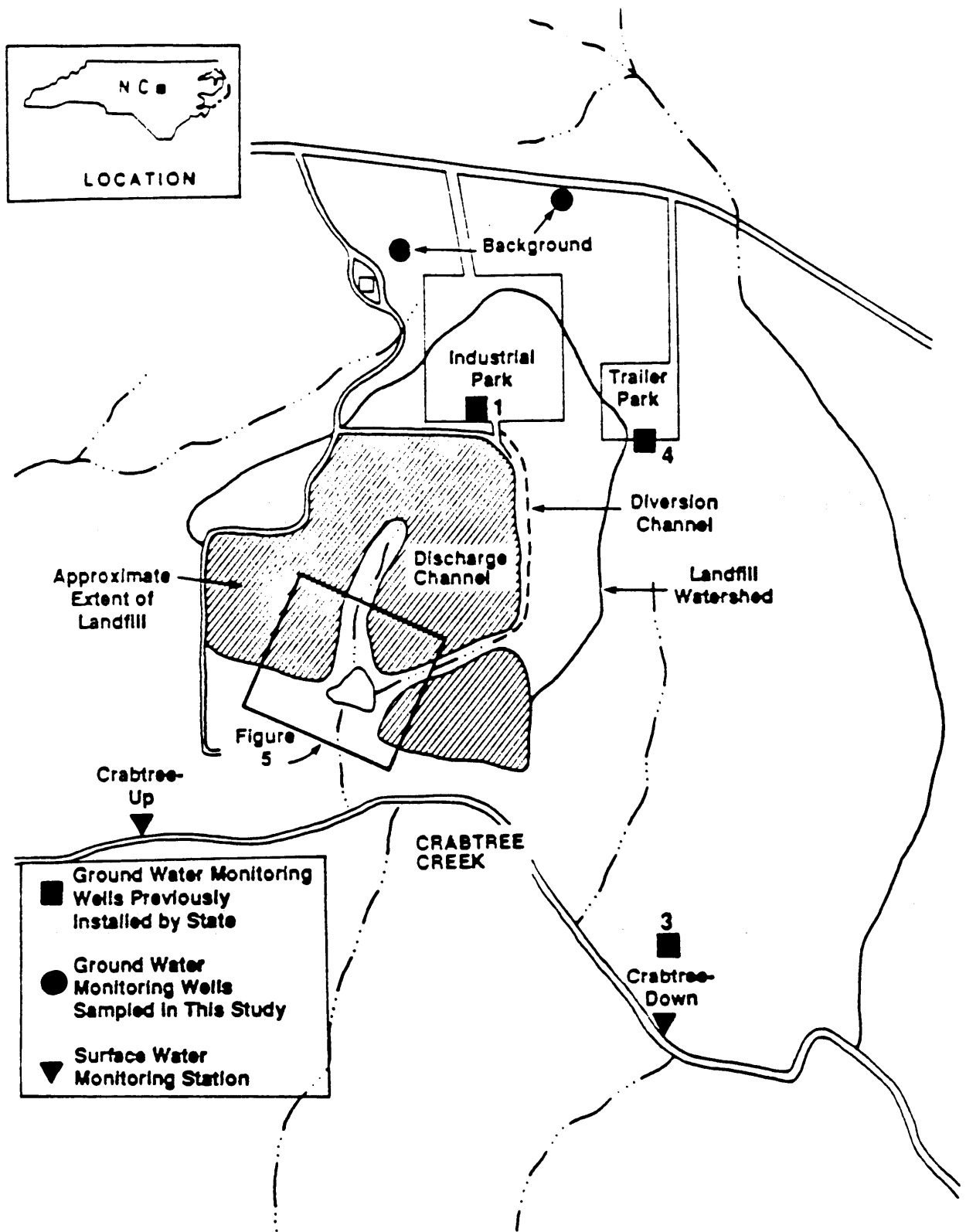


Figure 1. Landfill Location and Site Features.

topography. The refuse disposal area is divided into two regions by a small discharge channel passing through the center of the landfill. The portion of the landfill to the west of the channel was deposited between about 1972 and about 1982. Refuse on the eastern side is younger, having been buried from about 1982 to present. The landfill has recently received a permit for a vertical expansion. Surface water from upstream of the landfill is now being diverted around the fill area into a small diversion channel on the east side. This water mixes with landfill drainage in a sedimentation pond-wetland prior to discharge to Crabtree Creek. The larger discharge channel in the center of the landfill is now being filled with refuse.

Background Geology

Geologic mapping indicates that the landfill is located over a granitic intrusion known as the Rolesville Batholith (Parker, 1979). The Rolesville Batholith was probably formed during the Middle Paleozoic Era and is described as being an intrusion of Adamellite. Adamellite is a massive, gray granitic rock which is composed of subequal parts of sodic plagioclase and potassium feldspars with quartz, biotite, muscovite, and other accessory minerals. Diabase dikes are scattered throughout the batholith. To the immediate west are injected gneiss and schist, which may underlie a portion of the site. The gneiss and schist are described as being layered and having numerous dikes and sills of granite, pegmatite and aplite.

Core samples taken during drilling have revealed a granitic saprolite of varying thickness underlying the study site. A portion of the site near the drainage channel is underlain by 20 to 30 feet of colluvial sedimentary deposits. These deposits vary widely in composition, and are often similar in appearance to granitic saprolite. Part of the upper colluvium was noted to have a high organic fraction. The saprolite appears to have a high quartz grain content and a fairly low cohesiveness.

Soil Classification

Soils throughout the site are predominantly sandy loams and loamy sands. The U.S. Department of Agriculture soil survey report indicates that the predominant soil series are Appling sandy loam and Louisburg loamy sand (Lawthorn, 1970). A previous soil investigation (Soil & Material Engineers, 1986) showed that the landfill soil is predominantly composed of clayey sand to silty sand. Soil samples taken for this investigation confirm these findings.

Fracture Trace Analysis

A fracture trace analysis may be used to identify subsurface fracture patterns through the observation of surface features. Large fractures commonly have some geomorphological manifestation, such as a sudden directional change in a stream channel or a set of opposing tributary streams with the same directional trend. Fracture analysis was used in this study to identify the primary water bearing fractures in the bedrock aquifer. The placement of monitoring wells in the bedrock

was guided by the identification of fractured zones in the bedrock. As there is essentially no information in the literature regarding fracture patterns in the Rolesville Batholith, certain topographic features were assumed to reflect the subsurface fracture patterns. Figure 2 highlights the topographic features which may reflect major bedrock fractures. The drainage way running through the approximate center of the landfill appears to be a fracture trace. For this reason, the bedrock monitoring wells were constructed as closely as possible to this feature.

Ground water Hydrology

Ground water conditions at the landfill are believed to be typical of those common to the piedmont physiographic province. Three distinct geologic layers can be distinguished: a shallow regolith or saprolite layer, a transition zone and the underlying bedrock (Heath, 1980). The boundaries between the three zones are often difficult to clearly identify. Figure 3 illustrates the relationships between the layers.

The saprolite layer consists of material weathered in-place from the underlying bedrock. In this layer, most of the feldspars and heavy minerals have been weathered to clays. By definition, however, saprolite retains some of the parent material's structure, such as foliations and lineations. For the purposes of this study, the colluvium observed beneath the landfill was considered to be saprolite. Rising head or "slug" tests in the saprolite indicate a variable hydraulic conductivity, ranging between 3×10^{-5} and 1×10^{-3} cm/s (Britt and Van Tassel, 1988, Marshall, 1989).

The transition zone is composed of saprolite and partially weathered bedrock. In this zone, weathering has resulted in increased primary porosity but has not been so extensive as to block fractures present in the parent bedrock. Slug and pumping tests commonly indicate that the permeability of the transition zone maybe greater than the saprolite or bedrock layers. Pumping of a well screened in the transition zone suggests that this is true for the landfill examined in this study.

The bedrock layer is composed of unweathered parent material. Secondary porosity features, such as fractures, quartz veins, and intrusive dikes and sills, are primarily responsible for permeability in the bedrock. Pump test data from well RW-1 indicate that the permeability of the underlying bedrock is about 10^{-3} cm/s (Roudebush and Whitman, 1989).

There are also three discernible hydrogeologic zones: the unsaturated zone; the capillary fringe; and the saturated zone (Heath, 1980). The unsaturated zone extends from the surface to the capillary fringe. Void spaces in this region are only partially filled with water. In the capillary fringe, the material is saturated, but the water is under a negative pressure head (with respect to atmospheric pressure). Surface tension forces hold the water in the soil's void spaces (Freeze and Cherry, 1979). In the saturated zone the void spaces are filled with water, but water pressures are

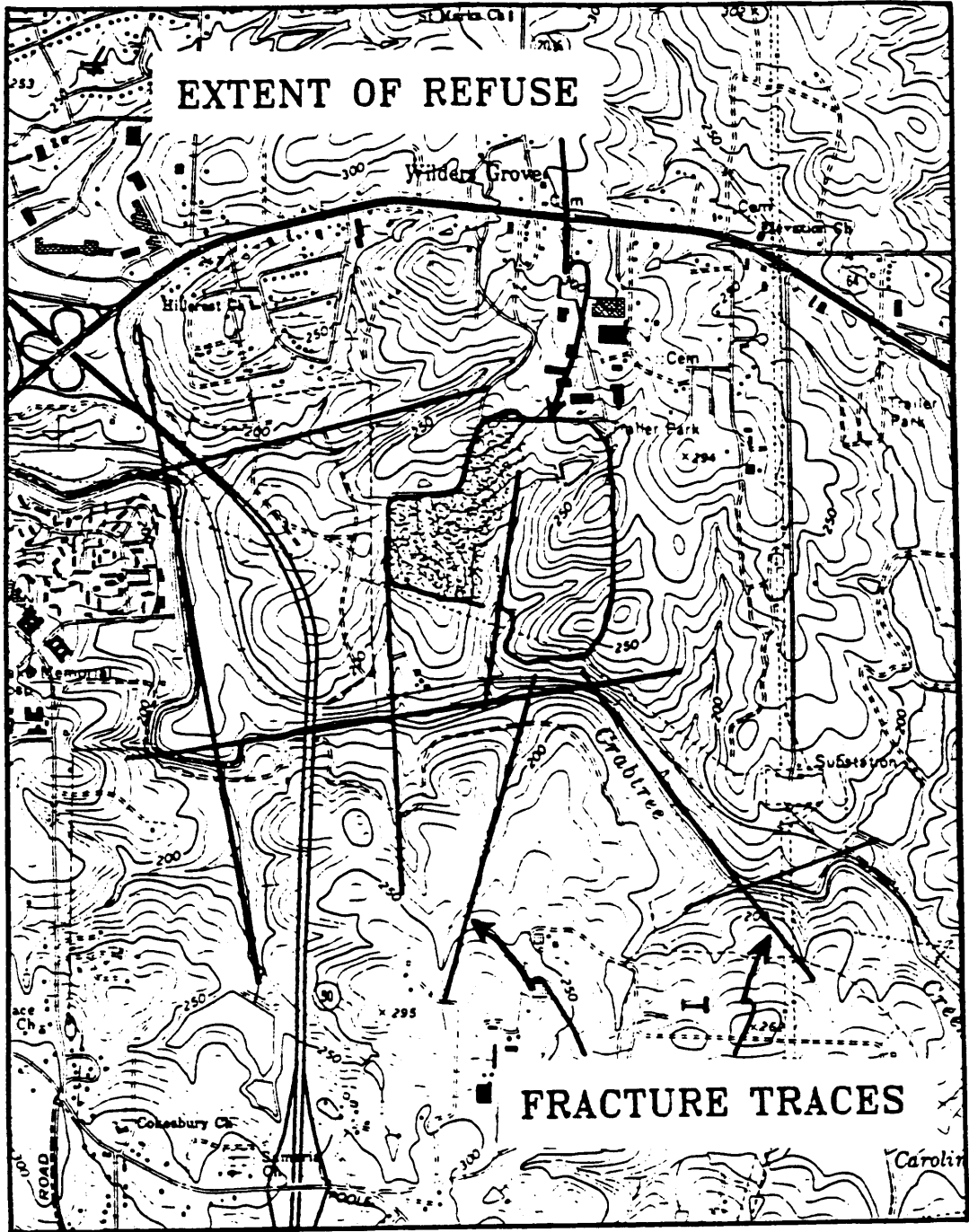


Figure 2. Fracture Trace Analysis.

HYDROGEOLOGIC ZONES

HYDROGEOLOGIC TERMS

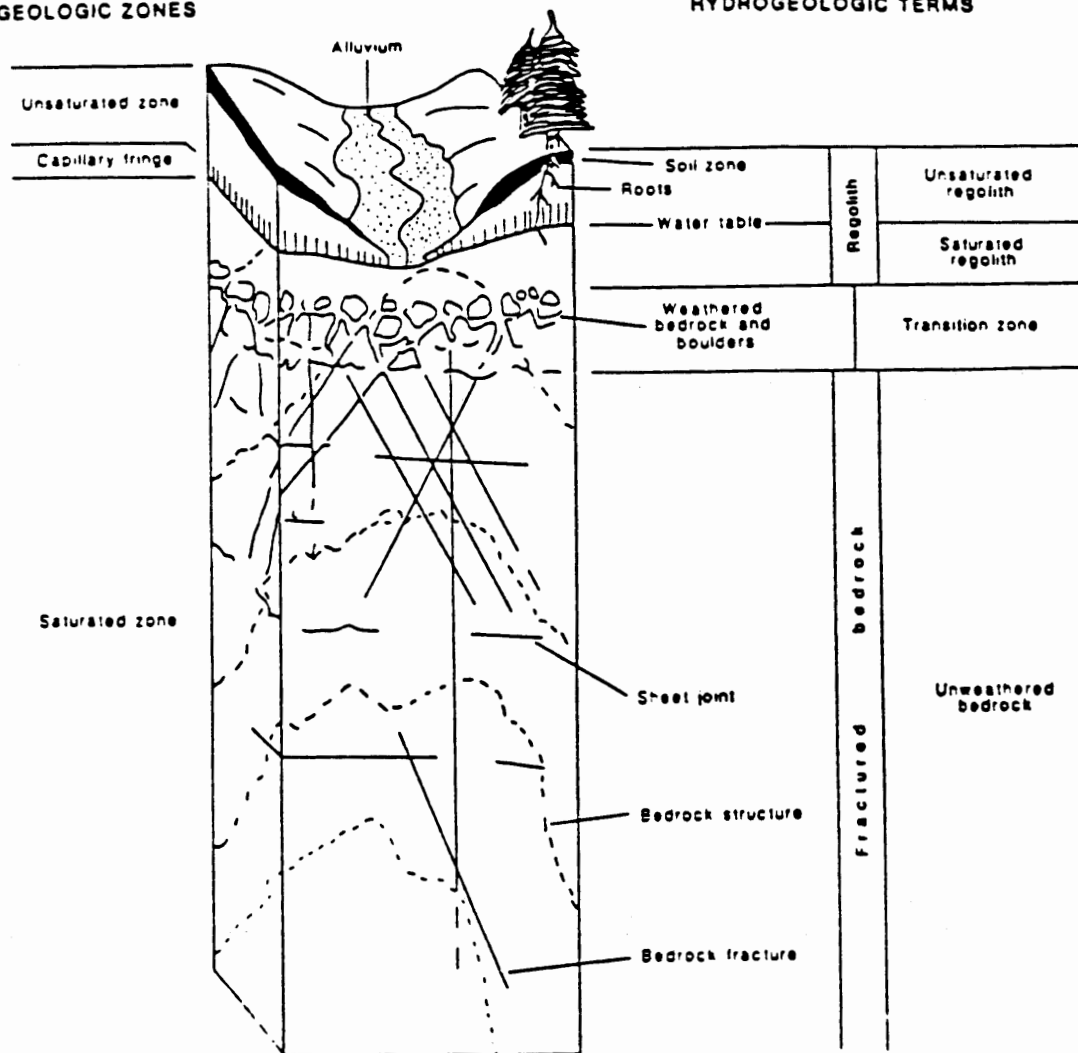


Figure 3. Typical Hydrogeologic Features in the Piedmont Physiographic Province (Source: Heath, 1980).

positive. Depth-to-water measurements taken in wells mark the top of the saturated zone.

Leachate released from the waste will flow vertically through the landfill base into the saprolite and/or transition zones. Leachate which has entered the transition zone, may flow laterally and discharge at the surface or pass through the transition zone and enter the underlying fractured bedrock aquifer. Generalized ground water migration paths are shown on Figure 4. We believe that most ground water at this site flows from elevated surface features, such as the refuse disposal area and other upland areas, through the saprolite and transition zones towards either the smaller drainage features or directly to Crabtree Creek. Water entering the bedrock is believed to discharge primarily to Crabtree Creek.

Decreasing hydraulic head measured in closely spaced monitoring wells support the existence of vertical flow between the saprolite and bedrock aquifers. Water levels in shallow wells were higher than those in adjacent rock wells (0.6 feet upgradient; 3.2 feet downgradient), suggesting the downward flow of water into the bedrock. The magnitude of these differences suggests that the saprolite-colluvium and bedrock zones are poorly linked hydrologically. This observation may suggest that lateral flow occurs through the saprolite at the lithic contact. It is likely that such lateral flow would be intercepted by the diversion channel (Figure 1). A good hydraulic connection between the saprolite and rock would probably result in a smaller vertical hydraulic gradient than was observed.

Surface Water Hydrology

Surface water runoff eventually discharges southward into Crabtree Creek, which joins the Neuse River a few miles to the southeast. The topography itself tends to reflect the underlying crystalline bedrock. Drainage features appear to be a combination of dendritic and fracture-controlled patterns. The Soil Conservation Service curve number calculated for the landfill watershed is about 79. [This curve number is based on noncultivated land use with no mechanical treatment, poor hydrologic condition, and group B hydrologic soil group (McCuen, 1989.) The curve number is an index that represents the combination of a hydrologic soil group and a land use and treatment class as related to stormwater runoff from the watershed. The curve number is empirically related to soil type, soil cover, and antecedent moisture conditions and can be used to calculate estimated runoff from the watershed (Soil Conservation Service, 1975).

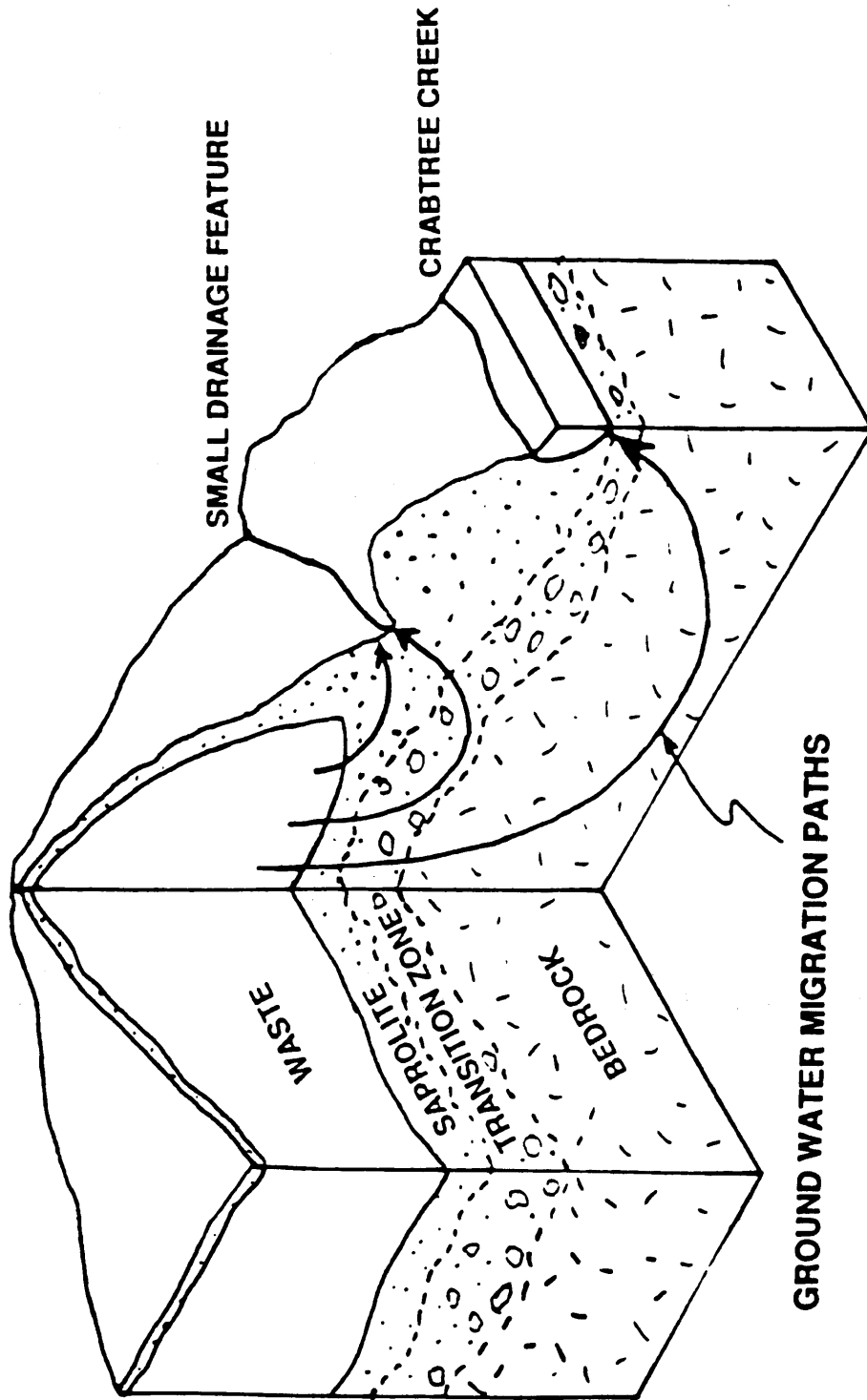


Figure 4. Potential Ground Water Migration Paths.

WATER QUALITY MONITORING PROGRAM

A water quality monitoring program was developed to characterize the chemical quality of surface and ground water at the Piedmont landfill. The different monitoring methods used are described in this section, along with any problems encountered.

Surface Water Monitoring

Four monitoring stations were installed to determine the effect of the landfill on surface water quality. Two surface water monitoring stations were located along Crabtree Creek in an attempt to monitor changes in water quality along the receiving stream (Figure 1). The monitoring stations were located along the creek in positions upstream and downstream of the landfill property and are referred to in Figure 1 as "Crabtree-Up" and "Crabtree-Down," respectively. Two additional stations were located on landfill property just upstream and just downstream of the sedimentation basin. The upstream sampling station is located in the central drainage channel separating the two sides of the landfill. This station is referred to as the "Inflow" station and is assumed to be representative of surface water quality shortly after discharge from the refuse. The sampling station downstream of the sedimentation basin is referred to as the "Outflow" station and is assumed to be representative of surface water quality as it leaves the site and enters Crabtree Creek. The locations of the Inflow and Outflow stations are shown in Figure 5.

Surface water at the sanitary landfill can be separated into two basic categories: stormflow and baseflow. Baseflow has been defined for this study as water discharging from the site during periods of dry weather. The source for this flow is believed to be primarily ground water. Stormflow has been defined as streamflow which occurs during and immediately after (within 24 hrs) a precipitation event.

Baseflow was monitored at both the Inflow and Outflow stations. A 60 degree V-notch weir was installed in the stream channel at the Outflow station to monitor baseflow. At the Inflow station, all flow discharges as a free outfall from a pair of reinforced concrete culverts. Discharge at the Inflow station was determined by measuring the time required to fill a bucket of known volume. Water quality samples were collected by submerging the sampling jars in the flow until full. The jars were then capped, placed on ice, and returned to the laboratory for analysis.

Storm flow was evaluated by measuring stream flow and collecting water samples at regular intervals during several precipitation events. The streamflow data were then used to generate flow-weighted average samples for later analysis and to calibrate a water balance simulation model of the landfill. Several different types of equipment were used during this study, with varying degrees of success, to collect water samples and streamflow data. Initially, fully automated and integrated sampling systems (Marsh-McBirney Flotote pressure and flow transducers coupled

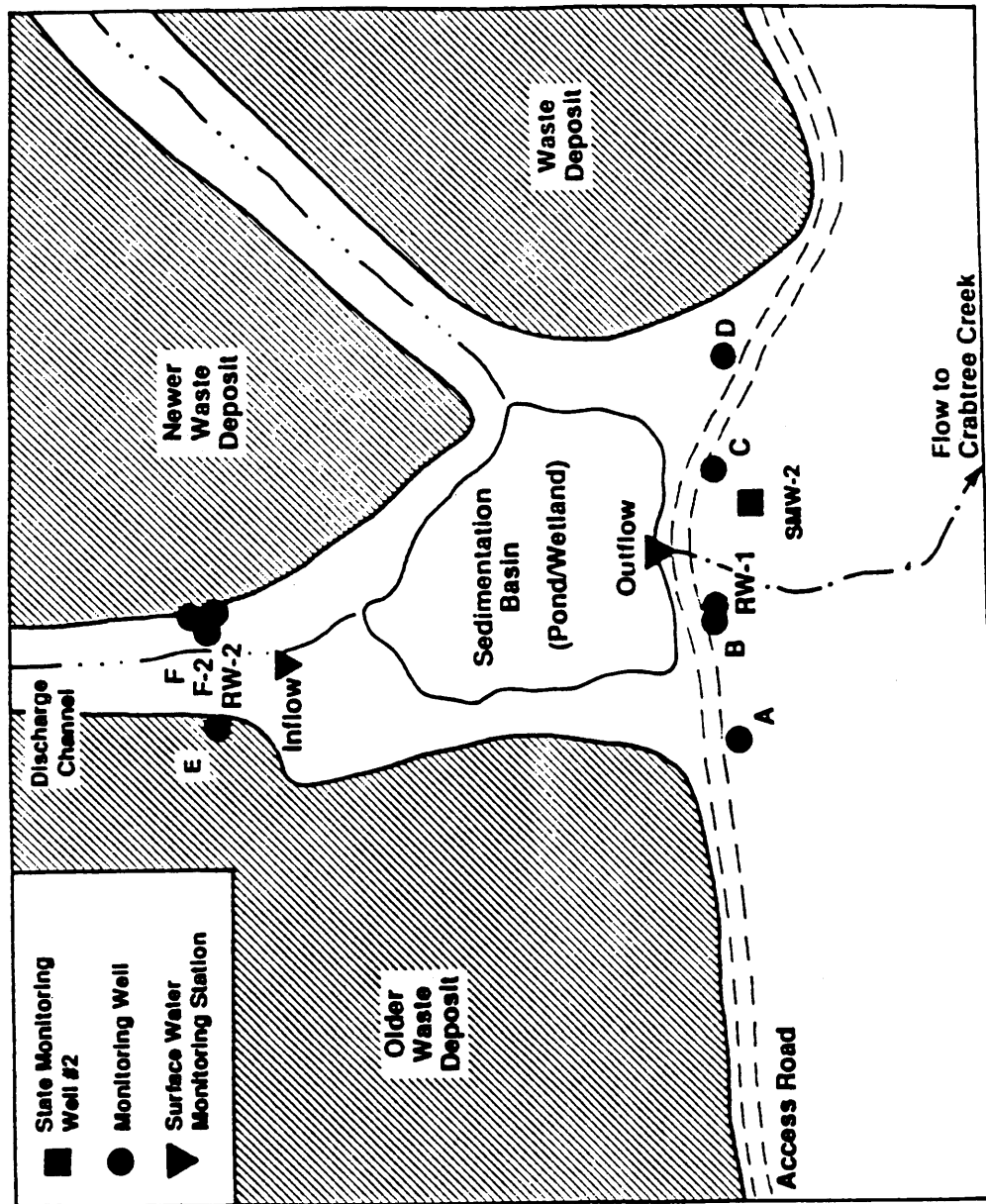


Figure 5. Ground Water and Surface Water Monitoring Stations in the Vicinity of Waste Deposits.

to Manning automatic samplers) were used for both the Inflow and Outflow stations. The intent was to be able to automatically monitor precipitation events without being present during the storm itself. Numerous technical difficulties occurred with this system including plugging of water sampling lines, loss of suction in the vacuum pump, defective batteries, and numerous other problems. Because of these problems, the automatic monitoring systems were eventually abandoned in favor of a more direct sampling approach. The final storm event monitored for this study (July, 1989) was measured using a scale, sample jars, and a rain gage. Depth-of-flow readings, water samples and precipitation measurements were taken at 15 minutes intervals for the duration of the event. Depth-of-flow was related to discharge using a previously developed stage discharge curve. This curve was developed using published stage discharge relationships (U.S. Department of Transportation, 1977). Field measurements of flowrate and depth matched the published relationships within the range of measurement error.

Sufficient depth/time data were collected to generate complete hydrographs for two storm events and a partial hydrograph for a third storm event. The first complete storm, October 1988, and the partial storm, July 1988, were recorded by the Marsh-McBirney Flotote system. The second complete storm, July 1989, was directly measured with a scale and a rain gage. The partial storm hydrograph was interpolated from Flotote data for the July 1988 storm event.

The total flow volume for a storm was found by integrating the area under the hydrograph. The amount of runoff occurring during each storm event was then computed by dividing the total flow volume by the watershed area. The October 1988 storm was found to have approximately 0.20 inches of runoff with 0.96 inches of precipitation. The July 1989 storm had 0.086 inches runoff with 1.10 inches of total precipitation. The July 1988 event was estimated to have 0.84 inches of runoff with 2.35 inches of precipitation.

Ground Water Monitoring

Nine monitoring wells were installed in positions adjacent to and downgradient from refuse disposal areas (Figure 5). Monitoring wells were arranged to determine water quality differences both horizontally across the site, and vertically between the saprolite-colluvium, transition and bedrock zones. Wells were placed in two general regions. Wells E, F, F-2 and RW-2 were located to evaluate ground water quality adjacent to waste disposal areas. These wells will be referred to as the "Upgradient wells" in reference to their hydrologic position with respect to the sedimentation basin. Ground water intercepted by these wells is assumed to originate as landfill leachate and to flow laterally and downward toward the drainage channel, the sedimentation basin or the bedrock aquifer. In this region, wells F, F-2 and RW-2 were located adjacent to relatively recent waste deposits (less than five years old, but not currently receiving waste), and well E was located adjacent to waste deposits more than five years old. In the second region, wells A, B, C, D and RW-1 were located hydrologically downgradient of the site. These wells

will be referred to as the "Downgradient Wells." Upgradient wells F, F-2 and RW-2 are located only several feet apart and provide a vertical profile of ground water quality, as do downgradient wells B and RW-1. Figure 6 is a cross section showing the location of the Upgradient wells in relation to waste deposits, geologic layers and the central discharge channel.

Two existing water supply wells hydrologically upgradient from the sanitary landfill were sampled for background water quality data (landfill supply well and Capitol Trailer Court supply well) (Figure 1). There are no water supply wells directly downgradient from the landfill.

The majority of the wells were constructed with casing and screen, while the remainder were built open to the bedrock. Permits required by the State of North Carolina Division of Environmental Management were obtained prior to installation of any wells. All wells were constructed in compliance with these permits. Data on the construction of these wells is provided in Table 3.

Wells A through F were installed in the shallow saprolite-colluvium soil zone. These wells were constructed using a 3.5 inch diameter solid-stem auger and a soil sampling hydraulic probe. The casing and screens were made of PVC which had a 2 inch internal diameter. The screens were #10 slot and were 5 feet in length. The screens were packed with native material, which formed as the borehole collapsed after drilling. Bentonite seals were placed at the top of the wells. Steel protective casings were set in concrete above the bentonite seals, and fitted with locking caps.

Wells RW-1 and RW-2 were set in the granitic bedrock, while well F-2 was set in the transition zone between the saprolite and bedrock. These wells were installed using a CME-75 drill rig with a 10.5-inch outside diameter, hollow-stem augers. The augers were used when drilling in the material above the bedrock. When hard rock was reached, 4-inch internal diameter steel casing was set in place with cement grout and an air rotary hammer was employed to continue drilling into the rock. The portion of the well in the bedrock was left as is without screen. Wells RW-1 and RW-2 were completed in this manner to a total depth of 109 feet below ground surface. Well F-2 resulted from a failed attempt to drill RW-2. This well has the steel casing to 25 feet, with a 5-foot long, 2-inch internal diameter screen set with packers between 25 and 30 feet below ground surface.

Soil caving and heaving presented significant problems during drilling. The shallow monitoring wells (A-F) were limited in depth by caving of the borehole sides once the auger was removed. Soil heave limited core sampling during the construction of the deep rock wells.

All monitoring wells were developed by repeated surging and bailing to remove fine-grained sediments which may have plugged the screens. Prior to sample collection, each well was purged of 3 to 6 well volumes. The 2-inch diameter wells were bailed, while the 4-inch diameter wells were pumped using a stainless steel

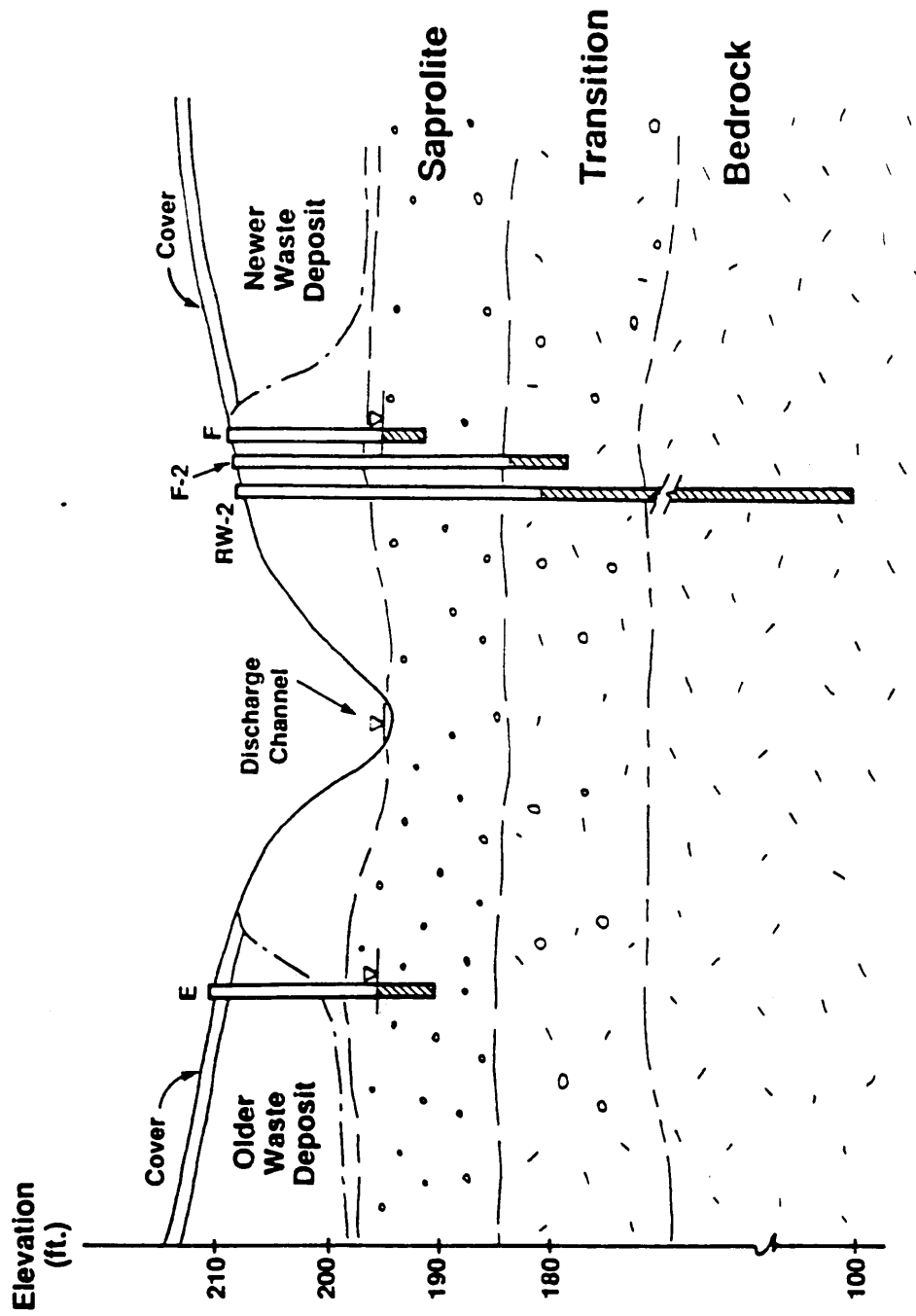


Figure 6. Landfill Cross Section Showing Upgradient Well Locations (Horizontal and vertical dimensions are not to scale).

Table 3. Summary of Monitoring Well Characteristics

WELL	LOCATION	TYPE	TOP OF CASING ELEVATION	SCREENED ELEVATION ¹	AVERAGE WATER TABLE INTERVAL ²
A	DOWNGRAD	SAPROLITE	188.99	10-15	179.27
B	DOWNGRAD	COLLUVIUM	188.83	10-15	179.76
C	DOWNGRAD	COLLUVIUM	186.69	10-15	180.06
D	DOWNGRAD	COLLUVIUM	188.36	10-15	180.50
RW-1	DOWNGRAD	BEDROCK	187.80	58-109	176.65
E	UPGRAD	SAPROLITE	211.33	15-20	195.07
F	UPGRAD	SAPROLITE	209.86	12-17	196.50
F-2	UPGRAD	TRANSITION	208.71	25-30	196.38
RW-2	UPGRAD	BEDROCK	208.86	28-109	196.60

¹ All units in feet

² Below Top-of-Casing

submersible pump. Upon collection, all samples were immediately iced and returned to the laboratory for later analysis.

Under state law, the landfill operators are required to monitor surface and ground water quality and to submit annual reports which include this data. Several ground water wells and surface water sampling points for this monitoring program were in place prior to this study. Locations of these sampling points are included in Figures 1 and 5. Although the monitoring data from these sampling points are not analyzed rigorously in this study, a brief discussion of this data is presented in a later section.

Sampling and Analytical Procedures

Surface and ground water were sampled on seven different occasions. Surface water samples were collected in July and October of 1988, February, March, July and November of 1989. Ground water samples were collected in January, March and November of 1989. Most chemical analyses were performed in the Environmental Engineering Laboratories at North Carolina State University (NCSU). The North Carolina Solid and Hazardous Waste Management Branch (S&HWMB) assisted in this project by performing some analyses for inorganic and organic parameters.

All chemical analyses performed at NCSU were based on procedures described in Standard Methods for the Examination of Water and Wastewater (American Public Health Association, 1985). Specific conductivity, temperature and pH were measured by use of standard electrodes. During the initial stages of this project, specific ion electrodes were used to measure nitrate and ammonia. Analysis of spiked samples indicated that these procedures gave inconsistent results, presumably due to interferences by other dissolved compounds. Standard wet chemical methods were then used to measure these compounds. Nitrate was measured by the cadmium reduction colorimetric procedure (Hach, Inc.). Ammonia was measured by the titrimetric procedure following initial distillation. Chloride concentrations were measured using the selective ion probe method. Chemical oxygen demand was determined by the ampule colorimetric procedure (Hach, Inc.). Analyses for iron, manganese and chromate measured at NCSU were measured using Chemetrics, Inc. portable colorimetric test kits. These results were then compared with analyses performed by the N.C. S&HWMB laboratory using the atomic adsorption method. In all cases, the colorimetric kits provided results comparable to the atomic adsorption procedure. Organic analyses performed by S&HWMB followed USEPA procedures (methods 624 and 625). A list of organic compounds which could be quantified by this procedure is shown in Table 4.

Quality Assurance/Quality Control

All water sampling devices were decontaminated between uses. Bailers and similar sampling equipment were disassembled, scrubbed in hot soapy water, rinsed, washed with hydrochloric acid, rinsed with deionized water, washed with isopropyl alcohol, rinsed with deionized water and allowed to dry. The bailers and other equipment were reassembled and covered with aluminum foil for transport to the

Table 4. Organic Chemicals which can be Identified by USEPA Methods 624 and 625.

<u>Purgable (Volatile) Compounds</u>	<u>Base/Neutral and Acid Extractables</u>	
chloromethane	N-nitrosodimethylamine	benz(a)anthracene
bromomethane	bis (2-chloroethyl) ether	chrysene
dichlorodifluoromethane	2-chlorophenol	3,3-dichlorobenzidine
vinyl chloridephenol	phenol	bis (2-ethylhexyl) pthalate
chloroethane	1,2-dichlorobenzene	di-n-octyl pthalate
methylene chloride	1,4-dichlorobenzene	benzo(b)fluoranthene
trichlorofluoromethane	1,2-dichlorobenzene	benzo(k)fluoranthene
1,1-dichloroethene	bis (2-chloroisopropyl) ether	benzo(a)pyrene
1,1-dichloroethane	hexachloroethane	indeno(1,2,3-cd)pyrene
1,2-cis-dichloroethene	N-nitroso-di-n-propylamine	dibenzo(a,h)anthracene
chloroform	nitrobenzene	benzo(g,h,i)perylene
1,2-dichloroethane	isophorone	aniline
1,1,1-trichloroethane	2-nitrophenol	benzoic acid
carbon tetrachloride	2,4-dimethylphenol	benzyl alcohol
bromodichloromethane	1,2,4-trichlorobenzene	4-chloroaniline
1,2-dichloropropane	napthalene	dibenzofuran
1,3-trans-dichloropropene	hexachlorocyclopentadiene	2-methylnapthalene
trichloroethylene	2,4,6-trichlorophenol	2-methylphenol
chlorodibromomethane	2-chloronapthalene	4-methylphenol
benzene	acenapthalene	2-nitroaniline
1,1,2-trichloroethane	2,4-dinitrotoluene	3-nitroaniline
1,3-cis-dichloropropene	4-nitrophenol	4-nitroaniline
2-chloroethyl-vinyl ether	fluorene	2,4,5-trichlorophenol
bromoform	4-chlorophenylphenyl ether	caprolactam
1,1,2,2-tetrachloroethane	diethyl pthalate	hexanoic acid
tetrachloroethene	4,6-dinitro-o-cresol	heptanoic acid
toluene	diphenylamine	octanoic acid
chlorobenzene	azobenzene	
ethylbenzene	4-bromophenyl ether	<u>Pesticides/PCB's</u>
acetone2-butanone (MEK)	hexachlorobenzene	PCB's
carbendisulfide	pentachlorophenol	Endrin
2-hexanone	phenanthrene	Lindane
4-methyl-2-pentatone (MIBK)	anthracene	Methozychlor
styrene	dibutyl pthalate	Toxaphene
vinyl acetate	fluoranthene	
xylenes (total)	pyrene	
Methyl-t-butyl ether	benzidine	
Isopropyl ether	butyl benzyl pthalate	

field. Blank samples collected during this study verified the effectiveness of this procedure. The stainless steel submersible pump was cleaned by allowing the pump to run for about 20 to 30 minutes in a bucket of soapy water, followed by tap water, a mixture of deionized water and isopropyl alcohol, and then deionized water. Vinyl gloves were worn during all decontamination procedures as well as while handling the decontaminated equipment.

Immediately after collection, all samples vials were tightly capped and placed on ice in insulated coolers. Samples for COD, metals, nitrate, and ammonia were acidified in the field with HCl. All vials for organics were filled without head space. Duplicate analyses were performed on at least one monitoring point from every

SUMMARY OF WATER QUALITY MONITORING RESULTS

A summary of the ground and surface water quality monitoring data is presented in tables 5 to 8. The sampling location, average value and range, average concentration, and number of observations are given for each parameter. Inorganic and organic parameters are listed separately. The two water supply wells (SUPPLY) sampled during this study did not show any evidence of contamination and have been assumed to be representative of uncontaminated groundwater. For some parameters, duplicate data were available from the S&HWMB for a particular monitoring point. Duplicate data from the NCSU and S&HWMB laboratories were averaged. In a few cases, individual organic compounds were identified, but the estimated concentrations were below the USEPA specified detection limit. These values were calculated by comparison with an internal standard and are listed as estimated values.

The data set for any given parameter is often limited to a few observations for a location. Analysis of duplicate and spiked samples indicates that the reported values are reasonably accurate.

The major uncertainty in analysis of the monitoring data is the temporal variation in pollutant concentrations. Ground water moves very slowly. Consequently, large changes in pollutant concentrations in ground water are unusual and the ground water quality data is expected to be fairly representative of long term trends. However,, previous surface water quality investigations have shown that pollutant concentrations in surface water are highly variable, especially during precipitation events. Because of the high concentration variability and small data set obtained in this study, there is considerable uncertainty in our estimates of average pollutant concentrations and mass loads discharging from the landfill. Average surface water pollutant concentrations listed in this report should be considered to be order-of-magnitude estimates. Nonetheless, we believe that the information contained in this report can be useful in accessing the water quality impacts of a typical sanitary landfill relative to other land uses.

Table 5. Summary of Inorganic Ground Water Quality Results

LOCATION ¹	Parameter ²	MONITORING POINT										SUPPLY NA						
		A DG	B DG	C DG	D DG	E UG	F UG	F-2 UG	RW-1 DG	RW-2 UG								
pH ³	Range	5.8-6.67	5.64-6.86	5.4-5.79	6-6.2	5.5-5.85	4.54-4.7											
	Ave.	6.23	6.25	5.60	6.10	5.67	4.62											
	#Obs.	2	2	2	2	2	2											
Temp	Range	13-20.3	12-20.2	13.5-19.8	12-18.6	16-18	13-17.3	17.3	16	16	16	16	16	16	16	16	16	16
	Ave.	16.4	15.8	16.6	14.9	16.7	14.8	17.3	16	16	16	16	16	16	16	16	16	16
	#Obs.	3	3	2	3	3	3	1	1	1	1	1	1	1	1	1	1	1
Specific Cond ⁴	Range	560-686	421-623	177-248	201-271	232-343	1150-1502	754	451	451	685	685	685	685	685	685	121-270	196
	Ave.	619	492	213	229	294	1351	754	451	451	685	685	685	685	685	685	196	196
	#Obs.	3	3	2	3	3	3	1	1	1	1	1	1	1	1	1	2	2
Chloride	Range	27-38.7	34-54.9	39.0	9-11.8	51-66	80-100	59	16	16	60	59	59	59	59	59	12	12
	Ave.	32.9	44.5	39.0	10.4	58.5	90.0	59	16	16	60	59	59	59	59	59	12	12
	#Obs.	2	2	1	2	2	2	1	1	1	1	1	1	1	1	1	2	2
Fluoride	Range	<0.1-0.13	0.14-0.22		<0.1-1.1	<.10	<.10	<.10	<.10	<.10	<.10	<.10	<.10	<.10	<.10	<.10	0.23	0.23
	Ave.	0.12	0.18		0.11	<.10	<.10	<.10	<.10	<.10	<.10	<.10	<.10	<.10	<.10	<.10	0.23	0.23
	#Obs.	2	2		2	2	2	1	1	1	1	1	1	1	1	1	2	2
Nitrate-N	Range	0.1	0.1		0.1	0.15	0.2	0.06	0.11	0.11	0.1	0.1	0.1	0.1	0.1	2-3.5	2-3.5	2
	Ave.	0.1	0.1		0.1	0.15	0.2	0.06	0.11	0.11	0.1	0.1	0.1	0.1	0.1	2.75	2.75	2
	#Obs.	1	1		1	1	1	1	1	1	1	1	1	1	1	2	2	2

Table 5. (continued)

LOCATION	MONITORING POINT										
	A DG	B DG	C DG	D DG	E UG	F UG	F-2 UG	RW-1 DG	RW-2 UG	SUPPLY NA	
Parameter	Range	6.5-8.5	2.74-3.7	1.3	0.06-1.37	0.39-1.5	0.78-0.5				<0.1
Ammonia-N	Ave.	7.5	3.22	1.3	0.72	0.95	0.64	ND	ND	ND	<0.1
	#Obs.	2	2	1	2	2	2				1
	Range	<3.0	<3.0		<3.0-3.0	<3.0	<3.0	<3.0	<3.0	<3.0	<3.0-23
Sulfates	Ave.	<3.0	<3.0	ND	<3.0	<3.0	<3.0	<3.0	<3.0	<3.0	13
	#Obs.	2	2		2	2	2	1	1	1	2
	Range	74-179	45-149	23	33-65	35	804-982	84	13	23	43-63
COD	Ave.	126.5	97	23	49	35	893	84	13	23	53
	#Obs.	2	2	1	2	1	2	1	1	1	2
	Range	75-96	29-60		19-23	25-101	380-391	60	25	60	<5-7
TOC	Ave.	86	44.5	ND	21	63	385.5	60	25	60	6
	#Obs.	2	2		2	2	2	1	1	1	2
	Range	248-288	212-360		136-336	112-232	1080	484	280	448	128-160
TDS	Ave.	268	286	ND	236	172	1080	484	280	448	144
	#Obs.	2	2		2	2	1	1	1	1	2
	Range	77-106.5	23-43	25-37.5	1.23-3.4	10.5-17.8	74-160	3.5	2.5	3.8	<0.05-.05
Iron	Ave.	96.2	30.7	31.3	2.2	13.9	109.8	3.5	2.5	3.8	<0.05
	#Obs.	3	3	2	3	3	3	1	1	1	

Table 5. (continued)

LOCATION	Parameter	MONITORING POINT											SUPPLY NA
		A DG	B DG	C DG	D DG	E UG	F UG	F-2 UG	RW-1 DG	RW-2 UG			
Manganese	Range	7.98-8	4.2-27.5	<0.1	5.6-6.7	0.63-3	34.4-106	19.7	8	8.3	<0.03-.1		
	Ave.	7.98	12.61	<0.1	6.1	1.9	64.11	19.7	8	8.3	<0.1		
	#Obs.	3	3	2	3	3	3	1	1	1	2		
Zinc	Range	<0.05-0.08	<0.05-.15	ND	<0.05	<0.05	<0.05-.05	0.16	<0.05	<0.05	<0.05-.29		
	Ave.	0.07	0.10	ND	<0.05	<0.05	0.05	0.16	<0.05	<0.05	0.17		
	#Obs.	2	2	2	2	2	2	1	1	1	2		
Barium	Range	0.20	0.2-0.6	ND	<0.10	0.2-0.3	0.20	0.20	0.10	0.15	<0.10		
	Ave.	0.20	0.40	ND	<0.10	0.25	0.20	0.20	0.10	0.15	<0.10		
	#Obs.	2	2	2	2	2	2	1	1	1	1		
Lead	Range	<0.03	<0.03-0.37	ND	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03		
	Ave.	<0.03	0.20	ND	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03	<0.03		
	#Obs.	2	2	2	2	2	2	1	1	1	2		
Mercury	Range	<0.002	<0.002-.0005	ND	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002		
	Ave.	<0.0002	0.00035	ND	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002		
	#Obs.	2	2	2	2	2	2	1	1	1	2		

1 UG=Upgradient DG=Downgradient (See Figure 6)

2 All parameter concentrations are in mg/l unless otherwise indicated

3 Standard pH units

4 Specific Conductivity, µmho/cm

ND Not Determined

Table 6. Summary of Organic Ground Water Quality Results - SOC's Identified in this Study

LOCATION ¹		MONITORING POINT								
		A DG	B DG	D DG	E UG	F UG	F-2 UG	RW-1 DG	RW-2 UG	SUPPLY NA
<u>Parameter²</u>										
Phenol	Range	<5.0	<5.0	<5.0	<5.0	10-53	<5.0	<5.0	<5.0	<5.0
	Ave.	<5.0	<5.0	<5.0	<5.0	32	<5.0	<5.0	<5.0	<5.0
	#Obs.	2	2	2	2	2	1	1	1	2
Diethyl Phthalate	Range	<5.0	<5.0	<5.0	<5.0	17-22	<5.0	<5.0	<5.0	<5.0
	Ave.	<5.0	<5.0	<5.0	<5.0	20	<5.0	<5.0	<5.0	<5.0
	#Obs.	2	2	2	2	2	1	1	1	2
4-Methylphenol	Range	<5.0	<5.0	<5.0	10-30	138-367	<5.0	<5.0	<5.0	<5.0
	Ave.	<5.0	<5.0	<5.0	<5.0	20	253	<5.0	<5.0	<5.0
	#Obs.	2	2	2	2	2	1	1	1	2
Vinyl Chloride	Range	<5.0	<5.0	<5.0	8-130	14-59	26	3E	8	<5.0
	Ave.	<5.0	<5.0	<5.0	<5.0-31	13	<5.0	2E	<5.0	<5.0
	#Obs.	2	2	2	2	2	1	1	1	2
Methylene Chloride	Range	<5.0	<5.0	<5.0	<5.0-31	13	<5.0	2E	<5.0	<5.0
	Ave.	<5.0	<5.0	<5.0	18	13	<5.0	2E	<5.0	<5.0
	#Obs.	2	2	2	2	2	1	1	1	2
1,1-Dichloroethane	Range	<5.0	<5.0	<5.0	25-52	86-98	60	3E	26	<5.0
	Ave.	<5.0	<5.0	<5.0	39	92	60	3E	26	<5.0
	#Obs.	2	2	2	2	2	1	1	1	2

Table 6. Continued

LOCATION ¹	Parameter ²	MONITORING POINT											SUPPLY NA	
		A DG	B DG	D DG	E UG	F UG	F-2 UG	RW-1 DG	RW-2 UG					
1,2-Dichloroethene	Range	<5.0	<5.0	<5.0	36-182	70-96	12	68	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	Ave.	<5.0	<5.0	<5.0	109	83	12	68	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	#Obs.	2	2	2	2	2	1	1	1	1	1	1	1	2
Trichloroethylene	Range	<5.0	<5.0	<5.0	5-14	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	Ave.	<5.0	<5.0	<5.0	10	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	#Obs.	2	2	2	2	2	1	1	1	1	1	1	1	2
Benzene	Range	<5.0	<5.0	<5.0	4-8	10-16	8	<5.0	<5.0	2E	<5.0	2E	<5.0	<5.0
	Ave.	<5.0	<5.0	<5.0	6	13	8	<5.0	<5.0	2E	<5.0	2E	<5.0	<5.0
	#Obs.	2	2	2	2	2	1	1	1	1	1	1	1	2
Toluene	Range	<5.0	<5.0	<5.0	16-106	610-1015	126	<5.0	<5.0	10	<5.0	10	<5.0	<5.0
	Ave.	<5.0	<5.0	<5.0	61	813	126	<5.0	<5.0	10	<5.0	10	<5.0	<5.0
	#Obs.	2	2	2	2	2	1	1	1	1	1	1	1	2
Xylenes (total)	Range	<5.0	<5.0	<5.0	<5.0-19	64-186	12	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	Ave.	<5.0	v	<5.0	12	125	12	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	#Obs.	2	2	2	2	2	1	1	1	1	1	1	1	2
Tetrachloroethene	Range	<5.0	<5.0	<5.0	5-23	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	Ave.	<5.0	<5.0	<5.0	14	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	#Obs.	2	2	2	2	2	1	1	1	1	1	1	1	2

Table 6 Continued

LOCATION ¹	Parameter ²	MONITORING POINT										SUPPLY NA	
		A DG	B DG	D DG	E UG	F UG	F-2 UG	RW-1 DG	RW-2 UG				
Ethylbenzene	Range	<5.0	<5.0	<5.0	<5.0-8.0	22-53	3E	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	Ave.	<5.0	<5.0	<5.0	7	38	3E	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	#Obs.	2	2	2	2	2	1	1	1	1	1	1	2
Acetone	Range	<5.0	<5.0	<5.0	<5.0	1524	181	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	Ave.	<5.0	<5.0	<5.0	<5.0	1524	181	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	#Obs.	2	2	2	2	1	1	1	1	1	1	1	2
2-Butanone (2-Buta)	Range	<5.0	<5.0	<5.0	<5.0-217	1035-2115	102	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	Ave.	<5.0	<5.0	<5.0	111	1575	102	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	#Obs.	2	2	2	2	2	1	1	1	1	1	1	2
2-Hexanone	Range	<5.0	<5.0	<5.0	<5.0	7-132	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	Ave.	<5.0	<5.0	<5.0	<5.0	70	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	#Obs.	2	2	2	2	2	1	1	1	1	1	1	2
4-Methyl-2-Pentanone (MIBK)	Range	<5.0	<5.0	<5.0	<5.0-7	38-129	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	Ave.	<5.0	<5.0	<5.0	6	84	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	#Obs.	2	2	2	2	2	1	1	1	1	1	1	2
Caprolactam	Range	913-1076	72-117	46-219	61-94	25-151	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	Ave.	995	95	133	78	88	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
	#Obs.	2	2	2	2	2	1	1	1	1	1	1	2

Table 6. Continued

LOCATION ¹	Parameter ²	MONITORING POINT										SUPPLY	
		A DG	B DG	D DG	E UG	F UG	F-2 UG	RW-1 DG	RW-2 UG	NA			
	Hexanoic Acid ³	-	-	-	+	+	ND	ND	ND	ND	ND	ND	ND
	Heptanoic Acid ³	-	-	-	+	+	ND	ND	ND	ND	ND	ND	ND
	Octanoic Acid ³	-	-	-	+	+	ND	ND	ND	ND	ND	ND	ND

¹ UC= Upgradient DG=Downgradient (See Figure 6)

² Units in mg/l

E Estimated

³ + means detected; - means undetected

Table 7. Summary of Inorganic Surface Water Quality Results

Parameter*	MONITORING LOCATIONS								
	INFLOW		OUTFLOW		CRABTREE-UP		CRABTREE-DOWN		
	STORM	BASE	STORM	BASE	STORM	BASE	STORM	BASE	
Specific Cond ¹	Range	423	1314-1712	171	244-389	89		73.4	
	Ave.	423	1513	171	317	89	ND ²	73.4	ND
	#Obs.	1	2	1	2	1		1	
Chloride	Range	32-260	13-70	23.1-31.3	6	1			
	Ave.	111.7	188.2	41.5	27.2	6	ND	6	ND
	#Obs.	3	1	2	2	1	1		
Fluoride	Range	0.18	0.25	0.14	0.10	<0.1		<0.1	
	Ave.	0.18	0.25	0.14	0.10	<0.1	ND	<0.1	ND
	#Obs.		1	1	1	1	1		1
Nitrate-N ³	Range	.15-3.33	<0.1	.05-2.48	<0.1	0.53		<1.0	
	Ave.	1.09	<0.1	0.70	<0.1	0.53	ND	<1.0	ND
	#Obs.	4	1	4	1	1		1	
Ammonia-N	Range	0.07-1.99	4.5-28	0.1-1.48	BDL-0.95	0.73-0.84		0.22	
	Ave.	1.77	16.27	0.58	0.47	0.78	ND	0.22	ND
	#Obs.	4	3	4	3	2		1	
Sulfates	Range	50	7	21	13	8		9	
	Ave.	50	7	21	13	8	ND	9	ND
	#Obs.	1	1	1	1	1		1	
COD	Range	87.5-620	166-962	11-142	23-99.8	68.4-165	23	160	33
	Ave.	264	506	91	48.6	116.7	23	160	33
	#Obs.	4	3	4	3	2	1	1	1

Table 7. Continued

PARAMETER*	MONITORING LOCATIONS											
	INFLOW			OUTFLOW			CRABTREE-UP			CRABTREE-DOWN		
	STORM	BASE	STORM	BASE	STORM	BASE	STORM	BASE	STORM	BASE	STORM	BASE
TOC	Range	328	205	21	16	9	9	9	9	9	9	9
	Ave.	328	205	21	16	9	9	9	9	9	9	9
	#Obs.	1	1	1	1	1	1	1	1	1	1	1
TOT Solid	Range	4515-60770	947-1847	700-3100	226-460	480	480	138	350	350	350	82
	Ave.	32643	1348	1564	353	480	480	138	350	350	350	82
	#Obs.	2	3	3	3	1	1	1	1	1	1	1
TSS	Range	4022-54967	60-240	3-600	<1-100	18	18	25	11	11	11	15
	Ave.	29495	125	268	40	18	18	25	11	11	11	15
	#Obs.	2	3	3	3	1	1	1	1	1	1	1
TDS	Range	493-5803	649-1444	100-2900	219-298	267	267	113	202	202	202	67
	Ave.	3148	993	1297	263	267	267	113	202	202	202	67
	#Obs.	2	3	3	3	1	1	1	1	1	1	1
Total Iron	Range	35-33.83	3.83-20	1.1-2.0	1.7-1.98	3.3	3.3	ND	2.16	2.16	2.16	ND
	Ave.	16.99	14.0	1.55	1.8	3.3	3.3	ND	2.16	2.16	2.16	ND
	#Obs.	2	3	2	2	1	1	1	1	1	1	1
Dissolved Iron	Range	0.1-18.8	0.6-10	0.46-0.8	0.4-1.0	<0.1-0.3	<0.1-0.3	0.35	0.2	0.2	0.2	0.35
	Ave. #Obs.	9.95 3	5.3 2	0.69 3	0.67 3	0.2 2	0.2 2	0.35 1	0.20 1	0.20 1	0.20 1	0.35 1
Manganese	Range	7.32	5.8-9.1	0.7	0.67-3.2	0.18	0.18	0.15	0.14	0.14	0.14	0.15
	Ave.	7.32	7.45	0.7	1.94	0.18	0.18	0.15	0.14	0.14	0.14	0.15
	#Obs.	1	2	1	2	1	1	1	1	1	1	1

Table 7. Continued

Parameter*	MONITORING LOCATIONS											
	INFLOW			OUTFLOW			CRABTREE-UP			CRABTREE-DOWN		
	STORM	BASE	STORM	BASE	STORM	BASE	STORM	BASE	STORM	BASE	STORM	BASE
Zinc	Range	0.4	<0.05	0.07	<0.05	0.06	0.06	ND	0.08	0.08	0.08	ND
	Ave.	0.4	<0.05	0.07	<0.05	0.06	0.06	ND	0.08	0.08	0.08	ND
	#Obs.	1	1	1	1	1	1	1	1	1	1	1
Barium	Range	0.40	0.40	0.10	0.10	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
	Ave.	0.40	0.40	0.10	0.10	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
	#Obs.	1	1	1	1	1	1	1	1	1	1	1
Lead	Range	<0.03	<0.005	0.03	<0.005	<0.03	<0.03	ND	<0.03	<0.03	<0.03	ND
	Ave.	<0.03	<0.005	0.03	<0.005	<0.03	<0.03	ND	<0.03	<0.03	<0.03	ND
	#Obs.	1	1	1	1	1	1	1	1	1	1	1
Mercury	Range	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	ND	<0.0002	<0.0002	<0.0002	ND
	Ave.	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	ND	<0.0002	<0.0002	<0.0002	ND
	#Obs.	1	1	1	1	1	1	1	1	1	1	1

* All parameter concentrations are in mg/l unless otherwise noted.

1 Specific Conductivity, $\mu\text{mho}/\text{cm}$.

2 Not Determined.

3 S&HWMB detection limit is 1.00 mg/l; NCSU limit is 0.1 mg/l.

STORM: Storm flow conditions.

BASE: Base flow conditions.

Table 8. Summary of Organic surface Water Quality Results - SOC's Identified in this Study

Parameter*	MONITORING LOCATIONS											
	INFLOW			OUTFLOW			CRABTREE-UP			CRABTREE-DOWN		
	STORM	BASE	STORM	STORM	BASE	STORM	STORM	BASE	STORM	STORM	BASE	STORM
Phenol	50	21	<5.0	<5.0	10	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
Diethyl Phthalate	25	27	10	10	10	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
4-Methylphenol	240	48	<5.0	<5.0	10	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
Vinyl Chloride	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
Methylene Chloride	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
1,1-dichloroethane	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
1,2-Cis-dichloro-ethene	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
Trichloroethylene	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
Benzene	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
Toluene	20	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
Xylenes (total)	7	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
Tetrachlorethene	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
Ethylbenzene	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
Acetone	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
2-Butanone (2-Buta)	487	628	93	93	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
2-Hexanone	11	<5.0	5	5	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
4-Methyl-2-Pentatone (MIBK)	20	13	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
Caprolactam	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
Tetrahydrofuran	<5.0	21	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
Hexanoic Acid ¹ (+/-)	+	-	+	+	+	+	+	+	+	+	+	ND

* Units are µg/l.

¹ + means detected; - means undetected

STORM: Storm water flow.

BASE: Base flow.

Ground Water Quality

The inorganic water quality data for the ground water monitoring points are summarized in Table 5. Organic compounds of commercial and industrial origin (synthetic organic chemicals or SOC's) detected in monitoring wells are listed in Table 6. Data gaps result from a variety of factors including analytical error and equipment malfunction. Wells C and E were damaged by bulldozer activity and could not be sampled during the second half of this study.

Monitoring well data show that the ground water is contaminated beneath and immediately downgradient from the sanitary landfill. Analysis of the site hydrogeology and water quality monitoring data indicates that a leachate plume is discharging from the landfill and is migrating downgradient through the saprolite, transition zone and bedrock aquifers towards Crabtree Creek. Total dissolved solids, iron, manganese, pH and eighteen SOC's exceed the concentration limits established for Class GA ground waters (North Carolina Department of Environmental Management, 1989) in wells inside the landfill immediately adjoining the waste (E, F, F-2, RW-2). Iron, manganese and pH were in violation of ground water standards in the downgradient saprolite wells (A, B, C, D) and the bedrock well (RW-1). No SOC's were above the analytical detection limit in the saprolite wells. Trace levels (less than 5 µg/l) of vinyl chloride, methylene chloride, and 1,1-dichloroethane were observed in the bedrock well (RW-1) which is below the USEPA detection limit. Based on estimated concentrations, vinyl chloride exceeds the standard while methylene chloride is below the standard. Ground water quality standards have not been explicitly established for 1,1-dichloroethane. Consequently, the analytical detection limit is the ground water standard for this compound. Since the estimated concentration of 1,1-dichloroethane was below the analytical detection limit, this is not a water quality violation. The only organic compound detected in RW-1 above the analytical detection limit was 1,2-dichloroethene (68 µg/l). The water quality standard for this compound is 70 µg /l.

Monitoring data from the two nearby water supply wells located upgradient from the landfill show no adverse impacts due to the landfill. All water quality parameters were within ground water standards. No SOC's were above the analytical detection limit in the landfill's own water supply well. The Capital City Trailer Park is immediately upgradient from the sanitary landfill. The water supply well at the trailer park could not be sampled directly so a sample was obtained from a faucet after the water was chlorinated. Trace concentrations of chlorination byproducts (chloroform and bromodichloromethane) were detected in the trailer park water. These compounds were not detected at any wells within the landfill.

Surface Water Quality

Surface water quality data collected in this study are summarized in Tables 7 and 8. Water quality data were collected during baseflow and stormflow conditions immediately downgradient from the deposited waste (Inflow station), just prior to

discharge from the site (Outflow station), immediately upstream (Crabtree-Up) and immediately downstream from the landfill on Crabtree Creek (Crabtree-Down). The ranges and averages of parameter values presented for the stormflow are the event mean concentrations (EMC) for storm events. The data are subdivided into baseflow and stormflow concentrations. Samples collected in March 1989 were classified as baseflow. These samples were collected during dry weather following a two week period of very wet weather. These samples did not contain direct surface runoff but are probably representative of an elevated baseflow period.

Water quality monitoring data were collected at the Inflow station to estimate pollutant concentrations in the absence of dilution or other attenuation mechanisms. Solid waste has been deposited over the entire drainage basin for the Inflow monitoring station at sometime in the past. The landfill operators are in the process of placing additional fresh refuse over much of this area. Consequently, water flowing in the discharge channel originated either as surface runoff from waste deposit areas and/or as shallow ground water which originated as landfill leachate. Therefore, there is no dilution of leachate released from the waste other than by precipitation which falls directly on the fill or groundwater flow directly beneath the fill. Because of the short travel time from the waste to the Inflow monitoring station, chemical or biological attenuation of the pollutants should be minimal. Elevated concentrations of the following pollutants were found during baseflow conditions: chloride, fluoride, ammonia nitrogen, chemical oxygen demand (COD), total organic carbon (TOC), dissolved solids, suspended solids, iron, manganese, barium, phenol, diethyl-phthalate, 4-methylphenol, acetone, 2-butanone (2-Buta), 4-methyl-2-pentanone (MIBK) and tetrahydrofuran. The concentration of most of these pollutants was also elevated during stormflow although to a lesser extent. The somewhat lower pollutant concentrations during stormflow were presumably due to dilution by surface runoff. The major exception to this trend was total suspended solids. Extremely high concentrations (4,000 to 55,000 mg/l) of suspended solids were observed during rainfall events. These high concentrations were apparently due to soil erosion. Smaller increases were observed in nitrate, sulfate, TOC, and iron. The observed increase in nitrate and sulfate may have resulted from dissolved solids present in rainfall. The increases in TOC and iron during stormflow were small and are within the range of measurement error.

Pollutant concentrations were also monitored at the Outflow sampling station (Figure 5) during baseflow and stormflow conditions. The Outflow station was monitored to determine pollutant concentrations at the sedimentation basin outfall which discharges to Crabtree Creek. Pollutant concentrations discharging from the landfill were generally lower than at the inflow station. However, it was impossible to determine the relative attenuation of various pollutants as they pass through the pond-wetland system because of the contributions of other land uses (commercial-industrial) which were not monitored. Detectable levels of several identifiable organic chemicals (phenol, diethyl-phthalate, 4-methylphenol, 2-butanone, and 2-hexanone) were observed in the landfill discharge (Table 8). With the exception of

diethylphthalate, most of these compounds are produced during anaerobic degradation and are readily biodegradable. Only 2-Buta was observed at a concentration above 10 µg/l in the landfill discharge. Average pollutant concentrations at the Outflow monitoring station are compared to monitoring data from Crabtree Creek downstream from the landfill for the period February 1980 to October 1984 (USEPA, 1990) in Table 9. The concentrations of COD, TSS, NH₄-N, NO₃-N, total iron and total manganese are higher in the landfill surface water discharge than in the receiving stream (Crabtree Creek).

Crabtree Creek was sampled on two occasions during this study (twice for inorganics, once for organics) immediately upstream and downstream of the landfill. The purpose of these samples was to examine the impact of the landfill on Crabtree Creek. A summary of the monitoring data is provided in Table 10. Within the limits of the available data, the landfill had no measurable effect on the receiving stream. The concentration of some pollutants increased slightly from upstream to downstream sampling stations while others decreased. No SOC's were detected upstream or downstream of the landfill.

Previous Monitoring Data

One water quality data set was available for a sampling date prior to this study for the four ground water and two surface water monitoring stations established by the state of North Carolina (See Figures 1 and 5 for station locations). Table 11 presents the analytical results for water samples collected in October 1987. Monitoring results show elevated levels in October 1987. Monitoring results show elevated levels of several metals at one or more sampling location for this sampling date. We believe, however, that the observed concentrations are an artifact of particulate matter remaining in the unfiltered water sample or are representative of background levels. It is the policy of S&HWMB labs not to filter water samples prior to analysis and the observed metal concentrations may be associated with sediment remaining in the sample. In addition, elevated levels of the same metals were observed in surface water samples collected from the upstream monitoring station which would not be impacted by landfill surface runoff or leachate. Therefore, the observed concentrations probably do not indicate contamination originating from the landfill.

The SOC Caprolactam was identified in ground water samples taken prior to this study as well as during this study. The source of Caprolactam in these water samples is not well understood. Caprolactam is present in plastisizers and textile fibers and, therefore, may be present in landfill waste. However, the concentration of Caprolactam does not appear to be correlated with other organic compounds identified in ground water samples, and it was detected in wells upgradient from waste deposits. Although Caprolactam was identified by gas chromatography-mass spectroscopy analysis at the S&HWMB laboratory, there was no independent confirmation of its presence. Caprolactam is structurally similar to by-products of anaerobic microbial activity, and it is possible that these by-products were misidentified as Caprolactam.

Table 9. Average Concentrations of Inorganic Pollutants in Crabtree Creek.

Parameter (mg/l)	Upstream	Downstream
Chloride	6.0	6.0
CO ₂ D	70.0	97.0
NH ₃ -N	0.78	0.22
NO ₃ -N	0.53	<1.0
Total Iron	3.3	2.2
Total Manganese	0.17	0.15
Total Solids	309.0	216.0

* S&HWMB has a detection limit of 1 mg/l, while NCSU has a limit of 0.1 mg/l. The Upstream value was determined by NCSU and the Downstream by S&HWMB.

Table 10. Comparison of Water Quality in Crabtree Creek and the Landfill Discharge.

Parameter	Landfill - Outflow				Crabtree Creek ¹			
	Mean	Standard Deviation	Maximum Value	Number Samples	Mean Deviation	Standard Value	Maximum Samples	Number
Chemical Oxygen Demand	95.0	79.0	199.0	5	22.0	11.0	59.0	29
Total Suspended Solids	190.0	240.0	600.0	5	47.0	101.0	540.0	51
NH ₄ -N	0.60	0.5	1.5	6	0.07	0.37	0.18	52
NO ₃ -N	0.83	1.1	2.5	4	0.30	0.22	1.4	52
Total Iron	1.5	0.6	2.0	5	1.0	0.28	1.2	2
Total Manganese	1.6	1.6	3.4	3	0.2	0.15	0.3	2

¹ Monitoring data from N. C. Division of Environmental Management Monitoring Station Number 37183, Crabtree Creek at the Neuse River, February 1980 to October 1984.

Table 11. Surface and Ground Water Quality Parameters for State Monitoring Stations (October, 1987).

LOCATION ¹	MW ² 1	MW2	MW3	MW4	UPSTREAM	DOWNSTREAM
Parameter ³						
Arsenic	ND	ND	ND	ND	ND	ND
Cadmium	ND	ND	ND	ND	ND	ND
Mercury	ND	ND	ND	ND	ND	ND
Zinc	0.018	0.068	0.078	0.040	0.023	0.023
Chromium	ND	ND	0.025	0.018	0.013	0.008
Copper	ND	ND	0.45	ND	ND	ND
Lead	ND	0.058	0.028	0.030	0.035	0.040
Silver	ND	ND	ND	ND	ND	ND
Selenium	ND	ND	ND	ND	ND	ND
Manganese	0.015	0.023	0.163	0.370	0.110	0.083
Iron	ND	1.50	12.6	14.3	3.45	3.07
Barium	0.038	0.035	0.156	0.108	0.040	0.055
Sulfate	1.81	3.63	2.55	2.68	1.24	1.94
Chloride	8.28	5.35	3.20	9.71	8.43	7.14
Flouride	ND	ND	ND	ND	ND	ND
pH	6.35	6.21	6.32	6.31	6.32	6.55
TDS ⁴	173	152	141	183	166	153
TOC ⁵	5.53	6.01	4.07	6.45	6.93	6.84
Nitrate	5.08	0.36	0.52	5.06	1.32	1.13
Conductivity, NTU	100	100	50	125	125	100

ND = below detection limit

¹ State Monitoring Locations (See Figures 5 and 6)

² MW = Monitoring Well

³ All units are mg/l unless otherwise specified

⁴ Total Dissolved Solids

⁵ Total Organic Carbon

SPATIAL VARIABILITY OF GROUND WATER CONTAMINANTS

Monitoring wells were installed at several locations around the landfill to examine the spatial variation in ground water contaminants and to determine if attenuation of these contaminants is occurring during transport through the subsurface. In most cases, ground water contaminants would be expected to become less concentrated with both depth and distance from the refuse, due to dilution by uncontaminated water and chemical and biological attenuation.

Effect of Refuse Age on Ground Water Quality

Differences in chemical concentrations in wells E and F were examined to identify changes in water quality between the older and newer portions of the landfill. Well E is situated immediately downgradient from the older portion of the site, which was filled between

about 1972 and 1982. Well F is located immediately downgradient from more recently deposited refuse. [Note that both wells E and F were previously described as "Upgradient" wells (Figure 5 and Table 3) with respect to their hydrologic position in relation to the sedimentation basin. In this section, well locations are described more specifically as downgradient of waste deposits of specific age.] The concentrations of most inorganic and organic pollutants are significantly higher in ground water downgradient from the recently deposited refuse (Table 12).

The differences in ground water quality between wells E and F are not surprising when the decomposition cycle of a landfill is understood. Refuse in a landfill typically proceeds through three general stages of decomposition: the aerobic stage; the anaerobic acid-forming stage and the anaerobic methanogenic stage (Crawford and Smith, 1985). Harmsen (1983) found that leachates from landfills in the acid phase typically had much higher concentrations of heavy metals and degradable organics (fatty acids, volatile amines, and ethanol) than landfills in the methane phase. The high concentrations of heavy metals in acidic leachate were assumed to be due to the low pH and complexation with free volatile fatty acids (Qasim and Burchinal, 1970).

Field observations at the piedmont landfill indicate that the older portion of the fill is generating large amounts of methane while little if any methane is being produced in the newer portion. This implies that the majority of the newer portion is probably in the acid-forming stage. Ground water downgradient from the newer portion of the fill had a low pH (4.6) and contained very high concentrations of metals, conventional pollutants (TOC, COD, TDS) and SOC's (4-methylphenol, toluene, xylenes, acetone and 2-butanone). Pollutant concentrations in well E (located adjacent to older waste deposits) were higher than in background wells. Yet, pollutant concentrations in well E were significantly lower than those in well F which was located adjacent to more recent waste deposits. The lower concentrations

Table 12. Comparison of Inorganic and SOC Concentrations in Ground Water Near Old and New Portions of the Landfill.

Parameter	Older Refuse Well E	Newer Refuse Well F
pH	5.67	4.62
Specific Conductivity ($\mu\text{mhos/cm}$)	294.0	1351.0
Chloride (mg/l)	58.5	90.0
NH ₃ -N (mg/l)	0.95	0.64
NO ₃ -N (mg/l)	0.15	0.20
COD (mg/l)	35.0	893.0
TOC (mg/l)	63.0	386.0
TDS (mg/l)	172.0	1080.0
Iron (mg/l)	14.0	110.0
Manganese (mg/l)	2.0	64.0
Barium (mg/l)	0.25	0.20
Phenol ($\mu\text{g/l}$)	<10.0	32.0
Diethyl Pthalate ($\mu\text{g/l}$)	<10.0	20.0
4-Methylphenol ($\mu\text{g/l}$)	20.0	253.0
Vinyl Chloride ($\mu\text{g/l}$)	69.0	37.0
1,1-Dichloroethane ($\mu\text{g/l}$)	39.0	92.0
Toluene ($\mu\text{g/l}$)	61.0	813.0
Xylenes ($\mu\text{g/l}$)	12.0	125.0
Acetone ($\mu\text{g/l}$)	<10.0	1524.0
2-Butanone (MEK) ($\mu\text{g/l}$)	111.0	1575.0
2-Hexanone ($\mu\text{g/l}$)	<10.0	70.0

in Well E may be due to biodegradation and/or more extensive leaching over time. The pollutant vinyl chloride (Vinyl Chl) was one exception to this general trend. The VC concentration in well E was approximately double the concentration in well F. Vinyl chloride may be formed during the anaerobic dehalogenation of chlorinated solvents such as tetrachloroethylene and trichloroethylene (Vogel and McCarty, 1985). The higher concentration of VC and lower concentration of 1,1-dichloroethane suggests that anaerobic dehalogenation may be occurring in the leachate as the landfill ages.

Horizontal Variations in Ground Water Quality

Monitoring wells have been previously grouped as hydrologically upgradient or downgradient with respect to the sedimentation basin. Water quality in upgradient wells was assumed to be representative of the landfill leachate. Water quality of downgradient wells was assumed to be representative of ground water as it flowed from the site. Average concentrations for selected inorganic and SOC parameters for the upgradient wells (E, F, F-2 and RW-2) and downgradient wells (A, B, C, D, RW-1) are shown in Table 13.

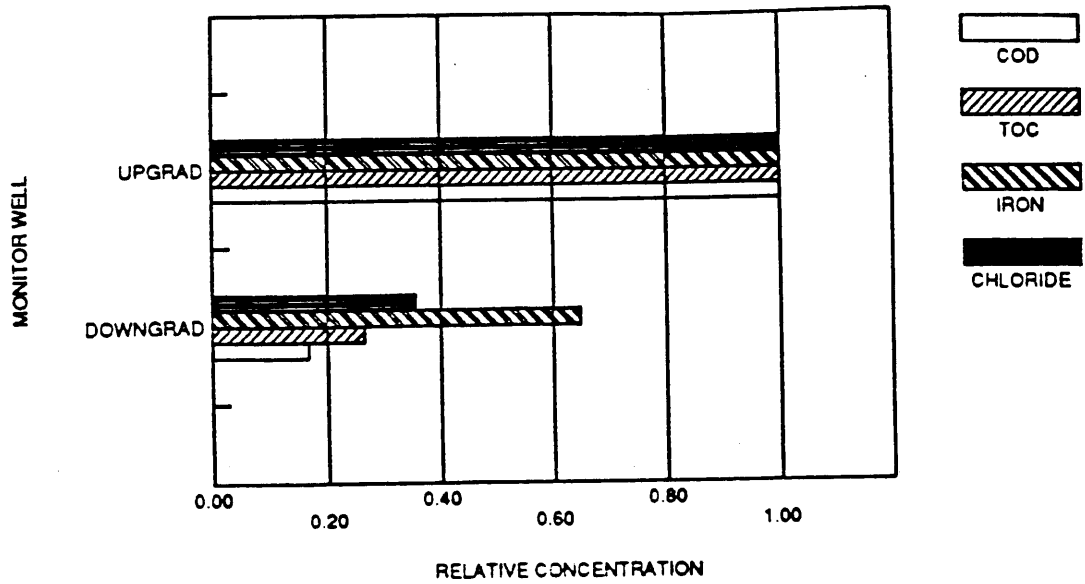
The relative concentration of most inorganic parameters at the downgradient cluster is similar to chloride suggesting that there is little or no attenuation of the inorganic pollutants beyond dilution with uncontaminated water. Ammonia-N is the exception with the average concentration increasing at the downgradient wells. This concentration increase is believed to result from degradation of organic compounds in the refuse.

In order to more clearly demonstrate the importance of chemical and biological attenuation processes relative to dilution, water quality differences between wells may be presented as relative concentrations. Relative concentrations are computed by dividing the observed concentration of a given parameter by the observed concentration at a reference well. The well containing the highest concentrations of contaminants is typically used as the reference well. Figure 7 illustrates the relative concentration changes for selected inorganic and organic pollutants between the upgradient and downgradient wells. The SOC's shown on Figure 7 are 4-Methylphenol (4-Methyl), Vinyl Chloride (Vinyl Chl), 1,1-Dichloroethane (1,1-Dichl) and 2-Butanone (2-But). Chloride is shown with the SOC's for comparison. Chloride is a non-reactive species whose concentration should be affected only by dilution. The upgradient wells are used as the reference point. Figure 7 shows that the average concentration of most parameters decreases by an amount far greater than would be predicted by dilution alone. The average toluene concentration drops from 253 mg/l to less than 5 µg/l; a decrease of approximately 98%. The average chloride concentration decreases by about 45%. These data suggest that some mechanism other than dilution is reducing the concentration of the hazardous organics as they migrate through the subsurface.

Table 13. Water Quality Comparisons Between Upgradient and Downgradient Wells.

Parameter	Average Concentrations		
	Upgradient Wells	Downgradient Wells	Background
Chloride	67.0	29.0	12.0
COD (mg/l)	259.0	62.0	53.0
TOC (mg/l)	142.3	44.1	6.0
NH ₃ -N (mg/l)	0.8	3.2	<0.1
NO ₃ -N (mg/l)	0.13	0.1	2.75
Iron (mg/l)	38.0	32.6	<0.05
Manganese (mg/l)	23.5	7.0	<0.01
Zinc (mg/l)	0.1	0.1	0.19
Toluene (µg/l)	252.5	<5.0	<5.0
Total Xylenes (µg/l)	38.5	<5.0	<5.0
Ethyl Benzene (µg/l)	13.8	<5.0	<5.0
1,1-Dichloroethane (µg/l)	54.3	<5.0	<5.0
Vinyl Chloride (µg/l)	35.0	<5.0	<5.0
Methylene Chloride (µg/l)	10.3	<5.0	<5.0
4-Methylphenol (µg/l)	70.8	<10.0	<10.0
1,2-Dichloroethene (µg/l)	52.3	<5.0	<5.0

INORGANIC PARAMETERS



SOC PARAMETERS

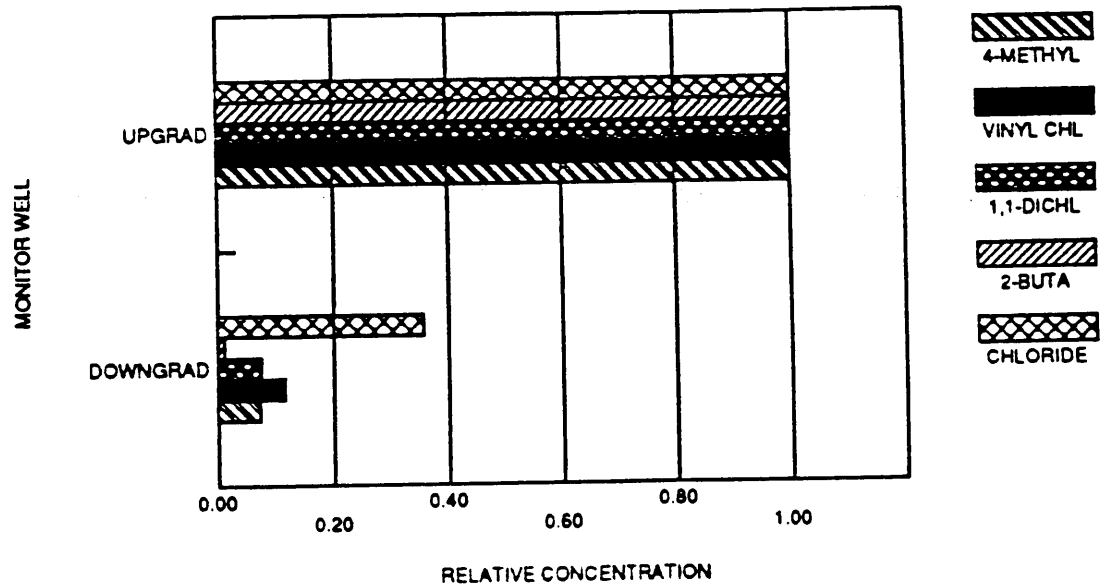


Figure 7. Comparison of Relative Pollutant Concentrations at Upgradient and Downgradient Well Clusters.

Variation in Ground Water Quality along a Flow Path

Previous studies have shown that contaminated ground water may be renovated by natural processes during transport through the subsurface (Borden and Yanoschak, 1989). The rate and extent of natural attenuation appears to be dependent on the site specific hydrogeology and geochemistry. Extensive research has been completed on the transport and attenuation of hazardous pollutants from landfills in glacial (Hughes et al., 1971, Barker et al., 1986) and coastal sedimentary deposits (Baedecker and Apgar, 1984). However, there have been no published reports on leachate transport in a Piedmont hydrogeologic setting. One objective of this study was to determine if attenuation of ground water contaminants was occurring during transport through the subsurface at the Piedmont landfill.

Leachate discharging from the landfill will flow vertically downward through the saprolite (well F) into the transition zone (well F-2). We believe that most ground water in the transition zone discharges at the drainage channel, while a portion of the flow probably enters the bedrock aquifer (well RW-2) and flows downgradient through the fracture system, discharging from the landfill near well RW-1. By comparing the changes in pollutant concentration between the saprolite and bedrock aquifers to changes in chloride concentration (a nonreactive tracer), it is possible to assess the relative attenuation of different leachate contaminants. By this method, the pollutants could be grouped into three general categories: pollutants which are rapidly consumed during transport through the subsurface; pollutants which decline to a limited extent and then remain constant; and pollutants which undergo little or no concentration change (other than dilution). Concentrations of selected pollutants in wells F, F-2, RW-2 and RW-1 are listed in Table 14. Figure 8 shows the variation in organic and inorganic compounds between these wells. During passage from the saprolite to the bedrock aquifer, the chloride and total dissolved solids concentrations decline by approximately half. There is a much more rapid decline in the concentration of 4-methylphenol (>96%), benzene (>85%), toluene (99%), xylene (>96%), acetone (>99%), 2-butanone (>99%), 2-hexanone (>86%) and 4-methyl-2-pentanone (>88%) during passage from the saprolite (well F) through the transition zone (well F-2) and into the underlying bedrock aquifer (well RW-2). The rapid decline in phenols and ketones is not surprising since these compounds are known to be biodegradable under anaerobic conditions (Boyd et al., 1983, Shelton and Tiedje, 1984). Somewhat more surprising is the rapid decline in benzene, toluene and xylene. Aromatic hydrocarbons have commonly been assumed to be resistant to degradation under anaerobic conditions. Recent studies by Grbic-Galic and Vogel (1987) have shown that benzene and toluene can be degraded by a mixed methanogenic consortia. A steady decline in 1,2-dichloroethane was also observed during transport from well F to RW-1 presumably due to anaerobic bio-oxidation (Bouwer and McCarty, 1983).

Sorption processes could account for some of the observed reduction in contaminant concentrations. However, the relative changes in contaminant concentrations cannot be explained by sorption processes alone. Typically, the most

Table 14. Change in Ground Water Quality along a Flow Path

Parameter	Upgradient Saprolite F	Upgradient Transition F-2	Upgradient Bedrock RW-2	Downgradient Bedrock RW-1
Chloride (mg/l)	90.0	59.0	59.0	16.0
COD (mg/l)	893.0	84.0	23.0	13.0
TOC (mg/l)	386.0	60.0	60.0	25.0
TDS (mg/l)	1080.0	484.0	448.0	280.0
Iron (mg/l)	110.0	3.5	3.8	2.5
Manganese (mg/l)	64.1	19.7	8.3	8.0
Nitrate (mg/l)	0.2	0.06	0.1	2.75
4-Methylphenol (µg/l)	253.0	<10.0	<10.0	<10.0
Vinyl Chloride (µg/l)	37.0	26.0	8.0	3.0 ^E
1,2-Dichloroethene (µg/l)	83.0	60.0	26.0	68.0
1,1-kDichloroethane (µg/l)	92.0	60.0	26.0	3.0 ^E
Benzene (µg/l)	13.0	8.0	2.0 ^E	<5.0
Toluene (µg/l)	813.0	126.0	10.0	<5.0
Xylenes (µg/l)	125.0	12.0	<5.0	<5.0
Acetone (µg/l)	1524.0	181.0	<10.0	<10.0
2-Butanone (2-Buta) (µg/l)	1575.0	102.0	<10.0	<10.0
2-Hexanone (µg/l)	70.0	<10.0	<10.0	<10.0
4-Methyl-2-Pentanone (MIBK) (µg/l)	84.0	<10.0	<10.0	<10.0

^E Estimated Value

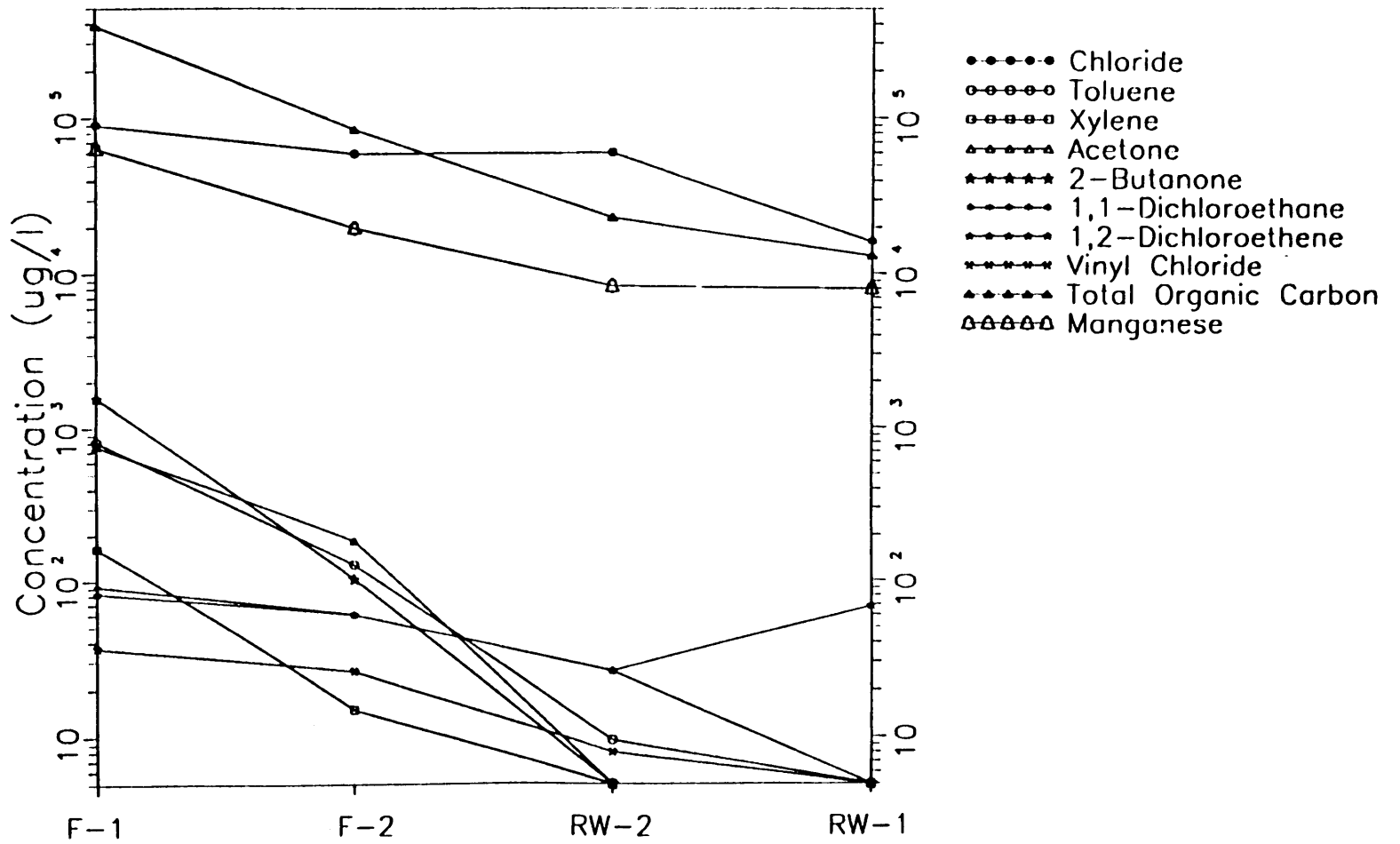


Figure 8. Variation in Ground Water Quality Along a Flow Path.

soluble components are sorbed the least and are more readily transported. However, it was observed that very soluble compounds such as acetone and 1,2-dichloroethene, which would not be strongly sorbed, were removed to the greatest extent. Vinyl chloride, on the other hand, has a very low aqueous solubility (and would be expected to sorb strongly), yet it was one of the more persistent compounds. These observations suggest that biodegradation is an important mechanism for the removal of organic compounds along the flow path.

Chemical oxygen demand (COD) and total organic carbon (TOC) concentrations decline rapidly during transport from the saprolite (well F) to the bedrock aquifer (well RW-2) and then remain essentially constant during transport through the fracture system to well RW-1. A similar pattern has been observed by other investigators studying the change in leachate composition with transport through the subsurface. During the early stages of decomposition, leachate contains high concentrations of COD, TOC and a wide variety of organic compounds including aliphatic and aromatic acids, phenols and terpene compounds (Reinhard et al., 1984). As leachate flows through the subsurface, these compounds are degraded with a resulting drop in COD and TOC. In the later stages of decomposition, the leachate is composed primarily of high molecular weight humic material which is not readily degradable (Harmsen, 1983). Sykes et al. (1982) found that the COD of contaminated ground water dropped rapidly from a maximum of approximately 2,000 mg/l at the edge of the fill to a nondegradable residual of 150 to 175 mg/l approximately 2 m away from the fill.

The distribution of iron and manganese in ground water is similar to COD and TOC. Dissolved iron and manganese concentrations decline by a factor of five during transport from the saprolite to the bedrock aquifer (well RW-2) and then remain essentially constant. This pattern can be explained if these metals are mobilized by low molecular weight organic acids or other complexing agents. Since the organic acids are biodegraded, a fraction of the iron and manganese remains in solution due to complexation with high molecular weight humics and the low Eh and pH of the ground water.

The chlorinated organics vinyl chloride and 1,2-dichloroethene (1,2-DCE) do not appear to be degraded to any significant extent during transport in the subsurface. The chloride concentration decreased at approximately the same rate as the VC and 1,2-DCE concentrations. Vinyl chloride and 1,2-DCE may have originated as degradation products from the reductive dehalogenation of more highly chlorinated solvents such as tetrachloroethylene and trichloroethylene (Vogel and McCarty, 1985). Further degradation of 1,2-DCE to VC is possible under anaerobic conditions present in the aquifer (Vogel and McCarty, 1985). Biological transformation of VC in the landfill aquifer is unlikely, however, since oxygen is required for the further conversion of VC (Vogel et al., 1987). Although VC is quite volatile and tends to sorb readily, the relatively small change in concentration implies that substantial volatilization and/or sorption of VC did not occur.

COMPARATIVE EVALUATION OF SURFACE WATER QUALITY IMPACTS

Water Quality in Urban Areas

Storm water monitoring data from several urban watersheds were found in published literature and used for comparison with the piedmont landfill. This comparison is made to provide a basis for evaluating the significance of pollutant discharges from the landfill relative to other pollutant sources. These urban watersheds are located in Winston-Salem, NC (U.S. Environmental Protection Agency, 1983), Durham, NC (Bryan, 1970), Charlotte, NC (Wu, 1989), Knoxville, TN (U.S. Environmental Protection Agency, 1983), Washington, DC and Toronto, Canada (Mills, 1977), Windsor, Canada (Hartt, 1973) and Tulsa, OK (Avco, 1970). The first four watersheds are in the same physiographic province as the landfill. The other watersheds are included to provide a larger data set for comparison. Table 15 summarizes the general characteristics of these watersheds.

The chemical quality of storm water runoff is often highly variable. The coefficients of variation for COD, nitrate-N, ammonia-N and total suspended solids between storms events at the landfill ranged from 75% to 125%. These high coefficients of variation suggest that the reported concentrations can be used only to provide a very rough estimate of the average storm water concentration and pollutant loads. The actual average pollutant concentrations from the landfill may be several times the reported average. Nonetheless, the available monitoring data can be used to provide order of magnitude estimates of stormwater quality.

Table 16 lists the site locations with ranges and average storm water concentrations. The values are given as Event Mean Concentrations (EMC). The EMC is defined as the total parameter mass discharged divided by the total storm event volume. The ranges and mean values for COD, ammonia-N, nitrate-N, and lead are shown in Figure 9.

Event mean concentrations for most pollutants at the landfill are comparable to the values reported for other urban watersheds. Ammonia-N concentrations appear to be somewhat greater at the landfill. Certain metals, such as lead and copper, appear to occur in greater concentrations in urban storm water. Only total solids, which is not illustrated, is an order of magnitude greater for the landfill.

Pollutant Load Estimation

Mass loading estimates were developed for selected water quality parameters to evaluate the landfill as a pollutant source to downstream surface water bodies (Crabtree Creek and the Neuse River) and to allow comparison with other urban land uses.

Table 15. Physical Characteristics of Urban Watersheds

LOCATION	ABBREVIATIONS	DRAINAGE AREA (ACRES)	LAND USES (%)		
			Commercial	Residential	Undeveloped
CHARLOTTE, NC	CHRLTE	437	0	100	0
DURHAM, NC ¹	DRHM	1069	30	60	10
WINSTON-SALEM, NC	WINSTON-SM, WTN-SM	23	100	0	0
WASHINGTON, DC ²	DC	unknown	80	20	0
KNOXVILLE, TN	KNXVLE	26	100	0	0
TULSA, OK	OK	558	144	46	10
TORONTO, CANADA	TRNTO	119	0	100	0
WINDSOR, CANADA	WNDSR	89	0	100	0
NURP-RESIDENTIAL	N-RES	N/A	0	100	0
NURP-COMMERCIAL	N-COM	N/A	100	0	0

¹ Public/Institutional land use grouped with Commercial/Industrial.

² Assigned categories based on land use descriptions given in report.

Table 16. Variation in Storm Water Quality Between Urban Watersheds.

PARAMETER (mg/l)		PIEDMONT LANDFILL	DURHAM NC	WINSTON-SM NC	CHARLOTTE NC	KNOXVILLE TN	TORONTO CANADA	WINDSOR CANADA	WASHINGTON DC	TULSA OK
CHLORIDE	Range	32-260	3.0-390	ND	ND	ND	8-5855	4-2580	ND	3-46
	Ave	112	12.6	ND	ND	ND	511	110	ND	12
COD	Range	88-620	40-600	25-411	ND	21-140	30-700	ND	29-1514	42-138
	Ave	264	179	120	ND	73	282	ND	335	86
NH ₃ -N	Range	0.07-1.99	ND	ND	ND	ND	0.07-1.30	0-1.80	ND	ND
	Ave	1.77	ND	ND	0.22	ND	0.56	0.09	ND	ND
NO ₃ -N	Range	0.15-3.33	ND	0.33-4.57	ND	0.16-1.60	0.43-1.56	0.05-6.30	ND	ND
	Ave	1.09	ND	1.12	ND	0.66	1.06	1.4	ND	ND
TOT SOLID	Range	4515-60770	274-13800	ND	ND	ND	ND	ND	338-14600	199-2242
	Ave	32643	2730	ND	ND	ND	ND	ND	2166	545
TSS	Range	4022-54967	ND	15-2381	ND	39-360	ND	ND	130-11280	84-2052
	Ave	29495	ND	163	135	123	ND	ND	1697	367
ZINC	Range	0.32-1.20	ND	0.17-2.08	ND	0.13-0.52	ND	ND	ND	ND
	Ave	.072	ND	0.55	0.07	0.32	ND	ND	ND	ND
LEAD	Range	<0.03	0.10-1.85	0.05-1.69	ND	0.07-0.37	0.175-1.39	ND	ND	ND
	Ave	<0.03	0.32	0.38	<0.1	0.16	0.62	ND	ND	ND

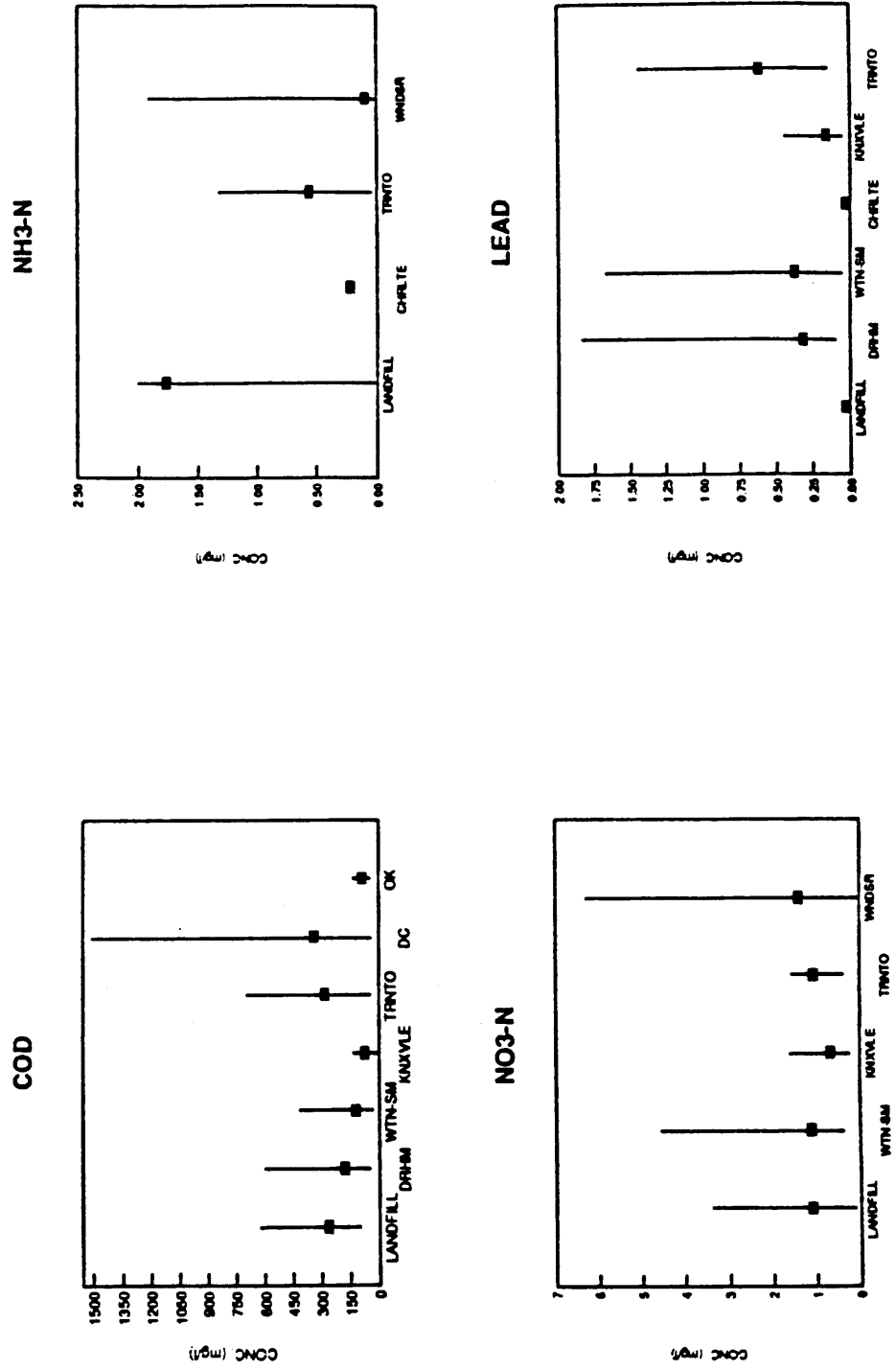


Figure 9. Variation in Stormwater Quality Between Urban Watersheds

Water Budget for a Piedmont Landfill

The mass loading estimates made in this study consider both the surface and subsurface flow of water from the landfill. The Hydrologic Evaluation of Landfill Performance or HELP computer model (Schroeder et al., 1983) was used to approximate the average annual surface and ground water flow volumes. Flow volumes were used in conjunction with average pollutant concentrations to develop average annual mass loading estimates.

The HELP model is a quasi-two-dimensional, deterministic, computer-based water budget model (Schroeder, 1983). HELP routes water through numerical representations of the landfill's physical components. The modified Penman method is used to calculate evapotranspiration. Darcy's Law is used to compute percolation under both saturated and unsaturated conditions. A linearized Boussinesq equation is used to analytically compute lateral drainage. Climatic data, vegetative cover, soil conditions and landfill design features are incorporated into the model. Daily, monthly and yearly water budgets are obtained as output.

Several steps were followed in calibrating the computer model to the landfill. Local climatic data were entered into the HELP data base. A series of layer types, soil textures and cover material were chosen to best represent the landfill in the model. A smaller watershed, located entirely within the waste disposal area, was selected for the calibration. The model was calibrated using data collected during stormflow and baseflow conditions.

Precipitation, temperature and solar radiation data for the Raleigh-Durham Airport from the years 1984 to 1988 were used in the simulation. The average annual precipitation volume for this period was 40.21 inches, which is below the long-term mean of 44.66 inches. Precipitation was also monitored at the landfill. This data was used, as needed, when modeling individual storm events. Rainfall from the first seven months of 1989 was used to calibrate the July 1989 storm. Average monthly solar radiation and temperature values for Raleigh, NC (Robinson and Easterling, 1980) were used.

Five design layers are used to describe the landfill in the HELP model. The number and types of layers were selected on the basis of the landfill's physical characteristics. Figure 10 illustrates the general layout of soil layers available for the HELP model, while Figure 11 shows how these layers are analogous to the landfill itself. The initial soil types for the layers were chosen with regard to the soils identified during subsurface exploration. Table 17 summarizes the engineering properties for each design layer. Engineering property data were derived from soil survey data (Lawthorn, 1970) and from the HELP model manual (Schroeder et al., 1983).

The watershed discharging to the Inflow monitoring point was used to calibrate the HELP model. This watershed is about 40 acres and lies entirely within the refuse disposal area. When calibrating the model, the upper layer parameters were

adjusted first, so that the runoff calculated by the computer was similar to that measured during actual storm events. The lower layer parameters were then adjusted until the annual average drainage volume was similar to that calculated from the baseflow measurements.

Overall, the HELP model provided reliable predictions of storm runoff volumes. The model was initially calibrated to the July 1989 storm (measured runoff = 0.086 inches; model runoff = 0.084 inches) and then compared with two other storms to check the calibration. For the October 1989 storm, the predicted runoff volume was 0.181 inches compared to an observed runoff volume 0.202 inches. For the July 1988 storm, predicted and observed volumes were 1.06 and 0.87 inches, respectively. The close agreement suggests that the model provides a fairly accurate prediction of runoff. Simulated and measured baseflows typically matched within 20%.

Once calibrated, the HELP model reports an average annual water balance for the landfill. This water balance is divided into the different components of the hydrologic cycle. The right side of Figure 11 describes the hydrologic cycle as defined by the HELP model. Surface runoff is equivalent to stormflow. Drainage values were used to estimate baseflow, and percolation from the base of the landfill was assumed equivalent to the ground water discharge. Figure 12 shows the average water budget for the years 1984-1988. During the study period, about 70% of the rainfall was removed by evapotranspiration.

Pollutant Budget for a Piedmont Landfill

The landfill was evaluated as a pollutant source to downstream surface water bodies (Crabtree Creek and the Neuse River) through calculation of average annual mass loads leaving the landfill. The mass load estimates were compared to pollutant loads from the Neuse River Waste Water Treatment Plant and to pollutant loads from other urban watersheds.

The water quality data and the HELP generated average annual flow volumes for the years 1984-1988 have been used to calculate average annual mass loads. A key assumption is that 100% of the ground and surface water leaving the landfill eventually enters Crabtree Creek as either baseflow or stormflow. Mass load estimates were made by multiplying the average concentrations for ground water, stormflow and baseflow by the average annual flow volume from the HELP model. Table 18 shows the predicted average annual mass loading to Crabtree Creek for the inorganic constituents monitored for each flow type. The resulting mass loads were then summed to generate a single total mass load for each water quality parameter. The data are presented as pounds/year and as pounds/acre/year.

The average concentrations used were selected in order to present a worst case scenario for the landfill. The data were chosen from monitoring points where attenuation was minimal. The average pollutant concentrations in ground water were based on the average of Wells E and F, both of which are immediately

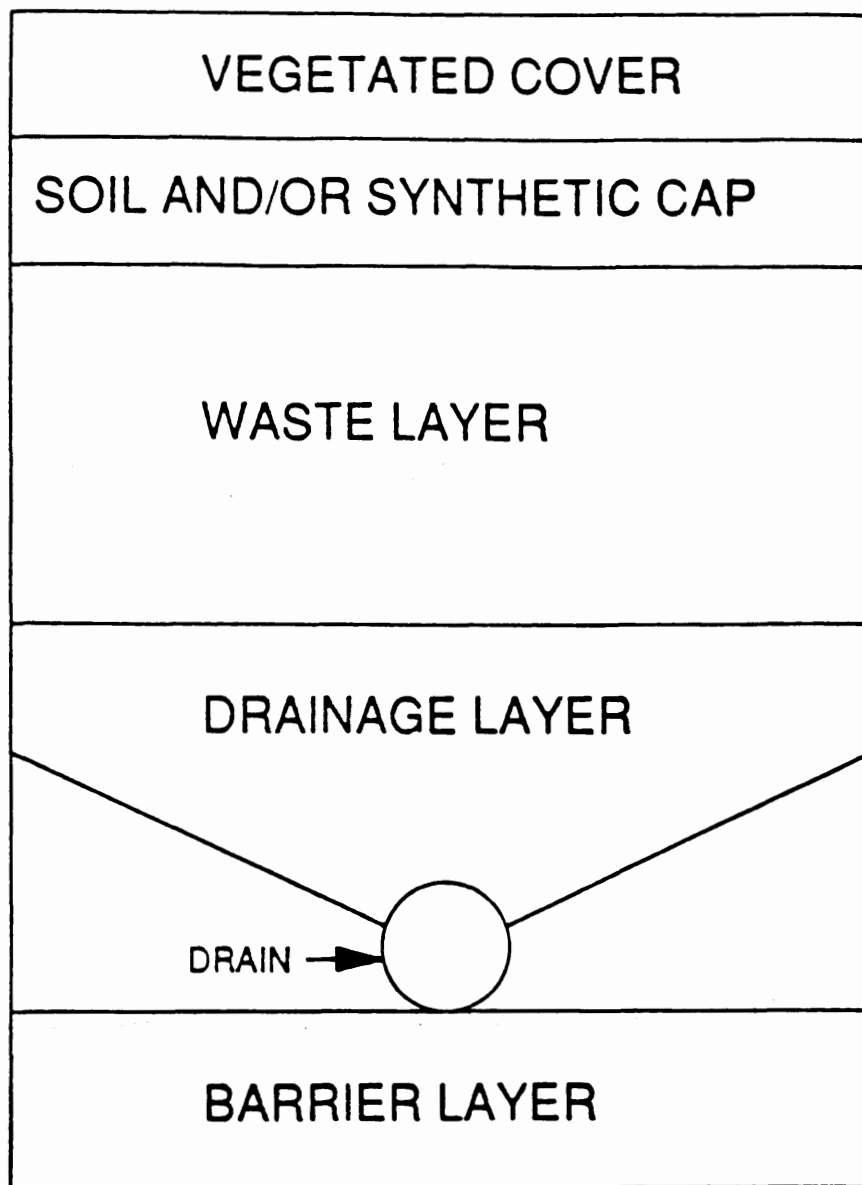


Figure 10. General Layout of HELP Model Soil Layers.

downgradient from the refuse. Baseflow and stormflow concentrations are from the Inflow monitoring point, upstream of the sedimentation pond-wetland.

Pollutant Budget for a Secondary Wastewater Treatment Plant

The Neuse River Wastewater Treatment Plant (WWTP) is located at the eastern edge of Wake County, North Carolina. It is currently a 40 MGD facility which serves much of Wake County, NC. Average flow for the time period studied (1987-1988) is 27 MGD, indicating that the plant was operating at about 68% of total capacity during this time. It is proposed to expand this facility to 60 MGD capacity.

The WWTP is an activated sludge-type treatment plant. The wastewater is pre-chlorinated, bar screened and pH-adjusted before entering the four primary settling tanks. The water is then aerated in three aeration basins and then passed through twelve secondary settling tanks. The effluent is then filtered, chlorinated, pH-adjusted and discharged to the Neuse River. Sludge created during these processes is land applied on cropland adjoining the WWTP. The water quality data suggests that the plant generally meets permit requirements.

Water quality data from the WWTP were used to estimate pollutant loads to the Neuse River for the years 1987 and 1988 (North Carolina Division of Environmental Management, 1987-1988). Average monthly flow volumes were multiplied by the measured concentrations and then summed to find total annual mass loads.

Comparison of Landfill, WWTP and Urban Runoff Pollutant Loads

Sanitary landfills have the potential to severely impact adjoining surface water bodies. In order to judge the relative severity of surface water impacts, pollutant loads from the Piedmont landfill were compared with pollutant loads from a WWTP and urban runoff.

Average annual mass loads for the Raleigh WWTP and the Piedmont landfill are shown in Table 19. Differences in total flow and pollutant loads between the landfill and the WWTP are illustrated in Figure 13. Although comparison of the pollutant loads at the Piedmont landfill and the WWTP provides some useful insight, the two facilities serve different fractions of the service area population. The WWTP receives wastewater from the entire City of Raleigh and some adjoining areas. The landfill examined in this study is one of three sanitary landfills in Wake County. In the past, most refuse generated within the county was disposed of in this landfill. In recent years, a larger portion of the total solid waste stream has been diverted to a new landfill located in northern Wake County. Currently, the landfill examined in this study receives about half of the total solid waste generated in Wake County. Pollutant discharges from the landfill result from both current and past disposal of refuse. Because of this, it is not possible to associate a specific fraction of the local

Table 17. Landfill Engineering Properties - Help Model

LANDFILL LAYER	1	2	3	4	5
HELP NUMBER	13	13	19	6	21
USDA Classification	SCL	SCL	WASTE	LFS	BAR
USCSII Classification	SC	SC	WASTE	SM	BAR
Maximum Infiltration Rate (in/hr)	0.110	0.110	0.230	0.340	0.001
Porosity (dimensionless)	0.453	0.453	0.520	0.401	0.520
Field Capacity (dimensionless)	0.319	0.319	0.320	0.129	0.480
Wilting Point (dimensionless)	0.200	0.200	0.190	0.075	0.400
Hydraulic Conductivity (in/hr)	0.084	0.084	0.283	1.000	1.42E-5
Evaporation Coefficient (mm/d) 0.5	4.7	4.7	3.3	3.3	3.1
Thickness (ft)	0.5	2.0	20.0	NA	NA

54

NA - not applicable

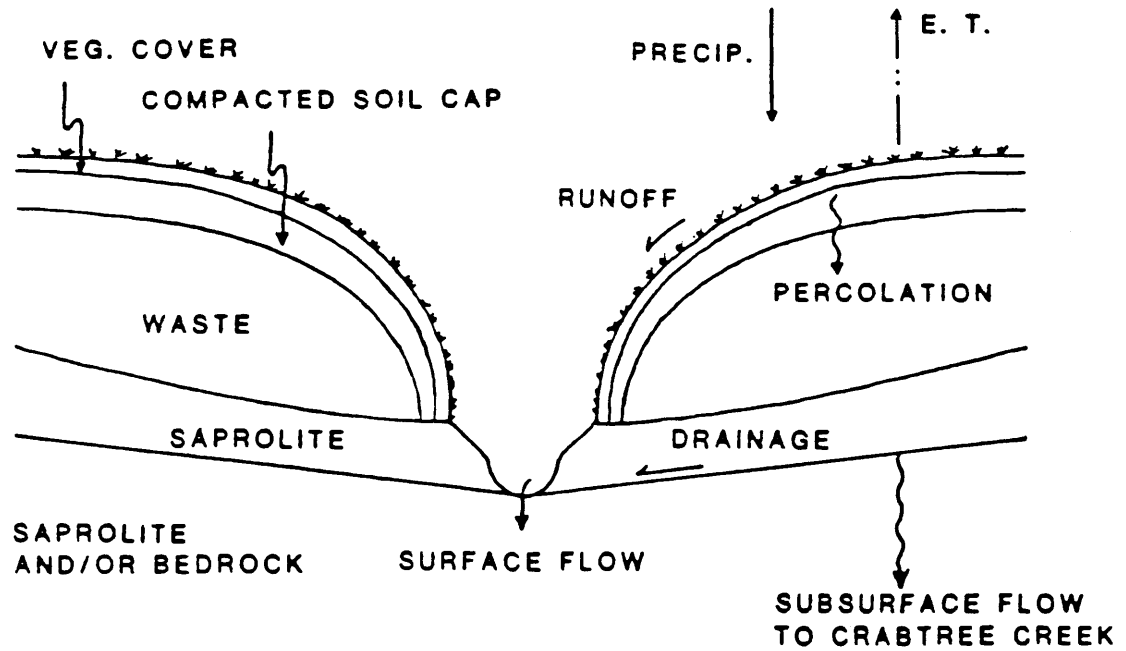


Figure 11. HELP Model Representation of Water Movement in a Landfill.

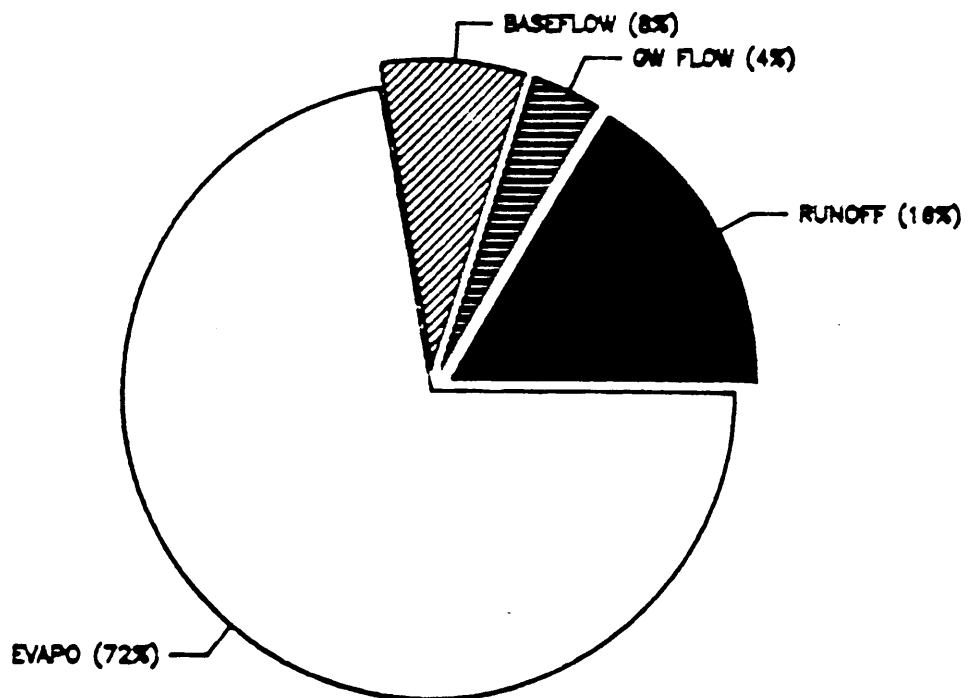


Figure 12. Predicted Average Annual Water Balance for 1984 to 1988.

Table 18. Predicted Average Annual Mass Loads for the Landfill¹

PARAMETER	Baseflow	Stormflow	Ground Water	Total	lbs/ac/yr
Average Annual Flow (cu. ft./yr)	1.40E+06	2.98E+06	6.97E+05	5.08E+06	NA ²
Chloride (lbs)	16436	20815	3230	40481	311
COD (lbs)	44239	49063	20183	113485	873
TOC (lbs)	17923	60957	9765	88645	682
NO ₃ -N (lbs)	9	203	0	219	2
NH ₃ -N (lbs)	1425	335	35	1795	14
Total Iron (lbs)	962	3159	2697	6818	52
Zinc (lbs)	27	223	49	299	2
Manganese (lbs)	640	1385	1436	3461	27
Barium (lbs)	35	74	10	119	1
Lead (lbs)	0	6	1	7	0.1
Total Solids (lbs)	117853	646546	0	6184399	47572
Total Suspended Solids (lbs)	10929	5481505	0	5492434	42249
Total Dissolved Solids (lbs)	86816	585041	27229	699086	5378

¹ based on 130 acres containing refuse

² NA - not applicable

population with the landfill. Overall, the WWTP may serve between one and two times the population that the landfill serves.

Average loads from the WWTP are much greater than from the landfill for most pollutants. The mass of nitrate-N, zinc and lead released from the WWTP is over two orders of magnitude higher than the mass released from the landfill. The higher pollutant loads are due to the much higher discharge from the WWTP. Total flow from the WWTP is roughly 3 orders-of-magnitude greater than that from the landfill. This results in higher pollutant loads even though the concentration of most pollutants is lower in the WWTP discharge. The major exceptions to this trend are iron, suspended solids and total solids. The high iron loading is in part due to the choice of monitoring stations selected for calculating average concentrations. Dissolved iron concentrations in ground water at Wells E and F and in surface water at the inflow station are an order of magnitude higher than at the downstream property boundary.

The pollutant which appears to pose the greatest risk to surface water quality is suspended solids. The total suspended solids loading from the landfill was estimated to be approximately twenty times the load from the WWTP. This high TSS load is believed to be primarily due to soil erosion. A significant fraction of the TSS may be removed in the sedimentation pond-wetland located downstream from the Inflow sampling station. Since the TSS load was calculated using data collected from above the sedimentation pond-wetland, it does not account for the beneficial impact of this erosion control feature.

Pollutant loads from the landfill and several developed urban watersheds are compared in Table 20. The total nitrogen (TN) value for the landfill was calculated by summing ammonia-N and nitrate-N. This assumes that the contribution of organic nitrogen is minor relative to ammonia and nitrate. Pollutant loads from the landfill COD, lead, and total solids are compared to other land uses in Figure 14. The National Urban Runoff Program (NURP) data represent the best estimates based on average median concentrations for the residential (N-RES) and commercial (N-COM) watersheds. The total load of COD, nitrate-N and lead released from the landfill each year is within the range of values observed for other urban land uses.

Suspended solids lost from the landfill during storm events may have the greatest adverse impact on surface water quality. The total suspended solids load from the landfill assuming zero removal in the sedimentation pond-wetland was estimated to be approximately 42,000 lb/ac/yr. This is an order of magnitude higher than for other developed land uses but is in the range of suspended solids loads reported for agricultural areas and construction sites. Chesters et al. (1978) reported TSS loadings from agricultural areas in the Great Lakes region ranging from 5 to 5000 lb/ac/yr. Suspended solids loads from uncontrolled construction sites may range from 10,000 to 50,000 lb/ac/yr (Novotny and Chesters, 1981).

Table 19. Comparison of Landfill and Wastewater Treatment Plant Average Annual Pollutant Loads

ANNUAL FLOW VOLUME (Cubic ft) LANDFILL	LANDFILL 5.08E+06	WWTP 1.29E+09	Ratio of WWTP to
WATER QUALITY PARAMETERS			
COD (lb)	1.13E+05	2.13E+06	18.9
NH ₃ -N (lb)	1.80E+03	2.97E+04	16.5
NO ₃ -N (lb)	2.19E+02	7.34E+05	3351.0
Total Iron (lb)	6.82E+03	1.20E+04	1.8
Lead (lb)	7.00E+00	1.39E+03	198.0
Zinc (lb)	2.99E+02	4.67E+04	156.0
Suspended Solids (lb)	5.49E+06	2.54E+05	0.05
Total Solids (lb)	6.18E+06	2.80E+07	4.5

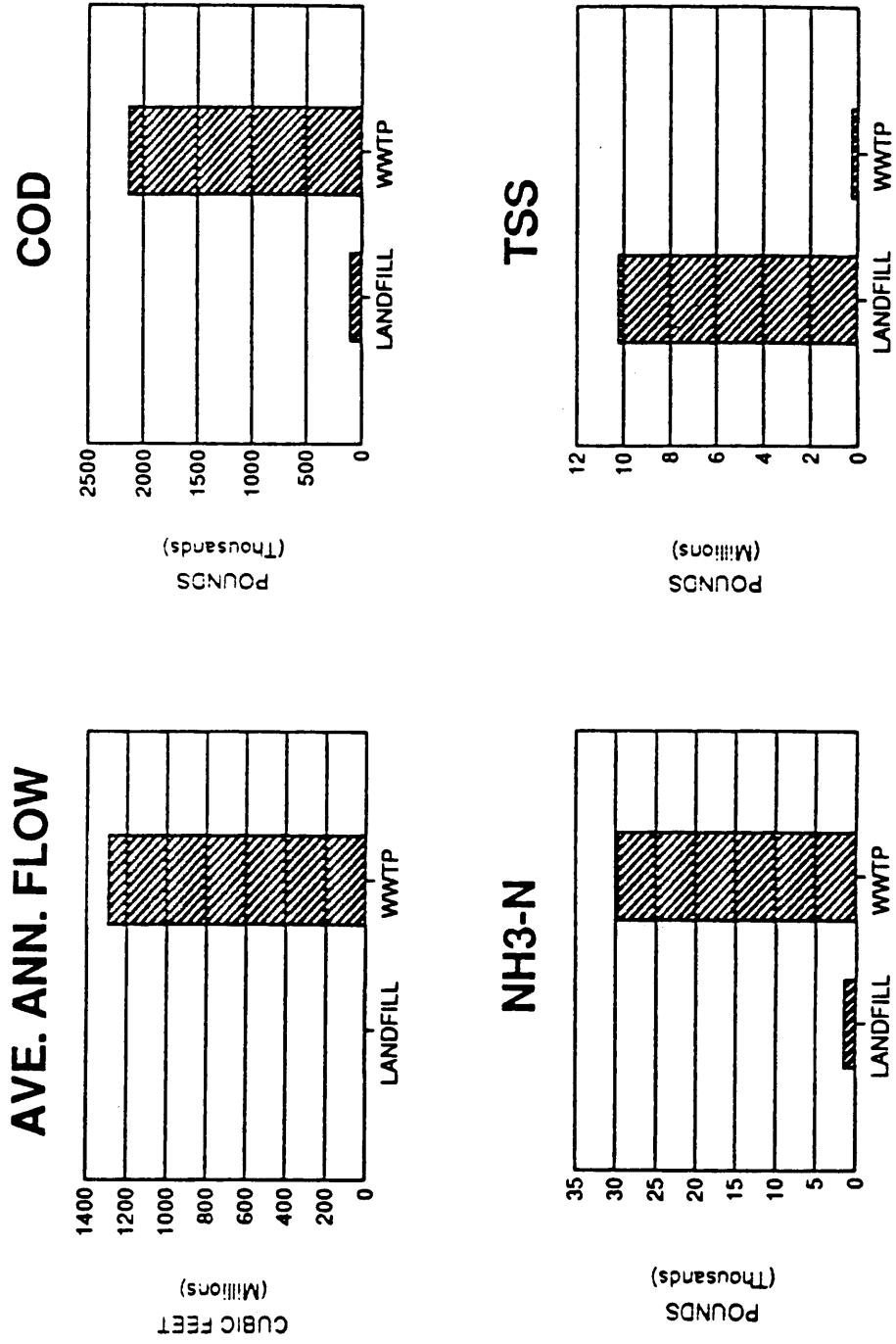


Figure 13. Average Annual Flow and Mass Loading Comparisons between the Landfill and Wastewater Treatment Plant.

Table 20. Mass Load Comparisons between Urban Watersheds and the Landfill

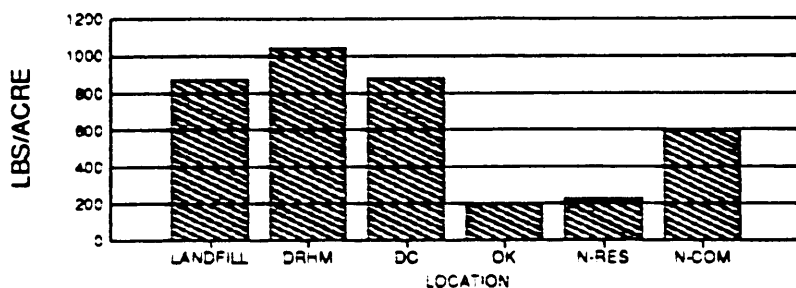
Parameter ¹	Piedmont Landfill	Durham NC	Washington DC	Tulsa OK	NURP Residential	NURP Commercial
Chloride	311	73	ND ²	ND	ND	ND
COD	873	1040	878	201	223	594
NO ₃ -N	2	ND	ND	2	6	ND
Total Nitrogen ³	16	ND	6	ND	ND	ND
Lead	0.1	1.9	ND	ND	0.5	1.3
TSS	42249	ND	4449	ND	491	1303

¹ Units are pounds/acre/year

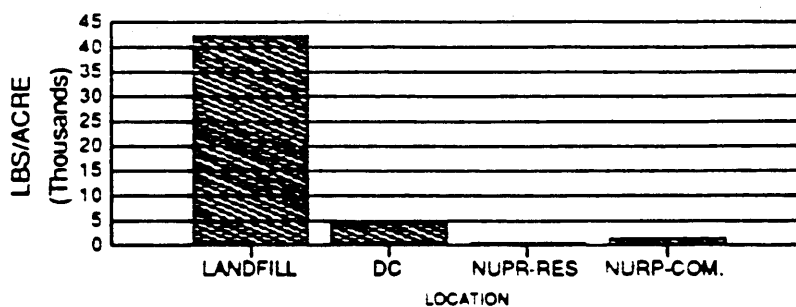
² Not determined

³ TN is total nitrogen; TN = NH₃-N = NO₃-N

CHEMICAL OXYGEN DEMAND



TOTAL SUSPENDED SOLIDS



LEAD

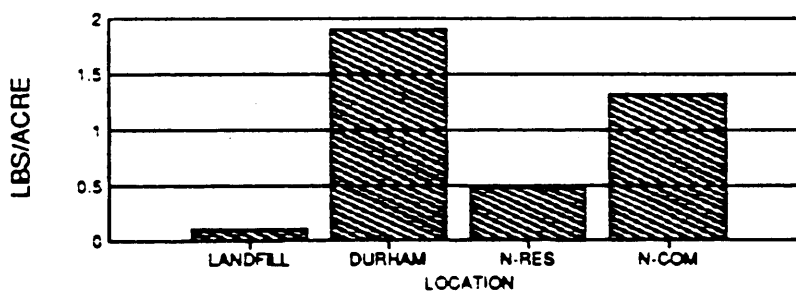


Figure 14. Comparison of Annual Pollutant Loads from the Landfill with other Urban Land Uses (Abbreviations are as identified in text).

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