

NORTH CAROLINA SANITARY LANDFILLS:
LEACHATE GENERATION, MANAGEMENT AND WATER QUALITY IMPACTS

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The most widely used method of solid waste disposal in North Carolina is burial in unlined municipally operated sanitary landfills (SLF's). These landfills typically range in size from 50 to 250 acres and receive from 100 to 500 tons per day of solid waste. Recently adopted state policy will require the installation of impermeable caps, liners and leachate collection and treatment systems at all new SLF's to prevent release of leachate to surface and ground water. These additional measures are expected to approximately double the overall cost of solid waste disposal. The greatest increase in costs will occur during the initial site development stage, increasing the cost of starting up a new sanitary landfill by roughly a factor of ten.

Sanitary landfills generate leachate, a complex mixture of organic and inorganic pollutants. The exact composition of the leachate will depend on site specific environmental conditions including the waste composition, pH, temperature, nutrients, and degree of decomposition. Current research suggests that landfills can be managed to reduce the production of leachate and enhance the attenuation of pollutants during transport through the subsurface.

A statistical analysis of existing surface and ground water monitoring data from 71 municipal sanitary landfills was performed to assess the impact of N.C. SLF's on surface and ground water quality. A total of 322 surface and 411 ground water quality records were analyzed using a computerized data analysis system. Almost all the landfill records included inorganic and heavy metal analyses while approximately half of the records also include organic scans by GC/MS.

Violations of ground water quality standards for organic and/or inorganic pollutants were detected at 53% of the landfill sites examined for which adequate monitoring data exists. The severity of ground water contamination at SLF's appears to be highly variable. At many of the landfills with ground water quality problems, a 50% reduction in pollutant concentration would eliminate all violations. At a few isolated landfills, extremely high concentrations of organic and inorganic pollutants have been detected. At these landfills, pollutant concentrations may exceed the existing water quality standards by a factor of 10,000 or more. These results suggest that (1) most landfills do contaminate ground water; (2) the severity of contamination is highly variable; and (3) the existing monitoring program is reasonably effective at detecting widespread trends. The existing monitoring program is not adequate to reliably detect ground or surface water contamination from individual landfills.

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CONCLUSIONS

1. Leachate generation in sanitary landfills (SLF's) and transport through the subsurface is a complex combination of physical, chemical and biological processes. A wide variety of toxic organic and inorganic pollutants are present in leachate which have the potential to render large volumes of ground and surface water unfit for use as a water supply without additional treatment. The actual extent of ground and surface water contamination due to a sanitary landfill will depend on the waste source, age, operating characteristics, and site hydrogeology. Current research indicates that landfills can be managed to reduce the production of leachate and enhance the attenuation of pollutants during transport through the subsurface. The current state of the art is not adequate to predict the extent of leachate attenuation prior to landfill construction, although significant advances are occurring in our ability to monitor and describe attenuation processes in the field.

2. Engineering measures, including impermeable caps, liners, and leachate collection and treatment systems, are available to control the loss of leachate from SLF's. There has been very little experience with the construction and operation of these systems, and their long-term reliability is unknown.

3. Regulations governing solid waste management in North Carolina are in a transition phase. The past regulatory approach depended heavily on qualitative site-specific evaluations. This approach provided the maximum flexibility in handling unusual conditions but also placed the greatest burden on state regulatory personnel. The state is now moving towards construction of engineered barriers to prevent the offsite migration of leachate.

4. The dominant method of solid waste disposal in North Carolina is burial in unlined municipally owned sanitary landfills. These landfills typically range in size from 50 to 250 acres and receive from 100 to 500 tons per day of solid waste. Installation of impermeable caps, liners and leachate collection and treatment systems at all new SLF's is expected to approximately double the overall cost of solid waste disposal. The greatest increase in costs will occur during the initial site development stage. The cost of starting up a new sanitary landfill is expected to increase by roughly a factor of ten.

5. Water quality analyses for inorganic pollutants and heavy metals were performed on 97% of surface water samples and 69% of ground water samples from sanitary landfills included in this study. Analyses for toxic organic pollutants were performed at least once on ground water samples from 52% of the landfills examined.

6. Violations of ground water quality standards for organic and/or inorganic pollutants were detected at 53% of the landfill sites examined for which adequate monitoring data exists. These results suggest that (1) most landfills do contaminate ground water; and (2) the existing monitoring program is reasonably effective at detecting widespread trends. The current level of monitoring is not adequate to define the nature, extent or severity of contamination at individual sites.

7. Pollutant concentrations in ground water directly impacted by N.C. SLF's are typically orders of magnitude lower than concentrations commonly reported nationally for landfill leachate. This difference suggests that significant pollutant attenuation and/or dilution is occurring during passage through the subsurface.

8. The severity of ground water contamination in N.C. SLF's appears to be highly variable. At most of the landfills with ground water quality problems, a 50% reduction in pollutant concentrations would eliminate all violations. At a few isolated landfills, extremely high concentrations of organic and inorganic pollutants have been detected. At these landfills, pollutant concentrations may exceed the existing water quality standards by a factor of 10,000 or more.

RECOMMENDATIONS

1. Past ground and surface water monitoring has shown that contamination by sanitary landfill leachate is a potential problem. These monitoring programs should be expanded to
 - (1) quantify the severity and extent of existing problems; and
 - (2) identify areas of contaminated ground water before public and private water supplies are affected.
2. The processes which control the generation and natural attenuation of leachate during passage through the subsurface are still poorly understood. If this attenuation capacity is to be used in the combined management of solid waste and water quality, improved methods will be needed to predict the extent of individual attenuation processes based on site-specific conditions. Additional research is needed on the natural attenuation of landfill leachate with particular emphasis on hydrogeologic conditions common to the Coastal Plain, Piedmont and Mountain regions of North Carolina. Emphasis should also be placed on developing methods to maximize the chemical and biological attenuation of leachate within a landfill before it enters surface and ground water supplies.

INTRODUCTION

Environmentally safe and economically efficient solid waste disposal is becoming increasingly difficult. As of 1986, there were 173 permitted sanitary landfills operating in North Carolina (N.C. DHR 1981). This number is expected to grow as our population increases resulting in increased potential for contamination of surface and ground water. At the same time, recent publicity over health risks associated with trace levels of organic 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 ground and surface water resources and solid waste disposal.

The most widely used method for disposal of solid waste is burial in sanitary landfills (SLF's). The dominance of this practice is due to two reasons. Sanitary landfills have been relatively inexpensive to own and operate, and until recently, SLF's were thought to present minimal hazard to the environment. 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 placed in trenches or cells and is covered by soil at the end of each day. Limiting the amount of waste exposed at any given time, greatly reduces problems associated with vermin, odors and burning trash. 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 containing high concentrations of dissolved organics and 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 resources.

Slightly over half of the existing 16,416 active landfills in the United States are owned by local governments. These landfills can be divided into the following waste classes: municipal (9,284), industrial (3,511), demolition debris (2,591), and "other" (1,030) (U.S. EPA 1986a). Recent studies have shown that at some landfills, ground and surface water has become contaminated by waste leachate.

Publicity over the hazards of heavy metals and organic compounds in drinking water has led to public hostility toward landfills. It is becoming increasingly difficult to locate and

permit solid waste facilities. The problem is magnified by increasing population and per capita waste generation rates and the absence of economical disposal alternatives. The purposes of this report are to (1) briefly describe how leachate is generated; (2) describe the technologies currently available for minimizing adverse impacts of SLF leachate; (3) provide an overview of the relevant regulations governing SLF's; (4) summarize the results of a survey of current solid waste disposal practices in North Carolina; and (5) describe the results of a statistical analysis of existing surface and ground water monitoring data.

SANITARY LANDFILL LEACHATE

The primary threat to surface and ground water quality from SLF's is contamination by leachate. Leachate is formed when water comes in contact with deposited waste. Water may be present in waste when it is deposited, or it may be generated as a product of decomposition. Percolating rainfall, runoff from adjacent areas, or ground water may also contribute to leachate formation. As water comes in contact with decomposing refuse, the more soluble components can be transported offsite and pose an environmental hazard. The greatest concern has been over toxic components such as heavy metals and synthetic organic compounds. Table 1 lists observed concentration ranges for commonly measured parameters. The lower values typically correspond to older landfills which are in the methane fermentation stage of anaerobic decomposition, while the higher values are typical of younger landfills in the acid fermentation stage of anaerobic decomposition.

Leachate composition is highly variable and dependent on the refuse composition and processing, landfill age, rate of water application, depth of leached bed and landfill temperature (Lu et al. 1985). Typical landfill leachate contains high concentrations of dissolved ions, ammonia nitrogen, phosphorus, dissolved solids and various metals. Biochemical oxygen demand (BOD) requirements 40 to 85 times higher than most raw domestic sewage have been reported (Qasim and Burchinal 1970). Typically leachate has a mean pH of 5.5. Organic compounds found in leachate include ketones, alkanes, ethers, short chain alcohols and short chain acids among others (Lu et al. 1985).

Chemistry and Microbiology of Leachate Production

The processes taking place within a sanitary landfill which result in leachate formation are highly dependent on site-specific environmental conditions such as moisture content, waste composition, pH, temperature, particle size, degree of mixing, oxygen and nutrient concentrations and stage of decomposition. Lu et al. (1985) identify several processes which influence leachate characteristics, production and transport in a sanitary landfill. These include:

1. Biological transformation of organic putrifiable material, either aerobic or anaerobic in character, and conversion to soluble and gaseous forms.
2. Chemical oxidation or reduction of organic and inorganic material.
3. Dissolution of organic and inorganic materials by water moving through the fill.

Table 1

Observed Ranges of Constituent Concentrations in Leachate from
Municipal Waste Landfills

(in mg/L unless noted)

Constituent	Concentration range	Constituent	range
COD	50-90,000	Hardness (as CaCO_3)	0.1-36,000
BOD	5-75,000	Total Phosphorous	0.1-150
Total Organic carbon (TOC)	50-45,000	Organic Phosphorous	0.4-100
Total solids (TS)	1-75,000	Nitrate nitrogen	0.1-45
TDS	725-55,000	Phosphate (inorganic)	0.4-150
Total Suspended Solids (TSS)	10-45,000	Ammonia Nitrogen ($\text{NH}_3\text{-N}$)	0.1-2,000
Volatile Suspended Solids (VSS)	20-750	Organic Nitrogen	0.1-1,000
Total Volatile Solids (TVS)	90-50,000	Total Kjeldahl Nitrogen	7-1,970
Fixed Solids (FS)	800-50,000	Acidity	2,700-6,000
Alkalinity	0.1-20,350	Turbidity (jackson units)	30-450
Total coliforms (CFU/100 ml)	0-10 ⁵	Cl	30-5,000
Fe	200-5,500	pH (dimensionless)	3.5-8.5
Zn	0.6-220	Ma	20-7,600
Sulfate	25-500	Cu	0.1-9
Ni	0.2-79	Pb	0.001-1.44
Total volatile acids (TVA)	70-27,700	Mg	3-15,600
Mn	0.6-41	K	35-2,300
Fecal coliform (CFU/1,000ml)	0-10 ⁵	Cd	0-0.375
Specific Conduct -ance (mho/cm)	960-16,300	Hg	0-0.16
		Se	0-2.7
		Cr	0.02-18

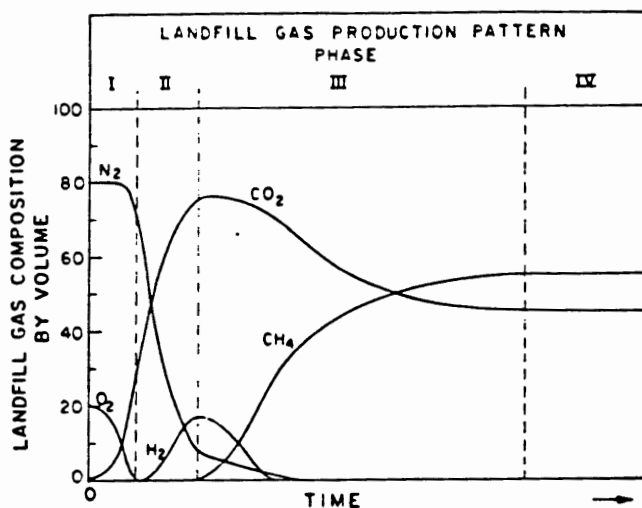
Source: U.S. EPA, 1986

4. Movement of dissolved materials due to concentration gradients and osmosis.

5. Entrainment of particulate matter in flowing water.

Lu et al. (1985) identify three stages of solid waste decomposition: aerobic decay, acid fermentation and methanogenesis while Farquhar and Rovers (1973) separate the methanogenic stage into unsteady and steady state phases. The gas production pattern from a "typical" landfill is shown in Figure 1. The aerobic decomposition phase is generally short in

Figure 1
Sanitary Landfill Gas Production Pattern



Source: Farquhar and Rovers, 1973.

duration due to the high biochemical oxygen demand (BOD), limited oxygen supply and rapid degradation of easily digested food used by the aerobic microbes. Leachate during the initial aerobic stage is characterized by the entrainment of particulate matter, the dissolution of highly soluble salts and generally a low concentration of organic compounds. Leachate formed during this stage is primarily a result of moisture squeezed out of the fill during placement and compaction of refuse.

The initial anaerobic decomposition stage is dominated by acid fermentation. The leachate formed has a low pH and a high volatile acids concentration. Complex organic compounds are converted into soluble organic acids (volatile fatty acids), amino acids and other low molecular weight compounds and gases

(H₂ and CO₂) through anaerobic fermentation. Johansen and Carlson (1976) have identified acetic, propionic and butyric acids as the dominant components at this stage, accounting for approximately 90% of the total organics present. The accompanying decrease in pH will influence chemical processes such as precipitation, dissolution, redox reactions and sorption. Acid conditions generally increase the solubilization of chemical constituents, decrease the sorptive capacity of the refuse and increase ion exchange between the leachate and organic matter. Inorganic ion concentrations increase due to biodegradation of organic matter and dissolution of particulate matter due to the low pH.

The acid fermentation stage is followed by methane fermentation. Free volatile acids are fermented by methanogenic bacteria yielding CH₄ (methane) and CO₂. Harmsen (1983) found a considerable portion of the organic carbon present during this stage (32%) to be in the form of high molecular weight compounds (molecular weight > 1000) while acids, amines and alcohols were not detected. These conditions indicate that the organic compounds present were end products of the degradation process. Lu et. al (1985) note that methane fermentation generally occurs within a year of refuse placement, but this is highly dependent on conditions within the fill. Harmsen (1983) studied leachate from a nine-year-old part of a landfill and found this portion of the fill to still be in the acidification stage. In order for a fill to convert to the methane fermentation phase, the environment must be suitable for methanogenic bacteria. Oxygen must be absent and the pH must be between 6.6 and 7.4. Suitable nutrients must also be available in readily assimilable form. The availability of nutrients is related to the redox potential of the refuse system, which also affects the solubility of metallic solids. Heavy metals in solution are inhibitory to methanogenic bacteria. Conditions favoring high methane production typically are found in older landfills which have already undergone considerable stabilization of the readily available organics.

Heavy Metal Mobility in Leachate

Important processes affecting the movement of trace metals are adsorption and complexation. Under oxidizing conditions, adsorption can effectively remove many of the trace metals in leachate. The metals can be adsorbed onto the lignin-type aromatic compounds common in sanitary landfills, certain metal oxides and clay minerals which may be present in the daily cover layers within the fill. Complexation involves the combining of metal ions with non-metallic compounds such as chloride, ammonia, phosphate, sulfate and many organic compounds by means of coordinate-covalent bonds. Complexation can increase the concentrations of metals in leachate far beyond their normal

solubilities. Sulfides produced during the anaerobic decomposition process can effectively compete with complexing agents, causing many heavy metals to precipitate out of solution. Harmsen (1983) notes that metals are more soluble during the acidification stage of biodegradation due to the lower pH and complexation with volatile fatty acids. The concentration of metals falls drastically once methane fermentation reduces the volatile fatty acid concentration.

Johansen and Carlson (1976) determined heavy metal concentrations in leachate were a function of complex formation with organics, the pH and the carbonate species present. Iron in the ferrous form comprised the bulk of the heavy metals in the leachate samples analyzed. The high iron content in landfill leachates has been attributed to the low pH and strong reducing conditions (Qasim and Burchinal 1970). Iron and manganese concentrations in leachate obtained from experimental waste columns by Raveh and Avnimelech (1979) were very high during the first year of operation but decayed after that. Iron concentrations were found to exceed what could be explained by solubility alone for iron in ionic forms. The authors suggest that this is an indication of large amounts of chelating agents which lead to the dissolution and stabilization in solution of heavy metals. Chelating agents contain nonmetal ions which can combine with metal ions through coordinate bonds to form ring compounds. Houle et al. (1972) found the concentrations of heavy metals in leachate were between 100 to 3000 times the solubility of the metals in pure water.

A comparison of leachates obtained from landfills in the acidification stage and methane fermentation stage was conducted by Harmsen (1983). Heavy metal concentrations were generally much greater for the leachate obtained during acidification due to greater solubility at a lower pH and to complexation with free volatile acids. However, lead also formed very stable complexes with humic acids present in the leachate obtained during methane fermentation. Leachate in the acidification stage was composed primarily of free volatile acids (over 95%), volatile amines (0.8%), and ethanol (0.7%) with high molecular weight compounds comprising only a small fraction. Leachate obtained during the methane fermentation stage had a much lower organic load, but about 32% of the TOC was in the form of compounds with a molecular weight over 1000. Acids, amines and alcohols were not detected in this leachate because the organics present represented end products of degradation processes. COD and BOD values were also much lower for leachate obtained during the methane fermentation stage.

The effect of industrial wastes on leachate composition is heavily dependent on the specific type and nature of the waste. Of most concern are those wastes which contain potentially toxic

components such as heavy metals or organics. Apparently, few studies have been initiated to examine this problem. Houle et al. (1977) studied the effect of landfill leachate on the solubility of heavy metals contained in electroplating waste, inorganic pigment waste and nickel-cadmium battery production waste. Results showed cadmium, copper and nickel concentrations in leachate ranging from 100 to 3000 times higher than the solubilities of these metals in pure water would allow. Chromium, however, showed no elevated concentration. The solubility of a particular metal in the leachate was found to be highly dependent on the type of waste present. These results demonstrate that the harsh environment within a landfill can result in high concentrations of metals and other toxics in the leachate.

Toxic Organic Compounds in Leachate

A wide array of organic compounds has been found in sanitary landfill leachate. Some of these compounds result from the natural decay of organic matter. Of more concern are the manufactured organic compounds, many of which are highly toxic to humans or are carcinogens. These compounds may be introduced into sanitary landfills as industrial or agricultural wastes or may originate as household hazardous wastes present in regular household garbage. Household hazardous wastes include a wide variety of home products including cleaners and disinfectants, and automotive, home maintenance, and lawn and garden products (SCS Engr. 1986). Waste sorting studies have estimated that between 0.0015 and 0.2% by weight of municipal solid waste could be classified as hazardous (LA County Sanit. Dist. 1979, 1984). The decomposition of finished products can also release manufactured organic compounds. Leaching of these compounds can continue long after the landfill appears to have stabilized. Table 2 lists organic compounds detected in municipal solid waste landfill leachate and gives the minimum, maximum and median concentrations observed (U.S. EPA 1986a).

Field studies have shown that sanitary landfills may leach low levels of toxic organic compounds even though the fill may not directly receive industrial wastes (Dunlap et al. 1976; Robertson et al. 1977). The source of the compounds is apparently finished products disposed of in the landfill. Leaching of the organic compounds appears to continue for long periods of time, even after the bulk of the waste has stabilized. The danger in this type of pollution is that the low quantities of pollutants do not noticeably affect the drinking water quality but can accumulate over time within an organism. The types and concentrations of organic compounds found in leachate are also dependent on the stage of decomposition within the landfill. Harmsen (1983) ran organic analyses of leachates from two Dutch

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
Bromomethane	10	170	55
1-Butanol	50	360	220
Carbon tetrachloride	2	398	10
Chlorobenzene	2	237	10
Chloroethane	5	170	7.5
bis (2-Chloroethoxy) 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
Toxaphene	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: EPA, 1986a (units in ppb)

landfills, one which was in the acidification stage of decomposition and the other in the methane fermentation stage. The former stage exhibited much higher organic compound concentrations while concentrations of most of the same organic compounds were below detection level in leachate from the latter stage.

Albaiges et al. (1986) analyzed leachate from the Barcelona, Spain, sanitary landfill for organic compounds. Carboxylic acids represented about 70 to 90% of the total organic extract and included fatty acids, proteins and lignins. The acid extractable fraction had p-cresol as its primary component. The neutral and basic extractable fraction was the least abundant but indicated the presence of many compounds; lindane and PCB's were the major components. Altogether, 17 carboxylic acids, 5 phenols and 28 neutrals and bases were detected.

The leachate from five Danish sanitary landfills was analyzed for organic compounds by Schultz and Kjeldsen (1986). Organic compounds detected in some landfill leachates were absent from others. This was apparently due to differing ages of the landfills and in part to the experimental procedures employed. Aliphatic carboxylic acids and aromatic carboxylic acids such as benzoic acid, methylbenzoic acids, phenylacetic acid and phenylpropionic acid were identified in all five leachates. Alkylbenzenes in particular were recognized as being potential ground water contaminants because of their high solubility in water.

Ground Water Contamination by SLF's

An early study of sanitary landfills in northeastern Illinois was conducted by Hughes, Landon and Farvolden (1971). The majority of the monitoring data was collected at the DuPage County Landfill, although some additional monitoring was performed at four other sites. The DuPage landfill is located on 10-to-21-foot-thick sands overlying a silty clay till. A number of monitoring wells penetrated the landfill but only three wells were installed down gradient from the fill. Monitoring data suggested that biotransformation and dilution were significant in reducing the concentration of contaminants in the groundwater. The chloride concentration decreased by approximately 75% from 30 ft to 650 ft down gradient from the landfill while the 20-day BOD and the COD both decreased by approximately 99%. Total manganese and iron also decreased with distance from the landfill but to a lesser extent.

Sykes et al. (1982) modeled the anaerobic degradation of a landfill leachate plume in a uniform sand aquifer at the Canadian Forces Base in Borden, Ontario. Microbial growth, decay and substrate utilization were simulated using Monod kinetics. The

nonlinear equations were generated using a Galerkin finite element approximation and solved using a Newton Raphson iteration procedure. Model simulations indicated that the majority of the degradable organics would be removed within a few meters of the landfill perimeter. This finding was confirmed by field studies at the site. The COD of the ground water rapidly dropped 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. Some further reduction was noted down gradient, presumably due to dilution by uncontaminated ground water

Reinhard et al. (1984) and Barker et al. (1986) studied the movement of two landfill leachate plumes near Toronto, Canada. The majority of organic compounds detected appeared to result from the natural decomposition of plant material and included aliphatic and aromatic acids, phenols and terpene compounds. Other compounds were of industrial and commercial origin and included chlorinated and non-chlorinated hydrocarbons, nitrogen compounds, alkylphenol polyethoxylates and alkyl phosphates. While a comprehensive list of specific organics was not provided, molecular weight distributions of the organic matter were included. Compounds with molecular weights <2000 were found to make up approximately 70% of the dissolved organic carbon (DOC), indicating the landfill was still in the acidification phase. This fraction was also thought to be the most biodegradable. The TOC concentration decreased more rapidly than chloride with distance from the fill, suggesting some type of biotransformation process.

DeWalle and Chian (1981) and Baedecker and Apgar (1984) studied the geochemistry of the Army Creek landfill near Wilmington, Delaware. Samples were taken from monitoring wells located within the fill and nearby to determine the extent of organic contamination from the landfill. Their results showed a decrease in leachate concentration with time and distance from the landfill. Total organic carbon concentrations decreased from 3700 mg/L to 260 mg/L in four years. Toluene, dichloroethane, butanol and acetone were present in the leachate at the highest concentrations. The observed decrease was in part due to the installation of several recovery wells near the landfill. A regular spacing of wells from approximately 100 m to 950 m from the fill showed a marked decrease in DOC and total volatile organic carbon (VOC) concentration. DOC values fell from 13 mg/l to 1.0 mg/l and total VOC values decreased from 242.2 mg/l to 15.4 mg/l with increasing distance from the landfill. The number of priority pollutants observed in the ground water also decreased from 21 to 11. Of all the volatile constituents, benzene was the most consistently reported, with a concentration of 6.9 µg/l near the landfill and 0.4 µg/l 950 m away.

ENGINEERING METHODS FOR CONTROLLING LEACHATE PRODUCTION AND MOVEMENT

The design and construction of sanitary landfills has advanced a great deal since the early 1960's when most solid waste was disposed of in open dumps. Newer landfills frequently incorporate special features to minimize the potential for ground and surface water contamination. These features can reduce the quantity and control the movement of leachate within a landfill. The most common features are improved grading and construction of an engineered cap to promote runoff and limit leachate production. The best cap designs are layered systems which include a top layer of uncompacted soil capable of supporting vegetation, a relatively impermeable layer of compacted clay and/or a geomembrane, and possibly a porous drainage layer above the impermeable layer to channel water away from the landfill. A healthy vegetative growth on the cap can reduce infiltration by promoting evapotranspiration. Liners function as nearly impermeable barriers which limit the amount of leachate leaving the landfill site. Liners can also be constructed of compacted clays and/or a wide variety of synthetic membranes. The relative impermeability of liners necessitates the inclusion of a leachate collection system in the landfill design to remove the leachate for treatment and disposal and to reduce the hydraulic head acting on the liner. The inaccessibility of the leachate collection system after construction means that considerable effort is required to insure it will not be damaged during landfill construction and operation. Leachate usually must be treated prior to discharge. Leachate recirculation and various traditional biological and physical/chemical processes have been shown to be effective treatment methods. The suitability of a particular process depends on the chemical composition of the leachate which in turn, depends largely on the age of the landfill. A leachate treatment system therefore must be flexible enough to effectively treat a range of expected leachate compositions.

Capping Systems

There are six major classes of capping systems (Thorsen 1982). These are multilayer systems, asphaltics, concrete, synthetics, natural soils and soil admixtures. The multilayer capping systems show the best overall performance and consist of a top layer of non-compacted soil which will support vegetation, a middle layer of gravel or crushed rock to serve as a drain layer and a bottom layer to serve as a relatively impermeable boundary.

The top uncompacted layer should be erosion resistant but should also promote runoff and the establishment of appropriate vegetative growth to increase moisture loss through evapotranspiration. Deep rooted plants should be avoided because

they may pierce the cap and actually be conducive to water flow. Increased thickness of this layer can result in more moisture being removed and less percolation.

The drainage layer should have a relatively high permeability (0.1 cm/s) and should be constructed of uniform sand or gravel. The required thickness of this layer is a function of the annual percolation rate, drain length, permeability and drain slope. The drain slope should be high enough to prevent standing water which could permeate the underlying clay.

Where native soils are not adequate for cover material it may be economical to blend imported materials with the on-site soil. Permeability can be significantly reduced by broadening the grain size distribution. For example, it was found that the addition of 20 to 65% gravel to a sand decreased the permeability to about one seventh of that for the sand alone (Lutton et al. 1979). The addition of sand and silt likewise can reduce the permeability of a soil by altering the grain size distribution. This procedure may have the added benefits of increasing strength, facilitating compaction and spreading, and helping to establish vegetative cover. Clay addition can also greatly reduce permeability but there can often be problems with mixing unless clay is in a processed form, such as sacked bentonite.

Synthetic materials may be added to soils to alter certain properties. Addition of about 8% of dry cement to granular soil can increase strength, and durability and decrease permeability. Likewise, 4 to 8% bitumen may be added to the soil. Both additions, however, can deteriorate over time due to contact with wastes or from settling of solid waste. Thus layers which contain these materials require periodic maintenance. Addition of as little as 1% of Portland cement can help stabilize granular soils, or a mixture of fly ash and lime can be used. Lime added to a cohesive soil can increase its strength. Dispersants in the form of soluble salts may be added to clay to increase its compactibility. Sodium chloride applied at 0.20 to 0.33 lb/ft² or sodium polyphosphate at 0.05 lb/ft² have been recommended. Such treatments, however, can increase susceptibility to erosion, shrinkage and swelling. If a clay exhibits excessive swell potential, lime may be added to reduce this potential.

Few quantitative analyses of cover performance appear in the literature. One study, however, was conducted by Emrich and Beck (1981) for the U.S. EPA involving a 25-acre sanitary landfill in Windham, Connecticut. Leachate from the site was threatening a municipal water supply. Contributing to the problem were the sands and gravels at the site which offered little resistance to leachate flow and provided poor cover material. Remedial actions recommended included regrading the landfill to maximize runoff and minimize infiltration, placement of a 20-mil Polyvinyl Chloride (PVC)

membrane, covering the membrane with approximately 18 inches of soil and establishing protective vegetation. A year after the cover placement, pits were excavated to test the integrity of the membrane. No evidence of puncture was evident but numerous indentations up to 1-1/2 inches deep (resulting from placement of the soil cover) were found. Later, membrane samples were taken from a few test pits located where substantial truck traffic had passed during construction. Several breaks were found, the largest being about one-quarter inch in length and all occurring from the top down. These breaks were judged to have little influence on the effectiveness of the barrier. Extensive ground water monitoring at the site had shown significant water quality improvement since cover placement. The leachate plume was shown to be retreating approximately one year after construction of the cover was completed.

Liner Systems

Liners are often employed at sanitary landfills to act as barriers to the migration of leachate to ground water. Liners may be constructed of natural soils, processed clays, geomembranes, soil additives and combinations of these materials. Where natural soils of sufficiently low permeability exist at a site, the liner must still be carefully constructed to eliminate discontinuities which may exist in the soil and which could act as leachate conduits. Construction of the landfill and subsequent operations should be planned to maintain the integrity of the liner. The relative impermeability of the liner will result in leachate ponding unless an outlet is provided. Without a means of drainage, the water table within the landfill could rise until it meets the land surface producing leachate seeps. This is known as the "bathtub effect". Large water pressures can also lead to failure of the liner.

Synthetic liners are manufactured membranes made from plastic or rubber based materials (see Yanoschak 1988). Polyvinyl chloride is one of the most commonly used liner materials. Unfortunately, the plasticizers often used with this material deteriorate due to biodegradation and weathering especially if exposed to ultraviolet radiation or atmospheric ozone. Therefore, this material must always be covered. Chlorosulfonated polyethylene is also a widely used liner material. It has good puncture resistance; is easy to seam with cement, solvents or heat; and is resistant to weathering, aging, oil and bacteria. Chlorinated polyethylene (CPE) is well suited for waste applications. It can withstand ozone, weathering and ultra-violet rays. It also resists many corrosive chemicals, hydrocarbons, microbiological attack and burning. CPE has a wide temperature range and is spliced by solvent welding. Butyl rubber, although suitable for clean water applications, has poor resistance to hydrocarbons and is difficult to splice. Ethylene-

propylene rubber (EPDM) is resistant to ozone, many chemicals, water adsorption and permeation. Some hydrocarbons have an adverse effect on it and splices are difficult to achieve. Asphaltic compositions can be used as liner materials. These include asphalt concrete, hydraulic asphalt concrete, preformed asphalt panels, sprayed emulsified asphalt, soil asphalt and asphaltic seals. The presence of some types of solvents, however, can have an adverse impact on these materials.

Concerns have been raised over the ability of soil liners to withstand the attacks of leachate over long periods. The Army Corps of Engineers Waterways Experimental Station (1978) examined the effects of percolating landfill leachate on soil properties. No changes in dry density, water content, permeability and percent fines were evident in soil beneath the landfill studied. In general, water extracts showed increased levels of leachable sulfate, chloride, organic carbon, nitrate and trace metals in the soil beneath the landfill. Calcium, iron and zinc were the only metals to show a decrease, probably due to the formation of soluble compounds or leaching by organic acids.

Finno and Schubert (1986) tested clay liner compatibility with various pollutants including landfill leachate. Clay has been found to be affected by high concentrations of cations. Ion exchange of the cations causes the soil fabric to expand, increasing the porosity and fluid conductivity. High multi-valent salt concentrations were found to significantly increase the conductivity of sand-bentonite mixtures. Acids and bases can increase the solubility of certain soil constituents thus increasing fluid conductivity. The effects of organic solvents on clay permeability have been studied, but the results of different studies are contradictory. Some studies show increases while others show decreases in permeability. Finno and Schubert performed laboratory tests on a Wadsworth till used as a liner for a landfill. Exposure of samples to leachate did not adversely affect fluid conductivity, even though the sample had been exposed to the leachate for three years in the field with hydraulic gradients of up to 300:1. Experimental evidence indicates that calcium sulfates precipitated within the pore spaces of the clay and contributed to a small decrease in fluid conductivity. A silt was similarly tested. Results showed that fluid conductivity increased by up to a factor of five before falling to a value below what had been initially observed for pure water. Clay samples from a surface impoundment were next analyzed to see if exposure to acidic industrial wastes ($\text{pH} < 1.0$) had altered their properties. The clay samples had been exposed for up to eight years in the field and were compared to stockpiled clay which had not been exposed. Index properties showed no significant changes, but a small decline in pH was observed. Metals present in the acidic wastes were found to have been removed from solution by chemical precipitation or cation

exchange on the liner. A brownish-red zone was observed in the soil, presumably caused by metal deposition. Inorganic salt concentrations were higher in the exposed samples, but a small decrease in fluid conductivity was observed. Apparently, the tendency of salts to produce a more flocculated and porous soil structure was offset by the deposition of heavy metals which blocked the pore spaces and caused the decrease in permeability.

Leachate Collection Systems

A conventional leachate collection system typically consists of a tile underdrain covered with crushed stone and placed at the bottom of the excavation but on top of the clay liner with a vertical monitoring standpipe placed at each end of the tile drain (Rakoczynski 1982). The standpipes are typically 2-to-3-foot diameter concrete sewer pipes with bell ends placed one on top of the other and are extended as the depth of the fill increases. The bottom segment is perforated and surrounded by crushed stone to serve as a filter. The leachate can be removed from the standpipes by either pump or vacuum truck. Numerous variations of this basic collection system design are possible. Scharch (1981) illustrated the importance of including a relatively permeable drainage layer above the liner to ease leachate flow to the collection system. A liner design utilizing a clay layer 5 feet thick was found to have a collection efficiency that was highly dependent on refuse density. A densely compacted landfill (1,200 lb/CY) was estimated to have an efficiency as low as 17% while a low density fill (800 lb/CY) was found to have an efficiency of 49%. The explanation for these differences involved the ability of water to flow freely through the waste. As water percolates through the waste and comes into contact with the sloped clay/waste interface, the dense refuse inhibits flow to the leachate collection pipe and thus creates a head acting on the liner causing an increase in leakage. A less densely deposited refuse allows better drainage at the interface so less of a head is formed. Various liner designs utilizing a sand blanket placed over the clay liner to allow free drainage above the clay were found to have calculated efficiencies of between 89 and 94%. The inclusion of such a sand blanket makes leakage independent of waste density.

Leachate Treatment

The design of leachate treatment systems is complicated by the variability of leachate composition among landfills and the changes in leachate strength with respect to time at a given site. Yanoshak (1988) provides a detailed summary of leachate treatment options and past operational experience. Normally, several processes will have to be incorporated in a system to treat different categories of constituents and to reduce pollutant levels to an acceptable level for surface discharge.

Leachate recirculation with residual treatment appears to hold promise. Recirculation greatly reduces the time required for waste stabilization by promoting the growth of bacterial populations. Recirculation accompanied by nutrient addition and pH adjustment can be used to promote methane generation for harvesting. Residual organics and inorganics can be removed by carbon adsorption, ion exchange or other processes.

There is a variety of traditional waste treatment processes which potentially could be incorporated into an integrated leachate treatment system. Biological treatment processes such as activated sludge systems, anaerobic filtration and anaerobic lagoons could be used for leachate treatment. However, the release of volatile organics into the atmosphere, the inability of some microorganisms to acclimate themselves to the leachate, and the need to control the pH and nutrient addition complicate matters. Chemical oxidation provides relatively poor removal of organic compounds but could be used to facilitate other processes, principally the removal of heavy metals and certain anionic species. Ion exchange can remove dissolved salts and other inorganics from aqueous solutions, but limitations in capability and pretreatment requirements prevent its widespread use. Air stripping might be used to remove ammonia and volatile organics from leachate.

Age-related variation in sanitary landfill leachate composition was found by Chian and DeWalle (1976) to influence the effectiveness of certain treatment processes. The authors concluded that leachates generated from young fills which are characterized by the presence of free volatile fatty acids were well suited for biological treatment while leachates from old landfills which consists mainly of poorly biodegradable humic and fulvic substances were more amenable to physical/chemical treatment. The reported COD removal for operating treatment systems ranged between 58 and 99% for aerated lagoon or activated sludge systems where the BOD/COD ratio varied from 0.45 to 0.81 for the leachate treated. Anaerobic digesters and anaerobic filters had COD removals between 87 and 99% for leachates whose BOD/COD ratios also ranged between 0.45 to 0.81. The leachates used in these studies came from recently installed landfills and experimental lysimeters and may not be representative of older landfills. Leachate from a 6-year-old landfill with ratios of COD/TOC and BOD/COD of 2.1 and 0.03, respectively, was not effectively treated by biological processes. The authors found the percentage of COD removal in an anaerobic filter decreased with decreasing BOD/COD ratios. Aerobic biological processes were found to be ineffective at reducing the COD of leachate previously treated anaerobically.

Chemical precipitation processes showed poor COD reductions (between 0 and 40% for leachates having BOD/COD ratios of 0.04 to

0.75), but they were effective in removing color and iron from the leachate (Chain and DeWalle 1976). Chemical precipitation was found to be effective primarily on organic matter with molecular weights larger than 50,000. This fraction is typically a small component of the organic matter found in young landfill leachates. Activated carbon treatment was found to be much more effective than chemical precipitation, with COD removals between 34% and 85%. The effectiveness of carbon adsorption is largely dependent on the type of acids present and, therefore, the age of the landfill. Leachates from old, stabilized landfills with low volatile fatty acid fractions have higher adsorption affinities. The use of oxidants to remove organic matter from leachates resulted in COD removals varying between 0 and 48%, with calcium hypochlorite showing better results than chlorine, potassium permanganate and ozone.

Recirculation has been suggested as a means of treating leachate and increasing the rate at which solid waste is stabilized within a landfill. Recirculation consists of collecting and reapplying leachate to the fill through spray systems, surface trenches or perforated pipes. The potential of leachate recirculation was demonstrated by Tittlebaum (1982). Several landfill simulators were set up and all but one, a control, had recirculation. Analysis of volatile acids concentration, BOD, COD and TOC showed that concentrations remained roughly the same when compared with the control during the first 100 days of the test but after that, the simulators with recirculation exhibited sharp decreases in these parameters. The average reduction after 500 days of operation for all four parameters was approximately 95%. These dramatic decreases demonstrate the effectiveness of leachate recycling as a means of biological stabilization. Throughout the test period, leachate was maintained at near neutral pH by the addition of sodium hydroxide as required. Neutralization enhanced stabilization by creating a suitable environment for pH sensitive methanogenic bacteria which break down short chained volatile fatty acids formed during the acidification phase of decomposition.

Robinson and Maris (1985) performed one of the few large-scale demonstrations of leachate recirculation at a landfill in Great Britain. The chemical oxygen demand (COD) of the leachate was reduced from approximately 60,000 to 30,000 mg/l by recirculation. This is a significant reduction, but additional treatment would still be required before discharge.

A problem which arises with leachate recirculation is the need to reduce the ammonia content of the leachate prior to disposal. Younger leachates with relatively high BOD₅ values usually have low ammonia levels due to incorporation in biomass. Older stabilized leachates, however, have ammonia concentrations which cannot be removed by uptake in new cells. In such cases,

treatment will have to provide some degree of nitrification. Knox (1985) examined the abilities of activated sludge and biological filter pilot plants to provide nitrification of ammonia. Sodium bicarbonate (for pH control) and sodium dihydrogen phosphate (as a nutrient) were added to both plants in slurry form. The pH was kept at or above 7.5 during operation. Both plants showed good nitrification capabilities, but fewer problems developed with the biological filter plant which was operationally simpler. The activated sludge plant effluent was of poorer quality with regard to BOD₅ values and suspended solids. The loss of solids in the effluent of this plant was considered a serious problem. Temperature had less effect on ammonia removal for the biological filter plant but the temperature was less variable for the activated sludge plant. By purely volumetric terms, the activated sludge plant was the more efficient system.

Construction and Operation Experience

Operational experience with liners and leachate collection systems, especially those used for municipal solid waste landfills, is limited. The first systems were installed in the early 1970's, and design and operating practices have changed significantly over the years. The costs associated with installing systems are highly variable and most are directly related to the size of the landfill. Polczynski and Romano (1982) estimated the 1981 costs of constructing leachate collection systems in Wisconsin. They determined the installed costs at that time for pipes, including fittings and valves, ranged between \$7.00 and \$9.00 per linear foot with the average cost being \$7.60 per linear foot.

Leachate collection systems can fail for a number of reasons. Even though systems constructed at sanitary landfills have been implemented rather recently, agricultural drainage systems have been in use for a much longer time and provide a means of predicting the performance of leachate collection systems. Bass (1986) has shown that many of the conditions which lead to failure of agricultural drains are just as likely to develop at sanitary landfills. The major causes of failure are clogging due to physical, chemical, biological, and biochemical processes and non-clogging mechanisms such as differential settling and deterioration of collection pipe. Sedimentation is probably the single most common failure mechanism.

Maintenance of leachate collection systems is required to ensure removal of the leachate that builds up over the liner. Bass (1986) notes there has been little experience with maintenance of leachate collection systems because typically they are only serviced when problems occur. The methods used to maintain leachate collection systems are generally the same as

those used for sewers. Maintenance, however, is more complicated because of fewer manholes and the use of plastic pipe at landfills. Rodding and cable machines can be used in runs up to 1000 ft and require access at only one end of the line being serviced. The primary disadvantage of these machines is that dislodged materials may not be removed from the line, and flushing is required afterward.

REGULATION OF SANITARY LANDFILL DESIGN AND OPERATION

Federal Regulation of Solid Waste Disposal

Over the past 20 years there has been a steady increase in the federal government's role in solid waste disposal. Prior to 1976, the federal government was restricted to a support role while the states and ultimately local governments were responsible for management of solid waste. Passage of the Solid Waste Disposal Act of 1965 established grant programs to support the application of improved disposal methods and the development of solid waste disposal plans by states (U.S. EPA 1986b). The Resource Conservation and Recovery Act (RCRA) increased the federal role by requiring the development of minimum technical criteria for solid waste disposal facilities (U.S. EPA 1986b). A grants program was also established to aid states in the development of Solid Waste Management Plans and to identify "open dumps". States were required to adopt these minimum technical criteria or develop more stringent standards of their own. The technical standards cover areas such as ground and surface water pollution, disease transmission, safety and floodplain management among others (U.S. EPA 1986b).

In 1984, the Hazardous and Solid Waste Act (HSWA) was passed which further increased federal involvement in solid waste disposal (U.S. EPA 1986b). HSWA specified that revised criteria must be developed for sanitary landfills that may receive household hazardous waste or small quantity generator hazardous waste. The U.S. EPA has assumed that essentially all municipal SLF's will receive some hazardous waste. The objective of the criteria is to protect human health and the environment. At a minimum, the new criteria require more extensive ground water monitoring, facility siting standards, and provide for corrective action as necessary. In May 1987, U.S. EPA issued draft regulations under HSWA which would require low permeability caps, liners and leachate collection systems for most sites in North Carolina. The draft regulations also include requirements for ground water monitoring to detect leakage, and some form of insurance or bonding to insure that financial resources are available for cleanup if ground water contamination is detected. Final versions of these regulations are expected in mid 1989.

Solid Waste Disposal in the Southeastern United States

Responsibility for solid waste disposal has historically rested at the local level. This will continue, although most states are significantly increasing their requirements for the design and operation of sanitary landfills. Capels (1986) has performed a detailed survey of past sanitary landfill siting and design requirements in six southeastern states: Florida, Georgia,

Maryland, North Carolina, South Carolina, and Virginia. The general results of her survey are summarized below.

In the past, most states have employed a qualitative approach to landfill regulation. In each of the six states surveyed, there are general provisions prohibiting any activity which will result in air or water pollution, a public health hazard or general nuisance. General operating guidelines are also provided to minimize fire hazards, soil erosion, and litter. The only specific requirements have typically been for waste compaction, final and daily cover, and runoff control. Each of the six states employed a multistep permitting process which included both an initial site investigation step and a detailed design and permitting step. The objectives of this multistep procedure have been to reduce costs by eliminating unsuitable sites during the site investigation step and to develop criteria for detailed designs. Some states have also required a public participation step.

The regulatory approach in more urbanized states of the Southeast is moving rapidly towards detailed prescriptive regulations on sanitary landfill siting, design, and operation. Florida has recently enacted and Maryland and Virginia are currently considering extensive regulations governing engineered measures for the management of solid waste in sanitary landfills. Each of these states requires all new landfills to construct an impermeable final cover, impermeable liner and leachate collection and treatment system. Detailed criteria are provided on the design and construction of these facilities. Provisions are also made for gas control systems and extensive surface and ground water monitoring. Florida has placed additional restrictions on areas where landfills can be located, excluding landfills from floodplains, certain geologic formations, areas open to public view, and immediately adjoining water bodies or water supply wells. Maryland and Virginia do not have specific siting criteria although a landfill on a poor site may be required to have more extensive control measures (Capels 1986).

Other southeastern states such as Georgia and South Carolina are moving more slowly in regulating sanitary landfills. General requirements have been developed for ground water monitoring, depth to water table, and buffer areas surrounding landfills. The approach in these states has been to locate landfills where they will have a lower impact. Less attention has been focused on specific engineering controls. In only a few cases have impermeable liners and leachate collection systems been required (Capels 1986).

Regulation of sanitary landfills in North Carolina is in transition from a policy focused primarily on site selection to a policy with a greater emphasis on construction of engineered

barriers to control leachate generation and transport. The past practice was to locate sanitary landfills near rivers whenever possible. The assumption behind this practice was that the regions adjacent to rivers are ground water discharge areas and the probability of major ground water contamination in these locations would be minimal. Also, any leachate that did enter the river from a landfill would be attenuated by natural dilution and biotransformation processes. In September of 1987, the State of North Carolina adopted a policy requiring the construction of impermeable covers, impermeable liners, and leachate collection and treatment systems at all new sanitary landfills (N.C. DHR 1987). Greater emphasis will also be placed on development and implementation of detailed ground water monitoring programs.

Overview of Solid Waste Regulation in North Carolina

The North Carolina Department of Human Resources (DHR) is designated by the state's Solid and Hazardous Waste Management Act as "the single State agency to promote the proper management of solid waste". The Commission for Health Services is authorized by the North Carolina Solid and Hazardous Waste Management Act to adopt and the DHR, to enforce rules for the establishment, location, operation, use, maintenance, and discontinuance of solid waste management sites and facilities. The rules are to be no less stringent than those most recently adopted under the federal act (RCRA) and are to be based on recognized public health practices and procedures, sanitary engineering research and studies, and current technological development in equipment and methods. Therefore, the DHR adopted EPA guidelines in 1982 with the establishment of the North Carolina Solid Waste Management Rules (10 NCAC 10G).

Current sanitary landfill design standards are listed in section of 10 of the North Carolina Administrative Code (10 NCAC 10G). Provisions are made for the protection of groundwater, but specific design standards are not established. Unless specified otherwise in the permit, a minimum distance of 4 feet is required between the bottom elevation of the solid waste and the seasonal high water table. The regulations establish a 50-foot minimum buffer requirement between all property lines and disposal areas; a 500-foot minimum buffer between private dwellings and wells and disposal areas; and a 50-foot minimum buffer between streams and rivers and disposal areas. Within this area "solid waste shall be restricted into the smallest area feasible" and "compacted as densely as practical into cells" (N.C. DHR 1985). "Solid waste shall be covered after each day of operation, with a compacted layer of at least six inches of suitable cover or as specified by the division. Areas which will not have additional wastes placed on them for 12 months or more shall be covered with a minimum of one foot of intermediate cover. After final termination of disposal operations at the site or a major part thereof or upon

revocation of a permit, the area shall be covered with at least two feet of suitable compacted earth" (N.C. DHR 1985). The landfill must also have controls for drainage of surface water, erosion, gas concentrations, and leachates. In most cases the state design standards closely parallel the federal technical criteria. For example, the state regulations establish specific buffer requirements and measures; whereas the Code of Federal Regulations recommends a buffer strip or berm as necessary to screen landfill activities (40 CFR 257).

North Carolina's solid waste regulations require that the proposed site "shall not contravene groundwater standards as established under 15 NCAC 2L, as amended through January 1, 1985." These groundwater standards are part of the state groundwater classification system which is an adaptation of the traditional surface water classification system. N.C. groundwater quality standards are currently (January, 1989) being revised.

The current standards establish a perimeter of compliance for existing and new facilities (N.C. EMC 1985). Exceedances of applicable groundwater quality standards within the perimeter of compliance of permitted waste disposal activities shall not be subject to the penalty provisions under the North Carolina Administrative Code. For new facilities, contravention of groundwater standards is prohibited beyond a distance of 250 feet from the source of contamination or within 50 feet of the property boundaries, whichever is the nearer. For existing facilities, the compliance perimeter is established at the property boundary or the point of contamination, whichever is nearer.

Current groundwater quality regulation is aimed at protecting all underground waters within the saturated zone "to a level of quality at least as high as that required under the standards established in Rule .202." (N.C. EMC 1985). This would allow the unsaturated zone of the subsurface to provide natural treatment and attenuation of pollutants enroute to the waters below.

Prior to construction of a new landfill, operators are required to complete a two part application process. The first step in the process is submission of a site plan application which must include aerial photographs of the site and appropriate scale maps showing nearby structures, roads, utilities, land use and zoning, wells and other potentially affected water resources. A conceptual site design and hydrogeological study is also required. This design must demonstrate that operation of the SLF will not contravene groundwater standards at the perimeter of compliance. Once the site plan is approved, a detailed construction plan must be submitted which includes construction

details, grading and soil erosion control plans. Ground and surface water monitoring requirements are established for each site by DHR and are dependent on the specific geological and hydrological characteristics of the site. Current policy requires that liners and leachate collection systems be constructed at all new landfills, although detailed guidance on the design and construction of these facilities has not yet been formulated.

The current regulations typically lack specific standards for the design of a proposed sanitary landfill. This lack of specific standards allows the permit applicant maximum discretion in the actual design of the facility. All other applicable state regulations must be complied with, however, before a permit will be issued. Permit applications must be supported with sufficient, site-specific data which demonstrate that, due to the site's natural conditions or through design feature, the probability of groundwater contamination will be slight. North Carolina has gone beyond the requirements imposed by most other states and has required an extensive system of buffer zones surrounding landfills. These buffers provide a safety factor in the design of a landfill and may allow for greater natural attenuation of leachate.

SOLID WASTE DISPOSAL IN NORTH CAROLINA

Solid waste disposal in North Carolina is a growing problem. Existing sanitary landfills are rapidly filling up while public concern over the potential for ground water contamination is making siting of new landfills more difficult. Information is needed on current solid waste disposal practices in order to plan for the future. As part of this research, a survey was mailed to each of the existing municipally operated sanitary landfills in North Carolina. Data were collected on site, operational, and environmental characteristics as well as engineering design data. Completed questionnaires were received from approximately 36% of the municipal sanitary landfills operating in North Carolina as of 1986. A copy of the survey form is shown in Appendix A. Values shown in the appendix are the averages of all responses. A complete listing of all survey responses can be obtained from the author.

The results of our survey indicate that the average municipal sanitary landfill in North Carolina covers a total area of approximately 119 acres, receives approximately 325 tons per day of solid waste, was started in the late 1970's, is slightly over half full, and has an expected future life of 13 yrs. While these statistics may accurately describe the average landfill, 16% of currently operating landfills are essentially full, with less than 2 years capacity remaining.

Approximately half of all solid waste originates in towns and cities while half originates in rural areas. The average haul distance to the landfill is 13 miles. Only one landfill was identified which has a liner-leachate collection system which would meet proposed EPA regulations for new SLF's although 30% of the survey responses indicated they had some form of liner, usually 4 to 6 inches of low permeability soil. Thirty % of the responses also indicated some form of leachate collection system, although only one tile drainage system is known to be in operation in the state. Leachate which is collected is typically treated in facultative lagoons prior to surface discharge. National Pollutant Discharge Elimination Systems (NPDES) permits have not typically been required for SLF's. Seventeen percent of all landfills are upstream of a surface water supply. The average distance to a private home is 1800 ft. Eighty-five percent of SLF's have a ground water monitoring system, usually consisting of three to four wells. These wells are typically sampled once or twice a year, while a few SLF's are sampled monthly. The results of this survey indicate that there has been little change in landfill operating practices since the last survey was performed as part of the preparation of the North Carolina State Solid Waste Management Plan in 1981. (N.C. DHR 1981).

The current policy of the N.C. Solid & Hazardous Waste Management Branch is to require all new landfills to install "engineered devices, such as liners and leachate collection systems" or obtain a variance from the N.C. Ground Water Standards (N.C. S&HWMB, 1987). These additional requirements are expected to raise the cost of developing a new landfill from \$5,000 to \$10,000 per acre to about \$125,000 per acre.

In order to evaluate the overall effect of these new requirements on solid waste disposal costs, a simple analysis of the total cost of solid waste collection, transportation and final disposal was performed for an average North Carolina landfill. Landfilling costs were obtained from an analysis prepared by Glebs (1988) on the expected impact these new requirements will have for sanitary landfills. Transportation costs were estimated from a statistical analysis completed by Hudson et al. (1981) assuming an average load of 4 tons/trip and approximately 15% of the day spent hauling. All costs were updated to 1988 using the Engineering News Record Construction Cost Index. Table 3 compares overall solid waste disposal costs for final disposal in an existing unlined landfill or in a new

Table 3

Comparison of Solid Waste Disposal Costs using Existing Unlined Landfill or a New State of the Art Landfill

Cost Item	Existing Unlined Landfill		State of the Art Landfill	
	\$/ton	%	\$/ton	%
1. Predevelopment Costs	0.26	2	1.37	6
2. Construction Costs	0.55	4	6.83	29
3. Operation Costs	3.36	27	6.83	29
4. Closure Costs	0.27	2	0.53	2
5. Collection and Transportation	8.10	65	8.10	34
Total Solid Waste Disposal Cost	12.54	100	23.66	100

Items 1 to 4 taken from Glebs, 1988.

Item 5 estimate from regression equations by Hudson et al., 1981.

state-of-the-art, double-lined landfill with leachate collection and treatment.

It is apparent from this analysis that the cost of solid waste disposal by sanitary landfilling is going to increase significantly. The overall cost of solid waste disposal is expected to increase by roughly 90% from approximately \$12.50 to \$23.50 per ton. The portion of the disposal costs which will increase the most is the cost incurred prior to any solid waste disposal. Predevelopment and construction costs are projected to increase by a factor of ten from \$0.81/ton to \$8.20/ton. The upfront costs will be the most difficult for small counties and municipalities to absorb. A typical small landfill (25 acres) could cost two to three million dollars to develop. This site could last 20 or 30 years but would have to be paid for upfront. Phasing the construction of liners and leachate collection systems is generally the most economical method but may not be feasible for small sites because of economies of scale.

STATISTICAL ANALYSIS OF WATER QUALITY DATA

The objectives of this portion of our research were to determine the nature and extent of current contamination problems at North Carolina municipal sanitary landfills and to identify any common characteristics associated with this contamination. This work involved the statistical analysis of monitoring data previously collected by the N.C. Division of Solid and Hazardous Waste.

Data Collection and Input

Only those active or recently closed (after 1984) sanitary landfills owned and operated by counties or municipalities in North Carolina were examined in this study. To insure a representative distribution of landfills across the state, a minimum of two counties were chosen from each of the 17 multi-county planning regions. Every effort was made to select counties representing the wide range of populations within the state; however, all counties with exceptionally large populations (greater than 200,000 in 1979) were included because of the potential impact large solid waste generators can have on local water resources. The sanitary landfills examined in this study are listed in Table 4. The approximate location of each landfill is shown in Figure 2.

The data used in this report came from organic and inorganic water quality analyses obtained from the Solid & Hazardous Waste Management Branch of the North Carolina Department of Human Resources. The vast majority of the analyses was performed by the state laboratory between 1980 and 1987. Occasional data from as far back as 1975 were included if more recent data were not available. A total of 322 surface and 411 ground water quality analyses from 71 different landfills were used. This sampling represents 57% of the permitted county or municipally owned sanitary landfills in the North Carolina. Of the 71 landfills, 69 (97%) included inorganic surface water quality data, 49 (69%) included inorganic ground water quality data, and 37 (52%) included trace organics analyses of ground water samples. Whenever possible, upstream and downstream surface water samples were collected as grab samples from the largest water bodies adjoining each landfill. In cases when a significant surface water body was not present, samples were collected from small rivulets discharging directly from the landfill property. The ground water samples were obtained from two or four inch PVC monitoring wells located on the landfill property. It was frequently impossible to clearly distinguish between upgradient versus downgradient monitoring wells in the sampling reports. Because of this, all ground water data was analyzed as one group. Also, the reader should be cautioned that the ground water samples obtained from some of the monitoring wells may not be

Figure 2
Sanitary Landfills Examined in this Study

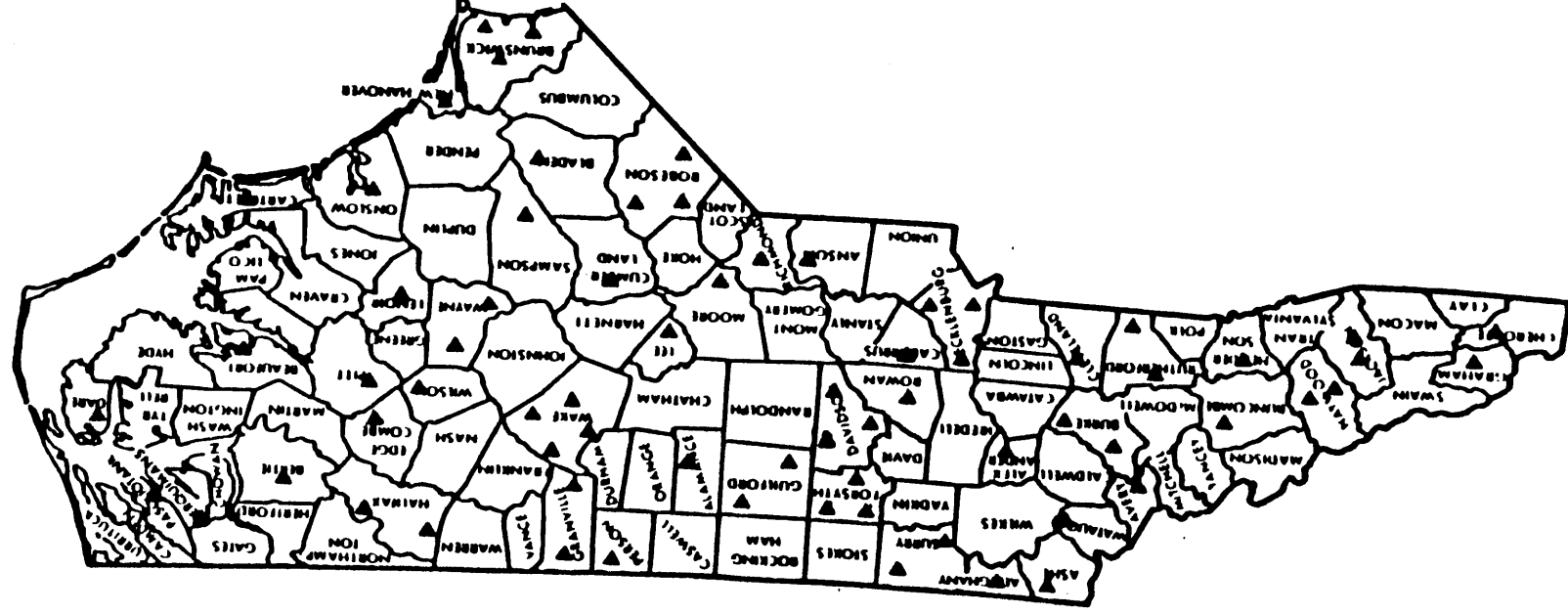


Table 4
Listing of Landfills Included in Study

Facility Name	Permit No.	Region	No. Wells	No. GW Analyses	No. SW
<hr/>					
Alamance Co.	01-01	P	2	2	1
Alexander Co.	02-01	P	1	2	2
Alleghany Co.	03-02	M	3	3	3
Anson Co.	04-01	P	1	2	3
Ashe Co.	05-01	M	2	2	3
Avery Co.	06-01	M	0	0	3
Beaufort Co.	07-02	C	6	24	4
Bertie Co.	08-01	C	2	5	4
Bladen Co.	09-01	C	3	12	5
Southport	10-02	C	4	19	8
Holden Beach	10-03	C	2	13	29
Brunswick Co.	10-07	M	4	12	1
Buncombe Co.	11-01	P	2	4	7
Burke Co. -Morganton	12-01	P	0	0	2
Burke Co. -Rhodhiss	12-02	P	3	5	3
Cabarrus Co.	13-02	P	0	0	3
Carteret Co.	16-02	C	3	10	2
Cherokee Co.	20-01	M	0	0	4
Cleveland Co.	23-01	P	3	16	4
Cumberland Co.	26-01	C	6	11	3
Dare Co.	28-02	C	0	0	3
Davidson Co.	29-02	P	3	12	8
Lexington	29-03	P	0	0	3
Thomasville	29-04	P	2	2	3
Edgecombe Co.	33-01	C	3	9	5
Winston-Salem	34-02	P	0	0	5
- Hanes Rd.					
Winston-Salem	34-03	P	0	0	9
- S. Dale St.					
Kernersville	34-04	P	0	0	4
Graham Co.	38-01	M	3	3	3
Granville Co.	39-01	P	0	0	2
- S.R. 1459					
Granville Co.	39-02	P	4	7	3
- S.R. 1004					
High Point	41-01	P	4	6	2
Greensboro	41-03	P	0	0	6
Halifax Co.-S.R. 1103	42-01	C	4	4	1
Halifax Co.-S.R. 1417	42-04	C	0	0	9
Haywood Co.	44-03	P	0	0	3

Definition of Region: P - Piedmont
C - Coastal
M - Mountain

Table 4 (continued)

Facility Name	Permit No.	Region	No. Wells	No. GW Analyses	No. SW
Canton	44-04	M	2	2	17
Henderson Co.	45-01	M	0	0	2
Western Carolina Univ.	50-01	M	0	0	4
Jackson Co.	50-02	M	0	0	4
Lee Co.	53-01	C	3	15	5
Lenoir Co.	54-03	C	2	14	2
Mecklenburg Co.	60-01	P	2	15	15
- Harris					
Mecklenburg Co.	60-02	P		6	6
- Holbrook					
Mecklenburg Co.	60-03	P		9	7
- York Rd.					
Moore Co.	63-01	C	1	6	3
New Hanover Co.	65-04	C	6	19	1
Onslow Co.	67-05	C	3	9	2
Pasquotank Co.	70-02	C	4	7	2
Perquimans/Chowan Co.	72-01	C	2	2	6
Person Co.	73-01	P	4	7	7
Pitt Co.	74-01	C	5	5	3
Richmond Co.	77-02	C	4	5	3
Robeson Co.-S.R. 1143	78-01	C	1	5	3
Robeson Co.-S.R. 1752	78-02	C	1	7	5
Robeson Co.-U.S. 74	78-03	C	4	9	0
Rowan Co.	80-01	P	4	4	5
Rutherford Co.	81-02	P	0	0	3
- S.R. 1815					
Rutherford Co.-U.S. 74	81-03	P	0	0	2
Samson Co	82-01	C	2	7	11
Surry Co.-S.R. 1815	86-02	P	0	0	9
Surry Co.	86-03	P	0	0	4
Raleigh	92-01	P	3	15	3
Wake Co.-U.S. 64	92-02	P	3	8	3
Wake Co.-N.C. 55	92-03	P	4	16	4
Wake Forest	92-05	P	3	11	3
Wake Co. S.R. 2006	92-09	P	4	4	0
Watauga Co.	95-02	M	0	0	11
Wayne Co.-S.R. 1203	96-01	C	7	9	3
Wayne Co.-S.R. 1320	96-02	C	3	6	2
Wilson Co.	98-01	C	0	0	1

Definition of Region: P - Piedmont
C - Coastal
M - Mountain

representative due to past problems with well construction and maintenance. Field inspections have shown some wells to be dented, cracked, poorly grouted or missing locking well covers. Because of these limitations, the analyses performed in this study have focused on identifying general trends. Use of this data set for identifying site specific violations will probably not be valid.

The analytical detection limits and water quality standards adopted for inorganic parameters examined in this study are listed in Table 5. The U.S. EPA primary and secondary drinking water standards were adopted for use in this study to provide a reference which could be used to evaluate the severity of contamination from sanitary landfills. In most cases these standards are identical to the North Carolina Ground Water Quality Standards for Class GA Waters as of 1987. The 1987 North Carolina ground water standard for synthetic, man-made organics is the analytical detection limit. The reader should be aware that the N.C. Division of Environmental Management recently

Table 5

Water Quality Standards and Detection Limits

Parameter	Standard (mg/l)	Detection Limit
Chloride (Cl)	250.	1.
Iron (Fe)	0.30	0.05
Nitrate (NIT)	10.0	0.05
Sulfate (SUL)	250.	10.
pH	6.5-8.5	NA
Conductivity (CON)	NA ¹	1.mmmhos
Total Dissolved Solids (TDS)	500.	1.
Arsenic (As)	0.05	0.05
Barium (Ba)	1.0	0.1
Cadmium (Cd)	0.010	0.005
Chromium (Cr)	0.05	0.01
Copper (Cu)	1.0	0.05
Fluoride (F)	1.5	0.10
Lead (Pb)	0.05	0.03
Manganese (Mn)	0.05	0.03
Mercury (Hg)	0.002	0.0002
Selenium (Se)	0.01	0.005
Silver (Ag)	0.05	0.05
Zinc (Zn)	5.0	0.05

¹NA = not applicable

proposed revised ground water standards for both organic and inorganic compounds. Chemical analyses reported to be below the detection limit were assumed to be equal to the detection limit in all statistical calculations. The Statistical Analysis System (SAS 1985) was used to analyze this extremely large and cumbersome data set. Analytical test results were first entered into a microcomputer by a spreadsheet program and then uploaded to a mainframe computer for processing.

Inorganic Water Quality Data

A summary of the ground water, upstream surface and downstream surface water quality data was prepared by calculating the mean for each landfill and then finding the mean, standard deviation, and maximum value of the data set containing all of the individual landfill means. These data are summarized in Table 6. This procedure eliminates biases toward landfills with more observations. The error introduced by simply summing observations would have been substantial for some parameter means (e.g. +38% for iron).

Examination of the surface water statistics in Table 6 yields several interesting observations. Many inorganic parameter means showed substantially higher values for downstream samples when compared with the upstream samples, although the downstream concentrations never approached the values reported in the published literature (see U.S. EPA 1986a) for sanitary landfill leachate. The greatest increase in the average concentration was observed for zinc which increased from 0.053 to 0.341 mg/l with a maximum concentration in any individual sample of 3.45 mg/l. This is a significant increase but does not approach the concentrations typically reported for landfill leachate (0.6 to 220 mg/l zinc reported by Pohland and Harper 1986). This difference may be due to dilution by other surface water or attenuation processes within the landfill. Significant increases in the average concentrations of manganese, turbidity and iron were also observed. These increases could be due to landfill leachate and/or soil erosion. Increases were also observed in the downstream means for the conventional dissolved pollutants such as chloride, total organic carbon, conductivity, alkalinity, total dissolved solids and fluoride, although the slight increases in arsenic, chromium, and lead may have a greater environmental impact. It should be noted that the vast majority of the heavy metal analyses were below the analytical detection limit. Consequently, it may not be appropriate to consider percentage increases in these parameters.

The general trends observed in the surface water data were also observed in the ground water quality data. If the upstream surface water quality is viewed as a control, zinc increased by 2600% while significant increases were also observed in

Table 6

Simple Statistics of Water Quality Data¹

Parameter	UPSTREAM SURFACE WATER			DOWNSTREAM SURFACE WATER			GROUNDWATER		
	Mean	Std	Max	Mean	Std	Max	Mean	Std	Max
Cl	12.9	17.1	56.0	44.0	105.3	645.0	22.1	41.8	208.6
Fe	3.068	4.503	16.540	12.493	26.539	122.750	14.318	16.329	91.032
NIT	*0.525	0.532	1.100	0.406	0.544	2.200	0.669	1.006	4.803
SUL	*18.	1.	18.	13.	11.	74.	19.	30.	178.
pH	6.63	1.08	7.60	6.88	0.53	7.80	6.37	0.58	7.47
ALK	38.9	53.4	182.0	96.7	113.2	400.6	42.2	45.4	130.2
CON	*141.3	6.7	146.0	414.1	519.4	2237.5	294.6	350.3	1579.2
TDS	*120.0	11.3	128.0	284.6	381.0	1812.0	203.6	220.6	924.1
TUR	42.2	84.2	300.0	209.9	392.7	1435.0	77.5	94.4	200.5
TOC	*11.0	NA	11.0	32.5	45.0	200.0	31.7	111.7	750.0
As	0.050	0.000	0.050	0.052	0.011	0.125	0.050	0.001	0.050
Ba	*0.13	0.05	0.20	0.15	0.08	0.45	0.20	0.15	0.75
Cd	0.0050	0.0000	0.0050	0.0050	0.0001	0.0060	0.0056	0.0023	0.0205
Cr	0.012	0.004	0.020	0.016	0.036	0.250	0.020	0.019	0.096
Cu	0.050	0.001	0.055	0.051	0.005	0.085	0.098	0.228	1.640
F	0.10	0.01	0.13	0.16	0.28	1.94	0.14	0.06	0.33
Pb	0.030	0.001	0.035	0.033	0.017	0.140	0.078	0.265	1.879
Mn	0.335	0.831	3.200	1.994	4.555	28.200	0.816	1.532	7.071
Hg	*0.00021	0.00003	0.00025	0.00020	0.00001	0.00028	0.00022	0.00005	0.00045
Se	*0.0050	0.0000	0.0050	0.0050	0.0000	0.0050	0.0050	0.0000	0.0051
Ag	*0.050	0.000	0.050	0.050	0.000	0.050	0.050	0.000	0.050
Zn	0.053	0.011	0.090	0.341	0.738	3.450	1.438	5.353	32.628

Notes

¹ All units in mg/l except pH (pH units), TUR (Jackson Turbidity Units, JTU) and CON (mmhos)

*these statistics based on data from less than ten landfills

NA = not applicable, less than two landfills had data for this parameter

Std = Standard deviation

Max = Maximum average concentration for an individual landfill

conductivity, total dissolved solids, and total organic carbon. The heavy metals barium, cadmium, chromium and lead also showed significant increases. Field checks performed at several landfills indicate that many of the high iron and manganese concentrations reported are due to collection of suspended clay with the ground water sample. The high concentrations of zinc, cadmium, chromium and lead in both ground and surface water could be due to solubilization under acidic anaerobic conditions or to complex formation with the organics present in leachate. Francis and Dodge (1988) found that Cd, Cu, Pb, and Zn could be solubilized through the production of microbial metabolites under reducing conditions similar to the environment of a sanitary landfill during the acid fermentation stage.

Correlation Analysis

Statistical analyses were performed on the upstream and downstream surface water data and the ground water data to determine if individual water quality parameters were highly correlated. For each case, the analysis was performed on the total data set consisting of information from all of the landfills to detect widespread correlation of parameters as well as by landfill to indicate if certain parameters might correlate at isolated landfills. It was necessary to treat those observations which were at or below the detection limit as missing values to avoid false correlation, especially among the heavy metals, caused by the grouping of large numbers of points at the detection limit. The correlations presented in this section therefore are based only on observations for which both parameters were present at levels above the detection limits. Only correlations which had a correlation coefficient (r) greater than 0.80 and were significant at the 99% level (based on F statistic) are reported. The calculated correlation coefficient is the non-weighted Pearson product-moment correlation (SAS 1982). A minimum of three observations containing both parameters were required in order for the correlation to be calculable.

Table 7 summarizes the results of the surface water and ground water analyses which were run on the total data set to indicate widespread correlation. No correlation of a significant magnitude was found to exist for the upstream surface water data. Once again, the results from this category of sample were probably adversely affected by the lack of data. Only 13 parameter pairs out of a maximum possible of 171 tested had more than three observations containing both parameters above the detection limits. The downstream surface water data had many more observations and exhibited significant correlation between six parameter pairs. Most of these correlations are not surprising. Chloride is shown to have considerable influence on conductivity and total dissolved solids. Conductivity and iron

Table 7

Summary of Correlation Analysis for Total Data Set

DOWNSTREAM SURFACE WATER

Parameters	r	n
Cl:CON	0.932	49
Cl:TDS	0.932	49
Fe:NIT	0.826	32
Fe:TDS	0.812	61
Fe:Ba	0.835	19
CON:TDS	0.916	61

GROUNDWATER

Parameters	r	n
Cl:CON	0.861	345
Cl:TDS	0.808	342
SUL:Cd	0.934	9
CON:TDS	0.894	381
Cr:Cu	0.865	25
Cr:Pb	0.893	35

r = Pearson product-moment correlation coefficient

n = number of observations

are also shown to be correlated with total dissolved solids. More surprising are the relationships exhibited between iron and nitrate and iron and barium. Note that only 19 observations contained both iron and barium above the detection limit; therefore, the presence of barium might indicate the presence of iron but the converse would not hold true. The ground water data yielded six significant correlations. As for the downstream surface water, correlations between chloride and conductivity, chloride and total dissolved solids, and conductivity and total dissolved solids existed. More unusual correlations existed between sulfate and cadmium, chromium and copper, and chromium and lead. It is interesting to note that unlike the surface water data set, no significant correlations existed for iron in the ground water.

Table 8 contains a summary of the correlation analyses which were run on individual landfills for both surface and ground

Table 8

Summary of Correlation Analysis for Individual Landfills

DOWNSTREAM SURFACE WATER

Parameters	# Observations	# Correlations	% Correlations
CON:TDS	7	2	29
Fe:pH	10	2	20
Fe:CON	8	2	25
Fe:TDS	7	1	14
Cl:CON	4	1	25
Fe:Mn	5	2	40
pH:Mn	4	1	25

GROUNDWATER

Parameters	# Observations	# Correlations	% Correlations
Cl:CON	41	8	20
Cl:Fe	39	2	5
Cl:TDS	40	8	20
Cl:NIT	20	2	10
Fe:SUL	12	2	17
Fe:Mn	35	4	11
Fe:Zn	24	2	8
Fe:CON	40	3	8
NIT:Mn	16	2	13
pH:CON	43	5	12
pH:TDS	42	4	10
CON:TDS	42	15	36
Cr:Zn	8	2	25
Cu:Pb	2	2	100
F:Mn	14	2	14

water data. There was an inadequate amount of data to run correlations for the upstream surface water data so only the downstream results are presented. As for the total downstream data set (Table 7), correlations exist for some landfills between conductivity and total dissolved solids, iron and total dissolved solids, and chloride and conductivity but these three correlations were present in less than 30% of the landfills for which there was adequate downstream data. Iron and manganese were found to correlate negatively with pH for downstream surface water in 20.0 and 25.0 percent, respectively, of the sanitary landfills. This negative correlation is expected since metals generally exhibit a higher solubility in acidic (lower pH) environments. What is somewhat surprising is that no significant correlations between metals and pH were found to exist for either

the total surface or total ground water data sets and that less than 30% of the individual landfills showed such correlations for the downstream surface water data set. The most significant correlation for surface water was between iron and manganese which appeared at 40% of the landfills. This correlation, however, was not present for either of the total data set analyses.

The second part of Table 8 lists correlations found for the ground water data from individual landfills. Only correlations which were found to exist at more than one landfill are listed. Twenty six relationships found to exist at only one landfill, most involving heavy metals, were not included because of apparent insignificance. Several of the most frequently occurring ground water correlations at individual landfills were also observed for the analyses of the total data sets. The exceptions were the relationships between chromium and zinc, and copper and lead. For the two times that copper and lead appeared at a landfill above the detection limits, they correlated well. Apparent correlations also existed between pH and conductivity, and pH and total dissolved solids for approximately 10% of the landfills but further examination of the SAS output showed that some of the correlations were positive while others were negative, therefore rendering these relationships meaningless.

The correlation analyses, in summary, did not appear to yield any unexpected parameter dependencies. The most notable relationships were between chloride and conductivity, chloride and total dissolved solids, and conductivity and total dissolved solids. Other relationships exist but appear to be highly dependent on specific conditions at individual sanitary landfills.

Ground Water Quality Violations - Inorganics

The previous analyses have shown that sanitary landfills are having a measurable impact on surface and ground water quality. Unfortunately, by averaging over the entire state it is impossible to identify critical areas or determine the magnitude of ground water contamination problems at individual landfills. One objective of our study was to identify monitoring wells where water quality standards were violated. This data could then be used to assess the relative importance of ground water contamination from sanitary landfills in North Carolina.

Ground water quality violations were assumed to occur when the ratio of the observed concentration to the water quality standard was greater than 1.0. A Student's t test was performed on the log transform of this ratio to determine if the mean value at a given sampling point was statistically greater than the standard at a 90% confidence level. The log transform was used

because a preliminary analysis of the sample skewness indicated that the sample was not normally distributed at the 90% level (Snedecor and Cochran 1967). Since a minimum of two readings for each parameter per sampling location are required to calculate the t statistic, those parameters with fewer than two were not analyzed. The reader should also realize that the calculation of the t statistic was based on a maximum of only six readings.

Table 9 summarizes the results of our analysis of the ground water monitoring data. Iron, manganese and pH were the parameters which most commonly violated standards. The high frequency of violations for iron and manganese is probably due to the collection of suspended clay particles with the samples and is not necessarily indicative of contamination from the landfills. Elevated levels of chloride, sulfate, total dissolved

Table 9

Summary of Monitoring Wells Exceeding Water Quality Standards¹

Parameter	<u>Wells Exceeding</u> <u>Standard</u>		<u>Wells Statistically</u> <u>Exceeding Standard</u> ²		<u>Landfills Exceeding</u> <u>Standard</u> ³	
	<u>No.</u>	<u>%</u>	<u>No.</u>	<u>%</u>	<u>No.</u>	<u>%</u>
Cl	2	2	0	0	2	6
Fe	85	94	52	57	32	97
Mn	58	64	34	37	27	82
NIT	0	0	0	0	0	0
SUL	2	2	0	0	2	6
pH	50	57	28	32	28	85
TDS	9	11	5	6	5	15
As	2	2	0	0	2	6
Ba	0	0	0	0	0	0
Cd	4	5	0	0	2	6
Cr	8	9	0	0	6	18
Cu	0	0	0	0	0	0
F	0	0	0	0	0	0
Pb	10	11	2	2	6	18
Hg	0	0	0	0	0	0
Se	0	0	0	0	0	0
Ag	0	0	0	0	0	0
Zn	5	6	1	1	2	6

¹ Only monitoring wells with two or more observations were considered.

² Wells statistically greater than the water quality standard at the 90% confidence level.

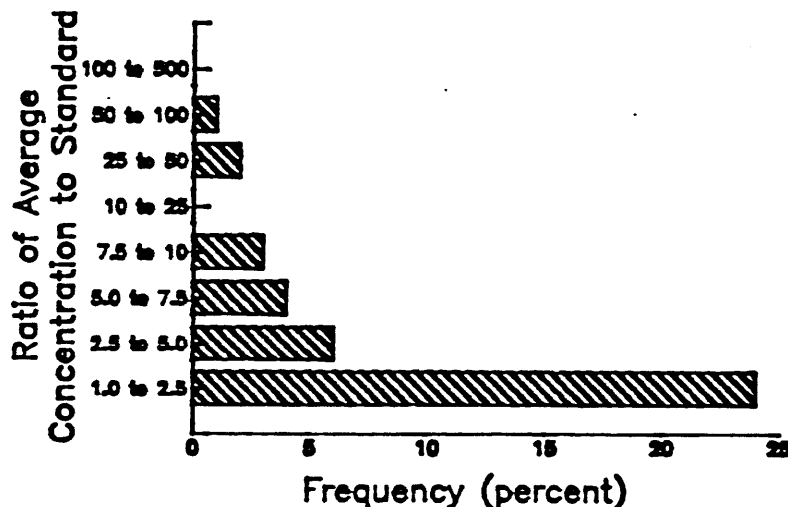
³ Landfill is assumed to be in violation when the average concentration on any monitoring well exceeds standards.

solids, arsenic, cadmium, chromium, lead and zinc were detected in ground water near a variety of landfills. For the heavy metals, only lead and zinc are shown to have at least a 90% probability of having a population mean greater than the standard at certain monitoring wells. The low frequency with which heavy metals were shown to statistically violate standards was due to two factors: (1) the small number of samples; and (2) the high variability in measured concentrations between samples.

Overall, approximately 30% of the landfills examined with adequate ground water data had heavy metal concentrations (excluding iron and manganese) exceeding the water quality standard at one or more monitoring wells. In order to judge the severity of contamination at these landfills, average concentrations for individual pollutants at monitoring wells which violated the standard were averaged and then divided by the water quality standard. The frequency distribution for this ratio is shown in Figure 3. In most cases, the water quality

Figure 3

INORGANIC GROUND WATER QUALITY VIOLATIONS



violations are not severe and the average concentrations exceed the standards by a factor of less than 2.5. In a few instances, the water quality violations are a major concern. For example, in one monitoring well the average concentration of lead was 4.24 mg/l, a factor of 85 increase over the water quality standard.

Ground Water Quality Violations - Organics

The water quality records of the N.C. Solid & Hazardous Waste Management Branch were reviewed for information on the detection of hazardous organic contaminants in the ground water near the sanitary landfills included in this study. The organic analyses performed by the state laboratory in Raleigh consisted of a gas chromatography/mass spectroscopy (GC/MS) scan for 106 organic priority pollutants. Occasionally this list was supplemented by other compounds when detected in high concentrations. The minimum detection limits of the equipment used ranged between 5 and 50 ppb for the various organic compounds. Only 36 out of 71 landfills (52%) included in this study had organic analyses. Typically, samples from all of the monitoring wells at a site were analyzed. Surface water samples were rarely analyzed for organics.

Since most landfills were monitored only once for organics, no attempt was made to perform a detailed statistical analysis. Instead, we have attempted to identify those compounds most commonly detected in ground water near sanitary landfills (Table 10). The North Carolina Administrative Code (1985) states that synthetic, man-made organics, or other substances that do not naturally occur are prohibited from entering ground water. Those compounds listed in Table 10 therefore represent violations of the code. A total of 14 out of 36 (39%) of the sanitary landfills with organic analysis data had organic priority pollutants detected in ground water samples. The compounds methyl ethyl ketone (MEK), tetrahydrofuran, and caprolactam were detected at some landfills but are not reported in Table 10 because of uncertainty over the source of contamination. MEK and tetrahydrofuran are PVC solvents which may have been introduced into the ground water during construction of older PVC monitoring wells. Caprolactam is used in the manufacture of nylon ropes and could have been introduced during bailing. Recently revised well installation and sampling procedures should eliminate future monitoring problems.

The organic compounds which were most commonly detected in ground water were the chlorinated solvents (three landfills), petroleum derived hydrocarbons (three landfills), and pesticides (two landfills). At two landfills there appeared to be extensive contamination from a wide variety of organic compounds. Two landfills also showed high levels of anaerobic degradation products such as propanoic, butanoic and pentanoic fatty acids, although synthetic organics were not detected at these two sites. These anaerobic degradation products are not toxic but due to their offensive odor, would render water supplies unfit for other uses. The high frequency of organics detected in ground water from landfills examined in this study suggests that sanitary landfills do pose a significant statewide threat to

Table 10

Organic Compounds Detected in Ground Water¹

<u>Compound</u>	<u>Detection Limit (µg/l)</u>	<u>% of Landfills Where Detected</u> ²
Benzene	5	5
2,4-D	1	5
Decane	1	3
1,1-Dichloroethane	5	5
Ethylbenzene	5	3
Heptane	1	3
Hexane	1	3
Total Hydrocarbons	NA	8
MIBK	2	3
Octane	1	3
Phenols	1	5
Tetrachloroethylene	5	3
Toluene	5	8
TOX	1	18
1,2-Trans- dichloroethene	5	3
Trichloroethylene	5	3
Xylene	5	5

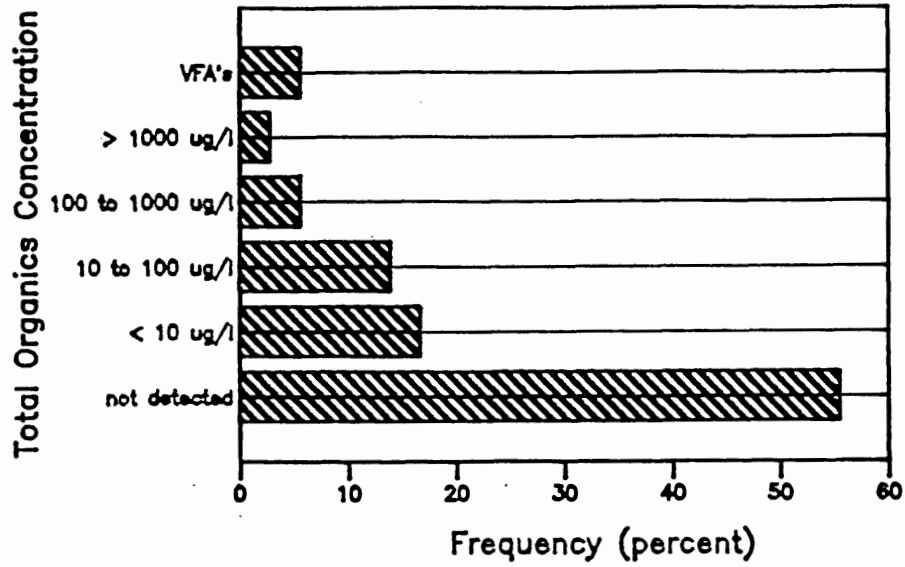
¹ Organics list excludes caprolactam, MEK, and tetrahydrofuran which may have been introduced during well construction or sampling.

² Based on a total of 37 landfills with organic analyses.

ground water quality. In order to judge the severity of contamination by organic chemicals, the total concentration of all organics detected in each monitoring well was summed and then separated into the following categories: (1) zero organics detected; (2) total organics less than 10 µg/l; (3) total organics between 10 and 100 µg/l; (4) total organics between 100 and 1000 µg/l; (5) total organics greater than 1000 µg/l; and (6) high levels of volatile fatty acids (VFA's) but zero synthetic organic chemicals. The results of this classification are presented in Figure 4. It is apparent from these data that the concentration distribution of organic chemicals in this study is strongly asymmetrical with observations of several thousand micrograms per liter made at a few sites and no detectable contamination at the majority of sites.

Figure 4

Sum of Organics Detected in Ground Water



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GLOSSARY

Ag	- silver
ALK	- alkalinity as CaCO ₃
As	- arsenic
avg.	- average
Ba	- barium
BOD	- five day biochemical oxygen demand
Cd	- cadmium
Cl	- chloride
COD	- chemical oxygen demand
CON	- conductivity
conc.	- concentration
CPE	- chlorinated polyethylene
Cu	- copper
CY	- cubic yards
Ds	- downstream
DOC	- dissolved organic carbon
EPDM	- ethylene-propylene rubber
F	- fluoride
Fe	- iron
FS	- fixed solids
GC/MS	- gas chromatography/mass spectrometer
Hg	- mercury
JTU	- Jackson turbidity units
lf.	- landfill(s)
max.	- maximum
MLVSS	- mixed liquor volatile suspended solids
Mn	- manganese
n	- number of observations
NA	- not applicable
NIT	- nitrate
NPDES	- National Pollutant Discharge Elimination System
obs.	- observation(s)
ORP	- oxidation reduction potential
P	- phosphorous
Pb	- lead
ppb	- parts per billion
PVC	- polyvinyl chloride
r	- correlation coefficient
Se	- selenium
SLF	- sanitary landfill
stat.	- statistically
SUL	- sulfate
TDS	- total dissolved solids
TOC	- total organic carbon
TOX	- total organic halides
TUR	- turbidity
US	- upstream
VOC	- volatile acids concentration
Vs	- volatile solids
Zn	- zinc

APPENDIX
Landfill Survey Form*

Landfill Owner _____
County _____
Landfill Name _____
Landfill Number _____

Sanitary Landfill Study

Please fill in each question with the answer that best states the actual condition at your landfill. If you operate more than one landfill, please check the Landfill Name and Number in the upper left-hand corner of this page to be sure that you are answering the questions for the correct landfill. If an answer is not known, write UNKNOWN in the blank space. All information collected by this survey is for study purposes only.

I OPERATIONS

- | | |
|--|----------------|
| 1) When did the landfill first start operation? | _____ 1976.6 |
| 2) How much of the landfill, as a percentage, has already been used? | _____ 52% |
| 3) What is the approximate size of the total landfill, in acres (the sum of the used and unused portions)? | _____ 118.6 |
| 4) What is the expected future life of the landfill, in years? | _____ 12.7 |
| 5) What is the approximate rate, in tons per day, which refuse is brought to the landfill? | _____ 326.1 |
| 6) How many acres of the landfill are used per year to dispose of the refuse? | _____ 5.6 |
| 7) What percentage of the solid waste is brought from towns, cities, and other municipal sources? | _____ 53.4% |
| 8) What percentage of the refuse is brought by private haulers or private individuals? | _____ 33.1% |
| 9) What is the average haul distance? | _____ 12.8 mi. |
| 10) What is the final depth of the refuse? | _____ 42 FT |
| 11) How is the refuse compacted? | _____ |
| _____ | |
| 12) What is the approximate acreage of uncovered refuse at any given time? | _____ 0.4 AC |
| 13) What is the depth of the intermediate or daily cover? | _____ 0.8 FT |
| 14) Does the landfill accept liquid municipal wastes (such as sludge from sewage treatment plants)? | _____ 20% yes |

* Tabulated values are average of all survey responses.

- 15) Does the landfill accept liquid industrial wastes? 0% yes
- 16) Does the landfill accept septic tank pumpings (septage)? 5% yes

II ENGINEERING DESIGN

- 17) Is the trench or area method used? 36% area, 24% trench
10% both, 18% other
- 18) What is the depth to the water table below the lowest point where wastes are placed? 16.6 FT
- 19) What is the depth to bedrock from the lowest point where wastes are placed? 41.5 FT
- 20) Is a low permeability soil or synthetic liner currently placed underneath all wastes? 29% yes
- 21) Is a double liner system used? 3% yes
If no soil liner is used go to question 24.
- 22) If a soil liner is used, please give the SCS classification or other appropriate description for the liner material.
- 23) How thick, in inches, is the soil liner? insufficient data
If no synthetic liner is used go to question 26.
- 24) If a synthetic liner is used, what is the brand name or chemical name of the liner material? insufficient data
- 25) Briefly describe the bedding or other practice used to protect the liner from damage during operation of the landfill. insufficient data
- 26) Does the landfill have a leachate collection system? 26% yes
If leachate is not collected, go to question 31.
- 27) Briefly describe the method used to treat the leachate prior to final disposal (facultative lagoon, trickling filter, etc). primarily facultative lagoons
- 28) Is the leachate treated at the landfill, or is it transported to another site for treatment? 100% onsite
- 28a) If treated offsite, how far must the leachate be moved and what transportation method is used?
- 29) How is the leachate finally disposed?
- 30) If leachate is discharged to surface water, has an NPDES permit been issued? 5% yes
- 30a) What is the NPDES number? NA
- 30b) What is the permitted discharge rate in gallons per day? 62,500 g/d

31) What is the SCS classification or other appropriate description of the cover material? _____

32) What is the depth of the final cover material? 2.3 FT

33) How is the cover material compacted?

34) Do you use temporary seeding on the cover material? 33% yes

34a) If temporary seeding is used, what type of plants do you seed? _____

35) What type of plants are used in final seeding of the cover material?

36) What is the slope of the landfill cover? 1.7%

III ENVIRONMENTAL CHARACTERISTICS

37) How far is the nearest stream from the edge of the fill (in feet)? range 4 to 5 miles, 1150 FT

38) What is the drainage area of the stream at the point nearest to the landfill in square miles? GEOMETRIC MEAN = 4.5 mi²

39) Is the stream used for a water supply downstream of the landfill site? yes 17%

40) How far are the nearest homes from the landfill (in feet)? 2000 FT

41) What was the natural topography of the landfill site (was it originally a ravine, a valley, a hillside, etc.)?

42) What is the natural soil type at the site based on the most recent county soil survey?

43) What is the geologic classification of the site (example: Triassic Basin, Stoneville formation)?

44) What would be a typical yield in gallons per minute for domestic wells in this area? 47 gpm

45) How many ground water monitoring wells are at the landfill site? 3.8

46) How many times per year are these wells sampled? 2.1

