

76-10-2

December 1981
Report No. Env. E. 67-81-2

Sanitary Landfill Leachate Collection and Treatment

Proceedings from the 1977 Symposium
University of Massachusetts, Amherst

Edited by

Aaron A. Jennings
Research Assistant

The research upon which this publication is based was supported in part by the
Division of Water Pollution Control, Massachusetts Water Resources Commission and
the U.S. Forest Service, Northeastern Forest Experiment Station.



ENVIRONMENTAL ENGINEERING PROGRAM
DEPARTMENT OF CIVIL ENGINEERING
UNIVERSITY OF MASSACHUSETTS
AMHERST, MASSACHUSETTS 01003

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Department of Civil Engineering
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Amherst, Massachusetts 01003

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PREFACE

Under the sponsorship of the Massachusetts Division of Water Pollution Control and the U. S. Forest Service Northeastern Forest Experiment Station, The University of Massachusetts has engaged in studies of sanitary landfill leachate. Specific areas of interest include its production rate, its interaction with soils, the characterization of mode of transport, and methods of treatment. This symposium was organized to disseminate some of the initial findings of the research project and to supplement the University's investigations through reports from other federal and state authorities. It was anticipated that detailed reports on the University's investigations would be made public through masters theses, technical reports, conference presentations, and journal articles.

The Symposium organizing committee consisted of Dr. Donald Dean Adrian and Dr. Tsuan Hua Feng, principal investigators on the University's research projects, and Mr. Ronald Lavigne, Research Assistant, all of the University of Massachusetts' Environmental Engineering Program of the Department of Civil Engineering. Mr. Leonard Martone of Martone Trucking Company, Barre, Massachusetts, is gratefully acknowledged for his support, encouragement, and cooperation in the University's leachate research. Dr. Aaron Jennings served as a researcher and as Editor of the Symposium Proceedings. Ms. Christina Moore organized the final publication of this report which was typed by Mrs. Dorothy Pascoe.

The encouragement of Dr. Clive L. Dym, Civil Engineering Department Head is acknowledged with thanks. The efforts of the following people, all of whom contributed to parts of the research reported in the Symposium Proceedings, are especially appreciated: Mr. Brian Payne, project officer for the U. S. Forest Service Northeastern Forest Experiment Station grant; Warren Kimball, Arthur Screpetis, John Elwood and Dr. Russell Isaac, Massachusetts Division of Water Pollution Control project officers, and David Standley, Commissioner of the Massachusetts Department of Environmental Quality Engineering.

Finally, the dedication and enthusiasm of the leachate team is acknowledged as they worked far harder and more enthusiastically than one had a right to expect. They included: Ronald Lavigne, Aaron Jennings, Franklin Tirsch, Robert Pease, George Lombardo, Paul Walker, Sandra Ferry, Paul Kozlowski, H. Patricia Hynes, Robert Drake, David Ferguson, Albert Narbonne, Elizabeth Johnson, and Phyllis Mayberg.

Donald Dean Adrian
December 1981

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LEACHATE PRODUCTION IN THE NEW ENGLAND ENVIRONMENT

Ronald L. Lavigne, Research Assistant

Environmental Engineering Program
Department of Civil Engineering
University of Massachusetts, Amherst

BACKGROUND

The generation of solid wastes has been a part of man's life style ever since his first aggregation of the earth's resources for his personal and public needs. C. G. Gunnerson (1), has even theorized that the rate of generation per century remains relatively constant from one populated area to another. Be that as it may, the methods of refuse disposal have also remained remarkably similar over the ages. There seems to be little difference between mounds of animal bones scattered at the mouths of caves, and mounds of "tin" cans scattered at the local dump.

More recently, man has advanced to the technique of covering wastes from time to time, but historically the motive here has not been one of environmental or sanitary concern. Rather, he has either desired the area for other purposes or has been forced to add cover material so that he might maintain an access to the dump's "face". Volume reduction has been and still is, in many parts of the country, accomplished by burning. For most communities, dump site selection was rarely a difficult problem. A parcel of land conveniently located and lacking utilitarian value was usually selected as a site. More often than not in New England, this site would be a swamp, swail, or river bank where people could neither build nor farm.

And so it was that communities chose and operated the "town dump" on a least-cost, least-inconvenience basis. With the dump came rodents, odor, flies, smoke, wind blown litter, and the local dump picker who salvaged metals, motors and miscellany as a marginal economic enterprise.

In the late 60's and early 70's serious questions were raised regarding methods of refuse disposal. On April 21, 1971, "Regulations for the Disposal of Solid Wastes by Sanitary Landfill" were filed with the Secretary of State, Commonwealth of Massachusetts.

Though the individual regulations in and of themselves appear to be reasonable from a common sense point of view, they imposed a drastic alteration of old established practices in the disposal of solid wastes. Implementation time was not provided for in the regulations but was instead left to the subjective determination of the Department of Public Health in each "Region". Larger cities and towns were usually made priority areas. Small towns were often ignored. Even today, some of them still operate the "town dump" in a manner not unlike that which was practiced prior to the issuance of state regulations.

Health officials have probably always been aware of the fact that waters leaching from dumps have undesirable qualities. Because these contaminated waters were usually restricted to the swamps the dumps were located in, little concern was ever given to the matter. Even when dumps were located along river banks, the accepted practice was to use the river as a receptacle for human externalities, and leachate discharge was no exception. More recently, landfill leachates have come under critical fire by health officials, researchers, and environmentalists. Areas of concern might be summarized as follows:

1. How much leachate is generated from landfill to landfill?
2. What are the chemical components and concentrations of leachate?
3. How much human sewage and how many pathogenic organisms leach from dumps into surrounding surface and ground water supplies?
4. Have changes in management practices over the years affected leachate composition i.e., do leachates from older parts of the landfill with large amounts of burned materials differ appreciably from those generated from current landfilling areas?
5. How much BOD is generated by unburned organic matter that is covered on a daily basis?
6. Does a 4 foot barrier of natural soils prevent leachate infiltration to the ground water?
7. How far and at what rate can leachate migrate from a landfill source?
8. If leachate does constitute a threat to surface and ground water quality, how might it be economically captured and treated?

9. How well is leachate attenuated in a soil media with respect to the various leachate components?
10. Are there any indications that sanitary landfilling practices in New England cause contamination of surface and/or ground waters?

One need not visit many sanitary landfills to conclude that refuse materials do indeed leach out in solution to surrounding areas.

Ironically, in New England where rainfall is heavy and leachates abound, very little research has been done to evaluate the effects of these leachates on public health and the environment.

A study by Apgar and Langmuir (2) started in 1967 at Pennsylvania State University detected large doses of leachate infiltration into ground water even though the landfill was located some 200 feet above the water table. Some 27 inches of rain falls annually on the site which has been in operation since 1962. Even after moving downward in the soil to a depth of 36 feet in 7 years, the leachate beneath the cells had a conductance of 6600 micro-ohms, 600 ppm $\text{NH}_3\text{-N}$, and 100 ppm of total Fe. It was also calculated that the rate of leachate movement was between 6-11 feet per year. Some areas of the landfill had leachates in similar concentrations to those mentioned above traveling as much as 80 feet over the seven year period. As a result of this study, researchers at Pennsylvania State concluded that serious ground water pollution can occur even when landfills are placed well above the water table.

The Illinois State Geological Survey has carried out extensive studies to determine what the optimal depth of impervious soil material should be between refuse and water table (3). Using typical clay, shale and silt, it was concluded that at least 30 feet of impermeable material is required between the bottom of the landfill and the shallowest aquifer.

When one considers that Massachusetts requires only 5 feet of separation, there is little wonder that leachate pollutants are increasing in ground and surface waters near their respective landfills.

If a landfill receives large amounts of industrial waste, the leachate is typically rich with inorganic chemical constituents. If domestic refuse and garbage find their way to the landfill, organic and microbiological constituents tend to be greater. With respect to the latter, it is

interesting to note that an ever increasing quantity of human body waste is finding its way to land disposal sites via disposable diapers.

Cooper, et al (8) attempted to evaluate the viability of microorganisms in landfill leachate by utilizing both natural and artificial conditions of growth. By using some 16 different lysimeters, comparisons were made between open dumps and sanitary landfills and between natural refuse and fecally seeded refuse. A select group of the fecally seeded lysimeters also received doses of laboratory cultured polio virus.

In comparing leachates from the different test units the following conclusions were evident:

1. Fecal coliform counts in leachate from both seeded and unseeded refuse were extremely high. This indicated a large amount of fecal material in the natural untreated refuse.
2. All coliform counts were higher in open dumps than in sanitary landfills. This was true not only for initial flows, but also for those leachates that were generated at later times during the collection period.
3. Fecal coliform counts were always as large as total coliform counts under all conditions. This suggests that a sizable amount of feces from warm blooded animals enters landfills with the typical daily refuse.
4. Though virus recovery was always higher in seeded refuse, considerably large amounts were also recovered from untreated refuse. The "open dump" units never generated viruses in their leachates, even when they had been added to the waste. Only the "sanitary landfills" were capable of sustaining polio virus and releasing them with their leachate.
5. As a result of these investigations, Cooper and his associates concluded that solid wastes containing disposable diapers, animal wastes and other forms of fecal contamination should be considered as a potential threat to public health even when disposed of by sanitary landfill methods.

Glantzbecker and Movello (5) utilized uniformly packed, anaerobic columns to evaluate the effectiveness of fines and clays in attenuating

bacteriological components of landfill leachates. The results of their study indicated that excellent attenuation of bacteria and virus (99%+) was possible if leachates could be forced to flow through 10 cm of silty clay. It should be noted that in a natural landfill environment this would be quite difficult in that silty clay would function more as a liner to direct flows horizontally across the surface. Though some leachate might initially percolate downward in a filter type manner, surface clogging and low specific yields for clay (6) would force almost all of the fluid to take the path of least resistance (i.e. horizontally through the over bearing layer of porous refuse). The hydrogeology of this phenomenon is best described by application of "Leaky Aquifer Theory" for conduction of water through a semi-confining stratum (7).

In actual field observations, the author has observed that leachates usually "break out" of the landfill face at the lower extreme of each lift. This observation concurs with the assumptions previously made. Figure 1 attempts to illustrate the flow pattern described.

In summary, it appears that landfill leachate poses a serious bacteriological and viral threat to ground and surface water quality. If leachate can be contained within the soil and is forced to flow through a semi-permeable stratum, screening and adsorption can occur, but all too often it has been found to break out at the face of the fill area and flow over land to adjacent surface waters. If these surface waters are used for drinking or contact recreation, there may be a need for increased attention to the microbiological health implication associated with leachate.

RESEARCH OBJECTIVES AND CONSIDERATIONS

As previously mentioned, very little research has been undertaken in New England to answer questions about surface water contamination from landfill leachate. The literature does, however, indicate a wide range of pollution parameters, as they relate to quantity and concentration at the leachate source (3). This suggests that most landfills are unique with respect to leachate production. This variability in production is probably a function of refuse composition, cover material type, annual precipitation, method of operation, and other geological factors. If this

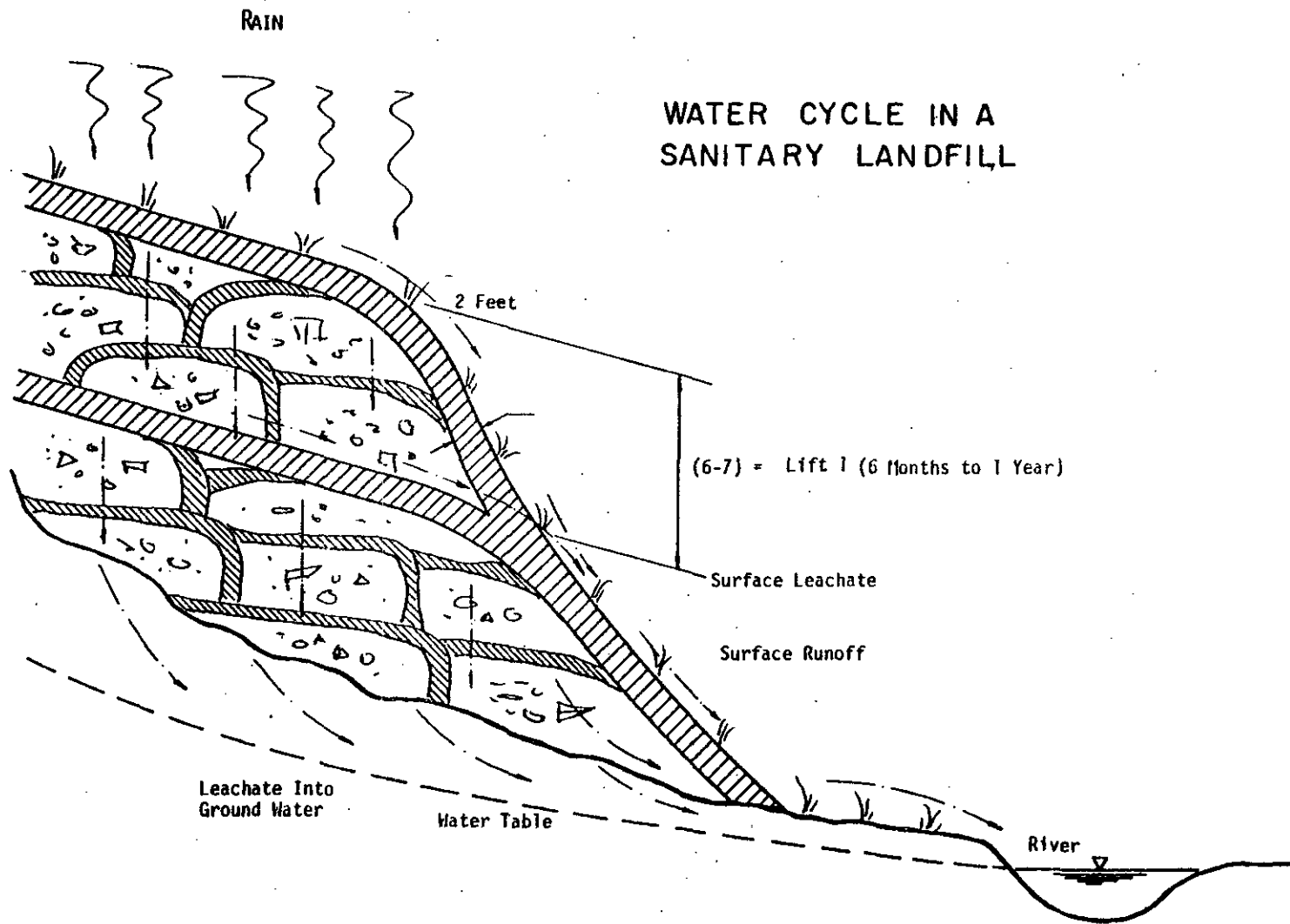


Figure 1. Leachate Movement through a Typical Sanitary Landfill.

is true, there may be shortcomings in prescribing a single methodology of landfill operation. Instead, perhaps what is needed is a site by site design plan that would attempt to consider the interacting components of leachate production mentioned above.

It was the objective of this project to determine if, in fact, leachate production and variability does occur in a New England environment and if individual site designs for leachate control are needed.

The determination of these unknowns was accomplished by attempting to answer the following questions:

1. Does the quantity of leachate production vary from landfill to landfill?
2. What are the chemical components and their concentrations in leachate waters?
3. Does human sewage in a landfill pose a public health problem with respect to possible pathogens in leachate?
4. How do location and method of operation affect leachate parameters in existing landfills?
5. Is there any indication that surface and ground water contamination is occurring in New England from landfill leachates?
6. How effective are present state regulations in controlling leachate production and movement?
7. If present regulations and methods of operation are inadequate, what are the alternatives available to properly control leachate production and movement?

Because of the wide variability of landfills in the Commonwealth, more than one was needed for this study. Some landfills are relatively new and follow State Regulations quite closely. Others are older and with respect to operation, are "open dumps". There is also a wide variability in refuse species. Rural landfills receive mostly domestic waste. Urban areas typically generate larger amounts of industrial waste.

In addition to these variables, topography is also an important consideration. Preliminary studies indicated that some landfills were in lowlands, either dumping into surface waters or very close to them. Situations like this are obviously a serious threat to surface water quality.

Other landfills were situated on high ground well away from surface waters. This condition, of course, poses a greater threat to ground water quality,

Complicating the problem even more are the geological factors that must be considered. When gravel type materials are used to cover refuse, the primary force at work in attenuation is filtration or screening. On the other hand, if clays and fines predominate one would expect adsorption and cation exchange to be principal forces. Though loam is not usually used as a daily cover material, if it were, one would have to consider biological activity as a major factor in the attenuation process.

All factors being considered, it is easy to see why the "typical landfill" just does not exist. It is because of the variability described that this project required an evaluation of several landfill sites. In choosing the sites, an attempt was made to represent as many of the variables as possible by isolating them in the respective landfill choices. This made each landfill unique and provided a basis for comparison of leachate production as a function of landfill type.

DATA COLLECTION AND SAMPLING ANALYSIS PROCEDURES

For each of the landfills selected, the following procedures were used.

1. Characterization of Incoming Wastes. A quantitative and qualitative determination of wastes was made on a gross scale. That is to say, weekly, monthly or annual loadings were estimated and a general descriptive characterization of wastes was compiled by site inspection. This also included an estimate of percent of total for each of the refuse types identified.
2. Characterization of Operation: Each landfill was described in terms of operational techniques being practiced. Typical considerations were cell structure, compaction technique, equipment types, depth of cover material, and frequency of cover. (Not all landfills are covered daily.)
3. Geologic, Topographic, and Hydrologic Characterization: For each landfill an attempt was made to characterize the above parameters. Standard sieve analyses were used to describe the cover materials. Topographic maps were utilized to describe grade and water proximity.

4. Characterization of Leachate. Regular sampling was carried out on a bi-weekly basis for four seasons of the year. Sampling sites attempted to represent (a) leachates being generated at the landfill "face", (b) upstream and downstream surface waters, and (c) well pumpings from ground water near the landfill.

Standard Methods were used to evaluate the following parameters.

- | | |
|-------------------------------|---------------------|
| 1. Temperature | 11. Ammonia |
| 2. Dissolved Oxygen | 12. Chlorides |
| 3. Biochemical Oxygen Demand | 13. Iron |
| 4. Carbonaceous Oxygen Demand | 14. Chromium |
| 5. Total Organic Carbon | 15. Zinc |
| 6. pH | 16. Copper |
| 7. Alkalinity | 17. Total Coliform |
| 8. Phosphate | 18. Fecal Coliform |
| 9. Nitrate | 19. Total Solids |
| 10. Sulfate | 20. Volatile Solids |

Not all parameters were evaluated for all samples. Generally speaking, the scope of analysis and intensity of sampling increased with time as new questions and new sites were identified.

SITE SELECTION

In an effort to represent the full spectrum of landfill types that were discussed earlier, the following sites were selected for consideration. Included is a brief description of their general characteristics and/or unique problems.

1. Easthampton, Massachusetts (Figure No. 2)

The Town of Easthampton has its landfill located in a relatively wet area. Poor engineering in the original design has resulted in a serious leachate problem. As Figure 2 illustrates, a drain tile was buried below the fill area to lower the natural water table.

Very soon after opening, a serious leachate problem developed at the outfall of the drain tile. Heavy flows of highly contaminated ground water flushed across the lower face of the dump area into a clean mountain brook.

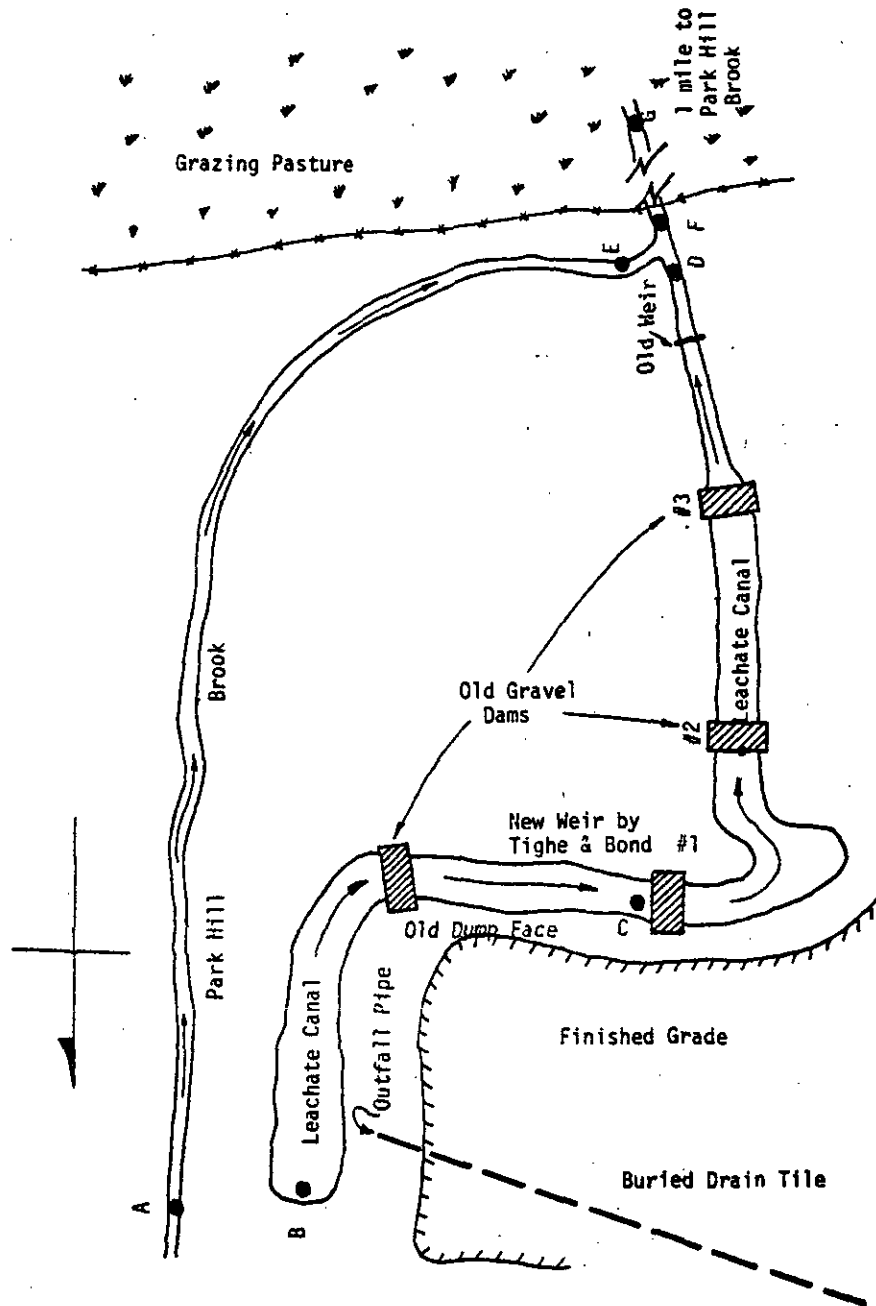


Figure 2. Easthampton, Massachusetts Landfill

Early efforts to solve the problem included the construction of a small ponding area next to the pipe. When the pond overflowed shortly after its completion, efforts were aimed at reducing the flow from the pipe itself. This was accomplished by extending the working face beyond the end of the pipe. It was expected that sealing of the end would greatly reduce flow rates into the ponding area. This was not the case and the overflow problem repeated itself during following storms. Over the past three years, the ponding area has been increased to many times its original size. It now forms a long meandering canal at the lower face, sectioned by a series of small gravel dams. In spite of all measures taken to date, when it rains the dams wash out and the canal empties directly into the brook. Recently, the town decided to leave the dams open so that leachate could escape 24 hours a day at a lower flow rate. Ironically, Easthampton designed, built, and operates its landfill by prescribed regulations.

2. Amherst, Massachusetts

Amherst, unlike Easthampton, has its landfill located on high ground, well away from any major surface water. Its waste is primarily domestic and commercial. It also receives a large amount of sludge from an over-taxed primary wastewater treatment plant.

Ground water is monitored with test wells located downgrade of the landfill. Increasing amounts of chloride in the ground water samples have raised considerable concern about water quality in nearby municipal wells. The wells are less than a mile away from the landfill and should they become contaminated Amherst will have to look elsewhere for water.

It should be noted that the Amherst Landfill has complied with all aspects of the State Sanitary Code. As with Easthampton, there are leachate problems in spite of this fact.

3. Athol, Massachusetts (Figure 3)

Athol has a large, poorly operated "dump". Unlike the previous two, there is a heavy inflow of industrial waste. Tully Brook flows within 100 ft of its lower face and when leachate flows are high, they pour directly into the brook.

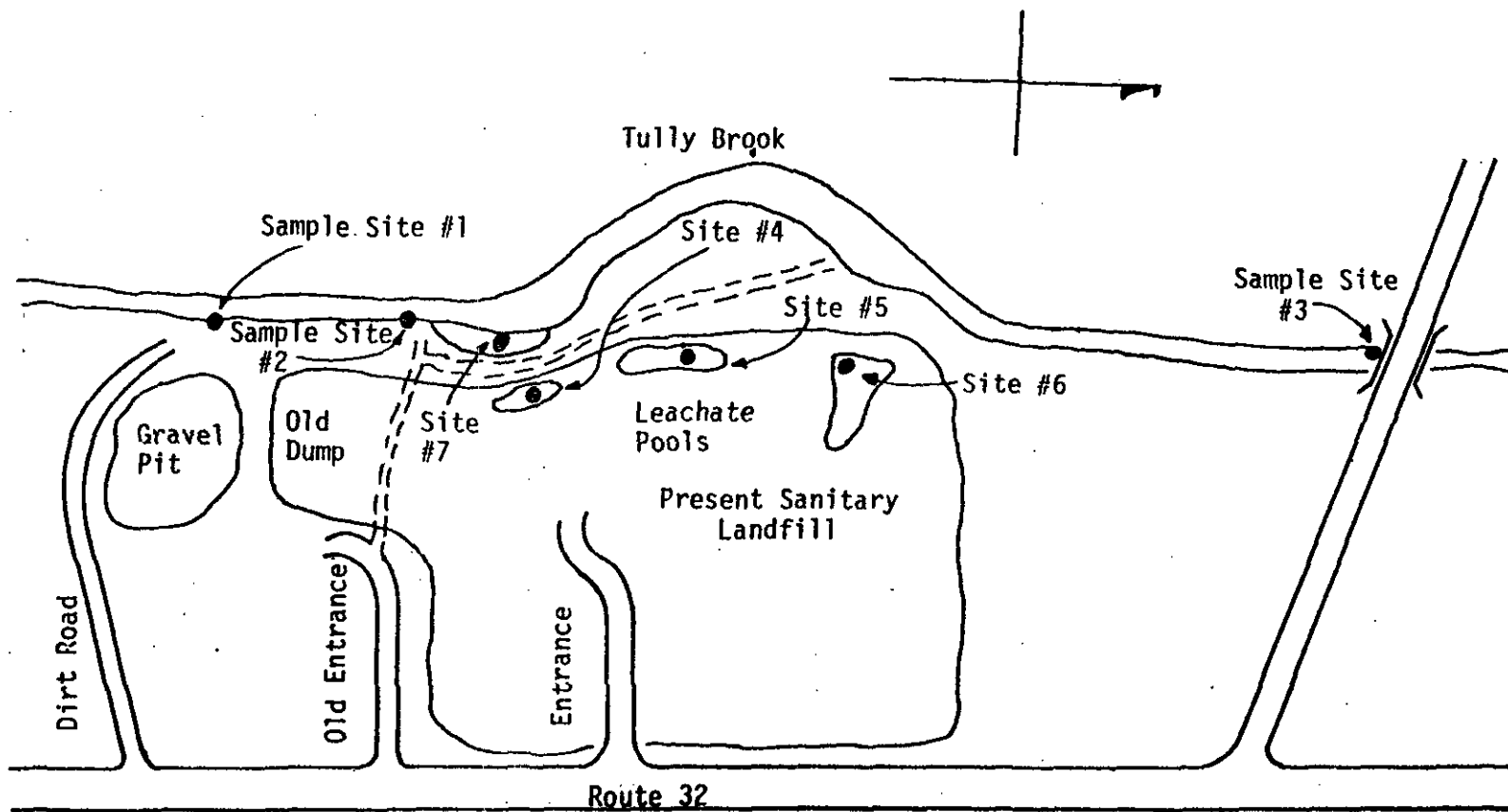


Figure 3. Landfill at Athol, Massachusetts

In talking with some of the residents, reports indicate very little control over the inflow of toxic wastes from local industries. One industry reportedly has been dumping between 10 to 20 fifty-five gallon drums of cyanide waste each month for the past few decades!

Sewage treatment sludge is not disposed of at the landfill.

An additional characteristic worth noting is that Athol has been using its present site since the turn of the century.

4. Petersham, Massachusetts (Figure 4)

Petersham does not have a landfill. Over the years it has been able to exist as an "open dump" with virtually no constraints. Cover material has been added from time to time, but only enough to maintain access to the "open face". The "lower face" is situated in a small swamp and a good part of the refuse is under water. As Figure 4 indicates, this swamp drains into a feeder brook of Quabbin Reservoir.

Characteristics unique to this study site are (a) the comparatively small volumes of waste received, (b) the total absence of industrial or sludge wastes, and (c) the placement of refuse directly into surface water resources.

5. Barre, Massachusetts (Figure 5)

The Town of Barre pays for the use of a privately owned landfill located within the town. In addition to owning the landfill, Martone Trucking imports refuse from several surrounding towns where it has refuse collection contracts.

The landfill is located within several hundred feet of the Ware River but an old railroad bed provides a solid barrier between the river and the refuse site.

Refuse is typically domestic with only a minimal input from small industry. There is no sludge disposal at the landfill. Local residents are permitted to use the facility on a limited schedule and the remaining waste enters via "packer truck."

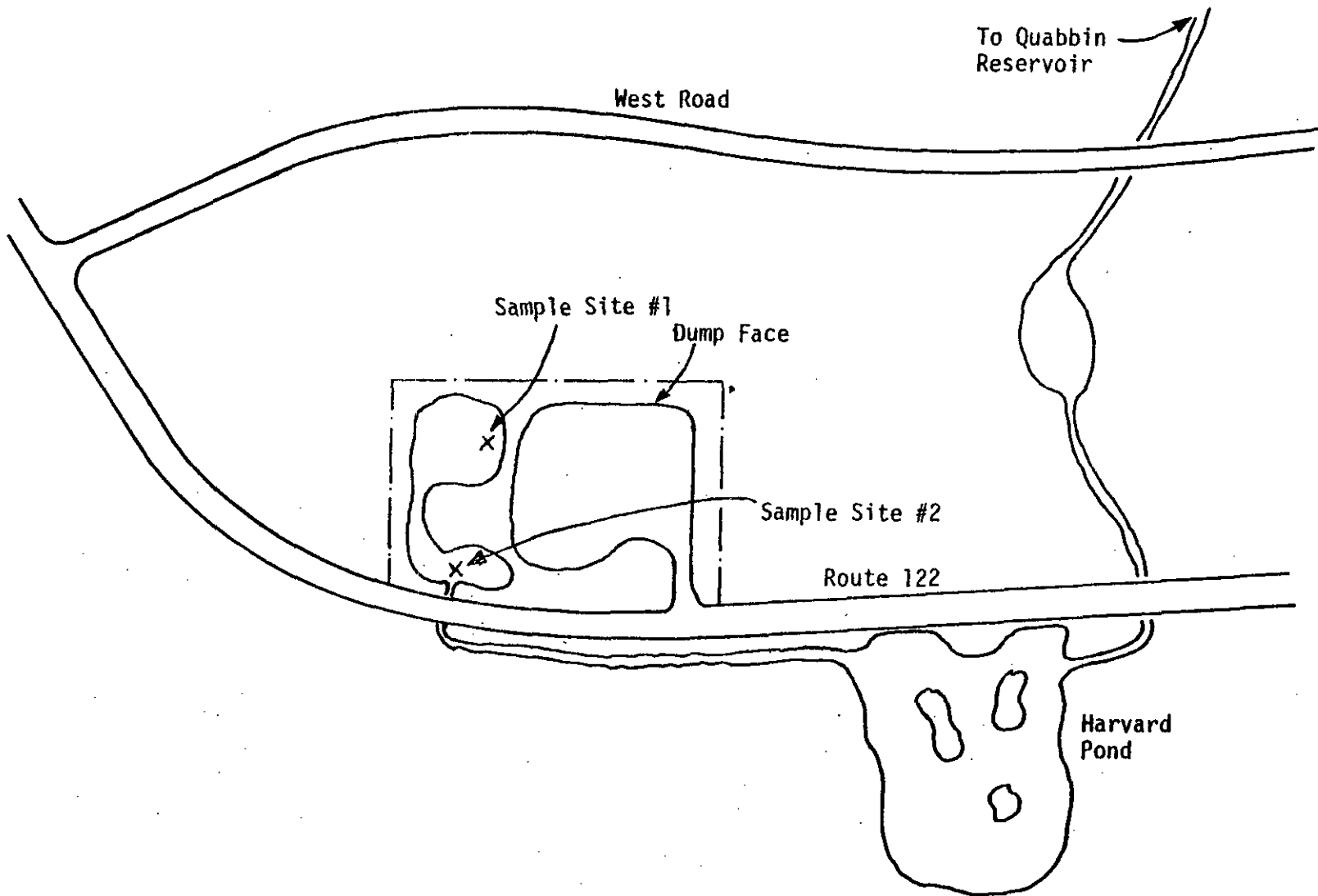


Figure 4. Petersham, Massachusetts Town Dump

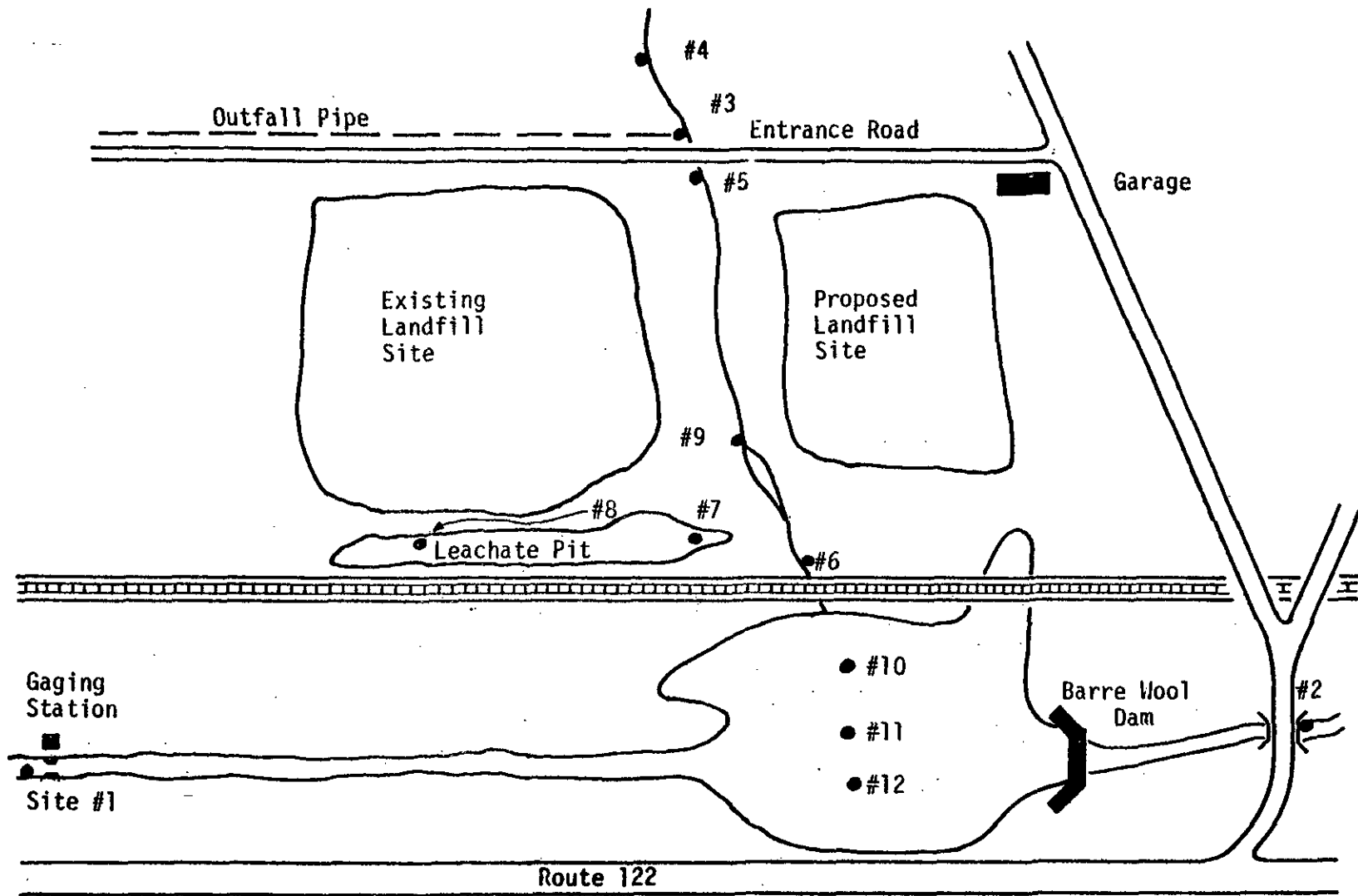


Figure 5. Landfill, Barre, Massachusetts

SITE ANALYSIS

Case 1: Easthampton, Massachusetts

Historical Overview

The Town of Easthampton is located in the flood plain on the west bank of the Connecticut River.

Much of the well drained farmland that once kept Easthampton very rural has now been developed into large housing tracts. This has resulted in a population growth rate of approximately 1.5 percent per year over the past twenty years. The present population is just under 15,000.

If we assume an annual per capita solid waste production of 2000 lb (8), Easthampton has some 15,000 tons of waste to dispose of annually. Table 1 illustrates past, present, and future population and solid waste growth patterns for the town.

The present sanitary landfill began operation in July of 1969. The original engineering plan was developed by Metcalf and Eddy and site preparation was carried out by Town forces.

Initial operations were marred by controversy relative to the placement of refuse in trenches containing water. On January 20, 1970, an order from the State Department of Public Health called for the Town to excavate all refuse located within four feet of ground water.

As an alternative to the removal of the rubbish, the Town decided to construct a drainage ditch (Figure 2) designed to lower the ground water level in the area of the rubbish. The ditch was constructed in August of 1970 and it was successful in lowering the ground water level.

Unfortunately, severe leachate problems developed in association with the dewatering operation.

In July of 1971, Tighe and Bond Engineering Consultants were engaged by Easthampton to develop a plan that would comply with water pollution orders being issued by the State Department of Public Health.

Their plan called for the laying of some 400' of 8" perforated pipe under a new fill area in the northwest corner of the site. The pipe was to be placed some 4-5' below the lower extreme of refuse placement. In addition to diverting ground water from the landfill 5' above, the drain pipe soon became an interceptor of leachates percolating through the refuse.

Table 1. Predicted Population and Estimated Refuse Generation
Easthampton, Massachusetts, September, 1971

Year	Population Estimate	Estimated per Capita refuse generation**	Tons per Week	Cubic Yard per Week	Acre Feet per Year
Previous Average*	13,000	3.5	160	860	
1971	13,200	4.9	226	1,200	38.6
1972	13,400	5.0	234	1,240	40.0
1973	13,550	5.1	242	1,285	41.4
1974	13,780	5.2	246	1,305	42.0
1975	13,950	5.3	259	1,375	44.3
1976	14,150	5.4	268	1,420	45.7
1977	14,450	5.5	278	1,475	47.5
1978	14,600	5.6	286	1,520	49.0
1979	14,800	5.7	295	1,565	50.5
1980	15,000	5.8	305	1,620	52.2

*Before open burning ban became effective.

**Pounds per capita per day.

NOTE: The estimated refuse generation rates were based on a short observation period. However, Phase II of the Report will reflect 9 continuous months of observation. The estimated per capita refuse generation rates reflect a direct proportional increase as realized during the observation period; i.e.,

$\frac{1200}{860} \times 3.5 = 4.9 \text{ \#/capita/day}$. Also, a uniform increase in both population and refuse generation has been established as illustrated in the above table.

NOTE: This table was prepared by Tighe & Bond Engineering for the Town of Easthampton.

This resulted in increased flows of highly contaminated leachate into the small brook crossing Park Hill Road (Figure 2). Following is a section of the original Engineering Study (9) that describes the under drain more clearly.

"The first and foremost consideration required before utilization of the Stage 1 parcel should be some minor grading and the laying of an under-drainage pipe. The laying of some 400' of 8" asphalt coated corrugated metal perforated pipe encased in a 30-inch envelope of 1/2" trape rock, and subsequently covered to approximately 5' above the invert of the pipe with fill material will assure adequate drainage of the area and maintain a minimum distance of 4' from the bottom of the refuse to the high ground water table. The usable depth of this parcel will be 8' bringing the finished Stage I elevation to approximately 236'."

It is evident from the above reference that a distance of 4' of leaching was considered adequate for leachate attenuation prior to its discharge. Considering the pollutorial strength of discharges from the pipe, especially during periods of heavy rain, it also seems evident that the prevention of rainfall infiltration into active sites is almost impossible and that leachate generation can begin very soon after refuse placement. In the case of Easthampton, BOD₅ breakthrough occurred in a few months, with values exceeding 200 mg/l by the end of the first year. After 18 months of operation, BOD₅ values were 500 mg/l. Because of the heavy dilution of leachate from intercepted ground water, the value of 500 mg/l BOD₅ must be interpreted as only a fractional part of the actual leachate strength at the source.

Contamination of Park Hill Brook

Since the opening of the Easthampton landfill in July of 1969, Park Hill Brook has been subjected to gross contamination by leachates. This contamination is usually evident more than a mile downstream of the outfall where the brook crosses Park Hill Road. Water samples (Site G, Fig. 2) have been taken from this crossing since October of 1969 and as Tables 2, 3 and 4 indicate, the quality of the brook water started to deteriorate in less than a year after the landfill's opening.

Upstream BOD's have continually remained in the 1-2 mg/l range, and COD values have rarely exceeded 50 mg/l (Tables 2, 3 and 4, Site A, Fig. 2). Conversely, the brook at Park Hill Road started to show signs of increased BOD and COD loadings by April of 1971 (Tables 2, 3 and 4, Site G, Fig. 2). By

Table 2. Water Quality Changes in Park Hill Brook (Easthampton, Mass)
Due to Leachate Addition Between Sampling Sites

COD mg/l			BOD ₅ mg/l								
Date Taken	Down Stream Site G	Up Stream Site A	Date Taken	Down Stream Site G	Up Stream Site A	Date Taken	Down Stream Site G	Up Stream Site A	Date Taken	Down Stream Site G	Up Stream Site A
10-23-69	-	-	7-17-73	219.0	30.3	10-23-69	2.4	1.9	7-17-73	137.0	1.2
11-4-69	-	-	8-14-73	84.9	17.2	11-4-69	1.0	.9	8-14-73	88.6	0
11-10-69	-	-	10-2-73	270.4	20.8	11-10-69	1.25	1.5	10-2-73	173.9	1.7
4-12-71	-	-	11-28-73	62.1	25.7	4-12-71	14.5	2.2	11-28-73	10.4	1.3
4-15-71	57.7	22.0	1-24-74	68.3	.17	4-15-61	24.4	.8	1-24-74	39.8	1.0
9-28-72	-	-	2-26-74	32.0	-	9-28-72	36.0	-	2-26-74	15.0	-
10-30-72	48.7	23.4	3-26-74	118.5	-	10-30-72	19.2	1.2	3-26-74	63.5	-
11-9-72	44.16	22.3	4-29-74	189.0	12.5	11-9-72	7.1	2.5	4-29-74	64.0	1.0
11-16-72	42.7	32.6	5-28-74	246.0	2.9	11-16-72	12.3	1.0	5-28-74	90.0	.8
11-21-72	70.6	19.22	7-9-74	127.0	-	11-21-72	20.9	1.0	7-9-74	56.0	-
12-14-72	52.0	48.0	8-27-74	303.0	-	12-14-72	26.0	.5	8-27-74	168.0	-
1-17-73	111.4	56.4	10-1-74	28.0	-	1-17-73	66.0	.7	10-1-74	1.0	-
2-22-73	99.0	34.6	12-10-74	47.0	20.0	2-22-73	54.0	.5	12-10-74	19.0	5.0
4-18-73	148.0	51.2	3-18-75	57.0	13.0	4-18-73	67.5	.2	3-18-75	55.0	1.8
5-24-73	178.5	27.7	9-11-75	114.0	19.0	5-24-73	79.9	.3	9-11-75	-	1.2
6-26-73	309.9	59.2	4-7-76	101.2	13.8	6-26-73	16.8	.9	4-7-76	63.0	2.8

Table 3. Water Quality Changes in Park Hill Brook (Easthampton, Mass)
Due to Leachate Addition Between Sampling Sites

Iron mg/l

Manganese mg/l

Iron mg/l			Manganese mg/l								
Date Taken	Down Stream Site G	Up Stream Site A	Date Taken	Down Stream Site G	Up Stream Site A	Date Taken	Down Stream Site G	Up Stream Site A	Date Taken	Down Stream Site G	Up Stream Site A
10-23-69	3.2	.10	7-17-73	10.1	.78	10-23-69	.48	-	7-17-73	2.96	.10
11-4-69	1.27	.17	8-14-73	110.5	.25	11-4-69	.12	.08	8-14-73	.82	.00
11-10-69	1.4	.32	10-2-73	10.5	.18	11-10-69	.32	.11	10-2-73	11.0	.02
4-12-71	2.52	.15	11-28-73	7.5	.62	4-12-71	-	-	11-28-73	.82	.07
4-15-71	3.58	.19	1-24-74	5.8	.40	4-15-71	-	-	1-24-74	1.9	.34
9-28-72	1.64	-	2-26-74	9.0	-	9-28-72	-	-	2-26-74	3.2	-
10-30-72	1.51	.44	3-26-74	8.5	-	10-30-72	-	-	3-26-74	3.0	-
11-9-72	1.65	.50	4-29-74	13.0	.10	11-9-72	.20	.00	4-29-74	5.7	.12
11-16-72	2.94	.30	5-28-74	13.0	.04	11-16-72	7.5	.00	5-28-74	5.4	.31
11-21-72	3.2	.44	7-9-74	7.3	-	11-21-72	1.55	.10	7-9-74	8.2	-
12-14-72	5.95	.44	8-27-74	3.4	-	12-14-72	.89	.02	8-27-74	12.0	-
1-17-73	8.95	.25	10-1-74	.58	-	1-17-73	3.68	.05	10-1-74	.60	-
2-22-73	10.0	.38	12-10-74	1.6	1.6	2-22-73	3.4	.08	12-10-74	1.2	.48
4-18-73	5.0	2.16	3-18-75	10	.5	4-18-73	4.65	.00	3-18-75	3.0	.45
5-24-73	10.6	.25	9-11-75	1.5	.5	5-24-73	4.2	.05	9-11-75	2.2	.10
6-26-73	17.2	.19	4-7-76	7.7	.24	6-26-73	7.8	.34	4-7-76	4.14	.05

Table 4. Water Quality Changes in Park Hill Brook (Easthampton, Mass)
Due to Leachate Addition Between Sampling Sites

Alkalinity mg/l						Dissolved Oxygen mg/l					
Date Taken	Down Stream Site G	Up Stream Site A	Date Taken	Down Stream Site G	Up Stream Site A	Date Taken	Down Stream Site G	Up Stream Site A	Date Taken	Down Stream Site G	Up Stream Site A
10-23-69	18	18	7-17-73	-	-	10-23-69	-	-	7-17-73	3.4	8.9
11-4-69	14	16	8-14-73	102.4	21.2	11-4-69	-	-	8-14-73	3.4	8.6
11-10-69	16	16	10-2-73	146.0	20.0	11-10-69	-	-	10-2-73	4.3	10.3
4-12-71	-	-	11-28-73	25.0	16.8	4-12-71	9.3	9.8	11-28-73	9.4	10.5
4-15-71	54	12	1-24-74	30.0	12.0	4-15-71	-	-	1-24-74	11.5	12.5
9-28-72	-	-	2-26-74	-	-	9-28-72	6.0	-	2-26-74	-	-
10-30-72	-	-	3-26-74	48.5	-	10-30-72	8.3	8.4	3-26-74	11.1	-
11-9-72	-	-	4-29-74	89	24	11-9-72	9.4	9.8	4-29-74	6.7	10.8
11-16-72	-	-	5-28-74	80	33	11-16-72	11.7	11.7	5-28-74	7.4	9.3
11-21-72	-	-	7-9-74	90	-	11-21-72	11.1	11.2	7-9-74	4.1	-
12-14-72	-	-	8-27-74	180.0	-	12-14-72	11.7	11.9	8-27-74	-	-
1-17-73	-	-	10-1-74	40	-	1-17-73	10.7	11.8	10-1-74	9.3	-
2-22-73	-	-	12-10-74	38	24	2-22-73	10.6	12.0	12-10-74	12.6	11.2
4-18-73	-	-	3-18-75	61	17	4-18-73	6.4	9.6	3-18-75	-	9.6
5-24-73	-	-	9-11-75	138.0	21	5-24-73	6.3	10.2	9-11-75	6.1	-
6-26-73	-	-	4-7-76	-	-	6-26-73	4.3	8.9	4-7-76	-	-

October of 1973, BOD values were approaching 200 mg/l and on occasion, COD values have exceeded 300 mg/l. Large fluctuations are explained by the repeated breaking of leachate dams in the leachate basin during periods of heavy rains.

In the Spring of 1975, the dams washed out for a final time. It was decided not to replace them, and since that time, leachates have run continually into Park Hill Brook.

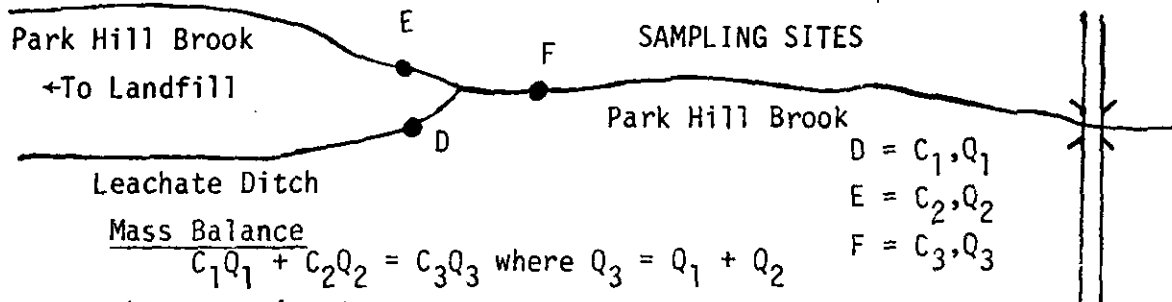
Leachate Analysis

With leachate flowing directly into Park Hill Brook on a continuous basis, concern was raised regarding the day-to-day loading of the stream. Town engineers constructed a weir and recorder in the leachate trench and daily flows were measured from July to September of 1975.

During the several months of data collection, flows usually averaged between 30,000-40,000 gallons per day (GPD). Several summer storms produced peak flows of 250,000-500,000 GPD. When flows exceeded 1 MGD, the weir washed out and was not replaced. This occurred on September 27th, and marked the end of flow recordings. Fortunately on September 11th, the Department of Environmental Quality Engineering at the University of Massachusetts Campus in Amherst collected an extensive series of samples in the leachate trench and in Park Hill Brook. The analyses of these samples, along with measured flow rates for that day, provide a useful means of determining typical leachate loading data for the Easthampton landfill. Unfortunately, this was the only sample series taken during the three month period of flow record keeping.

On September 11, 1975 leachate flow was measured at 30,000 GPD. At the weir, the COD was 476 mg/l. Several hundred feet downstream of the weir where the ditch spills into Park Hill Brook, the concentration of COD was measured at 382 mg/l. Using chloride and alkalinity mixing ratios at sample sites D, E, and F (see Table 5 and Figure 2) we can see that the flow rate in the brook prior to leachate addition was about 90,000 GPD and that the leachate added an additional 30,000 GPD for a total flow of $\approx 120,000$ GPD. Using a discharge COD of 400 mg/l and a flow of 30,000 GPD, we can estimate that ≈ 100 lbs/day of COD are added to Park Hill Brook during normal flow periods. With BOD_5 values averaging 60 percent of COD

Table 5. Flow Rate and Loadings Determination Based on Conservative Ion Concentration



Mass Balance

$$C_1 Q_1 + C_2 Q_2 = C_3 Q_3 \text{ where } Q_3 = Q_1 + Q_2$$

by arranging terms

$$\frac{C_1 - C_3}{C_3 - C_2} = \frac{Q_2}{Q_1} = \text{ratio of flows based on concentration ratios}$$

Typical Data

Date	Parameter	Concentration in mg/l			
		C_1	C_2	C_3	Q_2/Q_1
9-11-75	Alkalinity	375.0	36.0	125.0	2.8
9-11-75	Chloride	112.0	13.0	36.5	3.2
3-18-75	Chloride	95.5	9.0	24.0	4.73
3-18-75	Alkalinity	370.0	19.0	81.0	4.66
3-26-74	Alkalinity	208.0	11.0	60.0	3.02

Flow Data for 9-11-75

Leachate Flow from Recorder = 30,000 GPD

Estimated Flow at Site E based on $\frac{Q_2}{Q_1} \approx 3$

$$30,000(3) = 90,000 \text{ GPD @ E}$$

Estimated Flow at Site F

$$D + E = 90,000 + 30,000 = 120,000 \text{ GPD}$$

COD Loading for 9-11-75

COD 400 mg/l @ D 456.6 g = 1 lb
 Q_1 30,000 gal/day @ D .2642 gal = 1 l

$$\frac{30,000(.4)}{.2642(453.6)} \approx 100 \text{ lbs COD/Day}$$

values for leachate, we can estimate a BOD loading of 50-75 lbs/day.

It is worth noting that even though only one sampling occurred during flow measurement, samples taken for several years prior to September 11, 1975 compare quite favorably with the single sample values. If we assume a flow of 30,000 GPD to also be typical, the loadings of BOD and COD as well as other parameters can be calculated for any time during the life of the site when sampling was being done.

In considering loading values for the landfill, one must bear in mind that there is usually only a 3:1 dilution factor when leachates enter Park Hill Brook. Table 4 shows a considerable dissolved oxygen sag more than a mile downstream of the leachate addition. This is especially true in the warmer seasons of the year. Because BOD's are still very high at Site G it seems reasonable to assume that the depression of DO would continue several miles downstream.

Farmers once used the brook as a natural water supply for their cattle in pasture but now have fenced the area off to prevent the cows from drinking leachates.

Case 2: Amherst, Massachusetts

Historical Overview

Present landfill operations in Amherst are located west of Route 9, less than a mile from the intersection of Pelham, Belchertown, and Amherst. The fill, which has been in operation since 1971, is depicted in Figure 6. It is situated next to an older section used as a town dump prior to 1971 landfill regulations. Several well fields are located in the vicinity of the landfill. These municipal drinking water wells, and the threat imposed on them by landfill leachates, have been a point of concern for many authorities since the landfill's opening. The wells most subject to possible contamination are the Brick Yard Wells. Several monitoring wells were placed between the Brick Yard site and the landfill to detect possible leachate migration. Since February 7, 1973, the Department of Environmental Quality Engineering of the Commonwealth of Massachusetts at Amherst has been sampling and analyzing waters from these wells which are labelled A, B, and C in Figure 6. Their results are compiled chronologically by well letter in Appendix E of a previous paper by Lavigne (10).

Amherst is the only landfill in this study that utilized test wells for leachate monitoring. It is also the only landfill where surface

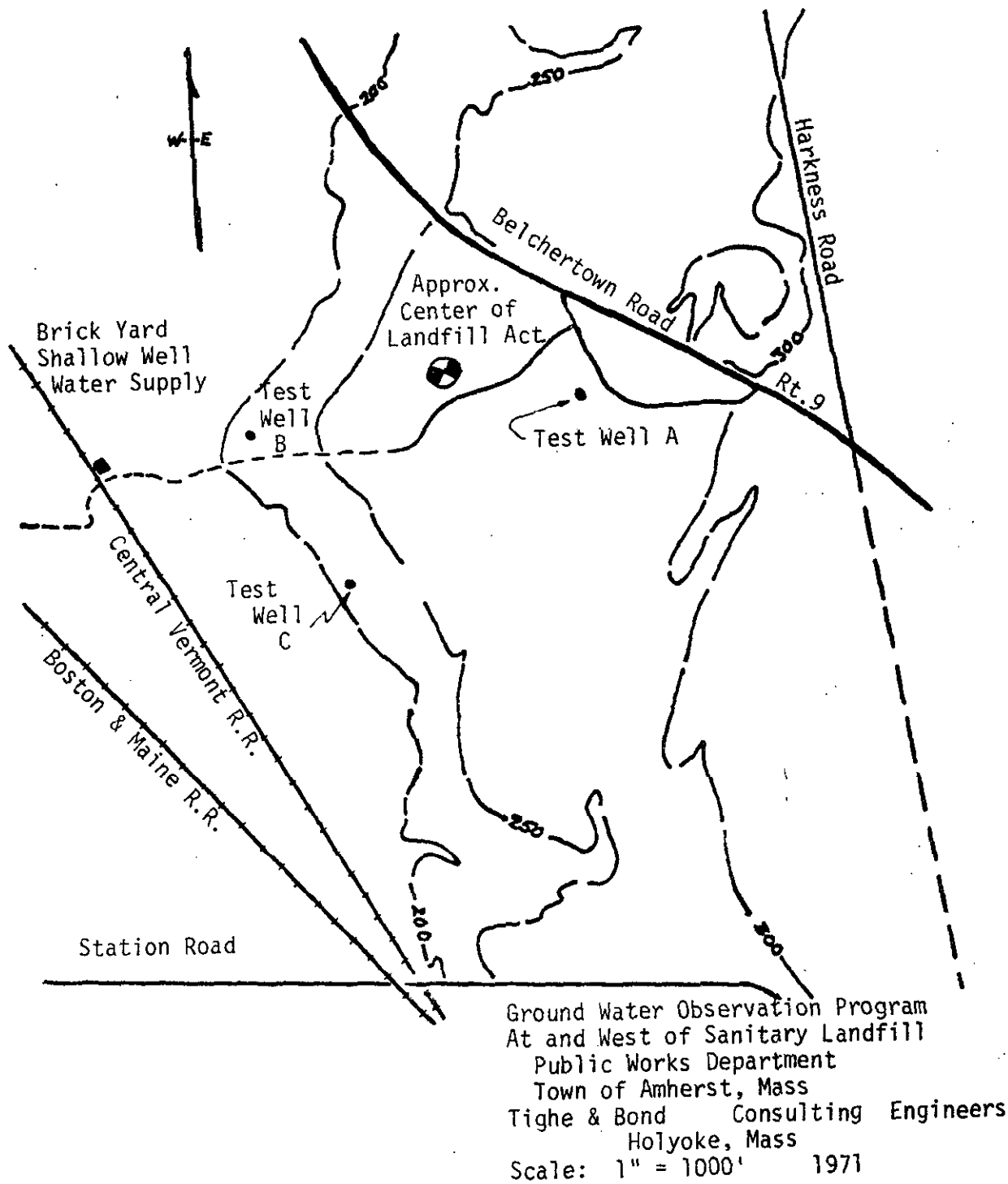


Figure 6. Amherst Landfill Showing Location of Brick Yard Wells and Test Wells A, B, and C.

pools of leachate have never been visible. In every respect, Amherst must be considered an exemplary landfill. Unfortunately, leachates are still being produced and their movement is evident from well data. Geologic, Topographic, and Hydrologic Considerations.

The landfill is located on high ground and infiltrating precipitation recharges the Lawrence Swamp area. The Brick Yard Wells are $\approx 100'$ down gradient of the fill site as shown in Fig. 6

The soil log for Test Well B shows many layers of sands and gravels to a depth of 107 feet. At this point, bedrock was encountered. Though seasonably variable, water level in the test well is usually 20-30 feet below ground surface. This means that there is $\approx 70'$ of saturation from the piezometric surface to the ledge. With this much water available there is little wonder that the Brick Yard Wells have been developed as extensively as they have.

Leachate Analysis and Data Presentation:

Test Well A is slightly up-grade of the fill area. There have been no significant changes in water quality parameters since sampling first started in February of 1973.

Test Well B is in the direct path of water flow from the landfill to the Brick Yard Wells. Unlike Well A, changes in water quality are evident here. In February of 1973, alkalinity and hardness were 25.4 and 44.0 mg/l respectively. By November of 1975, they had increased to 130.5 and 168 mg/l respectively. Iron concentrations in 1973 were 0.32 mg/l. By 1975 they had reached nearly 7 mg/l. Chlorides do not appear to have increased dramatically, but this could mean the chlorides front went by prior to the first sampling. Considering the conservative nature of chloride, this is very possible. It should be remembered that the landfill started operating in 1971. This theory is further supported by the lower chloride concentrations in Test Well A. Though not changing, chlorides in Well B are nearly twice that of Well A.

Test Well C, the farthest from the landfill and in the direction of the Lawrence Swamp wells, has also failed to show any signs of leachate movement in that direction.

The Brick Yard Wells have been sampled since 1968, with nearly 50 samples being taken over the seven year period.

Chlorides, alkalinity, and hardness have nearly doubled at the Brick Yard Wells since 1968. Though the values in and of themselves are not large, they may suggest that a leachate plume is approaching. The phenomenon of a "hardness halo" has been observed by many researchers in the field. A hardness front usually characterizes the leading edge of a leachate plume. The source of this hardness is calcium, which exchanges readily with leachate metals on soil particle surfaces. Once calcium is encountered, the breakthrough of metals is usually soon to follow. It is indeed unfortunate that heavy metals have not been analyzed for in the Brick Yard samples. The Brick Yard Wells are approximately 3000 feet from the landfill, and as such, the increase of iron at Test Well B during 1974 and 1975 may indicate that the metals front is half way to the Brick Yard area (note Test Well B is $\approx 1500'$ from the landfill and the well field).

Conclusions:

There seems little doubt that leachate is migrating from the Amherst Landfill. Estimates would suggest that the velocity and direction of travel of some parameters are $\approx 300'$ per year toward the Brick Yard Wells. If this is a reasonable approximation, increases in iron and similarly behaving cations would reach the drinking water wells by 1980. Though beyond the scope of this project, mathematical modeling might yield additional information regarding flow velocities of the leachate, and when it might be expected to reach the various pumping areas. Considerable work has been done in this area, and many researchers feel their models are now ready for general use (11,12,13,14).

From a more practical point of view discussions with Amherst's town engineer indicate that the town plans to abandon the Brick Yard Wells in the next few years. More recently, the town has decided to move its landfill, and to use a lined fill area at the new site. The type of membrane has yet to be determined, and there is still some question about what to do with the collected leachates.

Case #3: Athol, Massachusetts

Historical Overview

The Town of Athol has a current population of approximately 21,000 people. It is situated slightly northeast of Quabbin Reservoir, but only a small part of its surface water drainage enters this watershed area; Tully Brook and the Miller's River are two of the larger drainage basins for the town. L. S. Starrett Company and Union Twist Drill are the two largest industries.

The town's "landfill" is located some 3-5 miles north of the downtown section adjacent to Tully Brook. The site has served the town for many years, and dates back to the time when open dumps and burning were accepted practices. The lower face terminates in a swale, and is about 100 ft from the Brook.

Some 230 tons of waste are delivered to the site per week. With the newer landfill area having been in use for some ten years, it is estimated that more than a half million cubic yards of domestic, commercial, and industrial refuse are entombed along the bank of Tully Brook.

As might be expected, there was little difficulty in locating considerable amounts of leachate that were always present in open standing pools near the river. During periods of heavy ground and surface water flow these pools spill directly into Tully Brook (Figure 3).

Geologic, Topographic and Hydrologic Considerations.

Soils from the Athol fill area are typically fine sands, with virtually no clay or silt present. The work of Mahlock (15), Roulier (16) and Farquhar (17) would suggest that the absence of clays in Athol soil would minimize their adsorptive and ion exchange capabilities. This of course has resulted in an early breakthrough of metals and organics via ground water entering Tully Brook.

In spite of the leachate present around the site differences in upstream and downstream samples from Tully Brook were evident, but seldom large.

Mean flow in the Brook for the 1974-75 water year was 95.4 cfs. USGS data also indicate a total flow for the year of 34,816 SFD (second foot days). This is equivalent to \approx 22,500 MGY (million gallons per year).

Using Barre precipitation data for the same year, it is possible to estimate total rain and snow fall for the Athol landfill. Assuming that 48.8 inches of precipitation infiltrated to produce leachate, and that the landfill was at field capacity at the start of the water year (October 1), the maximum amount of leachate possible would be 13.25 MGY. With this amount of leachate entering Tully Brook, it would only amount to 0.06 percent of the brook's flow. It is felt that this is a major reason for the difficulty in detecting a gross contamination by the landfill of the stream. Athol's "solution to pollution is dilution"!

Leachate Analysis:

Heavy rains in February, June and July produced characteristic flushes of leachate, and corresponding changes in Tully Brook samples were evident.

During the months of June and July solids were considerably higher in the leachates from Site #4, and in the downstream brook samples (Site #1). This would indicate that leachate solids were flowing into the stream, and increasing downstream concentrations. Though BOD and COD values were always greater in the leachate samples than in the brook, Athol's leachate is still quite dilute. When one considers that the literature and data from Barre report BOD values greater than 10,000 mg/l for strong leachate, the values of 100-200 mg/l from Athol suggest extensive dilution from ground water. It is important to note here that we can dismiss the probability of better soils attenuation in Athol as the cause for our lower parameter measurements. This is documented by the fact that more conservative parameters (e.g. chlorides and alkalinity and sulfates) were also much more dilute in the Athol leachate samples. It is generally accepted that these materials are not attenuated appreciably by any of the soil types (18,19,20,21).

As with many of the parameters, iron also was usually higher in downstream samples, but concentrations of <10 mg/l must be interpreted as very low for leachate pollution. Chlorides showed the same behavior as iron with downstream values always higher.

Conclusion:

There can be little doubt that large amounts of leachate are being produced in the Athol Landfill. With much of the refuse remaining uncovered, and the failure to provide surface drainage control, virtually all of the precipitation falling on the site infiltrates.

From a public health point of view one is forced to question the wisdom of locating a municipal landfill just a few miles upstream of municipal water supply wells. The two wells in question are located on the banks of the Miller's River less than half a mile downstream of its confluence with Tully Brook. The total distance from the landfill to the wells is slightly more than two miles. With river velocities usually averaging more than two miles per hour, leachates can leave the landfill and travel to the wells in less than an hour's time. The town now plans to drill additional wells almost a mile closer to the landfill site. If reports of cyanide disposal in the landfill are valid, one must conclude that Athol's solid waste disposal facility poses a serious threat to the health of those Athol residents drinking municipal water. Should the proposed new wells be developed, the public health threat will be even greater.

It seems reasonable to conclude that the Athol Landfill should be relocated as soon as possible, and that a rigorous drinking water monitoring program should be undertaken immediately.

Case 4: Petersham, Massachusetts

Historical and Operational Overview

The Town of Petersham, has a population of approximately 1000 people. It is located on the Northeast edge of Quabbin Reservoir and is blessed with an abundance of ground and surface water resources. The community is primarily agricultural and residential, with little or no crowding of homes. There is no industry in Petersham and only a small commercial section.

Prior to the establishment of "sanitary landfill" regulations for the Commonwealth, Petersham disposed of its waste in an open burning dump that was located in a small recharge swamp of Quabbin Reservoir. Refuse is still dumped into the swamp, resulting in the slow but steady elimination of this wetlands area. Because the operation is contained in an area of approximately five acres, there has been very little objection raised by

the various state agencies concerned with either wetlands protection, solid waste disposal, or water quality.

Residents of Petersham deliver their waste to the site on an individual basis. Though records have never been kept, estimates would suggest that some 20,000-25,000 pounds per week are disposed of at the site. This is based on a daily per capita production of 3-3.5 lbs/day (8,22).

Cover material is placed on the refuse periodically but not on a daily or weekly basis.

Under present operating conditions, the dump is open 16 hours a week and residents are encouraged to separate glass at a small recycle shed at the entrance. The glass and bulky "white metals" are trucked out of town by independent recycle companies.

Geological, Topographic and Hydrologic Conditions

The Petersham Dump is located about 840 feet above sea level. The swamp drains to Brooks' Pond, and from there flows to Quabbin Reservoir at elevation 530 feet. The total travel distance from dump to reservoir is slightly less than six miles.

Soils in the area are composed primarily of gravel and sand, with many large cobbles. Fragipan layers (23) are also quite common, and probably account for many of the small perched water holes that dot the landscape. Based on the work of Tirsch, Jennings, and Lombardo (24,25,26) it seems reasonable to conclude that Petersham soils are probably ineffective in reducing leachate pollutants by adsorptive processes.

Leachate Characterization

In October of 1974 sampling began at the Petersham Dump in the small swamp situated at the lower face of the fill area. This swamp is actually two swamps connected by a narrow channel of water. Samples were collected from the sub-section nearest the refuse, and from a second point at the extreme end of property where water drains out through a storm culvert (see Figure 4). As with the two other towns under study in this area, storms in February and June produced considerable increases in almost all leachate parameters. Because of comparatively

lesser dilution from surface waters, Petersham leachates were much stronger than Athol's.

With an absence of flow data from the Petersham site, it is impossible to determine loading rates into the swamp or ground water next to the landfill. It is, however, evident from the data, that samples collected at the dump face (Site #1, Figure 4) were consistently of poorer quality than those taken from Site #2 (Figure 4), some 300 feet from the refuse fill area. Total and volatile solids, alkalinity, COD, sulfates, ammonia, iron, and chlorides showed the greatest change in concentrations as a result of the February flush. Two heavy storms in June produced a similar flush of BOD from the Petersham Dump (10).

Of considerable interest during the ten months of study was the continual reduction in concentrations for most parameters, as swamp water flowed from the dump face to the storm culvert at the road's edge. Associated with this were extensive algal blooms that often covered the entire surface of the swamp water. Other aquatic plants, such as the common "duck weed" (lemna), also seemed to thrive in the presence of leachate enriched waters. These field observations have stimulated subsequent investigations into the area of leachate-algae interaction. Laboratory work by Walker (27), and field studies at the Barre Research Landfill have indicated that algae may provide an excellent "living filter" for solid waste leachates.

Conclusions

The Petersham Dump seems to generate a minimum of environmental contamination. Biological activity within the swamp itself seem to be playing a major role in attenuating leachate. If Petersham can be considered as a realistic type of solid waste disposal facility, there may be merit in considering wetland areas as a possible sink for solid waste leachates. It should be remembered, however, that many pollution parameters associated with leachate in the past have not been considered in this study. Should conservative materials constitute a major part of the original leachate, they might well find their way into the Metropolitan District Commission's water supply.

Case #5: Barre, Massachusetts

Historical Overview

Martone Landfill presently handles the solid waste from seven communities, and several small industries. The Town of Barre with a population of 4000 accounts for about 50 percent of the domestic waste being landfilled. The other seven communities contribute wastes from another 4000 residents, so that the total effective population being served by the facility is about 8000. Mr. Martone estimates that he is receiving about 100 tons/week of solid waste for disposal.

The landfill is relatively new, having gone into operation during the summer of 1970. Being a private landfill, there has been a more than average surveillance by local and state officials at the site. In spite of this Mr. Martone has been able to operate his business profitably, and with a minimum number of confrontations by officials.

In June of 1976 the fill area used for this study was closed and no longer receives refuse. Since that time refuse has been placed in a newly lined fill area, and leachates are being collected and treated in algal lagoons.

Geologic, Topographic, and Hydrologic Considerations

Barre's soils are quite variable in physical structure. Gravel and sand predominate, with layers of clay and hard pan supporting many small perched water tables. Mr. Martone's operational scheme usually involves the stripping and stockpiling of clay soils, while using the sands and gravels for daily cover material.

The site has been cut into a relatively steep ridge on the south bank of the Ware River (Figure 5). Over a million cubic yards of stone and gravel were removed from the site prior to its sale for landfill purposes. Because of the porous nature of the cover material used, it seems reasonable to assume that virtually all of the precipitation falling on the fill area proper is transported vertically into the refuse. If this is a reasonable assumption, then there will be approximately 1 million gallons per year per acre of water available for leachate production. Once field capacity is reached, there would be a potential mean flow of

2800 gallons/day/acre of leachate. It is interesting to note that if one compares these hypothetical values to several field sites actually measured, there is reasonable agreement of values. Easthampton, Massachusetts has measured average flows of about 30,000 gal/day of leachate. With a landfill area of approximately 10 acres, this flow agrees quite closely with those theorized. At the Bucks County, Pennsylvania, landfill owned by Waste Resources Corporation the activated sludge leachate treatment plant is processing about 30,000 gallons per day. This landfill is also about 10 acres in size.

During the period of this research project (October 1974-August 1975), the old fill area was in active use, and it was generating considerable leachate flows at its working face. Figure 5 attempts to illustrate the large pit or pool of standing leachate at the face of the fill. Rapid increases in the elevation of this pool after storms, and the subsequent drop to normal levels within a few days, strongly suggested that large amounts of leachate were entering the soil system in the area.

Leachate Analysis and Data Presentation

The Martone Landfill in Barre, was selected as a study site because of its proximity to the Ware River. This landfill, more than any of the others, was subjected to an extensive sampling program for nearly a year. Samples were collected from above and below the landfill in the Ware River. Periodically samples were collected from Powder Mill Pond which is the section of river impounded by Barre Wool Dam.

Within the landfill itself, five sampling stations were selected along the brook at sites before, during, and after its traverse of the fill area. So that actual concentrations of leachate could be measured, two additional samples were regularly taken from the standing leachate pool. The Ware River flows by the Martone Landfill with little or no change in water quality. The entire reach of the river has been identified as Class B by the Massachusetts Division of Water Pollution Control. It should be noted however, that as with Athol, a considerable dilution of leachates could be the reason for the absence of measurable leachate pollution. The rapid change of leachate pool levels in the landfill

strongly suggests that they are flowing toward the river. As stated earlier, the short life of the fill area, and the natural attenuation capacities of the separating soils may be holding the leachate front at some point back from the river. If leachate flows had been allowed to continue, there is a good chance that breakthrough into the river would eventually have occurred. For this reason Martone closed this section of the landfill, and is now collecting and treating leachate in a new section.

Within the landfill proper, there was no difficulty in locating an abundance of highly contaminated leachate. As illustrated in Figure 1, steady streams of the brightly colored material ran out of the face at various lift levels and collected in a pool at the toe.

Total solids at the landfill face were always in the tens of grams per liter. In moving from site #8 to #7, there was always a considerable reduction in total solids concentration. Site #7 solids never exceeded 400 mg/l. In comparing the two river sampling stations #1 and #2, it would be difficult to suggest that there were any upstream-downstream differences in solids concentration. As with Petersham and Athol, Barre leachate also responded to heavy rains in February, April and June. The June storm pushed total solids concentrations to 80,000 mg/l in the landfill face leachate.

BOD and COD values of leachate from Barre were always high. Concentrations of 10,000 and 20,000 mg/l respectively, were not uncommon. Slight increases in downstream concentration of COD in the Ware River were measurable during periods of heavy rain.

Sulfates in samples from sites #7 and #8 were always greater than Ware River concentrations. This parameter also shows rapid increases in concentration after storms. The melt and storm in February pushed concentrations to 1200 mg/l in the leachate. River concentration never exceeded 10 mg/l and upstream downstream differences were extremely small.

During the year of sampling, dissolved oxygen measurements and temperature behaved normally. Strong leachates from sampling site #8 were always septic, except for the "unique" month of April. Samples during April from Sites #7 and #8 had a distinct green color to them. Upon microscopic examination, it was learned that dense populations of Euglena-like algae were thriving in this vile and putrid material. As a result of their photosynthetic activity considerable amounts of dissolved oxygen were present in both samples. From this phenomenon it was theorized that algae or their metabolic activities might provide an inexpensive method of leachate aeration and treatment. This theory has been utilized in the new leachate treatment facility at the Martone Landfill. Given adequate time, the process is capable of renovating leachate with a 99 percent treatment efficiency.

The production of iron in landfill leachates is evident when one compares ambient river concentrations to those found in samples from sites #7 and #8. Iron was seldom found to be less than 1000 mg/ℓ in strong leachate samples. Values as high as 5000 mg/ℓ have been measured. In spite of these high values, iron concentrations in the Ware River were always less than 1 mg/ℓ.

Coliform bacteria measurements were always unpredictable in leachates. Difficulties in measuring coliform bacteria have also been reported by other investigators (28, 29). During certain stages of flow, especially the early ones when concentrations of volatile organics are high, the leachate seems to be toxic or it seems to inhibit the growth of coliform bacteria. At other times counts reach exponential proportions. Some of the problems associated with coliform measurement in leachates have been investigated more thoroughly by Cooper (4) and Peterson et al. (30).

Work is presently underway at the University of Massachusetts to investigate more thoroughly the amounts of coliform and other pathogenic organisms that may be present in landfill leachate (31,32).

Chloride is perhaps the most mobile ion found in leachates. In some studies, chlorides have been found to greatly exceed 1000 mg/ℓ (41, 42,43). Evidently, domestic wastes do not contribute heavily to chloride production. During February, chlorides in Barre leachate reached 1000 mg/ℓ. More often they were less than 500 mg/ℓ. As with

hardness the conservative nature of chloride is often used as the early warning of an advancing leachate front.

Conclusions

Without a doubt, leachate concentrations measured in Barre were the highest of the three towns studied in this area of Massachusetts. It is interesting to note that this landfill site is capable of meeting all state regulations regarding land disposal by sanitary landfilling.

It is indeed fortunate that Mr. Martone had the insight to suspend normal operations, and to develop a catchment and treatment fill area.

SUMMARY

Though most of the disposal facilities considered in this study are operated as "sanitary landfills", the leachate being produced by them will most likely result in their being re-classified as "open dumps" under the new "Resource Conservation and Recovery Act of 1976 (PL 94-580) (33). It also seems reasonable to conclude, that in New England, where 40 or more inches of rain fall annually, leachate production will continue to be a way of life.

Most landfill codes include provisions to minimize infiltration of rainwater, but this works better in theory than in practice. Many landfill sites lack nonporous materials to use for cover, and those that do have it usually avoid its use because of the difficulties of working with it in wet weather.

Though biologically treatable, practical experience has shown leachate to be toxic to activated sludge organisms (34).

A more promising alternative to activated sludge treatment is being operated in Barre, Massachusetts at the Martone Landfill. Collected leachates are treated in gravity flow lagoons, using algae and aquatic plants for aeration purposes and heavy metals uptake. Early indications are that treatment efficiencies exceed 99 percent and are safe for discharge into a small trout pond. The advantages of this system are: low capital costs, no power costs, and a minimum of operational care. There are limitations to this type system where land is in limited supply or overly expensive. Approximately one-third of the landfill area must be set aside for collection and treatment of leachates. In small New England towns like Barre, these land requirements are rarely a problem

and the system is capable of treating leachate wastes in a cost-effective manner. A peripheral benefit of this type of system is that many more sites may be considered for landfill use when it is known ahead of time that leachates will be collected and treated.

In any case, present landfill regulations in Massachusetts are not an effective deterrent to surface and ground water pollution from leachates. The requirement of only four feet of separation between refuse and high water table is totally inadequate for the control of leachate. While some components of leachate adsorb or exchange well on clay soils, many more move with the flow and are attenuated by the forces of diffusion and dilution only. Lead and PCB's are typical of those materials that adsorb well. Chromium, selenium, and arsenic, however, show very poor attenuation in soils. Other materials like zinc, cadmium, copper and mercury are greatly affected by soil pH. This means that they may be adsorbed or exchanged at one pH, and then released at another.

In conclusion, there is a strong indication that the sanitary landfilling of wastes in areas receiving 40 or more inches of rain yearly is not an effective method of waste disposal. If land disposal is to be used in areas where waste recovery is inefficient or uneconomical then leachate collection and treatment should be required.

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OVERVIEW OF KEY LAND DISPOSAL PROVISIONS OF THE
RESOURCE CONSERVATION AND RECOVERY ACT OF 1976

Ira W. Leighton, Sanitary Engineer
Air and Hazardous Materials Division
U. S. Environmental Protection Agency
Region I, Boston, Massachusetts

RCRA PROVISIONS IMPACTING ON LAND DISPOSAL
OF NON-HAZARDOUS WASTES

On October 21, 1976, the Resource Conservation and Recovery Act of 1976 (P. L. 94-580), was signed into law amending the Solid Waste Disposal Act (P. L. 89-272). Section 1008(a) (3) of the Act authorizes EPA to publish minimum criteria to be used by the States to define what constitutes open dumping of solid or hazardous wastes. Section 4004(a) of the Act authorizes EPA to promulgate regulations for determining what constitutes an open dump ("Criteria for the Classification of Solid Waste Disposal Facilities"). After such classifications are made, an inventory of facilities will be made pursuant to Sec. 4005(b) of the Act. All facilities which do not meet the above "Criteria" will either be closed or upgraded by the States pursuant to Sections 4003(2) and (3) and 4005(c).

The thrust of these sections of the Act is to ensure that adverse effects upon air, land, and water resources, and public health, safety, and nuisance problems are prevented or minimized. Human health dangers resulting from improper waste disposal (i.e., open dumping) and consideration for the increasing amounts of pollution control residuals destined for land disposal as a result of the Clean Air Act, Water Pollution Control Act, and other Federal and State laws are also concerns of Sections 1008(a) (3) and 4004(a).

The Act has as its goals the regulation and control of wastes and waste disposal operations by prohibiting future open dumping on land and requiring upgrading or closing of existing open dumps. This concept defines "environmentally sound" solid waste management practices and, in effect, plugs loopholes left by other environmentally-oriented Federal Laws and regulations. As the criteria under consideration are promulgated and guidelines are issued, the States will have the responsibility to close down or force upgrading of existing open dumps.

The specific adverse effects of environmentally unsound disposal facilities can be grouped into seven categories of impacts: environmentally sensitive areas, air, water (ground and surface), soil, disease vectors, safety, and aesthetics. In essence, the "Criteria" under consideration delineate both performance and operational requirements to eliminate a "reasonable chance of adverse effects on health and environment from disposal of discarded material" at a waste disposal site.

For purposes of this conference the focus will be on those provisions of the "Criteria" likely to influence the use of leachate treatment and collection systems, namely environmentally sensitive areas (wetland and flood plain siting), surface water, and ground water impacts.

Environmentally Sensitive Areas

Environmentally sensitive areas in the Criteria are wetlands, floodplains, permafrost areas, critical habitats of endangered species, and recharge zones of sole source aquifers. These areas were selected for coverage because EPA feels these areas are natural assets (have beneficial qualities) which are not adequately protected by the other six criteria.

Wetlands

In the past, solid waste disposal sites have frequently been located in wetland areas, generally because there has been less public resistance to locating them there, the land is cheaper, and when completed the land is often sold or used for a direct economic purpose.

The alteration and destruction of the wetland resources through draining, dredging, landfilling and other means has had an adverse cumulative impact on wetlands and other aquatic resources. Recent estimates indicate over 40 percent of the 120 million acres of wetlands in the United States that existed 200 years ago have been irrevocably destroyed.

The intent of the criterion is to prevent the destruction of wetlands. New disposal sites may not be placed in wetlands and existing operations may not be continued in wetlands unless an NPDES permit has been obtained (under Section 402 of the Federal Water Pollution Control Act (FWPCA) Amendments of 1972 (Pub. L. 92-500, 86 Stat. 880, 33 U.S.C. 1342)), and if a levee, dike, or other type of containment structure is to be placed in the water as part of the disposal activity, an Army Corps of Engineers permit has been obtained (under section 404 of the FWPCA) according to 33 CFR 320-329. There is a strong presumption against the issuance of an NPDES permit for the discharge of solid waste into wetland areas. Only upon a showing of extraordinary circumstances--including a demonstration of alternative methods of disposal, an assessment of environmental impact for each alternative, an assessment of the technical and economic feasibility of each alternative, and a justification for the wetlands disposal alternative in view of the environmental impact and feasibility--will an NPDES application be considered and an NPDES permit issued. Any NPDES permit issued for the discharge of solid waste into wetland areas must assure that the facility utilizes appropriate technologies and/or best management practices to minimize any adverse effects.

This approach conforms with the intent of Executive Order 11990 dated May 24, 1977, concerning Protection of Wetlands.

Floodplains

Disposal of solid waste in floodplains may have several significant adverse impacts: (1) if not adequately protected from flooding, wastes in a disposal site may be inundated by water and flow from the site, affecting downstream waters; (2) since floodplains generally have direct hydraulic connection to wetlands, surface water, and ground water,

location of disposal sites in floodplains may result in leachate contamination of ground water; (3) filling in the floodplain may restrict the flow of flood waters, causing greater flooding upstream; and (4) filling in the floodplain may reduce the size and effectiveness of the flood-flow retaining capacity of the floodplain which may cause a more rapid movement of flood waters downstream, resulting in higher flood levels and greater flood damages downstream.

For purposes of these "Criteria" the floodplain is defined by the 100 year flood level. This level is considered adequate to minimize the chances for site inundation and increased flood levels and damages. This level is considered conservative in many parts of the country because construction activities (buildings, roads, storm sewers, etc.) continue to increase runoff, thereby increasing flood levels for similar precipitation incidents. The 100 year floodplain has been mapped for many areas of the country by the U.S. Geological Survey, Army Corps of Engineers, and the Department of Housing and Urban Development.

The intent of this criterion is: (1) to require an assessment of any new disposal site or expansion of any existing site in a floodplain to determine the potential impact of the disposal site on downstream and upstream waters and land, (2) to prohibit such disposal activities if the site as designed may cause increased flooding during the base flood, and (3) if the disposal site is located in a floodplain, to require the use of available technologies and methods to protect against inundation by the base flood and minimize potential for adverse effects on water quality and on the flood-flow capacity of the floodplains.

Surface Water

The criteria seeks to help achieve the objective of the Federal Water Pollution Control Act Amendments of 1972 (Pub. L. 92-500) of restoring and maintaining the integrity of the surface waters of the United States. Accordingly, all point source discharges of pollutants, including surface runoff, surface leachate, or leachate treatment effluent, must comply with an NPDES permit issued for the facility according to Section 402 of Pub. L. 92-500.

The criterion also requires, where possible, the prevention of direct discharges into surface waters of non-point sources of pollutants (unchannelled leachate seepage and surface runoff which may contain leachate, waste materials, or erosion sediment). Non-point source discharges should be prevented or minimized through facility design, operation, and maintenance, (e.g. by artificial or natural barriers, liners, or dikes,) and by collection of such waters if produced (e.g. by ditch or trench). When collected, such waters become a point source which requires an NPDES permit if discharged to off-site surface waters. Flow of surface runoff from adjacent and surrounding lands should be channelled away from the disposal site to avoid contamination.

Ground Water

Ground water is often a high quality, low cost, readily available source of drinking water. At least one-half of the population of the United States depends upon ground water as a source of drinking water. Approximately ninety-seven percent of the Nation's water resource (excluding the oceans) is ground water. In many regions, ground water is the only economic and high quality water source available. In others, ground water can be developed at a fraction of the cost of surface water. Ground water in aquifers across the nation is generally suitable for human consumption with little or no treatment necessary. Some large cities rely exclusively on ground water for drinking water.

Ground water has been contaminated by solid waste disposal facilities on a local basis in many parts of the nation and on a regional basis in some heavily populated and industrialized areas, precluding its use as drinking water. Serious local economic problems have occurred because of the loss of ground-water supplies. The degree of contamination ranges from a slight degradation of natural quality to the presence of toxic concentrations of substances. Effective monitoring of potential sources of ground-water contamination is almost nonexistent, and many known instances of contamination have been discovered only after ground-water users have been affected.

Proper site location (including avoidance of aquifer recharge zones), ground-water and land use planning, and proper design, construction, operation and maintenance of facilities are the principal techniques available for minimizing potential ground-water contamination problems. Where economics or other factors dictate that sites be located in areas of critical ground-water use (such as existing or potential drinking water sources), such technology as physical containment (liners), collection, and treatment of leachate may be necessary.

The ground-water criterion seeks to protect current users of the ground water and to protect other designated ground water for future use. Of primary concern is protection of current and future ground-water drinking water supplies.

The criterion uses the "endangerment" approach proposed for the Underground Injection Control Program (41 FR 36726) which prohibits contamination that would require additional treatment of current or future drinking water supplies or otherwise makes the water unfit for human consumption.

The ground-water criterion provides for application of "endangerment" at the property boundary of the disposal site. However, prevention of contamination of ground water within or under the site is often the only means to effectively achieve this goal at the property boundary. Monitoring ground-water quality at the "waste boundary" (or within or under the site) may be desirable in order to anticipate potential "endangerment" at the property boundary and to measure effectiveness of control technology. Monitoring only at the property boundary may not provide ample opportunity for appropriate corrective actions because of time, economic, and technical constraints. Extending the property boundary would only postpone and aggravate the problem and would evade the intent of this criterion.

Under the ground-water criterion it is necessary to assess the impact of disposal facilities relative to the current and planned future utilization of the ground water. Utilization is divided into two categories: Case I addresses ground water currently used or designated for use as

drinking water supplies or undesignated water containing 10,000 mg/l total dissolved solids or less; and Case II addresses ground water designated for other uses. Thus, the criterion seeks to recognize and encourage definitive ground-water planning decisions at the State and local levels.

In the case of ground water currently used or designated (planned) for use as a drinking water supply, the quality of the ground water beyond the disposal facility is to be maintained for that use. That is, the disposal facility is not to "endanger" the ground water beyond the property boundary.

In certain situations, conscientious resource management and societal needs may dictate that ground water be maintained at a quality either higher or lower than that provided by the "endangerment" concept which is based on the water quality needed when the water is used for drinking purposes. Such resource management decisions are appropriate at the State and local planning levels and should include participation of the public (e.g. public notice and hearings), involving the users of both the ground water and the facilities which may affect ground-water quality. Thus, if after specific determinations states designate ground water for uses other than drinking water, they should establish the quality at which the ground water is to be maintained consistent with the designated use. Consideration must be given to the finality of the designation and the potential impact on other water resources. The impact of a disposal facility would then be assessed against that quality specified by the State.

In order to predict, as early as possible, the potential for ground-water endangerment or the impact on ground-water quality, the disposal facility should be monitored so as to indicate the movement of contaminants from the disposal facility into the ground water. Contingency plans should be formulated for corrective actions to be taken in the event that an adverse impact is indicated by the monitoring.

RCRA PROVISIONS IMPACTING ON THE
LAND DISPOSAL OF HAZARDOUS WASTE

Subtitle C of the Solid Waste Disposal Act, as amended by the Resource Conservation and Recovery Act of 1976 (Pub. L. 94-580), creates a regulatory framework to control hazardous waste. Congress has found that such waste presents "special dangers to health and requires a greater degree of regulation than does non-hazardous solid waste" (Section 1002(b) 95). Because of the seriousness of this waste problem, Congress intended that the States develop programs to control it. In the event that States do not choose to operate this program, EPA is mandated to do so.

Section 3004 addresses standards affecting owners and operators of hazardous waste storage, treatment, and disposal facilities. These standards define the levels of environmental protection to be achieved by these facilities and provide the criteria against which EPA (or State) officials will measure applications for permits. Facilities on a generator's property as well as off-site facilities are covered by these regulations and do require permits. The leachate controls required for land disposal facilities accepting hazardous wastes are likely to be more stringent than the Section 4004 Criteria in the following areas:

1. Ground water protection requirements may necessitate design, location, construction and operational approaches for hazardous waste facilities which will prevent any discharge leachate to the groundwater.
2. Ground water and leachate monitoring will be required in the zone of aeration, and zone of saturation.
3. The operator and owner of a hazardous waste disposal facility will be required to formulate a contingency plan for corrective action in the event there is a discharge of hazardous materials to the surrounding air, surface, or subsurface environments.

The Section 4004 Criteria were published as proposed regulations in the February 6, 1978 Federal Register and should be finalized some time in the summer of 1978. The Section 3004 Regulations should appear as proposed regulations during April of 1978 and should be finalized during the fall of 1978. Table 1 provides a list of key EPA contacts involved in various aspects of land disposal.

December, 1981I. PROGRAM CONTACTS

Dennis Gagne US EPA Solid Waste Program Room 1903, J.F. Kennedy Bldg. Boston, MA 02203 (617)223-5755	Coordination of Region I Implementation of Land Disposal Provisions
Ken Shuster U. S. EPA Office of Solid Waste 401 M Street, S.W. Washington, D. C. 20460 (202)755-9125	Disposal Program Land Disposal Division
George Garland (202)755-9190	Open Dump Inventory
Barry Stoll (202)755-9116	Landfill Guidelines
Bruce Weddle (202)755-9107	Landspreading Guidelines

II. RESEARCH CONTACTS

Ron Hill Solid & Hazardous Waste Research Division Municipal Environmental Research Laboratory Cincinnati, Ohio (513)684-7861	Disposal Site Design & Operation
Mike Roulrier (513)684-7871	Pollutant Transport (Migration, Attenuation)
Robert Landreth (513)684-7871	Landspreading, Deep Well Injection
Dan Sanning (513)684-7871	Test Methods Development

LEACHATE ATTENUATION BY SOILS

Franklin S. Tirsch, Research Assistant
Environmental Engineering Program
Department of Civil Engineering
University of Massachusetts, Amherst

INTRODUCTION

In his review of the literature on groundwater pollution and sanitary landfills, Zanoni (1972) makes reference to "the almost miraculous capability of most soils to attenuate the leachates generated from sanitary landfills." It has generally been assumed that a few feet separation of refuse from underlying groundwater is sufficient to prevent water quality problems. This assumption is reflected in the present Massachusetts regulation requiring a minimum distance of four feet of earth from the lowest point of refuse deposition to the maximum groundwater elevation at any portion of the site during the highest groundwater period of the year.

Is four feet sufficient for attenuation? What are the mechanisms of attenuation? What methodology should be used to estimate the attenuative capacity of the soil? The remainder of this paper will address these questions.

LEACHATE GENERATION AND MOVEMENT

The problem of leachate generation and movement is summarized in Figure 1. Precipitation infiltrates and percolates through the landfill and becomes highly contaminated. Some of the leachate may "break out" through the sides of the cells and collect in surface pools open to the atmosphere. Most of the leachate, however, penetrates downwards through the unsaturated zone and into the groundwater of the saturated zone. Because of the highly reducing conditions in the landfill, this leachate is anaerobic. The difficulties in maintaining anaerobic conditions during collection, analysis, and experimentation have been discussed by Korte, Niebla, and Fuller (1976), Griffin et al. (1976), and Chian and DeWalle (1976).

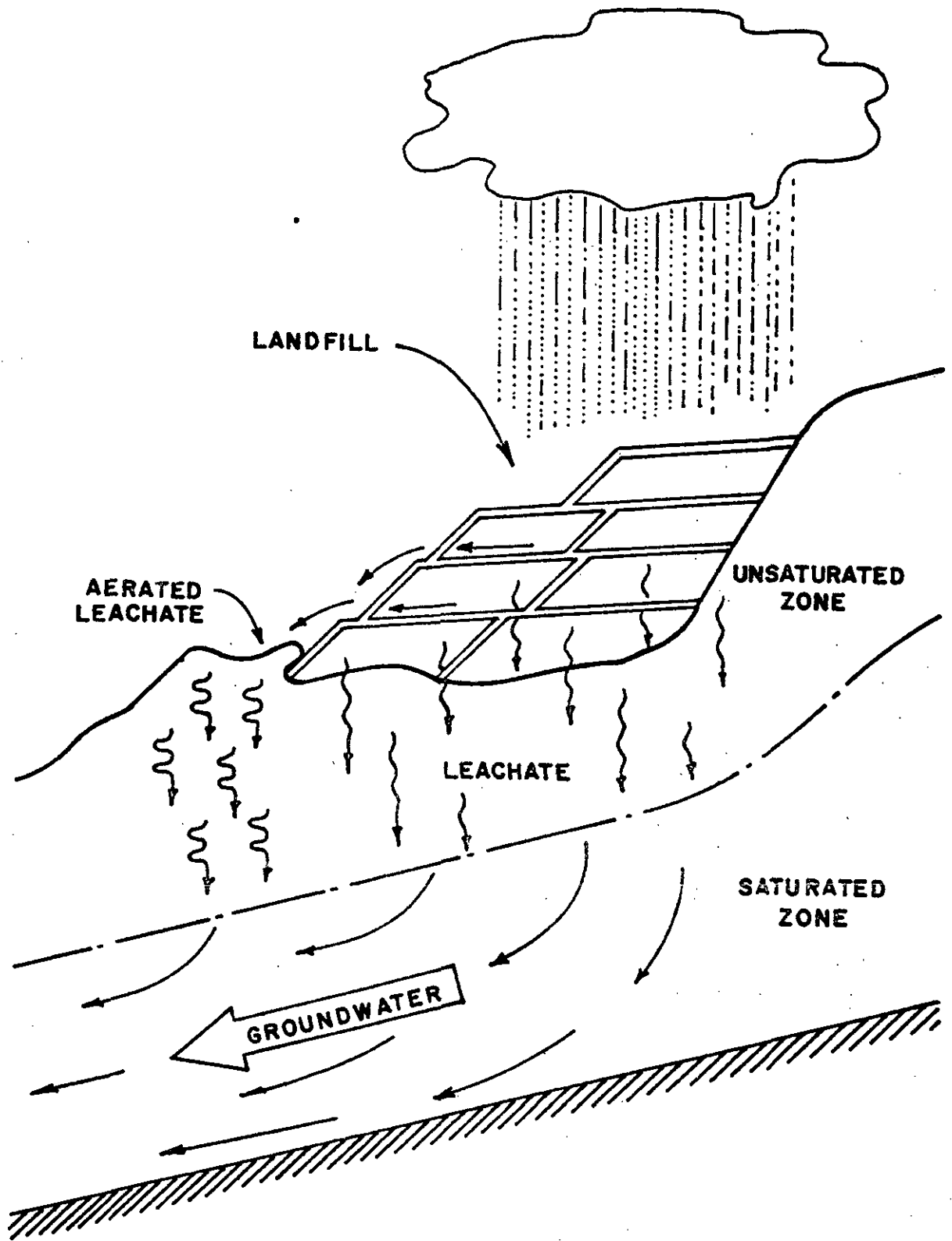


FIGURE 1
LEACHATE PATHWAYS

MECHANISMS OF ATTENUATION

Leachate may be attenuated in soils by chemical, physical and biological means. These mechanisms have been discussed in detail by Fuller and Korte (1976) and Philips and Nathwani (1976) and will only be discussed briefly here.

According to Philips and Nathwani (1976) adsorption is probably the most important process attenuating both organic chemicals and trace and heavy metals in soils. It is a physical chemical process that holds or immobilizes contaminants migrating in the soil system and is dependent on many factors including: character of the adsorbent, character of the absorbate, pH, and temperature.

Cation and anion exchange may be responsible for the replacement of more harmful constituents with more common ions. However, it must be noted that "removed" constituents may be available for re-exchange later on and for this reason Fuller and Korte (1976) and Philips and Nathwani (1976) consider ion exchange to be a temporary or transitory mechanism of attenuation. It should also be noted that ion exchange reactions do not effectively lower the total soluble salt concentration. Only a readjustment between solid and solution phases occurs. There are organic substances in the leachate as well as the soil and complexation reactions involving these organics play an important role in attenuation.

Precipitation reactions and pH are important in attenuation. Precipitates are considered as a well-defined separate phase but in soils it is sometimes difficult to distinguish between what is an adsorption reaction and what is a precipitation reaction.

Oxidation-reduction reactions are significant because reduced forms of many substances are generally more soluble. These reactions may be chemical oxidations and reductions or they may be mediated by microorganisms. Microorganisms are also responsible for mineralization and immobilization of constituents and for changes in organic constituents.

Philips and Nathwani (1976) point out that a clear-cut distinction between physical and chemical mechanisms of attenuation is not always possible. Particle size distribution, pore size distribution, moisture

relations, and temperature effects are several physical factors that play a role in attenuation. The importance of physical filtration may be questioned because many of the constituents are in their most soluble form when in the reduced state.

METHODOLOGY FOR STUDYING LEACHATE REACTIONS WITH SOILS

The investigation of leachate reactions with soils must be conducted in several steps including:

- (1) chemical characterization of the leachate
- (2) physical and chemical characterization of the soil
- (3) characterization of soil-leachate interactions
 - (a) batch equilibrium studies
 - (b) dynamic column studies
 - (c) field studies

The following sections will discuss these steps using examples selected from the study of three natural leachates and two natural soils collected at the Martone Landfill in Barre, Massachusetts.

Chemical Characterization of Leachate

Leachate analysis is a difficult matter (and is the subject of a later presentation). Special modifications to standard analytical techniques such as those described by Chian and DeWalle (1975), Environment Canada (1975), and Pease (1975) must be made. The Barre leachates were analyzed for alkalinity, NH_4^+ , COD, Cl^- , Fe, Mn, Ca, Mg, Cu, Zn, NO_3^- , pH, PO_4^{3-} , SO_4^{2-} , total carbon, inorganic carbon, and specific conductance. In addition, microorganisms were enumerated using agar plate techniques. The results of these analyses are summarized in Table 1. (It is recommended that ORP and total sulfur measurements be included in any leachate characterization.)

Physical-Chemical Characterization of the Soil

The Barre sand and Barre clay soils used in this study were analyzed for particle size distribution, organic content, cation exchange capacity, exchangeable cations (Ca, Mg, K), exchangeable acidity, pH, and manganese

TABLE 1
 Characterization of Leachate Samples

Parameter	Batch "A"	Batch "B"	Batch "C"
Nature	Winter Leachate	Summer Leachate	Summer Leachate
Color	Clear straw yellow	Clear pale yellow	Clear Green-Yellow
Odor	Intense	Moderate	Moderate
Specific conductance ($\mu\text{MHOS cm}^{-1}$)	>18,000	11,800	ND
pH	5.50	6.00	6.22
Alk (mg/l as CaCO_3)	2,100	1,850	4,150
COD (mg/l)	11,100	7,700	13,534
TOC (mgC/l)	ND	2,700	4675
NH_3 (mgN/l)	225	175	378
SO_4 (mg/l)	128	60.5	ND
NO_3 (mg/l)	2.60	ND	ND
PO_4 (mg/l)	0.18	ND	ND
Cl (mg/l)	ND	450	ND
Fe (mg/l)	1,020	362	1095
Mn (mg/l)	32.5	16.2	22.2
Ca (mg/l)	680	480	776
Mg (mg/l)	173	133	117
Cu (mg/l)	2.65	-	-
Zn (mg/l)	0.71	1.00	-
Total Plate Count (Organisms/ml)	60,000	200,000	ND
Source	Surface Pool	Surface Pool	Super Funnel

ND = Not Determined.

coatings. The results of these analyses are summarized in Table 2.

Characterization of Soil-Leachate Interactions

Batch Soil Equilibrations

One method for investigating soil-leachate interactions is the use of batch equilibrations. Two approaches are shown in Figure 2.

The first approach utilizes a series of dilutions of leachate in order to obtain a wide concentration range for the constituents of interest. A series of reactors of equal volume, V , containing the same weight of soil, W , are prepared. The diluted leachate is added to each reactor and the contents are equilibrated at constant temperature. After equilibration the sample is filtered or centrifuged to remove the fine soil particles and the concentrations of the constituents of interest are determined. The removal (or releases) may be calculated as follows:

$$X = \frac{C_I - C_F}{(W/V)}$$

where: X = amount of constituent removed per unit weight of soil, mg/g

C_I = initial concentration of constituent, mg/l

C_F = final concentration of constituent, mg/l

W = weight of soil used, g

V = volume of diluted leachate, l

This method provides for a wide concentration range, but it has its drawbacks. Aeration during preparation of the dilutions may result in precipitation of ferric hydroxide and changes in leachate composition. In addition, the leachate is a complex mixture of species and the interaction between the species may be affected by the dilution.

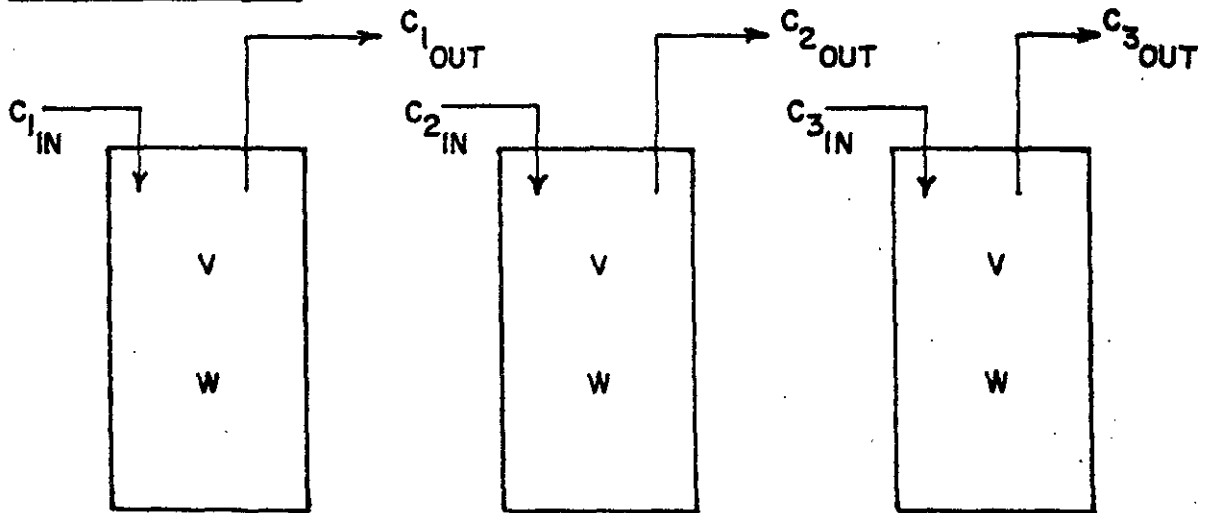
The second approach avoids these problems. A series of reactors with varying weights of soil are prepared. Different volumes of full strength leachate are added to obtain a range in the soil/leachate ratio. The samples are equilibrated at constant temperature and after equilibration the final concentrations of the constituents are determined and the removals are calculated as shown above. With this approach only a limited range for the soil/leachate ratio may be covered. Also, a certain quantity of leachate is taken up in wetting the soil and it may be

TABLE 2
Summary of Soil Analysis

Soil	Barre Sand	Barre Sand	Barre 'Clay'
Flow	Saturated & Washout	Unsaturated	Unsaturated
pH 10 g + 30 ml .01 M CaCl ₂	5.10	4.83	4.82
CEC meq/100 g	1.692	1.478	4.714
Ca meq/100 g	.069	.070	1.644
Mg meq/100 g	.023	.016	.666
Al meq/100 g	.038	.042	.064
Acidity meq/100 g	.088	.103	.082
Organic Matter %	.107	.109	.103
Mn mg/100 g	ND	2.51	9.83
Sand %	96.9	95.0	36.0
Silt %	2.1	1.8	36.8
Clay %	1.0	3.2	27.2

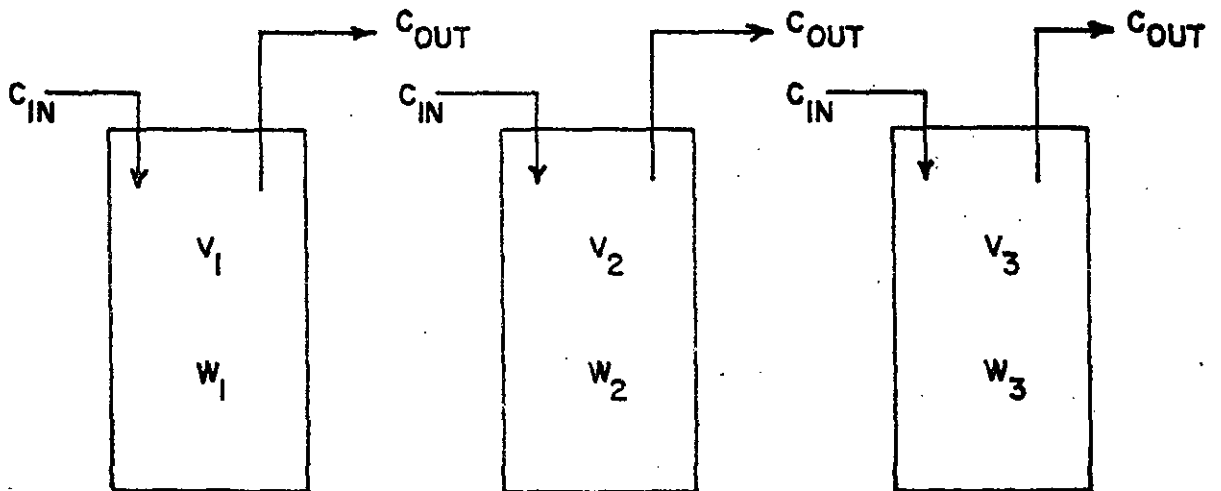
ND = Not Determined.

BATCH EQUILIBRATIONS

DILUTED LEACHATE

$$C_1 > C_2 > C_3 \dots$$

VOLUMES OF LEACHATE AND WEIGHTS OF SOIL ARE EQUAL

UNDILUTED LEACHATE

LEACHATE CONCENTRATIONS ARE EQUAL

$$W_1 > W_2 > W_3 \dots$$

$$V_1 < V_2 < V_3 \dots$$

EQUILIBRATE AT CONSTANT TEMPERATURE

Figure 2

difficult to obtain sample sizes large enough for all analyses, especially at high soil/leachate ratios.

For both approaches the use of a control, to which no soil has been added, helps to account for removals due to filtration or centrifugation of the samples and also helps to account for biological activity during the period of equilibration. This assumes that the soil microbiological population is inhibited or destroyed by the leachate and that all microbial activity is due to acclimated microorganisms in the leachate.

The accepted method of presenting the results of such equilibrations is the removal isotherm. Because it is difficult to distinguish between different mechanisms of removal the term removal isotherm is used rather than adsorption isotherm. The removal isotherm, in terms of the variables described above, is a plot of X as a function of C_F .

An example for iron removal is shown in Figure 3. For the Barre sand over the concentration range 650 to 1000 mg/l removal was relatively constant at .11 mg/g. For the Barre clay, the iron removal could be described by the Langmuir Equation

$$X = \frac{.0035 C}{1 + .0023 C}$$

An example for ammonia removal is shown in Figure 4. For NH_4^+ concentrations in the range 220 to 400 mg/l there is a sharp distinction in removal between the two soils. For the Barre clay, removal in the range .08 to .11 mg/g is predicted while for the Barre sand removal in the range .02 to .03 mg/g is predicted.

It should be noted that removals are not observed for all constituents. Figure 5 is representative of COD changes during the equilibrations. There is a wide scatter in the points and no obvious distinction between the two soils. A final example, Figure 6, demonstrates release rather than removal of a constituent. Manganese is released by both soils. This release is probably due to the reducing nature of the leachate and the solution of manganese coatings on the soil particles.

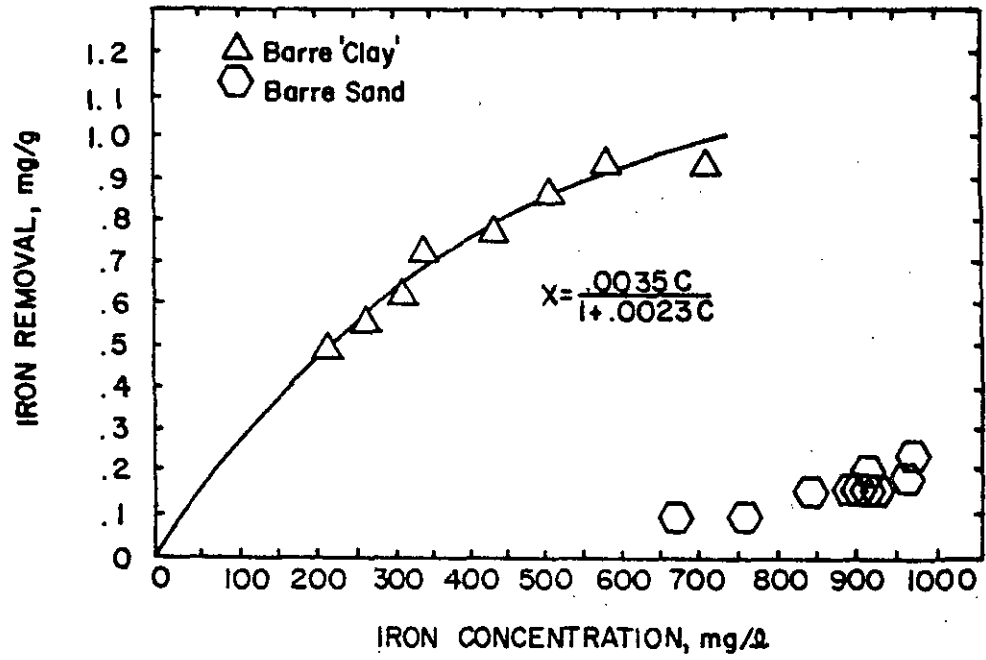


Figure 3. Removal In Batch Equilibrations

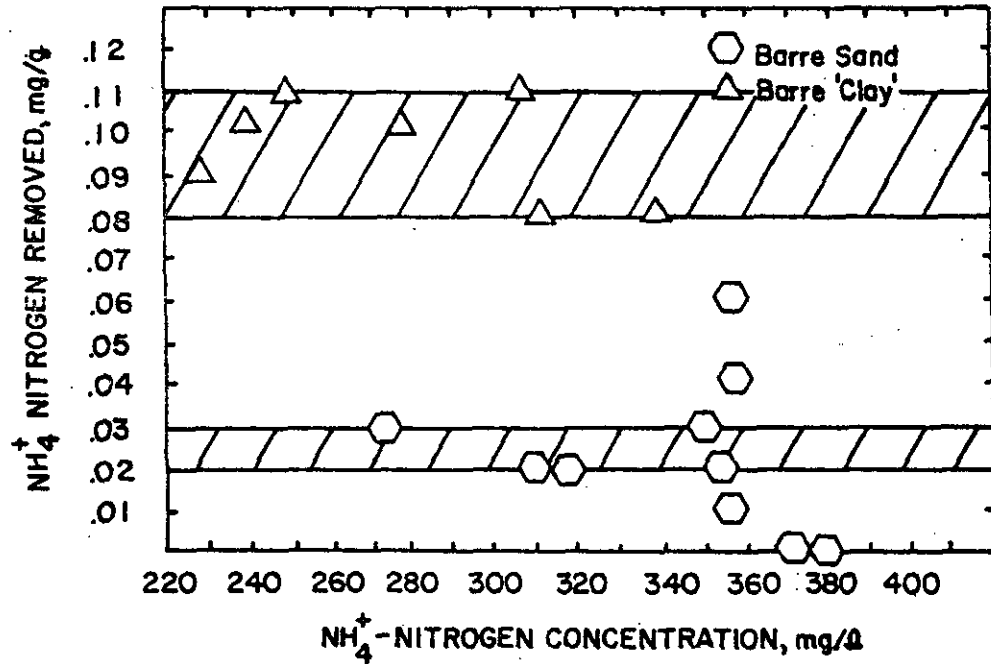


Figure 4. Ammonia Removal In Batch Equilibrations

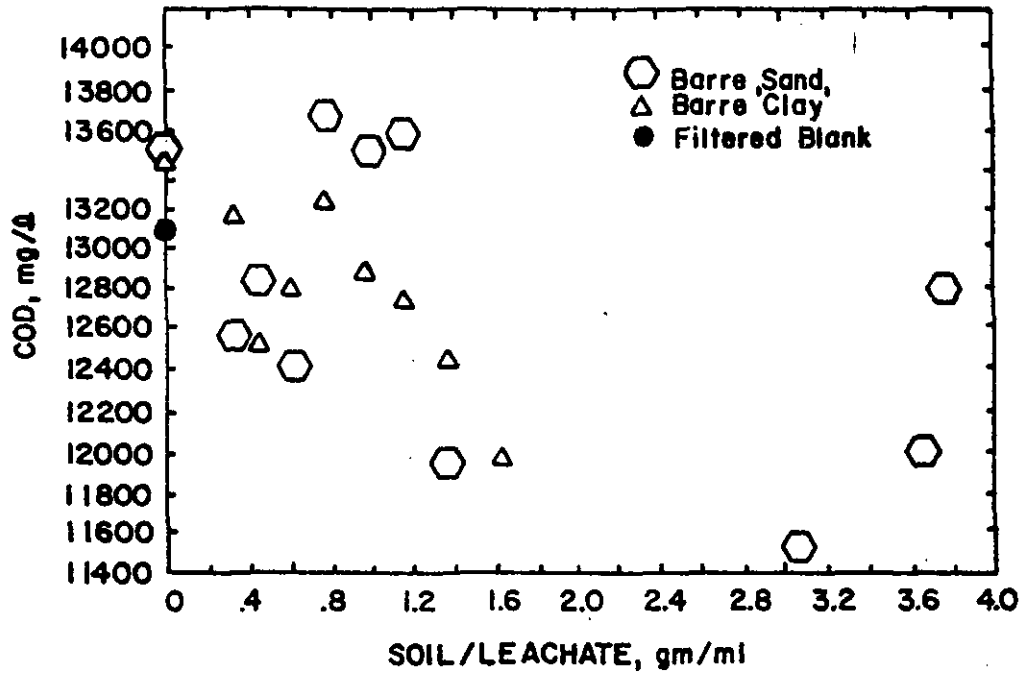


Figure 5. COD Changes In Batch Equilibrations

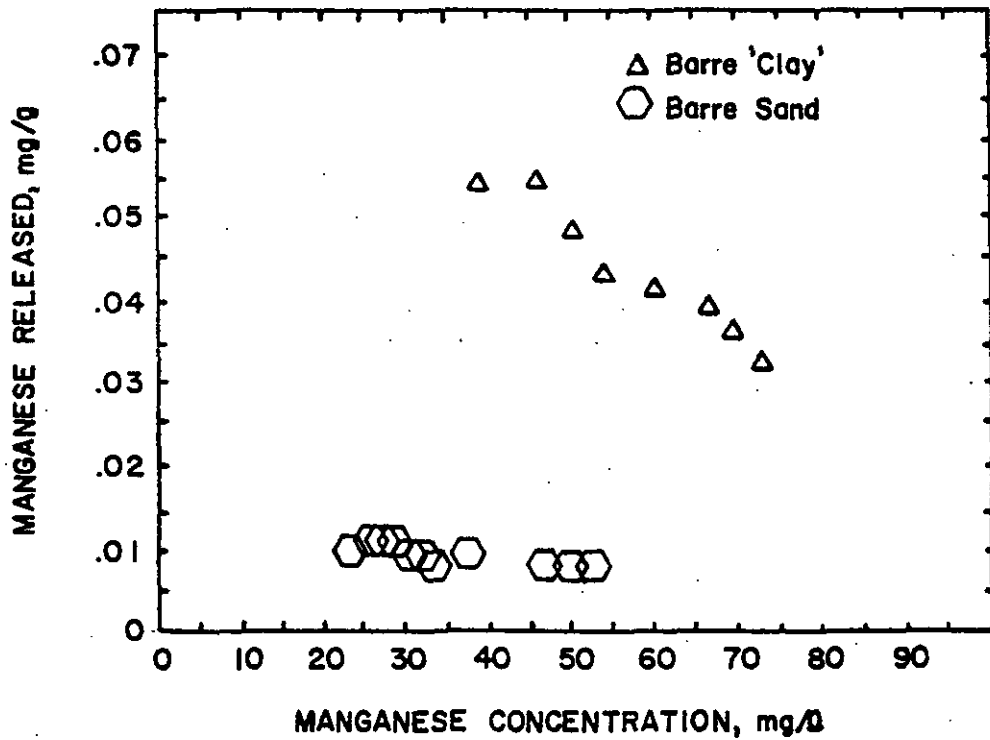


Figure 6. Manganese Release In Batch Equilibrations

Dynamic Column Studies

Dynamic column studies may be used to investigate attenuation under both saturated and unsaturated flow conditions. In addition, the permanency of removal may be examined during follow-up "wash out" experiments.

Column studies yield data in the form of breakthrough curves. Constituent concentrations are measured at a given time for various lengths or at a given length for various times. The multiple column apparatus shown in Figure 7 yielded both sets of information. Details of the column operation are discussed by Jennings (1977) and Tirsch (1977).

Figure 8 presents some generalized breakthrough curves. Curve #1 serves as a reference. For a conservative substance and a column with no soil, the result will be "piston" or "plug" flow. When a porous medium is introduced curve #2 results. The spreading of the front is due to molecular diffusion and hydrodynamic dispersion or simply as dispersion. Curve #3 represents a case with greater dispersion.

If the constituent reacts within the columns, curves #4, 5, and 6 may result. If the material undergoes a sorption reaction the breakthrough curve is delayed as demonstrated by curve #4. The constituent is removed until the finite capacity of the medium is exhausted. If biological uptake or chemical reactions remove a certain amount of the material the final concentration might only be a fraction of the original concentration as shown in curve #5. Finally, if reactions in the column result in the formation or release of the material curve #6 might result.

An understanding of the soil-leachate interaction and removal mechanisms may be gained by comparing the actual breakthrough curves to the generalized curves. In addition, the breakthrough curve data may be used in mass balance calculations to determine the capacity of the soil to attenuate particular constituents.

The saturated flow breakthrough curves for NH_4^+ are shown in Figure 9. These are all delayed behind the predicted curve for a conservative material indicating that sorption reactions are significant and that the finite sorptive capacity of the soil is eventually exhausted. The curves

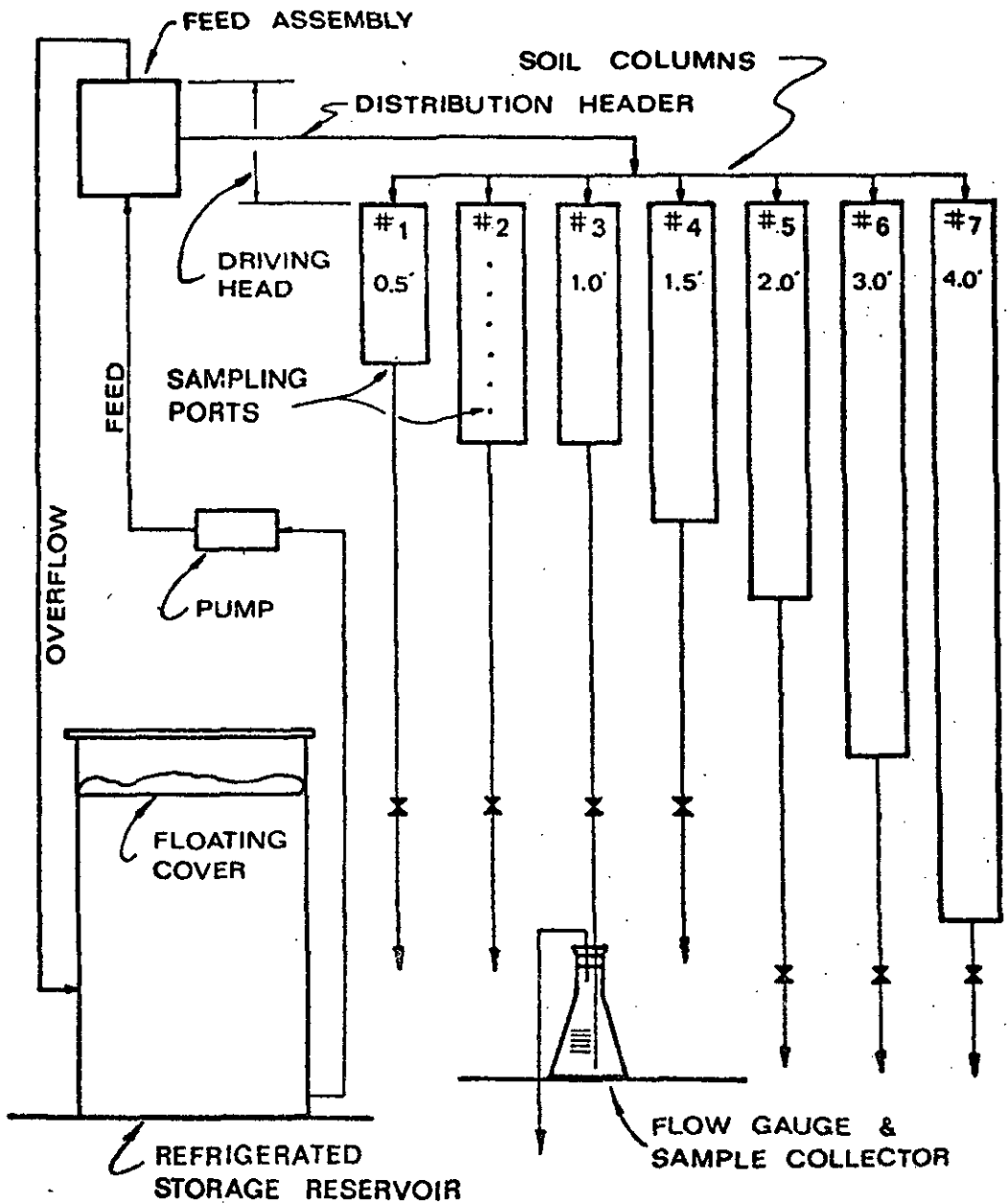
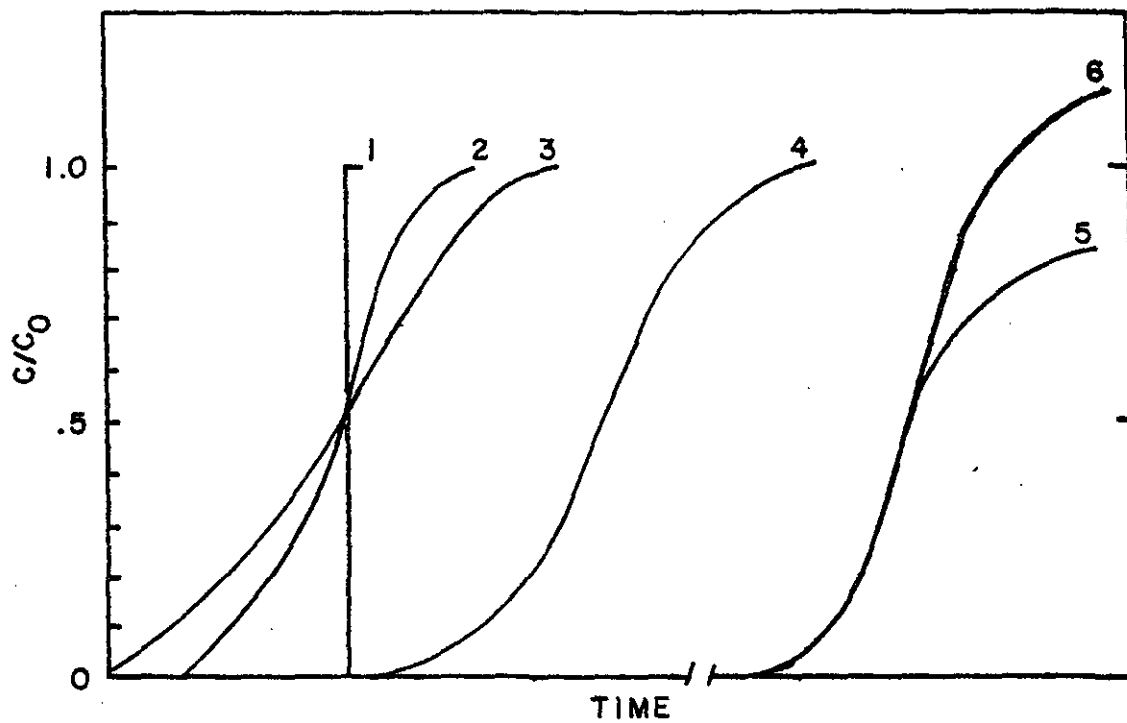


FIGURE 7
SOIL COLUMN APPARATUS

**CURVE TYPES**

- 1 Plug Flow
- 2 Dispersive
- 3 Dispersive
- 4 Sorptive
- 5 Reactive (Consumption)
- 6 Reactive (Formation)

Figure 8 Generalized Breakthrough Curves

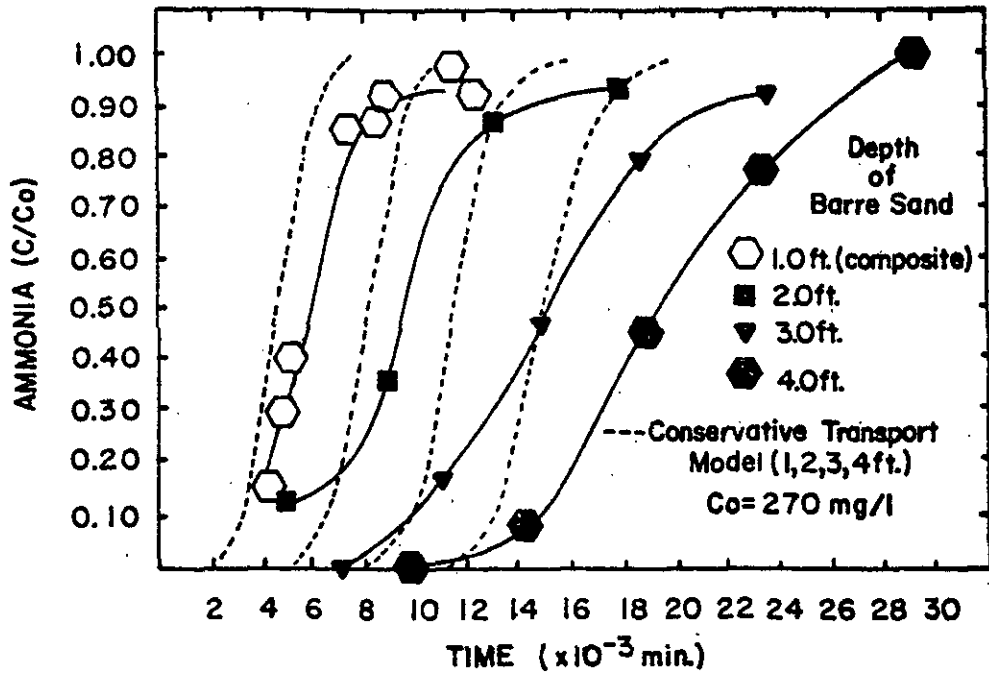


Figure 9. Ammonia Breakthrough Curves (Saturated Flow)

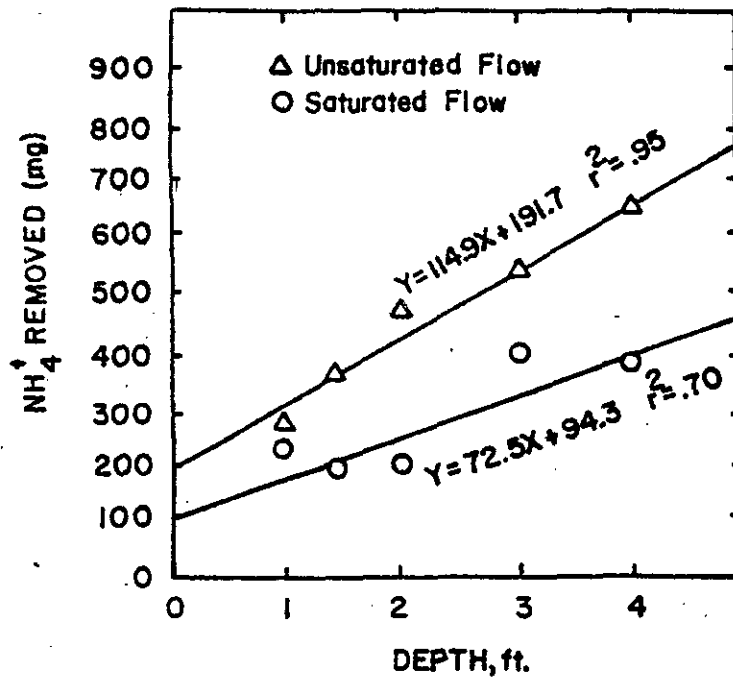


Figure 10. Ammonia Removal in Saturated and Unsaturated Flows

for the 1-, 2-, and 3-foot columns level off at C/C_0 values of .95 and the 1 1/2 (not shown) and 4-foot columns approach a C/C_0 of 1 making it difficult to determine if some removal is due to biological activity (general curve #5). The mass balance calculations for NH_4^+ are used to construct Figure 10. Ammonia removal under saturated flow conditions was 72.5 mg/ft or .016 mg/g, and under unsaturated flow conditions removal was 114.9 mg/ft or .026 mg/g. These values compare quite well with the .02 to .03 range predicted by the batch equilibrations.

The batch equilibrations for COD (Figure 5) showed that COD was not attenuated. There was also no evidence of any COD attenuation in the column studies. These COD results are discussed in greater detail by Jennings (this volume).

The batch equilibrations revealed that manganese was released by the soil. The resulting saturated flow breakthrough curves are shown in Figure 11. The curve for each depth resembles general type curve #6. The mass balance calculations for manganese are shown in Figure 12. The saturated flow curve has a slope of 51.1 mg/ft or .012 mg/g which is in excellent agreement with the .011 mg/g predicted by the batch equilibrations.

The results for Ca (Figure 13) are representative of general type curve #5. The curves level off at C/C_0 values less than 1 indicating that some consumptive reaction is taking place. In fact, it is possible that calcium carbonate may have precipitated at the surfaces of each column. This precipitation removal was not predicted during the batch equilibrium experiments.

The information from column studies is not limited to breakthrough curves and mass balance calculations. At the end of the experiment the columns may be sectioned so that soil samples from different depths may be analyzed. This information was particularly useful in understanding the Ca reactions. The unsaturated flow results are shown in Table 3. Soil exchangeable calcium increased throughout the sand columns. Surficial values were especially high and an observed crust at the surface supports the premise of CaCO_3 precipitation. For the Barre clay, on the other hand, exchangeable Ca is of general type #6 as shown in Figure 14.

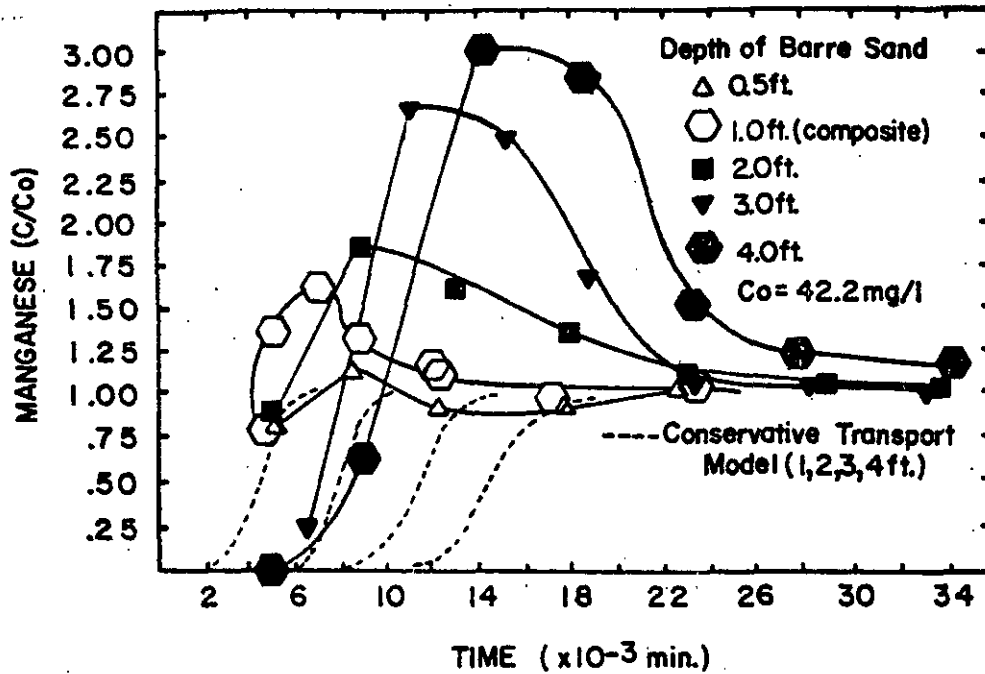


Figure 11. Manganese Breakthrough Curves (Saturated Flow)

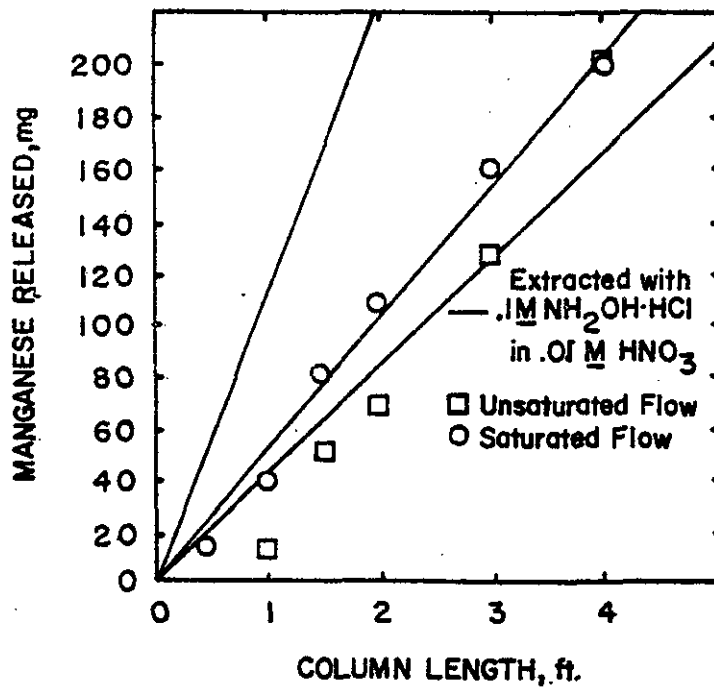


Figure 12: Manganese Release in Column Study

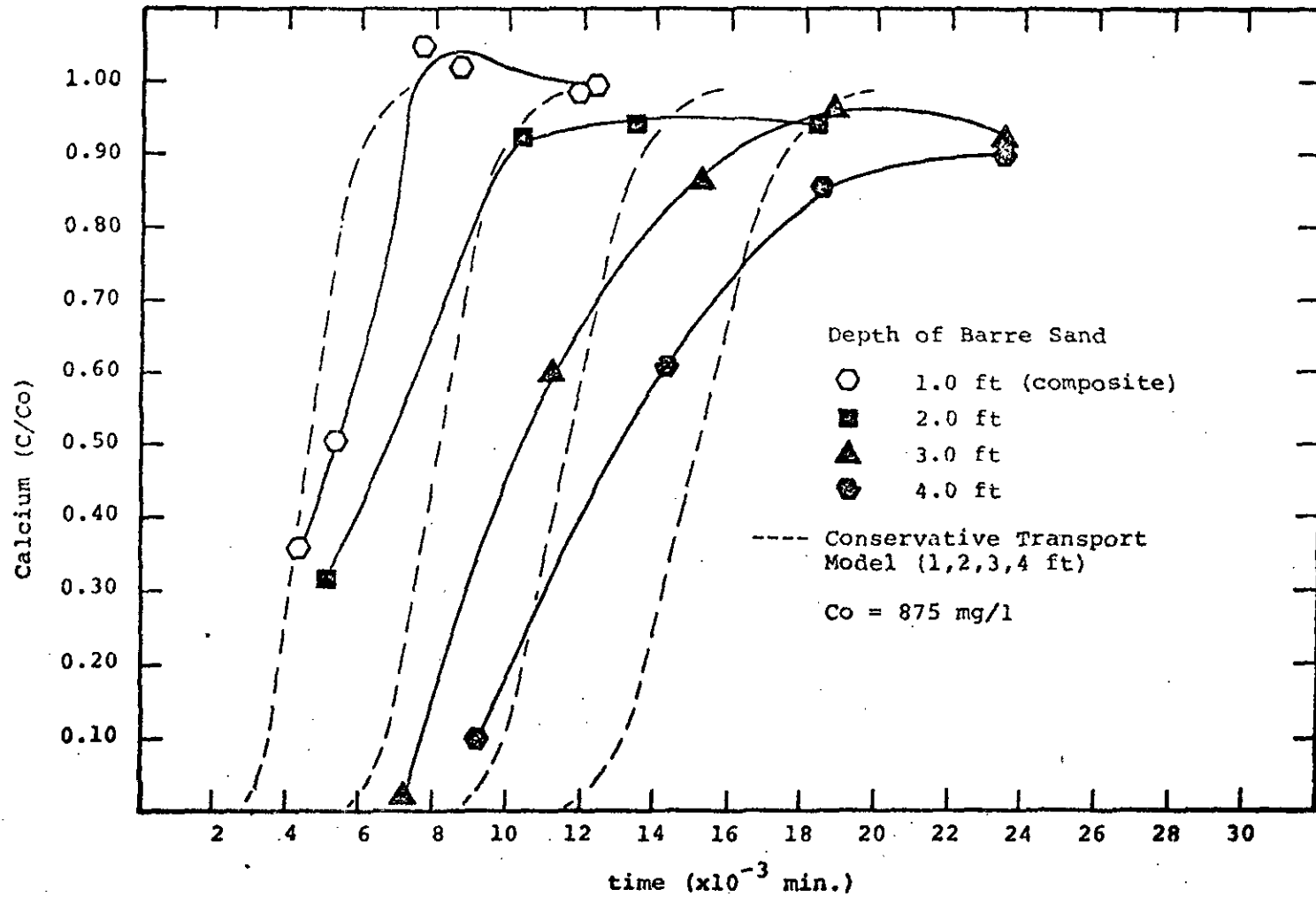


FIGURE 13 - Calcium Breakthrough Curves (Saturated Flow)

TABLE 3
Soil Analysis for Columns Sectioned After Unsaturated Flow

Column.	Depth ft.	Color	pH 10 g Soil + 30 ml .01M CaCl ₂	CEC meq/100 g	Ca ⁺² meq/100 g	Mg ⁺² meq/100 g	Al ⁺³ meq/100 g	Organic Matter %	Fe ⁺² meq/100 g	Acidity meq/100 g
Sand	-	Brown	4.83	1.478	.070	.016	.042	.109	-	.103
2	0	Black	7.18	2.089	1.460	.239	-	.285	.509	.052
2	.5	Black	6.84	1.50	.620	.217	-	.178	.349	.158
5	0	Black	7.10	1.714	1.416	.223	-	.331	.554	.054
5	.5	Black	6.98	2.187	.739	.187	-	.165	.498	.073
5	1	Black	6.88	1.411	.632	.178	-	.168	.406	.054
5	2	Black W/Brown	6.63	1.214	.525	.218	-	.142	.355	.166
6	0	Black	7.09	2.804	1.510	.227	-	.264	.724	.049
6	.5	Black	6.80	1.464	.673	.160	-	.185	.448	.056
6	1	Black	6.61	1.482	.543	.151	-	.180	.489	.052
6	2	Black	6.11	2.375	.382	.136	.114	.132	.586	.291
6	3	Brown	5.95	2.196	.307	.169	.217	.118	.371	.726
7	0	Black	6.93	1.375	1.003	.186	-	.235	.445	.056
7	.5	Black	6.62	1.464	.720	.145	-	.144	.406	.075
7	1	Black	6.49	1.161	.606	.134	-	.182	.512	.036
7	2	Black	6.32	1.321	.470	.132	.028	.149	.471	.136
7	3	Brown W/Black	6.00	1.429	.449	.156	.145	.129	.066	.319
7	4	Brown	5.91	1.196	.287	.136	.209	.103	.012	.547
Clay			4.82	4.714	1.644	.666	.064	.103	-	.082
1	0	Black	7.0	4.107	1.387	.623	-	.215	1.827	.104
1	.5	Gray- Black	6.37	4.054	.991	.520	.043	.177	1.277	.257

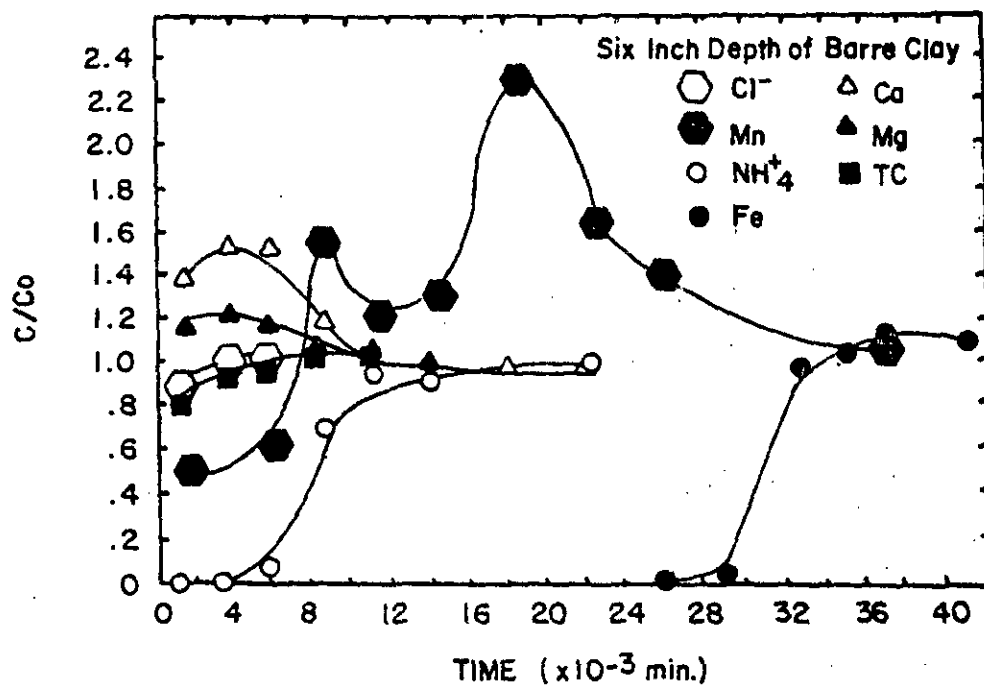


Figure 14. Relative Attenuations By Barre Clay
In Unsaturated Flow

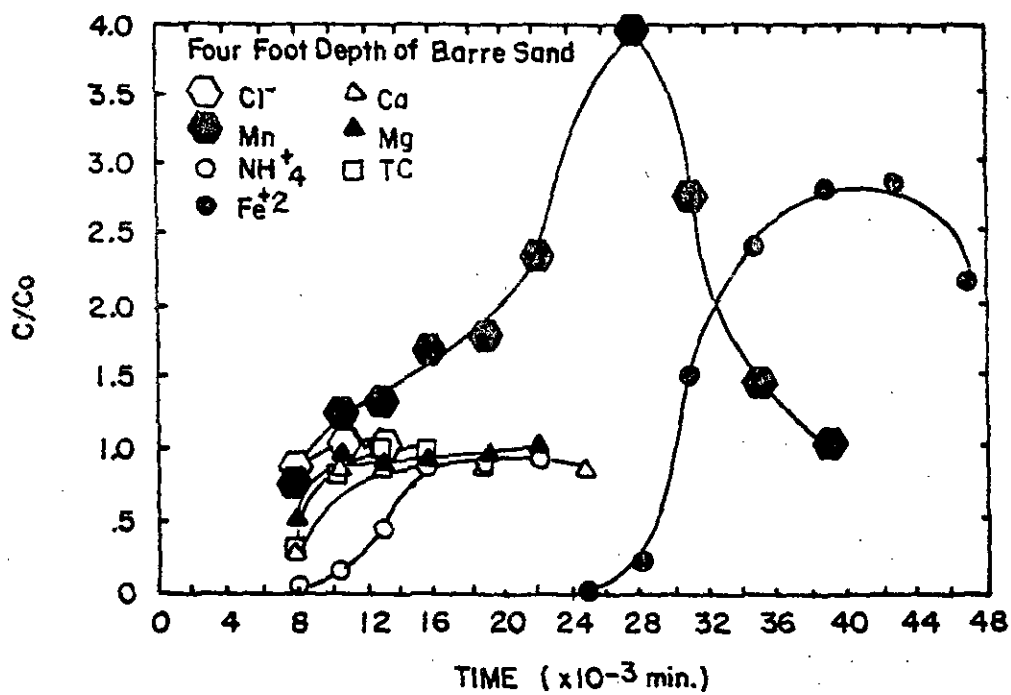


Figure 15. Relative Attenuations By Barre Sand
In Unsaturated Flow

The importance of the dynamic nature of the column studies cannot be overemphasized. The columns provide information on conditions that change with time that cannot be obtained using batch studies. The calcium reactions discussed above serve as one example. The unsaturated flow results for iron serve as an even more important example. Initially iron is attenuated as shown in Figure 15. The iron breakthrough curve is significantly delayed behind the curve for chlorides, a conservative constituent. A researcher would be misled if he stopped the experiment when C/C_0 values approached 1. It can be seen that a significant iron release takes place after this initial "attenuation." In fact, more iron was eventually released than was initially removed!

This iron example also helps to illustrate different results with different soils. The Barre clay column received the same leachate for the same time period. As shown in Figure 14, there was no evidence of a similar release of iron. If the experiment had been run longer perhaps this would have been observed.

Field Studies

Field studies were listed earlier as the final step in studying leachate reactions with soils. The design of the Barre Research Facility (Lavigne, 1976) provided sampling devices to measure vertical attenuation through the Barre sand and Barre clay soils and horizontal attenuation through the gravel dike. Unfortunately, little useful information was obtained because the first samples were essentially full strength leachate, i.e., the soil had little attenuative capacity.

CONCLUSIONS

A methodology for examining leachate-soil interactions was presented and the complex nature of these interactions has been demonstrated for three natural leachates and two natural soils collected at the Martone Sanitary Landfill in Barre, Massachusetts. Generally speaking, the Barre soils were ineffective in attenuating the leachate. A 4-foot depth of this soil would not provide any safeguard to underlying groundwaters.

It must be concluded that given the complex character of leachates and soils it is nearly impossible to extrapolate measurements for one leachate-soil system to predict interactions in a different leachate-soil system. An investigation of the type described is a necessary step in the design of a sanitary landfill facility.

ACKNOWLEDGMENTS

This research forms part of a Master of Science Thesis in Environmental Engineering submitted to the University of Massachusetts Graduate School in September, 1977. The work was funded by the Massachusetts Division of Water Pollution Control, Grant Number MDWPC 76-10(2) and by the U.S. Forest Service's Pinchot Institute Consortium for Environmental Forestry Research Grant USDA 23-591. The author gratefully acknowledges the efforts of his co-worker Aaron A. Jennings, his advisors, Dr. Donald Dean Adrian, Dr. John H. Baker, and Dr. Enrique J. La Motta, and the Environmental Engineering "Leachate Crew".

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LEACHATE TRANSPORT THROUGH SOILS

Aaron A. Jennings, Research Assistant

Environmental Engineering Program
Department of Civil Engineering
University of Massachusetts, Amherst

INTRODUCTION

Since the pollution potential of sanitary landfill leachate has been rigorously documented, the prospect of its transport away from a landfill site and through the groundwater regime is worthy of concern. The degree to which concern is justified is obviously related to how well leachate is attenuated by the soils through which it passes, but is more directly related to how well it is transported in opposition to attenuative influences. Transport is the specific subject of this paper, but before continuing, it is convenient to propose a few definitions. These are intended to be working vehicles rather than exhaustive enunciations and should be consumed for their intent rather than their letter.

Leachate - waters physically, chemically, or biologically altered to an analytically-significant degree by association with solid wastes.

Attenuation - the effect of a physical, chemical, or biological mechanism altering the liquid phase mass of a leachate constituent

Soil Attenuation - an attenuation initiated, participated in, catalyzed, or otherwise accomplished by the interaction of leachate and soil

Leachate Transport - the subsurface displacement of any leachate constituent via groundwater.

These are proposed here and have been specifically formulated to buttress the two following points.

First, although leachate often occurs in very high "strengths", a high concentration criterion should not be imposed on its definition. To evaluate leachate's full transport potential, the total mass of material released from a point of origin must be considered and all influenced areas delineated before areas of high concentration or what Palmquist and Sendlein (14) have termed "malenclaves" may be identified. Also, the apparent corollary that "as goes leachate strength, so goes the strength of its constituents" must not be applied. Leachate is a composite of many elements with its overall strength being the product of their interactions. It is altogether possible for leachate's overall strength to decline at the same time that concentrations of selected components remain unchanged or even increase. Conversely, it is possible for a specific component to be attenuated leaving overall strength virtually unchanged.

Secondly, attenuations must represent a change in the liquid phase mass of a leachate constituent and not simply its concentration. Positive attenuation indicates a removal of mass from solution, but negative attenuations are also possible indicating addition of mass to the liquid burden. Also, attenuations other than those initiated by the soil are possible. Examples of these include consumptive or formative chemical reactions occurring solely in the liquid phase. Biological transformations may also fall into this category if the organisms do not reside on the solid matrix. Additionally, attenuations are not restricted to those accomplished permanently, and time variations in the magnitude and direction of their effect are allowed. It is also important to note the definition of attenuation quite purposefully excludes effects such as dilution, dispersion, and diffusion since these only alter concentration without affecting liquid phase mass.

Given these definitions, leachate transport and leachate attenuation may be visualized as being in direct opposition to one another. Transport embodies the hydraulic and molecular forces attempting to displace leachate materials from their place of origin while positive attenuations resist the distribution of this material by transferring mass to the

stationary solid phase or by mitigating its liquid phase strength directly. Phenomena such as dilution and dispersion are supplemental to this system and their effects may be allowed for in the mathematical statement of attenuation or transport. As an example, dilution water inserting itself into the distribution regime would decrease liquid phase concentrations. This would alter the degree to which kinetically-limited or concentration-dependent attenuations were accomplished, but their mathematical formulations should automatically accommodate the effect. This would also create new hydraulic forces tending to expedite transport but a flexible hydraulic simulation would easily make the required adjustments.

In many studies of this complicated system of leachate/soil interaction, the effects of attenuation have been given detailed consideration. Several investigations have been performed to determine the nature and capacities of attenuations (with the work of this author being no exception) and much of this is of considerable interest. However, and this is a very big however, it must be remembered that attenuations acting on leachate's constituents do so with a wide range of effectiveness. Attenuation capacities for specific constituents vary from values so highly positive as to preclude the material's transport to capacities of zero and even values in the negative range which actually accelerate transport. This paper presents a discussion of work done to date on leachate attenuation with a strong emphasis placed on its transport ramifications.

EXPERIMENTAL METHODS

The results of soil column flow simulations by Jennings (11) and Tirsch (16) will be heavily relied upon. In this work, a multiple-column apparatus was used to anaerobically contact leachate with soil under several flow conditions. Saturated and unsaturated leachate flows were simulated as well as the flow of clean groundwater through leachate-equilibrated soils. Details of this experimental apparatus and the column-packing procedures used have been reported elsewhere. Additional soil column work by Griffin and Shimp (6, 7, 8, 9), and Farquhar and Rovers (3, 15), Ham (10), and Fuller *et al.* (4, 5, 12) will also be considered.

Soil column data may best be interpreted by the preparation of effluent breakthrough curves (plots of the normalized parameter C/C_0 versus time) where C_0 represents the column influent concentration of a leachate solute and C represents its time-varying effluent concentration. For rigorous analysis, an almost universally-omitted but quite necessary initial step is the evaluation of the porous medium's transport of a conservative (non-attenuative) solute. In a soil column this transport may be described by the equation

$$\frac{\partial C}{\partial t} = -V \frac{\partial C}{\partial Z} + D \frac{\partial^2 C}{\partial Z^2} \quad (1)$$

where C_0 = initial concentration (ML^{-3})

C = concentration (ML^{-3})

Z = depth (L)

V = interstitial velocity (LT^{-1})

D = dispersion coefficient (L^2T^{-1})

t = time (T)

and, subject to the following initial and boundary conditions,

$$C(Z,0) = 0 \quad (2A)$$

$$C(0,t) = C_0 \quad (2B)$$

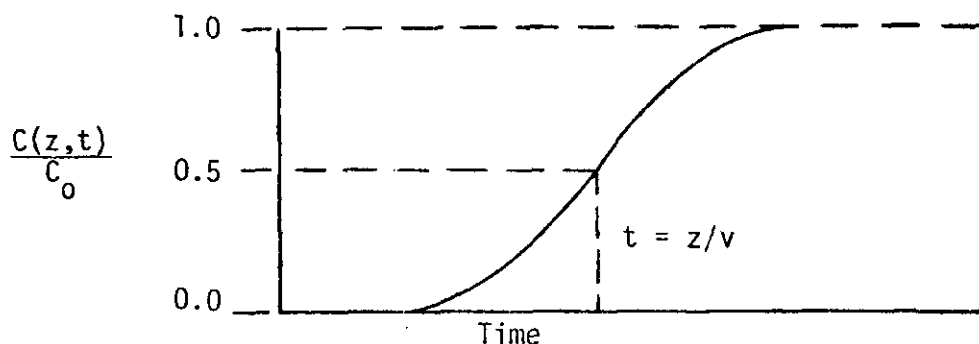
$$C(\infty,t) = 0 \quad (2C)$$

the resulting solution for C as a function of time (t) is well known (1).

SOLUTION

$$\frac{C(z,t)}{C_0} = \frac{1}{2} \left[\operatorname{erfc} \left(\frac{Z - Vt}{2 \sqrt{Dt}} \right) + \exp \left(\frac{VZ}{D} \right) \operatorname{erfc} \left(\frac{Z + Vt}{2 \sqrt{Dt}} \right) \right] \quad (3)$$

SOLUTION FORM (EFFLUENT BREAKTHROUGH CURVE)



Equation (3) may be calibrated to the soil under investigation by empirically generating a breakthrough curve for a conservative tracer at a known depth (Z) and for a known interstitial velocity (V). A 500 mg/l NaCl solution serves this purpose well. The formulation may then be forced to fit the observed data by varying the magnitude of the dispersion coefficient (D) until a "best fit" is achieved. Given this value, theoretical convective-dispersive transport curves may be plotted for any depth of soil and by comparing these to observed leachate breakthrough curves, concentration changes due to true attenuation rather than simple dispersion may be identified.

It is worth noting that this technique may be extended to identify and quantify attenuative mechanisms. "Type curves" may be generated for the solution of equation (1) to which active attenuation terms have been added. Once these have been developed, their "typical" characteristics identify the mechanisms in operation and a fit-forcing procedure may be used to extract the value of the required coefficients. These techniques were employed in the author's research and were found to be of considerable benefit in sorting out the results of what is, undeniably, a very complicated system (11).

In addition to the chemical analysis, microorganism enumerations were attempted for both column effluents and soil column section samples. Solid phase organism counts were accomplished by elution as described by

Clark (2), followed by agar plate enumeration. Both standard plate counts and soil extract agar plate counts were done but higher quality results were achieved from standard plate count agar plates incubated to maximum counts (up to two weeks). Reinoculation was not found to be a problem.

RESULTS AND DISCUSSION

Based on a careful interpretation of the published attenuation studies, the transport potential of leachate presents a grim prospect for the quality of adjacent groundwaters. Mechanisms of attenuation, with the possible exception of filtrations or consumptive reactions, are limited by finite capacities. Therefore, at any given soil location, all renovation will eventually cease and full strength leachate will pass the point unaltered. Quite aside from this, there are three additional considerations which may abrogate reliance on any soil attenuation at all.

- (1) The bulk of leachate organics may escape all attenuation and be transported intact.
- (2) The net effect of all soil interactions may not decrease leachate's overall ionic strength.
- (3) The reversability of attenuation mechanisms may eventually allow the total leachate mass to be released into the groundwater.

Leachate Organics

Using the chemical oxygen demand (COD) as a measure of the organics in leachate overestimates their burden since inorganic reduced cationic species also exert an oxidation demand during the test. Therefore, a demonstration of COD attenuation does not conclusively demonstrate organics removal. However, a failure to demonstrate COD attenuation strongly enforces the hypothesis of zero attenuation. It is the cationic species that are most likely to be attenuated so if little or no change in COD occurs one may be confident that the organics have remained.

This was the case in the work carried out by this author. Figure 1 presents COD breakthrough curves for the Barre sand. From this, and by comparison to Figure 2 which presents curves typical of an attenuated solute, it may be seen that no transport retardation (hence no attenuation) occurred. This result is not unique. Fuller *et al.* (5) actually reported initial increases in the effluent COD of soil columns contacted anaerobically with leachate. These increases slowly reduced to the initial C_0 concentration but no values below C_0 were measured. The conclusion drawn was that more organics were leached from the soil by the leachate contact and this added to the overall liquid phase burden. Results of Griffen and Shimp *et al.* (8) showed a minor attenuation of COD, but aside from sodium and chloride, no parameter measured was less positively attenuated. The fact that their calculations predicted any chloride attenuation at all suggests that their methodology failed to completely compensate for dispersional and diffusional effects. The actual degree of attenuation may have been even lower than their analysis indicated. Farquhar and Rovers (3) also reported "very little removal" of COD in soil column experiments. However, they went on to speculate that this might be due to the experimental procedure. They felt that their apparatus provided enough contact time to evaluate the potential for chemical removals but may not have allowed time for biological renovation. Considering their procedure, this author concurs with their analysis, but their result of no chemical removal remains unimpeached.

Actually, work by Ham (10) and Lombardo (13) has shown some potential for biological assimilation of leachate organics during soil transport. The one very important distinction is that both procedures used columns exposed to the atmosphere so anaerobicity was not maintained. Ham showed that attenuations of approximately 20 percent could result but that the effect was concentrated in the first 12 inches (30.42 cm) of soil. He used microorganism enumeration to establish the cause of organics reduction and showed that organism numbers were highest in the first 8 inches (5 cm) of soil and rapidly decreased (in his case, by 5 orders of magnitude) over the first foot.

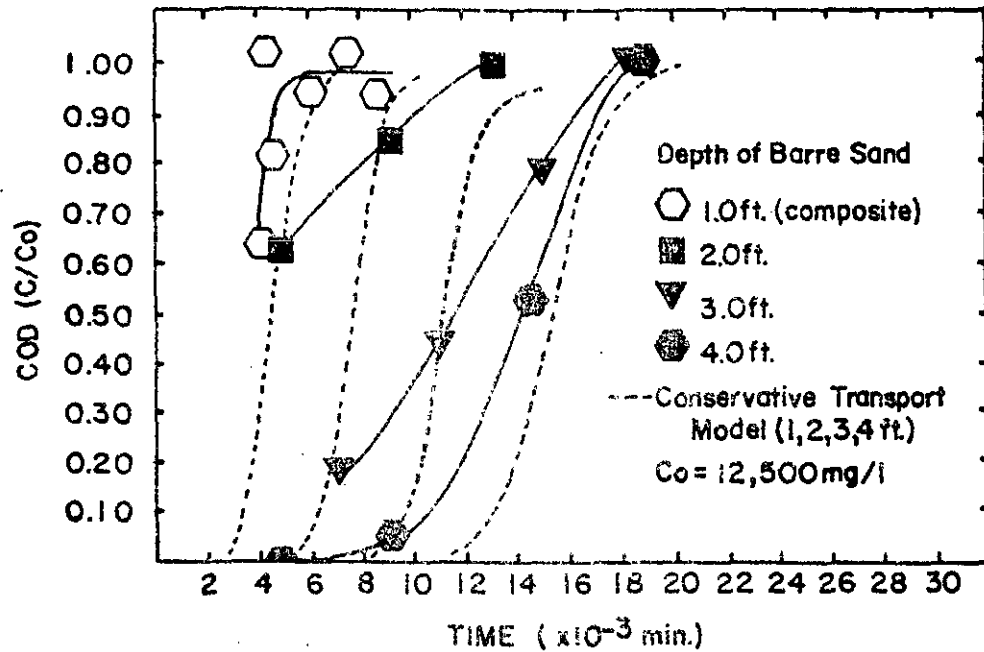


Figure 1. Chemical Oxygen Demand Breakthrough Curves (Saturated Flow)

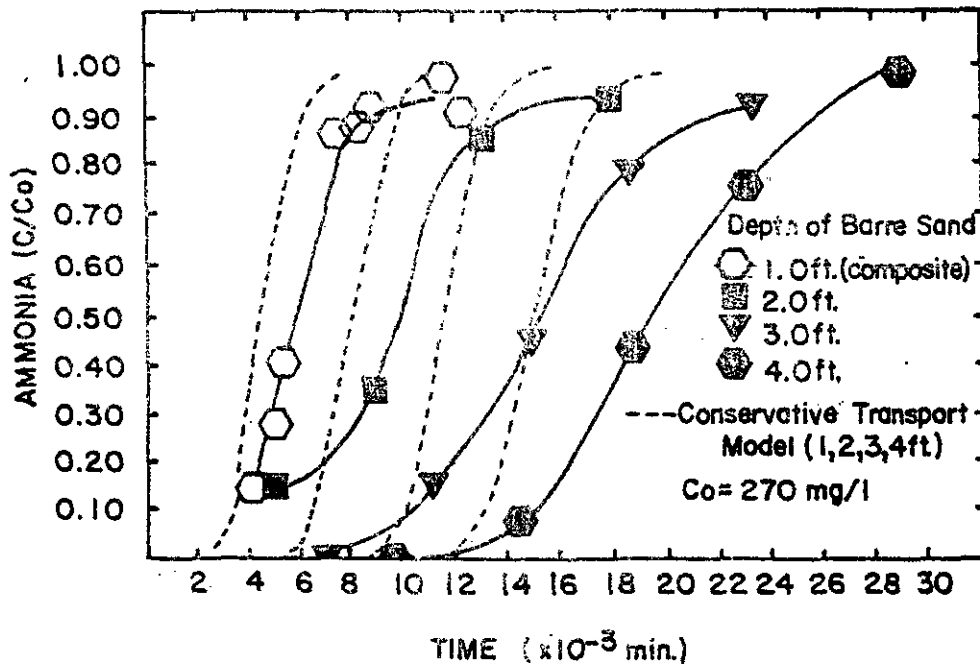


Figure 2. Ammonia Breakthrough Curves (Saturated Flow)

To investigate the potential for biological renovation of organics in the truly anaerobic situation, microorganism enumeration was incorporated into the experimental work done at the University of Massachusetts. Both liquid phase and soil population counts were attempted as a more direct indication of how viable the biological communities in the system were. Due to leachate inhibitions, only agar plate techniques proved successful in this effort. This was attributed to the fact that they isolate organisms from the hostile leachate elements and force good nutrient contact. Results of that research (Ref. Tables 1 and 2) are presented here with some reservations. The scatter of the data attests to the difficulty with which it was gathered and the significance of absolute values must be de-emphasized. Nevertheless, analyzed as trends only, the data still propound some very interesting conclusions. Based on these data, the following series of events was proposed:

The strong leachate appeared to inhibit unacclimated soil organisms so, upon initial contact, biological activity decreased. Significant biological activity was not possible until reinoculation of the soil by acclimated organisms indigenous to the leachate and this reinoculation proceeded slowly as a filtration breakthrough front. However, since no additional toxicity was observed in latter stages, leachate solutes building up on the solid phase may not promote further inhibition.

Beyond this, two additional points must be made. First, anaerobiosis is not as rapid or complete as its aerobic counterpart and there is little evidence to indicate that much biological activity will occur deep in the soil system. Secondly, the "acclimated" population consists of very specialized microorganisms and they may act selectively on only a small portion of leachate's organic species. Evidence from the studies by the author indicated that even after long periods of continuous flow, the biological populations never revived enough to accomplish significant COD attenuation.

As a final word on the fate of organics, in latter stages of the author's work, total organic carbon measurements were added as a better measure of organics but no significant attenuation was uncovered.

TABLE 1
 SOLID-PHASE MICROORGANISM POPULATIONS
 (organisms per gm soil x 10⁻⁴)

Operation		Saturated Leachate Flow		Saturated Groundwater Flow		Unsaturated Leachate Flow		
Dry Soil		20 x 10 ⁴ gm ⁻¹		20 x 10 ⁴ gm ⁻¹		14 x 10 ⁴ gm ⁻¹		
Feed		5.4 x 10 ⁴ ml ⁻¹		0.84 x 10 ⁴ ml ⁻¹		22 x 10 ⁴ ml ⁻¹		
Date		7/1	7/1	9/3	9/3	11/13	11/13	11/15
Column No.		#2	#7	#3	#6	#2	#6	#7
Depth (ft)	0.00	-	19	150	49	650	690	3200
	0.50	630	-	59	-	320	138	132
	1.00	40	2.4	75	40	8.5	23	63
	2.00	-	91	-	15.1	-	7.8	470
	3.00	-	2.3	-	19.6	-	9.0	22
	4.00	-	2.6	-	-	-	-	8200

TABLE 2
LIQUID-PHASE MICROORGANISM POPULATIONS
(organisms per ml x 10⁻³)

Operation		Saturated Leachate Flow		Saturated Groundwater Flow		Unsaturated Leachate Flow	
		5/4	5/28	7/14	8/18	10/15	10/26
Reservoir		71	54	5.9	8.2	240	210
Column Depth (ft).	0.00	70	55	18	42	-	-
	0.125	-	-	148	92	-	-
	0.375	5.3	34	208	133	-	-
	0.625	8.4	-	21	570	-	-
	1.00	2.9	59	53	21	90	100
	1.50	2.6	240	-	-	81	23
	2.00	.08	1050	920	81	9.8	2.34
	3.00	-	184	238	244	.04	43
	4.00	-	-	-	-	.62	92

Ionic Strength

Evaluation of the fate of leachate's gross ionic strength has been omitted from most studies but it is of considerable importance. Ionic strength is the result of the interactions of all leachate's solutes and, since they cannot all be quantified individually, a measure of their overall magnitude should be attained. Any water-borne solute can become a pollutant if it occurs in high enough concentrations and, even if a solute is benign, the total dissolved solids it represents may exceed acceptable levels.

In the column studies done at this University, specific conductance was used as the measure of ionic strength. Results of saturated flow experiments showed no attenuation of this parameter. Experimental results were described very well by the conservative mass transport model when the magnitude of the dispersion coefficient was increased. Experimental results and the corresponding model fits are presented in Figure 3. These have led to the interpretations that the difference between the observed data and the original theoretical breakthrough curves was the result of a significant but previously ignored diffusional mass transport component. This indicates that at high concentrations and low transport velocities, diffusion should be allowed for but that zero attenuation of ionic strength may well result.

This result is quite understandable if the proposed mechanisms of attenuation are examined. If ion exchange occurs between the soil and leachate, ions are simply replaced by counter ions supplied by the soil. The pollutional nature of the counter ion supplied may be far less harmful, as when heavy metals are exchanged for calcium or magnesium, but little reduction of ionic strength results. Also, positive attenuations of the non-exchangeable type may be counter-balanced by the negative attenuations occurring. Manganese is an excellent example, falling into this last category. As discussed by Tirsch in the preceding paper, the column effluent data showed that the Barre sand released a considerable amount of manganese as the result of leachate contact.

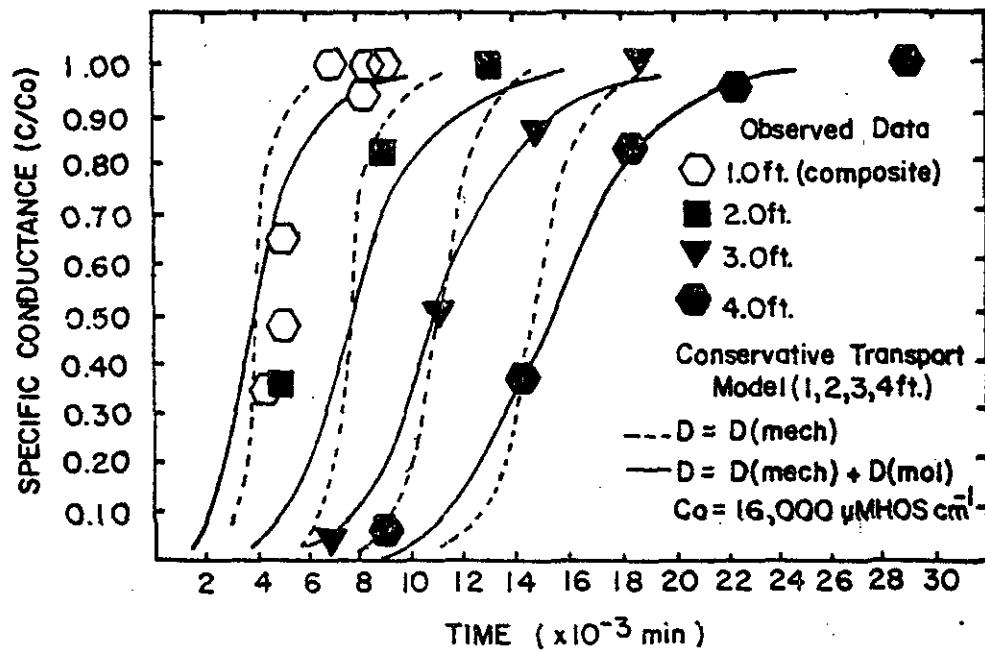


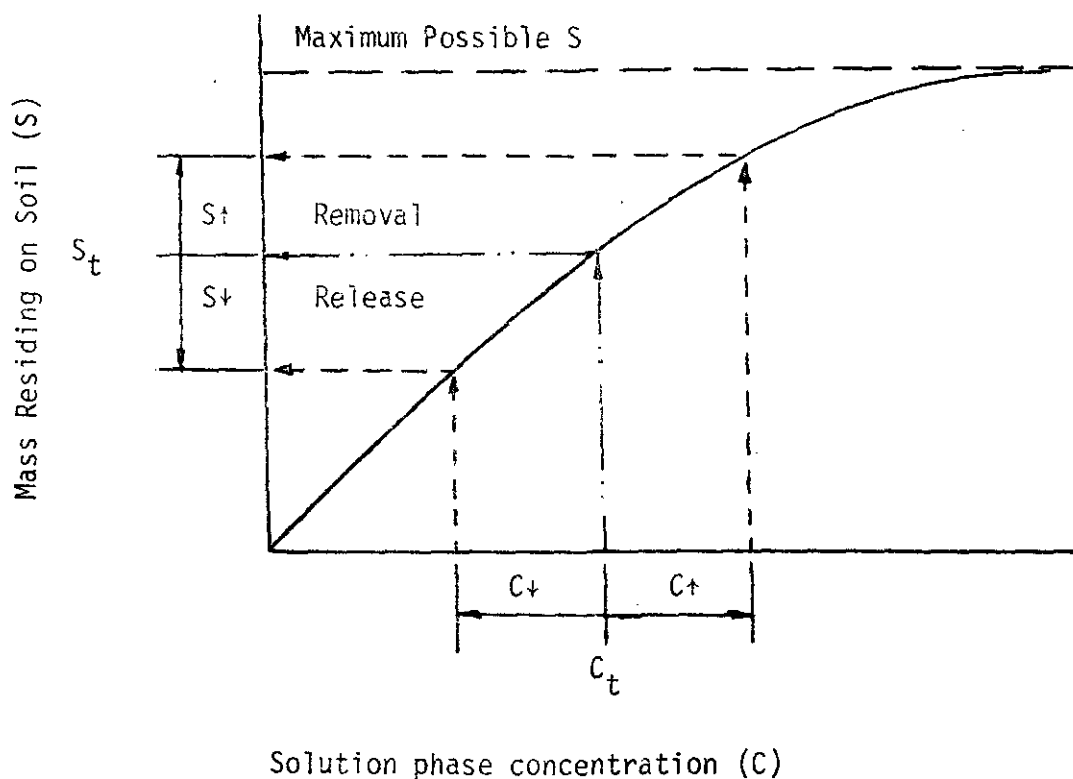
Figure 3. "Best Fit" Diffusion-Affected Conservative Transport Model for Specific Conductance (Saturated Flow)

Certainly the results of a single experiment do not warrant the conclusion that soils will not attenuate ionic strength. Experiments done with a weaker leachate and unsaturated flow actually demonstrated that under some conditions even the Barre sand can accomplish a small amount of attenuation. The results do indicate that this is a question worthy of further research and considerable concern.

Reversibility

The ramifications of reversible attenuations may be the most insidious of all leachate's malfunctions. Sorptive attenuations are probably described by some form of removal isotherm. This is a relationship that, for a constant temperature, describes the amount of solution phase mass that will be driven onto the soil in response to a given concentration.

GENERALIZED "REMOVAL" ISOTHERM



Initially, this relationship is favorable for resisting leachate transport. As leachate attempts to move, the local concentrations in its path increase. As these concentrations rise, more and more mass is taken out of the transport system as it is driven onto the soil. This is the effect to which most soil attenuation studies have addressed themselves. This, however, is favorable only as long as concentrations increase or remain high. If the mechanism is reversible, it will actually resist any future improvement in water quality.

As leachate generation at the source decreases, or with the injection of diluting waters, leachate concentrations will tend to decline. Reversible attenuations will respond to this by a shift in the equilibrium relationship that sends leachate mass back into solution. In this way, leachate equilibrated soils may serve as a future source capable of degrading groundwater quality far past the period of active leachate generation.

At the University of Massachusetts, reversibility of attenuation was investigated as a second phase of the saturated flow column studies. Once the columns had been equilibrated with leachate, a clean groundwater influent feed was stepped in. For this system, it is again important to evaluate the effects of dispersion. This is easily accomplished by assuming an instantaneous step change from $C=C_0$ to $C=0$ at a new zero time value. The conservative effluent breakthrough may then be modeled by a simple complementation of the original effluent model. This new effluent model predicts the shape of the curve produced as the liquid leachate remaining in the soil pores is washed out by the clean groundwater. This is the key in identifying reversibility. If leachate solutes persist in a column effluent after they have theoretically been washed out, then it provides strong evidence of material being supplied by reversible attenuation.

Figures 4 and 5 provide examples of two such results. Ammonia, previously identified as being attenuated, was shown to persist long after it should have disappeared from the effluent. Similarly, the

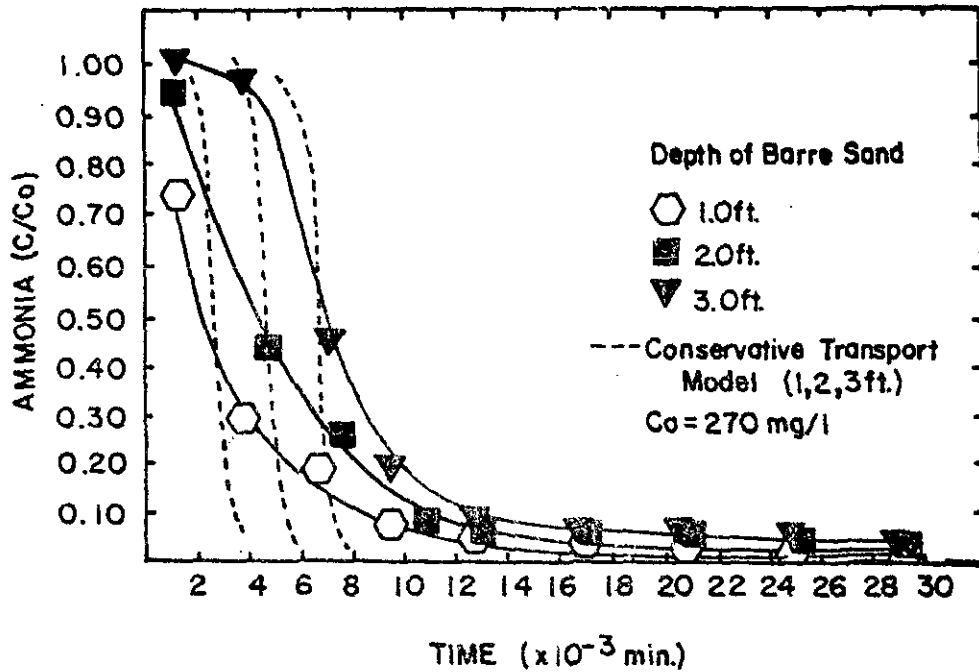


Figure 4. Ammonia Breakthrough Curves
(Saturated "Washout" Flow)

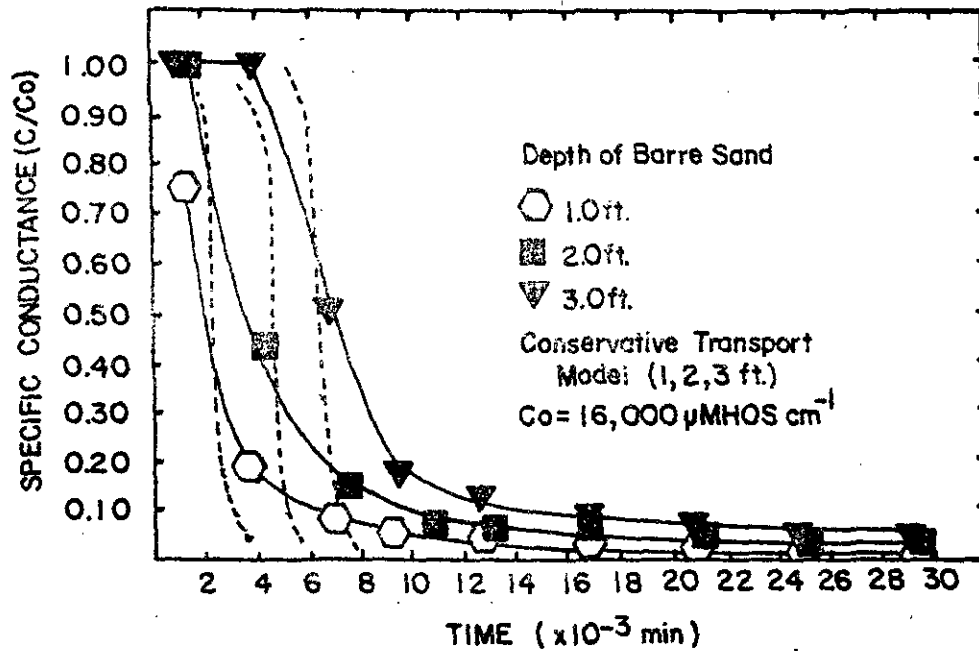


Figure 5. Specific Conductance Breakthrough Curves
(Saturated "Washout" Flow)

specific conductance of the effluent failed to converge rapidly to that of the groundwater providing more evidence of the soil's supply of pollutants exerting itself.

Figure 6 shows that even after extended washout flow (8.3+ weeks), increased specific conductance was still observed. The "blip" on this Figure was produced by exposing the system to oxygen and is an indication of the amount of material still present waiting to join its liquid phase counterparts.

The ramifications of these results are devastating. They imply that permanent attenuation may not result at all but that liquid phase transport and a more subtle solid phase migration may eventually move all the leachate mass off site and into the environment.

CONCLUSIONS

Based on the research cited, it seems wise to conclude that the maximum groundwater pollution potential of leachate is fixed by the bounds of conservative transport. Unquestionably, several of its components are readily attenuated but enough remain unaltered to pose a serious threat to whatever waterbody they reach.

For an initial evaluation, this reduces the transport modeling problem to a very simple case. If the potential for a hydraulic connection exists between a point of interest and a proposed or existing landfill site, then leachate generated at the site should be expected to eventually degrade water quality at the point of interest. Furthermore, the time of arrival may be approximated by the ground-water advective flow velocity and the maximum concentration by leachate's original strength. Certainly this neglects several of the nuances of conservative transport modeling, but it provides a sound point of departure for further evaluation. This also may seem a very pessimistic approach to take, but without sound experimental evidence in hand to demonstrate that the soils at a site will irreversibly attenuate not only selected anions and cations but also organics and gross ionic strength, then this conclusion is not only justifiable but is the most credible response to the available experimental evidence.

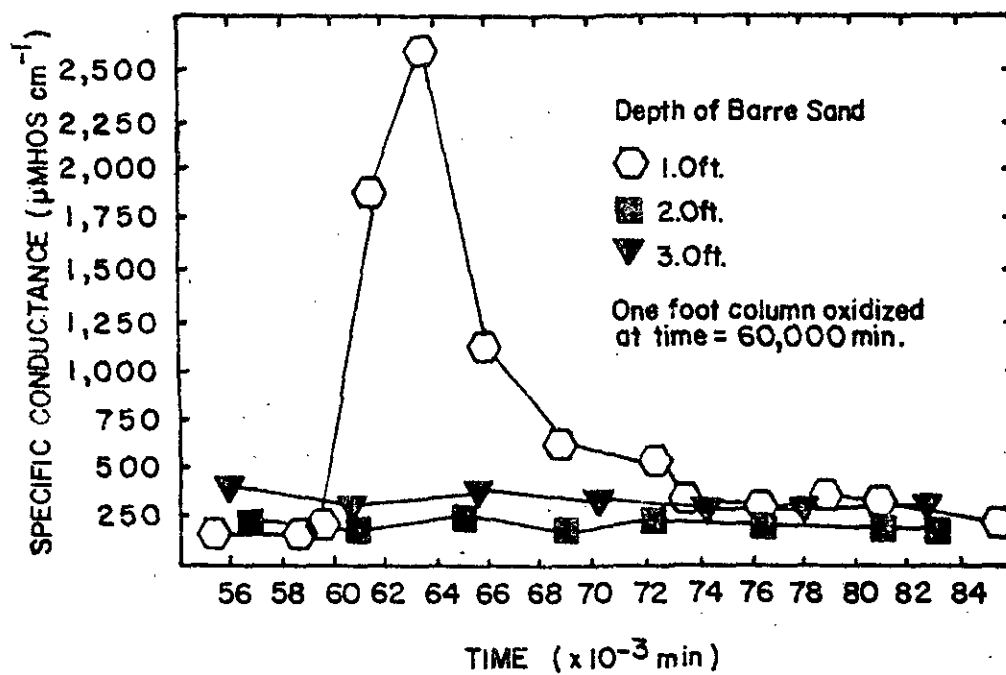


Figure 6. Specific Conductance Quasi-Equilibrium (Extended Washout Flow)

ACKNOWLEDGMENTS

This research forms part of a Master of Science in Civil Engineering thesis submitted to the University of Massachusetts Graduate School in August, 1977. The work was funded by the Massachusetts Division of Water Pollution Control, Grant Number MDWPC 76-10(2) and by the U. S. Forest Service's Pinchot Institute Consortium for Environmental Forestry Research Grant USDA 23-591. The author gratefully acknowledges the efforts of his co-worker Franklin Tirsch, Advisor Dr. Donald Dean Adrian, and colleagues of the Environmental Engineering Program's "Leachate Crew" for their frequent contributions.

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DEMONSTRATION OF A COMPLETELY MIXED UPFLOW ANAEROBIC TREATMENT
METHOD FOR SANITARY LANDFILL LEACHATE AT ENFIELD, CONNECTICUT

Robert Nichols, Engineer Intern
Solid Waste Management
Connecticut Department of Environmental Protection

A method for the anaerobic treatment of leachate using an upflow completely mixed reactor was first developed by Chian and DeWalle of the Civil Engineering Department at the University of Illinois prior to 1973. Down flow reactors have proven to be particularly susceptible to a wash out of the biomass during shock loading conditions. The system was designed as upflow to correct this problem. Mixing is achieved by recycling treated effluent quality liquor from the top of the reactor into the influent stream to dilute the highly toxic raw leachate and buffer its acidic pH. Anaerobic reactors typically have the advantages of high degrees of waste stabilization, low production of biological solids, the potential for recovery of energy in the form of methane gas, and reduced operating costs over aerobic systems in that aeration is not required.

The Environmental Protection Agency became interested and decided to fund a demonstration project in 1974. The State of Connecticut was asked to submit a proposal to conduct the demonstration at the Enfield Landfill and shortly thereafter contracts were prepared with A.W. Martin Associates in King of Prussia, Pennsylvania, to design the collection and treatment facilities.

Construction of the facility was completed late last year; however, numerous mechanical problems developed during the preacceptance testing period that have delayed the startup. Most of these problems have now been corrected and we are anticipating startup later this winter.

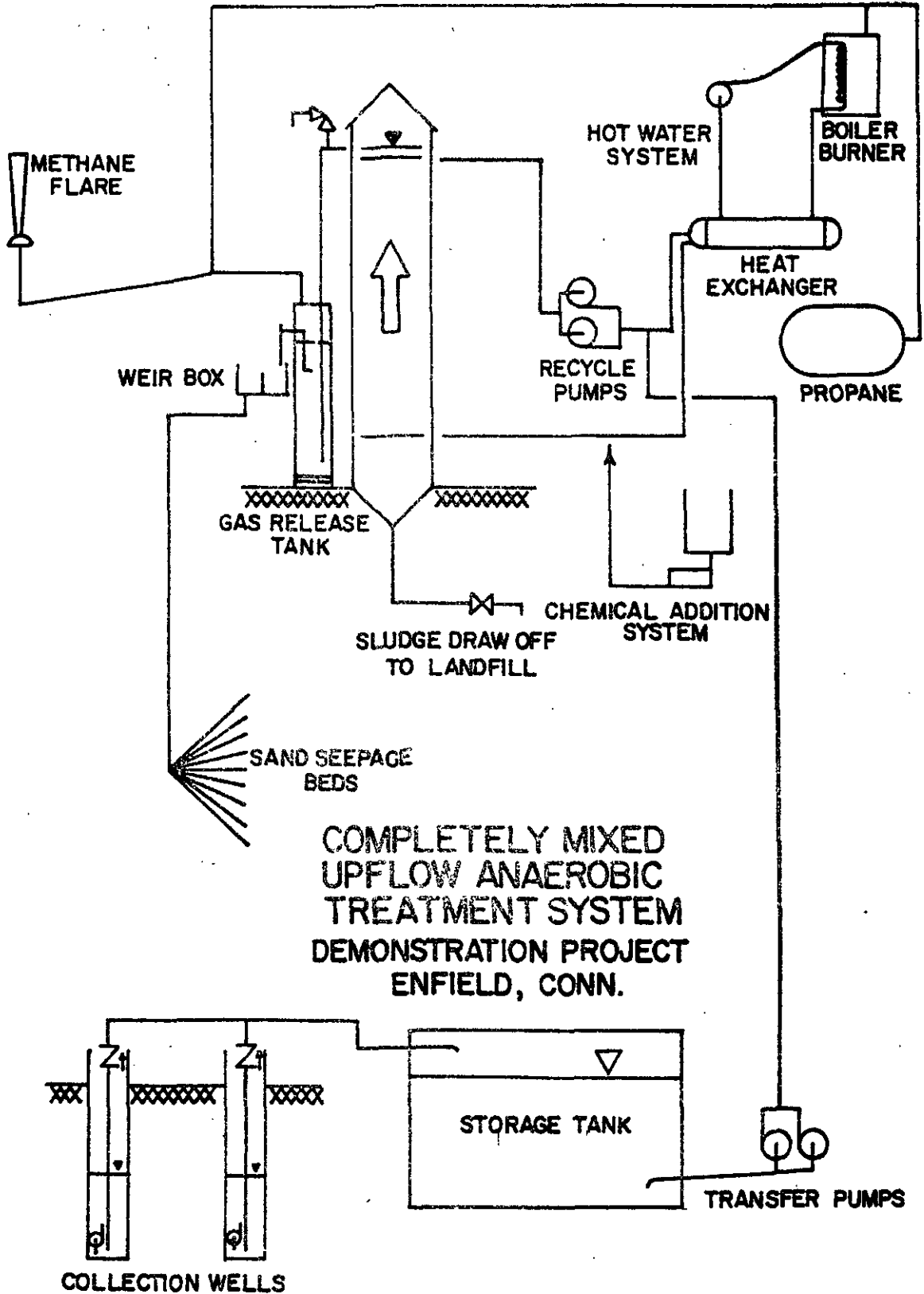
The Enfield Landfill serves a residential and commercial population of about 50,000 people and accepts approximately 150 tons of refuse per day. The site is located along the east side of the Scantic River in the Connecticut River Valley and is arbitrarily divided into an old section of approximately 12 acres used between 1967 and 1972 and a new section of 12 acres used since 1972.

The soils native to the site are fine sandy loams, silty loams, loamy fine sands or sand and clay terrace escarpments. These soils have all been used as cover material in the landfilling operations and account for a high surface permeability and high infiltration of rainfall.

Glacial Lake Hitchcock occupied the northern section of the Connecticut River Valley during the glacial epoch. Fine sediments, which accumulated to a depth of up to 150 feet, now underlie the new and old sections of the landfill. Infiltration, therefore, moves rapidly down through the highly permeable cover material and saturated refuse to the top of the lake clay and then horizontally over the clay until it surfaces at the base of the landfill. This setting provided an ideal natural system for the economical collection of the leachate. The absence of industrial waste at the landfill was another factor that contributed to the selection of Enfield for this demonstration.

The treatment method was developed for "young" leachate - leachate from a new landfill which has a high percentage of free volatile fatty acids and a COD of typically 25-30,000 mg/l. Leachate from older landfills contains lower percentages of fatty acids and more refractory organics such as humic and fulvic acids and COD's often less than 10,000 mg/l. COD removal efficiency on old leachate might be as low as 70 percent compared to the 95 percent removal we can expect on Enfield's young leachate.

Overall leachate production from the new landfill is estimated at 12 gpm. Five collection wells were drilled along the top of the first lift and were fitted with 18 inch diameter galvanized pipe, the lower seven feet of which were slotted. The wells are gravel packed and extend about 18 feet into the saturated refuse. Yield tests were conducted on all five wells and based on the results of these tests, two were selected as having sufficient capacity to provide the design 5 gpm requirement of the system. Because of the high permeability of the refuse, the cones of depression will be large and flat. The area of influence of the two wells will include nearly all of the toe of the landfill. A submersible pump, controlled by floating level control switches, is installed in each of these wells. These discharge into a common header which empties into the leachate storage tank (See Figure).



The leachate storage tank is a precast concrete tank with a 50,000 gallon capacity. This storage capacity is designed to provide for the needs of the treatment facility during an extended dry period when leachate production will decrease and also to absorb the dilution effects after periods of high rainfall. The inside of the storage tank is coated with a self-curing rubber sealant to make the tank leaktight.

Two variable speed leachate transfer pumps intake from the bottom of the storage tank and discharge to the raw leachate feed header that supplies the leachate to the facility. Here the raw leachate is mixed with recycled liquor in a ratio of about 1:10. The mixture is then heated in a shell and tube-type heat exchanger with leachate in the tube side and hot water on the shell side and then discharged into the inlet distributor at the bottom of the reactor vessel. Fifty-two outlets in the feed distribution manifold provide an even distribution of the influent and prevent short-circuiting around the media. Plugging of this manifold is an inherent hazard and the operating procedures provide for the reversal of the recycle flow periodically to control this problem.

The reactor vessel is a gas-tight structure 35 feet high and 12 feet in diameter. Three inches of insulation are installed around it and the piping to and from the building to minimize heat losses. The insulation is designed to keep the temperature drop between the inlet and outlet at less than 1^oF at 0^oF ambient temperatures.

The heat exchanger and heat source system were sized to allow operation at 100^oF for nine months of the year and above 80^oF at midwinter ambient temperatures as low as -20^oF.

A PVC media product, Koro-Z-Media 64, manufactured by the B.F. Goodrich Company, is installed between the inlet distributor and the recycle manifold to a depth of 24 feet. The material provides sites for the anaerobic organisms to live and has a specific surface of 64 square feet per cubic foot. The void volume is in excess of 90 percent.

Laminar flow conditions predominate so that only the dead organisms will slough off and these will be dense enough to fall against the current into the sludge pit at the bottom. An accumulation of about 100

gallons per day of precipitated metal sulfides, carbonates, and hydroxides, and dead organisms is expected. These will be discharged to the surface of the landfill. The reactor is protected from over-pressure and vacuum by a pressure vacuum relief valve and will normally be at 2.8 psi during system operation.

The fully treated effluent is tapped off the vessel above the recycle manifold and flows to the gas release tank at ground level. Here the gas by-products of the reaction - 70 percent methane and 30 percent carbon dioxide - are separated from the liquor. The effluent line from the reactor outlets in the gas release tank at a depth of 6 feet 6 inches. This submergence pressure maintains the pressure of 2.8 pounds in the vapor space of the reactor. An inverted U pipe at the top of the gas release tank maintains a constant fluid level in the tank and the back pressure required for the process gas system. The inverted U pipe extends to 36 inches below the nominal zero pressure fluid level and is vented to the atmosphere to prevent siphoning of the contents of the gas release tank.

The system is designed to be able to use the process gas as a fuel in the boiler-burner and gas production estimates indicate this can be done for all but the coldest months of the year. An independent propane gas system will be used during startup, in the winter months, and until the heat value of the process gas has been verified. During these months, the methane produced will be burned off in a waste gas burner flare.

After the gases have been stripped from the treated effluent, the liquor is directed to one of six sand seepage beds. The life of these beds will be a function of the suspended solids content of the effluent and is expected to be about seven days before switching to a standby bed and raking of the top crust is required.

Three biological reactions take place in the reactor. Acid fermenting bacteria act on the complex waste organics (carbohydrates, tannins, and proteins) to produce free volatile fatty acids (acetic, butyric, and propionic acids). These, in turn, are acted on by the methane-fermenting bacteria which generate methane and carbon dioxide as waste gases. Sulfate-reducing bacteria reduce sulfate to sulfur which plays an important role in the control of toxic metals.

The sampling parameters will include flow, temperature, pH, ORP (oxidation-reduction potential), conductivity, color, turbidity, COD, fatty acids, total organic carbon, organic nitrogen, inorganic nitrogen, suspended solids, chloride, the alkali metals: sodium, potassium, calcium, and magnesium, the toxic metals: iron, zinc, nickel, cadmium, lead, chromium, and copper, and finally gas composition. Sampling taps are located at the collection wells, at the reactor inlet and outlet, in the recirculation line, and at the eight and sixteen foot elevations of the reactor. Analyses will be run at daily, weekly, and bi-weekly intervals depending on the parameter and sampling point in question.

Evaluation of the analytical results will be performed to determine overall COD removal percentage and mixing efficiencies in the storage tank and reactor vessel. COD removal is expected to be about 95 percent and should correspond with the amount of gas produced. Coagulation and precipitation are expected to occur in the storage tank and this will be evaluated by consideration of the change in the ratio of the parameter in question to chloride. Since chloride will remain relatively inert throughout the system, its concentration should not change between the inlet and outlet of any of the subsystems. A change in iron, for example, between the inlet and outlet of the storage tank relative to chlorides will indicate the precipitation of iron hydroxide and carbonate in the storage tank. An unacceptable mixing efficiency in the reactor will be indicated by a substantial difference in quality between effluent and recirculation streams. This would require a redesign of the recycle manifold.

Bench scale studies conducted at the University of Illinois indicate that removal efficiencies for metals should range between 65 percent for zinc and 83 percent for iron. These results are not as good as can be expected from aerobic or physical-chemical systems. An 88 percent removal of suspended solids was achieved using Enfield leachate. Fatty acid removal was 95 percent.

The recirculation system will pump the liquor at a nominal rate of 14 gallons per minute which will provide one complete volume turnover in a 24 hour period. The breakdown of fatty acids causes an increase in alkalinity with height in the vessel. This alkalinity is used by the

recirculation system to buffer the acidic pH of the incoming raw leachate, and eliminates the need for the addition of costly buffer solutions.

A chemical addition system is provided for the addition of sodium sulfide in case of a metal toxicity condition. This could result from either a drop in pH, which would resolubilize some of the metal hydroxides and carbonates, or from depletion of the sulfide reserve, which could occur if iron concentrations increase. Gas production, which is the single most important parameter in evaluating system performance, will give the first indication of metal toxicity. Copper, because of its high toxicity, is expected to be the most limiting toxic metal. Soluble copper concentrations as low as 10^{-12} mg/l can inhibit gas production.

Startup of the reactor will involve seeding of the vessel with one thousand gallons of digester supernatant. This will provide the first colonies of anaerobic organisms as well as a rich supply of nutrients. Low buffering capacity in the recycle system during startup limits the rate at which raw leachate can be fed. A minimum pH of 6.8 should be maintained and 7.0 is preferable. Gas production must stabilize at certain calculated theoretical values before feed rate can be increased. A time period of sixty days is expected before the microorganisms become fully acclimatized to conditions in the reactor and feed rate can be increased to the design rate.

In order to determine the optimum operating conditions for this system, the testing program has been set up to allow variation testing of each of the three system variables: operating temperature, detention time and recirculation flow rate. Operating temperature will be maintained at each 5°C interval from 35°C (95°F) down to 15°C (59°F) for a two month period to determine the effect of temperature on the biological reactions. These tests will take 10 months to complete and will be followed by feed rate testing. Feed rate will be varied from 1.4 gpm (10 days detention time) up to 3.5 gpm (4 days detention time). Conditions will be allowed to stabilize for one month at each feed rate or at each one day change in detention time. If the testing schedule permits, the recycle rate will also be varied from 14 gpm down to 2 gpm. If operation at these low recycle flow rates proves to be feasible, substantial savings in operating costs could be realized.

Manpower requirements at this facility are expected to be one man day per day during the first six months of operation. After that time, equilibrium conditions should allow a reduction in this cost to about one half man day per day for the remaining project period. General surveillance, housekeeping, and maintenance requirements would seem to prohibit further reductions in this cost.

The treatment of landfill leachate using this anaerobic method has the advantages of low solids production, lower operating cost than aerobic methods, and the potential for recovery of energy in the form of the methane gas. A full evaluation of the system's potential, however, must await completion of the testing program at Enfield.

ACKNOWLEDGMENTS

This paper contains much information from unpublished reports by E. S. K. Chian and F. B. DeWalle of the Department of Civil Engineering, University of Illinois/Urbana and A. W. Martin Associates of King of Prussia, Pennsylvania. The author gratefully acknowledges their contributions.

THE CHEMICAL NATURE OF LEACHATE

Robert W. Pease, Research Assistant
Environmental Engineering Program
Civil Engineering Department
University of Massachusetts, Amherst

A landfill containing buried municipal refuse acts like a large anaerobic digester, converting complex vegetable and animal material to organic acids and various gases such as methane and carbon dioxide. The prevailing reducing environment determines the chemical state of the various leachate constituents. Iron is found in its more soluble ferrous form, nitrogen appears as high ammonium concentrations, the dissolved oxygen is essentially non-existent, and the organics have a low degree of oxygenation. Pollutational aspects include odor, oxygen demand, solids, heavy metals, and harmful organics. Table 1 displays the average values of chemical analysis on leachate samples from the University of Massachusetts Research Facility, located at the Martone Landfill in Barre, Massachusetts. It is evident from these data that leachate is unlike other "natural" pollutants, such as wastewater or urban runoff, due to the coexistence of metals and organic compounds in high concentration. This material is better regarded as an industrial waste whose treatment and regulatory concerns include not only the removal of bulk organics, metals, and nutrients, but also trace toxic compounds that might leach out of the landfill site.

This paper attempts to present a description of the chemistry of leachate as it pertains to those involved in its analysis, treatment, and regulation. Discussion must necessarily be in general terms due to the natural variability of samples caused by differences among landfill sites in precipitation, soil characteristics, refuse content, depth, and age. Conclusions drawn from analysis of Martone Landfill leachate are specific to that particular site, but similar methods may be employed for other locations.

TABLE 1

Composition of Leachate from Barre, Massachusetts
 No Filtration Employed; Collected from Surface Pools
 all concentrations in mg/l

	Average of 21 Samples (5) from 10/74 to 8/75	Winter Leachate(9)	Summer Leachate (9)
BOD ₅	3,200	ND	ND
COD	5,620	11,100	13,534
TOC	2,940	ND	4,675
TS	11,350	ND	ND
VS	3,240	ND	ND
Cond (μ mho/cm)	ND	18,000	ND
pH	6.1	5.5	6.2
Alk (CaCO ₃)	1,400	2,100	4,150
Inorganic P	3.3	0.2	ND
SO ₄ ²⁻	172	128	ND
NH ₄ ⁺ -N	200	225	378
NO ₃ ⁻ -N	2.6	2.6	ND
CL ⁻	189	ND	ND
Fe (Total)	993	1,020	1,095
Mn (Total)	ND	33	22
Ca (Total)	ND	680	776
Mg (Total)	ND	173	117
Cu (Total)	0.4	2.7	ND
Zn (Total)	22	0.7	ND
Cr (Total)	0.9	ND	ND

ND: Not determined.

CHEMICAL COMPOSITION

Organic Constituents

A study by Chian (1) and Chian and DeWalle (2) shows that the age of a landfill greatly affects the distribution of the organic constituents. This is not an unexpected result because active anaerobic fermentation would occur in only unstabilized or "young" landfills, producing large quantities of volatile organic acids. It is the odor of these compounds that makes leachate such an unpleasant substance to work with. Table 2 gives the results of volatile organic acid analysis of leachate from various researchers. Only two ages of the landfills are known: Chian used a laboratory lysimeter in place less than a year and the author's sample is from the Martone Landfill, 18 months active.

TABLE 2

Volatile Organic Acid Content of Leachate from Various Researchers

<u>Researcher</u>	<u>Reference</u>	<u>Volatile Organic Acid</u>
Chian	1	49% of TOC
Mao and Pohland	1	80% of COD
Burrows and Rowe	1	75% of COD
County of Sonoma	1	40% of COD
Hughes, <u>et al.</u>	1	20% of COD
Dunlap, <u>et al.</u>	3	67% of mass
Johansen and Carlson	4	56% of TOC
Pease	5	36% of TOC as acetic
		<u>or</u> 73% of TOC as butyric*

* Analysis accomplished by adsorption-elution-titration procedure with results expressed in mass equivalent of specified acids; this is not molecular identification.

Chian and DeWalle (1,2) analyzed a single leachate sample from a 13 year old landfill and found no organic acids present indicating

TABLE 3
Composition of Leachate from Older Landfills
From Chian and DeWalle (2)

	All Concentrations in mg/l		
	DuPage MM 61	DuPage LW 6B	Winnetka
Age (yrs)	14.5	16.0	16.1
COD	360	40	153
BOD ₂₀	125	225	105
TOC	ND	ND	ND
pH	ND	7.0	ND
Alk. (CaCO ₃)	1,630	2,250	1,450
Hardness (CaCO ₃)	690	540	7.0
Total P	0.5	8.9	1.3
NH ₄ -N	ND	ND	ND
NO ₃ + NO ₂ -N	0.14	1.6	0.2
Ca	156	102	109
Cl	205	135	70
Na	63	74	34
K	85	100	39
SO ₄	1	2	5
Mn	0.24	0.06	0.2
Mg	110	90	75
Fe	106	0.6	11
Zn	0.10	4.5	0.1
Cu	0.5	0.5	0.5
Cd	0.05	0.05	0.05
Pb	1.0	1.0	1.0

ND: Not Determined

a stabilized situation with only water soluble refractory organics washing out. The total organic carbon (TOC) was a low 644 mg/l and 94 percent of the organics had a molecular weight (MW) of 500 or less (determined by membrane filtration). Table 3 shows the composition of leachate from older landfills indicating reduced loadings in all parameters except alkalinity. However, it will be subsequently shown that the source of the alkalinity undergoes change.

The molecular weight distribution of organic molecules in leachate reflects the predominance of the low carbon number organic acids. Table 4 shows the results of fractionation of samples by Chian (1) and this author (5); each investigator used membrane ultrafiltration techniques including prefiltration with a 0.45 μ filter for solids removal.

TABLE 4
Molecular Weight Fractions of Leachate Samples

<u>Molecular Weight Range</u>	<u>TOC, mg/l</u>	<u>% TOC</u>
	Researcher: Chian Initial TOC = 17,060	
less than 500	12,454	73.0
500-3000	3,156	18.5
3000-30,000	426	2.5
greater than 30,000	1,024	6.0
	Researcher: Pease Initial TOC = 7,000	
less than 1000	5,740	82
1000-10,000	280	4
10,000-50,000	210	3
50,000-100,000	280	4
greater than 100,000	490	7

To date there has been no adequate investigation of the higher molecular weight non-volatile acid components of leachate, but possible compounds include fatty acids, polypeptides, and humic substances. Dunlap, et al. have isolated and identified 42 specific compounds, the most significant being plasticizers leached from synthetic polymers in the refuse. These materials constituted 7 percent by mass of the compounds identified (but not necessarily 7 percent of the total organics).

TABLE 5

Composition of Leachate Used in Present Study
 Suspended Solids Removed by 0.45 μ Filter
 Collected by Landfill Tap System
 All Concentrations in mg/l

TOC	7000	Fe (Total)	1225
Inorganic C	13	Na (Total)	695
Total Volatile		Ca (Total)	1000
Organic Acid (mg/l Acetic)	6360	Zn (Total)	2
Conductivity (μ mho/cm)	30,000	Mn (Total)	19
pH	6.2	SO ₄ ⁼	1080
Cl ⁻	460		

Inorganic Constituents

This discussion of inorganic constituents refers to the chemical analysis of leachate from the Martone Landfill listed in Tables 1 and 5. The landfill was constructed to be a research and treatment facility as well as an active solid waste disposal site. Included in its design are leachate collection funnels underneath the refuse connected by pipes to valves for anaerobic sampling. The leachate was passed through a 0.45 μ filter before analysis for suspended solids removal. Investigation of Tables 1 and 5 yields the following observations:

1. Leachate is a solution of high ionic strength (I) requiring the use of activities rather than concentrations for all equilibrium calculations. Direct calculation of ionic strength is impossible because of the undetermined ionic species, but estimation may be made from the measured conductivity and its equivalence in moles/l of sodium chloride. Linear extrapolation of conductivity measurements of known sodium chloride solutions (determined by colleague Franklin Tirsch) gives a value of 8400 ppm for 30,000 μ mho/cm. This is equal to 0.14 moles/l NaCl which, for a simple salt, is also equal to the ionic strength. All subsequent activity coefficient calculations are based upon an estimate of $I = 0.14$.

2. Iron is the predominant metal measured on a mass basis* and due to the anaerobic nature of leachate, is present in the ferrous form. The amount of oxygen required for converting 1000 ppm to the ferric oxidation state is 143 mg O₂/l representing a very small fraction of COD measurements common for leachate. The concentration of iron when carbonate species are present is limited by the formation of solid ferrous carbonate. The measured inorganic carbon concentration of 13 ppm is equivalent to 1.1×10^{-3} moles/l. The maximum solubility of ferrous iron associated with that alkalinity is 312 ppm. This number is calculated from the solubility product of ferrous carbonate ($K_{so} = 2.1 \times 10^{-11}$; source: Stumm and Morgan (6), p. 186), using an activity coefficient, γ , equal to 0.35 for both ionic species. The activity coefficient is determined by the Davies Equation which is a function of only ionic charge and ionic strength of the solution (6). A comparison of the measured concentration of iron with its predicted solubility shows either an apparent super-saturation or an increase in solubility by 300 percent due to complexation with the leachate organics. The former value is 1225 mg/l while the latter is 312 mg/l; it is shown in a later section that organic complexation is responsible for the enhancement of iron's solubility.
3. Chromium and copper concentrations (Table 1) are low due to limited solubility at pH = 6.2.
4. Nitrate concentrations are low due to redox considerations: the ammonium ion is the stable form of nitrogen at low oxidation-reduction potentials (ORP). The source of the nitrogen is most likely to be organic rather than from nitrate ions, reflecting anaerobic degradation of proteinaceous materials. Since the average pH of leachate is well below the pK of NH₄⁺, the nitrogen present contributes little to its alkalinity. The

*Using a molar basis, the order becomes: Na(0.030M), Ca (0.025M), and Fe(0.022M).

dichromate oxidant of the COD test will not oxidize the ammonium ion, and therefore will not indicate its presence.

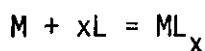
Ammonium ions are biologically oxidized and can be a significant contribution to the ultimate biochemical oxygen demand (BOD) of leachate. The results from the 21 samples averaged in Table 1 show that NH_4^+ accounts for 22 percent of the ultimate BOD using a stoichiometric equivalence of 457 mg/l O_2 per 100 mg/l NH_4^+ and assuming ultimate BOD = $\text{BOD}_5 + \text{NOD}$ (nitrogen oxygen demand).

5. The alkalinity of leachate is caused primarily by the volatile organic acids whose values for pK are all less than 5, indicating almost complete dissociation at pH = 6. Using a value of 6000 mg/l acetic as the acid concentration and assuming 90 percent ionization at pH = 6, an alkalinity of 2250 mg/l CaCO_3 results. This is calculated by determining the amount of acid that would protonate 50 percent of the moles of acetic acid (the endpoint of the alkalinity titration is close to the pK's of the acids) and determining its equivalence as calcium carbonate. Unfortunately, an alkalinity determination was not attempted on the leachate of Table 5 for which there are volatile acid measurements, but examination of Table 1 shows that this calculated alkalinity value is similar to the analytical results listed there.
6. Sulfur is present as the sulfate ion and can be readily reduced to the sulfide form with a lowering of the ORP. This has occurred with the author's samples after nitrogen sparging to produce a black ferrous sulfide precipitate. The iron in leachate is stable within a very narrow ORP range: air exposure causes ferric hydroxide precipitation and nitrogen sparging causes sulfide precipitation.
7. Dissolved phosphate concentrations are low due to the insolubility of ferrous phosphate. Singer (7) has calculated the K_{so} to be 1.3×10^{-29} indicating limited phosphorous solubility; relatively high concentrations of phosphorous determined by analysis may be due to colloidal or organic species hydrolyzed by the digestion methods employed.

METAL-ORGANIC INTERACTIONS

Metal ions are well known to form molecular associations, called "complexes," yielding species possessing altered chemical behavior. Metal ions do not complex with just organic molecules, but will seek any type of association that will result in the lowest, thus most stable, energy state possible. The free energy of the metal is reduced by the contribution of electrons from the other partners in the complex; this is not simply due to the neutralization of the metal's positive charge, but to the formation of new molecular orbitals with definite spatial orientations. A molecule with the ability to complex a metal is known as a "ligand" and must contain an electron-donating atom, such as oxygen, nitrogen, sulfur, or one of the halogens.

The extent of any complexation is quantitatively described by the equilibrium or "stability" constant for the reaction between a metal and a ligand. For one mole of metal, M, complexing with x moles of ligand, L, the chemical equation and resulting stability constant are expressed as for any other reversible reaction:



$$K = \frac{ML_x}{M \cdot L^x}$$

Higher values of K correspond to more metal being complexed at equilibrium.

Knowledge of stability constants of the complexes formed between metals and organics in landfill leachate would provide useful information for those engaged in the following:

1. Treatment System Design

Metals are usually removed from a wastewater by lime precipitation, oxidation, or ion exchange. These methods may have only limited success if extensive complexation exists to cause increased solubilities at high

pH, protection against oxidizing environments, or reduction of free metal ions. Likewise, effective metal removal may carry along a significant organic content, producing a sludge that requires stabilization.

2. Subsurface Modeling

Ion exchange sites within soils are in equilibrium with uncomplexed metal ions and the value of the stability constants for the predominant cationic species will enable the calculation of soil exchange capacity and attenuation. Lack of such information requires difficult and tedious experimentation with anaerobic soil columns (8,9).

3. Chemical Analysis

Analytical tests, such as those in Standard Methods (10), have been developed with consideration for natural waters and municipal wastewaters. The special nature of leachate requires evaluation of the applicability of these techniques and the extent of complexation is necessary knowledge for such a review.

Stability constants between the metals iron, zinc, manganese, and calcium and anaerobic leachate have been determined by the author (5) utilizing an ion exchange technique as outlined in Martell and Calvin (11). Special glassware and handling methods were designed and constructed to eliminate atmospheric contact.

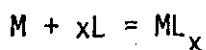
Calcium and manganese exhibit no interaction with the leachate organics while the stability constants for iron and zinc were determined to be 47 and 225 respectively ($\log K_{Fe} = 1.67$ and $\log K_{Zn} = 2.34$). The magnitude of these values is relatively quite small, indicating that very weak complexes are formed. The conclusion to be drawn is that the complexation which exists in leachate will have no significant effects on treatment or modeling efforts.

The experimentally determined stability constants are numerically equivalent to those previously listed for the same metals and volatile organic acids (12) indicating that they are the ligands of consequence and there is no contribution from other sources, such as the humic

substances (Schnitzer and Skinner have measured $\log K_{\text{Fe(II)}} = 5.77$ for a fulvic acid fraction (13)). This conclusion is supported by work of Chian and DeWalle who determined that greater than 90 percent of the metals in their leachate sample associate with organics of a molecular weight less than 500 (14).

An example of the use of the stability constant is the calculation of the free metal concentration of iron at equilibrium to determine whether it is indeed super-saturated in the presence of $1.1 \times 10^{-3} \text{ M}$ carbonate.

The chemical reaction and mass action equations are:



$$K = \frac{a_{m1}^x}{a_m a_1^x} \quad (1)$$

or

$$K = \frac{\gamma_{m1}^x C_{m1}^x}{\gamma_m C_m (\gamma_1 C_1)^x} \quad (2)$$

where:

M = moles of uncomplexed metal

L = moles of uncompleted ligand

x = stoichiometric coefficient

a = activity of ionic species

γ = activity coefficient of ionic species

C = molar concentration of ionic species

For the iron-leachate interaction, $x = 1$ and $\gamma_{m1} = \gamma_1$ because of their equal ionic charge.* The equation simplifies to:

$$K = \frac{C_{m1}^x}{\gamma_m C_m C_1^x} \quad (3)$$

*The ligand, L, has a definite charge of -1 (ionized monoprotic acid), but the designation of a +1 charge to the complex is questionable due to the low value of K indicating weak interactions.

Using Table 5 as a basis, the following quantities may be substituted in equation (3) for the evaluation of C_{m1_x} , the moles/l of complexed metal.

$$\text{let } C_{m1_x} = z \text{ moles/l}$$

then at equilibrium:

$$C_m = \left(\frac{1.225 \text{ g/l Fe}}{55.847 \text{ g/mole}} - z \right)$$

or

$$C_m = (0.219 - z) \text{ moles/l}$$

and

$$C_l = \left(\frac{6.360 \text{ g/l acetic acid}}{60.05 \text{ g/mole}} - z \right)$$

or

$$C_l = (0.106 - z) \text{ moles/l}$$

from the Davies Equation for +2 charged ionic species:

$$\gamma F_e = 0.35$$

and from experimental results:

$$K = 47$$

The solution of the quadratic equation produced by the above substitutions shows that the amount of complexed iron in the sample is 0.0132 moles/l or 737 mg/l. This means that 60 percent of the iron in the leachate sample of Table 5 is complexed and 488 mg/l exist as free metal. The previously determined solubility limitation for iron is 312 mg/l and there is no real disparity between the two values considering the estimations involved in activity coefficient calculations.

CHEMICAL ANALYSIS

Leachate is a difficult material to analyze because the types of compounds present cause interferences with standard chemical methods. Chian and DeWalle have extensively evaluated the applicability of various chemical tests for leachate analysis and published their recommendations in an Environmental Protection Agency (EPA) report (15). It is evident

from their work that very few procedures are free from interference and the method of standard additions must be employed for accurate results. This makes analysis very time-consuming and so the author has organized the chemical constituents of leachate into five categories with suggestions of representative compounds to facilitate routine analysis. The chemical methods for analysis of the suggested parameters are all relatively interference-free when used for leachate analysis.

1. Heavy Metals

Perhaps all members of the transition elements could be found by trace analysis, but iron is predominant and presents no special analytical problems. Acidification to below $\text{pH} = 3$ is all that is required to eliminate organic acid complexation (by completely protonating the acids) and standard additions is suggested to evaluate any matrix interferences.

2. Anaerobic Metabolites

Volatile organic acids, the ammonium ion, and the oxidation-reduction potential are representative of this classification. Organic acid analysis can be accomplished by specific GC identification or the column-partition chromatographic method described in Standard Methods (10). Neither one is subject to interference from leachate constituents. The ammonium determination may be adequately performed by either the specific ion electrode or the Nesslerization method. The former determines the activity of the ion and so dilution is necessary for concentration readings. The latter requires pretreatment with strong base to $\text{pH} = 10$ for metals precipitation with subsequent decantation of the clarified liquor. Nessler's reagent is added without distillation and the resulting solution should be free of turbidity, indicating complete metals removal. The ORP is a simple electrode measurement whose analytical interferences involve nonreversible redox reactions which are entirely unavoidable.

3. Nutrients

The ammonium ion is the only nutrient present in significant quantity for consideration. Analytical techniques are described in the preceding section.

4. General Organics

Organic loading is most easily accomplished by a TOC determination in conjunction with the COD test for approximate oxygen demand. Biochemical oxygen demand (BOD) analysis is not recommended because of extreme dilutions (up to 2000X) and leachate seed necessary to overcome toxicity.

5. General Inorganics

The best determinations for inorganic loading are conductivity, total solids, pH, and sodium. None are subject to leachate interferences and they provide accurate indication of leachate intrusion. Calcium, sulfate, and chloride analyses have been determined by the author to be particularly susceptible to components in leachate, requiring careful pretreatment.

SUMMARY

Landfill leachate is characterized by odor, oxygen demand, heavy metals, solids, and a conglomeration of organics. Ferrous iron is the predominant metal within a very narrow ORP range. Its presence limits the concentration of phosphorus and its complexation with the volatile organic acids enhances its solubility. The organic acids contribute strong odor to leachate samples and are found to be the principal organic constituent of young landfills. These acids are easily treated by both aerobic and anaerobic systems and the weak complexation formed with iron ions will not affect these methods. Chemical analysis of leachate is difficult and time-consuming, but several parameters have been suggested for routine work.

ACKNOWLEDGMENTS

The author wishes to express sincere appreciation to Dr. D. D. Adrian, for his advice and guidance, to Drs. E. J. La Motta and O. T. Zajicek for their suggestions on experimental design and data analysis, and to Dr. E. E. Lindsey for the loan of his ultrafiltration apparatus. The efforts of the researchers and analysts of the "Leachate Crew" are also gratefully acknowledged. This research forms part of a M. S. Environmental

Engineering Thesis submitted to the University of Massachusetts Graduate School in December, 1977. The work was funded under the Massachusetts Division of Water Pollution Control, Grant Number MDWPC 76-10(2).

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ROLE OF ALGAE IN LEACHATE TREATMENT

Paul A. Walker, Environmental Engineer
Hollingsworth & Vose Company
East Walpole, Massachusetts 02032

Donald Dean Adrian
Professor of Civil Engineering and Director of the
Environmental Engineering Program
University of Massachusetts, Amherst Massachusetts 01003

INTRODUCTION

Recently leachate generated at sanitary landfills has been recognized as a carrier of high concentrations of pollutants. Under pressure from state and Federal regulatory agencies the solid waste industry has been searching for low cost leachate treatment methods. Field observations by University of Massachusetts environmental engineers of algal blooms in dilute leachate pools prompted the authors to study leachate treatment using algal lagoons. A modified version of the Algal Assay Procedure Bottle Test, AAPBT (1), was adapted to study the effects of sanitary landfill leachate on algal growth.

MATERIALS AND METHODS

Unialgal cultures of the green alga Scenedesmus dimorphous were used in this study. These cultures were maintained in 3.78 liter (1 gal.) glass culture vessels containing AAPBT synthetic algal nutrient medium. Because several recommended nutrients were unavailable, the following substitutions were made: 12.170 mg/l $MgCl_2 \cdot 6H_2O$ for 5.700 mg/l $MgCl_2$, 415.543 $\mu g/l$ $MnCl_2 \cdot 4H_2O$ for 264.264 $\mu g/l$ $MnCl_2$, 1.429 $\mu g/l$ $CoCl_2 \cdot 6H_2O$ for 0.780 $\mu g/l$ $CoCl_2$, and 0.01073 $\mu g/l$ $CuCl_2 \cdot 2H_2O$ for 0.009 $\mu g/l$ $CuCl_2$. Cultures were incubated in a constant temperature control room at a temperature of $24 \pm 2^\circ C$. Illumination was provided by two ceiling lamps, each equipped with two 48-inch cool white fluorescent bulbs. Light intensity adjacent to the vessels at liquid level was approximately 400 foot-candles*, as recommended by Trainor (2). The light cycle was

*Measured with a Weston Illumination Meter, Model 756.

14 hours of light and 10 hours of darkness, as suggested by Myers and Graham (3). Culture vessels were rotated daily in an attempt to correct for any local differences in light intensity. All other test procedures are outlined in the AAPBT. Samples of sanitary landfill leachate were taken from the Barre Massachusetts landfill site. A test vessel was made up for each of four dilutions of leachate. Three controls were prepared in a manner similar to that used for the test vessels except that no leachate was added.

A complete physical-chemical analysis of the leachate was made immediately before the test period. Two liter volumes of each of the test vessels, together with the control vessels, were inoculated with an appropriate volume of Scenedesmus dimorphous cells to result in a starting concentration of 1×10^4 cells/ml. All vessels were incubated in a climate control room under test conditions described above. To permit adequate gas exchange, each of the vessels was hand swirled twice daily, 20 times in the morning and 20 times in the evening. Growth of algae was monitored over a two week test period. The Sedgewick-Rafter counting procedure was adapted from Standard Methods (4). Counts were obtained at a magnification of 100X using a 10X ocular and a 10X objective and a Whipple disc placed in the eyepiece. Algal cells were counted in 10 or more random fields of the Whipple grid, as recommended by Moore (5). For selecting locations of fields to be counted, a procedure similar to that of Ingram and Palmer (6) was followed. Cell counts were made on days 3, 5, 7, 9, 11, and 14, as suggested by the Inter-Laboratory Precision Test (7).

RESULTS AND DISCUSSION

For the two preliminary assays, samples of pure leachate were taken at the Barre landfill site during the weeks of January 5, and January 9, 1976. Results of physical-chemical analyses of these samples are shown in Table 1. Constituent materials of leachate dilutions for Runs #1 and #2 immediately before the start of the two week test period appear in Table 2. Growth curves for Runs #1 and #2 are presented in Figures 1 and 2, respectively. In Run #1, both

Table 1. Analyses of Pure Leachate for Run #1 and #2

<u>Parameter</u>	<u>Run #1</u>	<u>Run #2</u>
Total Solids (mg/l)	10,800	13,200
Volatile Solids (mg/l)	3930	4480
COD (mg/l)	16,300	17,000
pH	5.30	5.75
Alkalinity (mg/l CaCO ₃)	4000	6500
Chloride (mg/l)	575	10.2
Phosphate-Phosphorus (mg/l)	0.0280	0.0500
Ammonia-Nitrogen (mg/l)	362	488
Nitrate-Nitrogen (mg/l)	2.42	4.47
Specific Conductance (μmho/cm)	-	>8000

undiluted leachate and 1/2 (i.e., 1 part leachate:2 parts total volume) leachate were toxic to algae and resulted in a zero cell count by day 3. 1/10 leachate and 1/100 leachate were also toxic to algae. The cell concentration first decreased then appeared to stabilize significantly below the starting concentration.

In Run #2, 1/100 leachate again proved toxic to algae. Number of cells decreased during the first week from the starting concentration, then stabilized at approximately 200 cells/ml by day seven. In both 1/1000 leachate and 1/10,000 leachate, the number of cells increased over a period of 14 days to numbers comparable to the controls. 1/500 leachate, however, was inhibitory to algal growth. Based upon the results of these two exploratory assays, dilution factors of 1/100, 1/500, 1/1000, and 1/2,000 were used to make up leachate dilutions for most of the subsequent assays.

For Run #3, the sample of pure leachate, taken during early February, 1976, (see Table 3) was significantly different than the samples used in the exploratory assays. 1/100 leachate was definitely inhibitory to algae, as manifested by a lag period of three days (see Figure 3). Inhibitory effects declined with time, and a fairly normal growth pattern was established except for an apparent die-off

Table 2. Constituent Materials of Leachate Dilutions for Run #1 and #2

Parameter	Run #1				Run #2			
	Undiluted	1/2	1/10	1/100	1/100	1/500	1/1000	1/10,000
Dilution Ratio	Undiluted	1/2	1/10	1/100	1/100	1/500	1/1000	1/10,000
Total Solids (mg/l)	10,700	5390	1110	178	158	61.0	33.0	19.0
Volatile Solids (mg/l)	3910	1990	415	62.0	60.0	33.0	26.0	10.0
COD (mg/l)	16,200	8240	1690	269	214	150	11.9	0.0
pH	5.30	5.35	5.35	5.05	5.75	6.00	6.20	6.55
Alkalinity (mg/l CaCO ₃)	3850	1950	293	32.5	65.0	20.5	15.0	13.0
Chloride (mg/l)	525	275	67.5	12.0	8.00	7.00	7.00	6.00
Phosphate-Phosphorus (mg/l)	0.00700	0.00800	0.275	0.288	0.165	0.0550	0.140	0.0300
Ammonia-Nitrogen (mg/l)	370	115	27.5	2.70	3.52	0.820	0.550	0.220
Nitrate-Nitrogen (mg/l)	3.07	0.182	1.92	2.77	1.94	1.82	1.98	1.92
Specific Conductance (μmho/cm)	-	-	-	-	450	200	160	140

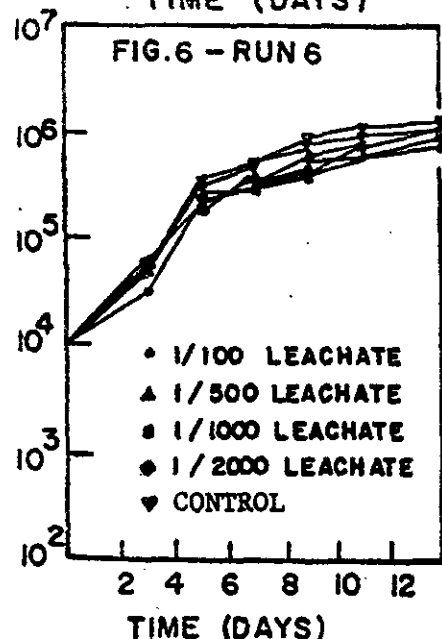
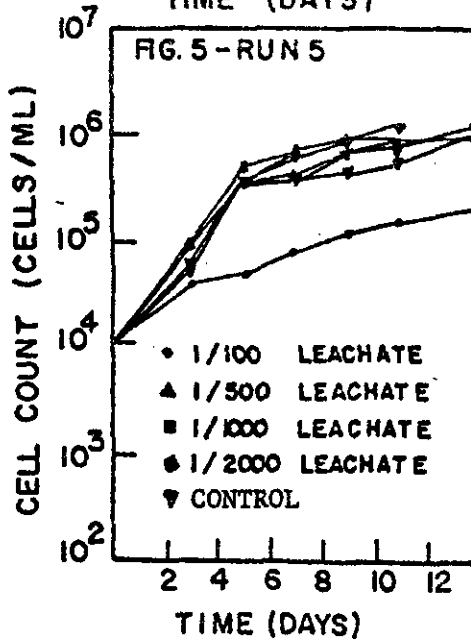
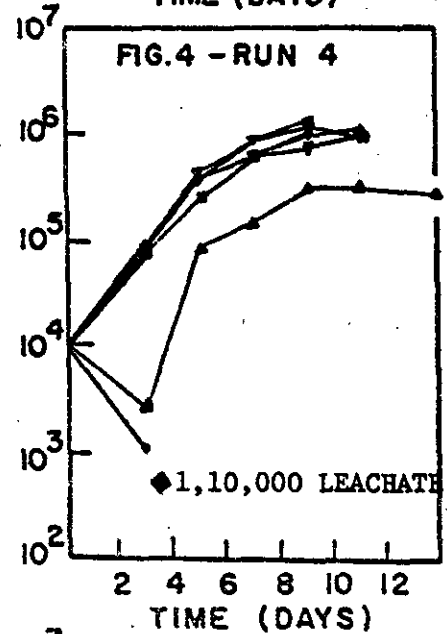
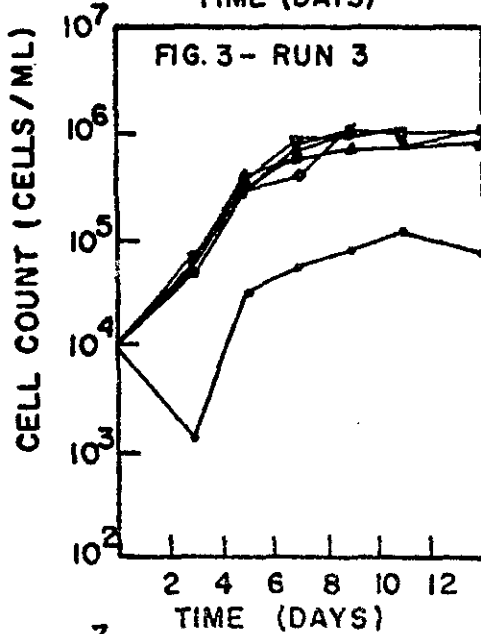
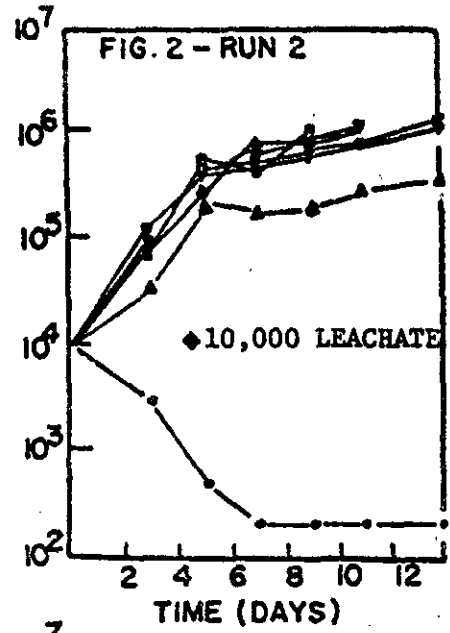
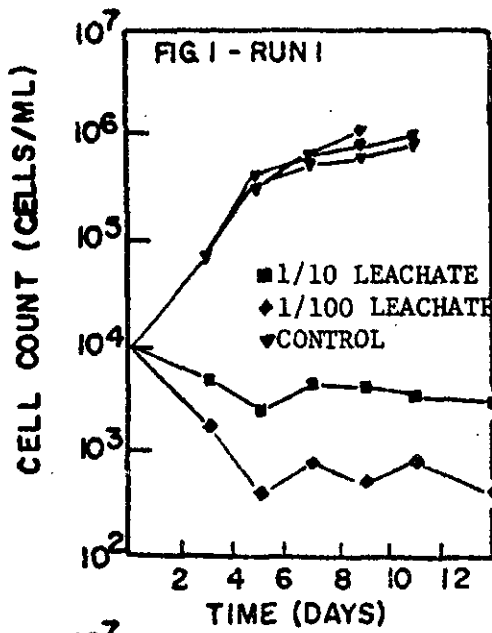


Table 3. Constituent Materials of Leachate Dilutions for Run #3

Parameter	Concentration				
	1/1	1/100	1/500	1/1000	1/2000
Dilution	1/1	1/100	1/500	1/1000	1/2000
Total Solids (mg/l)	4330	85.00	80.0	87.0	72.0
Volatile Solids (mg/l)	1610	63.0	62.0	54.0	52.0
COD (mg/l)	5760	40.0	12.0	4.00	0.0
pH	5.75	5.80	5.20	6.75	6.80
Alkalinity (mg/l CaCO ₃)	1300	25.0	5.50	13.5	12.5
Chloride (mg/l)	57.6	9.50	7.35	7.50	7.25
Phosphate-Phosphorus (mg/l)	0.220	0.360	0.110	0.0800	0.0450
Ammonia-Nitrogen (mg/l)	85.0	1.50	0.800	0.0420	0.0560
Nitrate-Nitrogen (mg/l)	1.69	2.03	4.97	2.65	2.49
Specific Conductance (µmho/cm)	8000	230	170	150	140

between day 11 and 14. The growth patterns exhibited by 1/500 and 1/1000 leachates were similar to each other. In contrast to 1/1000 leachate, however, maximum standing crop in 1/500 leachate was much lower than those of the controls. The growth curve in 1/2000 leachate followed the normal growth pattern of the controls. On day 14, cell density was so high that clumping occurred, making an accurate count impossible. A similar phenomenon was observed in each of the controls.

Analyses of constituent materials of the sample of pure leachate for Run #4, which was taken during late February 1976, revealed that this leachate possessed greater pollutional strength (refer to Table 4), and a leachate dilution of 1/10,000 was included.

1/100 leachate was toxic to algae (see Figure 4). Number of cells decreased from the starting concentration of 1×10^4 cells/ml to 1200 cells/ml by day 3 and 0 cells/ml by day 5. Algal growth in 1/500 leachate demonstrated inhibition as manifested by a lag period of three days. After this initial lag period, a fairly normal growth pattern was observed as in the case of 1/100 leachate in Run #3. Again, a slight die-off took place toward the end of the two week test period. The growth patterns followed by 1/1000 and 1/10,000 leachates were very similar to each other. In both of these leachate dilutions, clumping took place by day 14, indicating maximum standing crops in excess of 1×10^6 cells/ml.

Table 4. Constituent Materials of Leachate Dilutions for Run #4

Parameter	1/1	1/100	1/500	1/1000	1/10,000
Dilution	1/1	1/100	1/500	1/1000	1/10,000
Total Solids (mg/l)	18,800	159	127	103	104
Volatile Solids (mg/l)	6150	87	45	54	53
COD (mg/l)	24,900	252	51.2	19.7	3.90
pH	5.40	5.60	5.90	6.25	6.60
Alkalinity (mg/l CaCO ₃)	5900	66.0	19.5	16.0	12.0
Chloride (mg/l)	800	14.0	8.80	7.80	6.00
Phosphate-Phosphorus (mg/l)	0.220	0.215	0.330	0.0390	0.187
Ammonia-Nitrogen (mg/l)	340	4.00	0.800	0.420	0.200
Nitrate-Nitrogen (mg/l)	3.19	1.84	1.80	1.84	1.87
Specific Conductance (μmho/cm)	16,000	540	225	185	160

The constituent materials of the pure leachate for Run #5 which was taken late in March 1976, appear in Table 5, together with the constituent materials of diluted leachate. Unlike previous runs, leachate dilutions of 1/500 and greater seemed stimulatory to algae (see Figure 5). Growth rate appeared to increase with an increase in leachate concentration.

Table 5. Constituent Materials of Leachate Dilutions for Run #5

Parameter	1/1	1/100	1/500	1/1000	1/2000
Dilution	1/1	1/100	1/500	1/1000	1/2000
Total Solids (mg/l)	6190	143	71.0	56.0	58.0
Volatile Solids (mg/l)	2250	41.0	18.0	11.5	15.0
COD (mg/l)	8670	88.3	28.1	20.1	12.0
pH	5.45	5.85	6.40	6.60	6.70
Alkalinity (mg/l CaCO ₃)	2500	31.0	14.0	2.0	11.5
Chloride (mg/l)	175	9.00	7.50	7.50	6.75
Phosphate-Phosphorus (mg/l)	0.380	0.300	0.230	0.210	0.340
Ammonia-Nitrogen (mg/l)	128	1.66	0.530	0.290	0.230
Nitrate-Nitrogen (mg/l)	4.29	2.03	2.18	2.30	2.03
Specific Conductance (μmho/cm)	12,000	300	150	150	130

In terms of pollutional strength, the sample of leachate for Run #6, which was taken during the week of April 19, 1976, was the weakest of all samples tested (refer to Table 6). As shown in Figure 6, leachate seemed to be inhibitory to algae over the entire range of dilution factors as manifested by smaller maximum standing crops.

Table 6. Constituent Materials of Leachate Dilutions for Run #6

Parameter	Concentration				
	Dilution	1/1	1/100	1/500	1/1000
Total Solids (mg/l)	2780	92.0	82.0	74.00	88.0
Volatile Solids (mg/l)	474	53.0	52.0	50.0	51.0
COD (mg/l)	43.3	7.90	7.90	0.0	0.0
pH	6.10	6.65	6.70	6.80	6.20
Alkalinity (mg/l CaCO ₃)	1200	12.5	10.5	10.5	6.50
Chloride (mg/l)	10.0	6.25	5.75	6.50	6.50
Phosphate-Phosphorus (mg/l)	0.640	0.280	0.250	0.160	0.190
Ammonia-Nitrogen (mg/l)	0.900	0.360	0.190	0.200	0.220
Nitrate-Nitrogen (mg/l)	0.390	1.68	1.80	2.02	18.7
Specific Conductance (μmho/cm)	660	148	140	138	135

In summary, where leachate concentrations were greater than those that algae could tolerate, the effects were dramatic. Growth curves exhibited rapid decay, as in the cases of all the leachate dilutions for Run #1, 1/100 leachate in Run #2, and 1/100 leachate in Run #4. In two other instances, 1/100 leachate in Run #3 and 1/500 leachate in Run #4, leachate concentrations were obviously inhibitory to algal growth, as manifested by observable lag periods.

Little has been said, however, about the cases where leachate concentrations did not seem so obviously inhibitory, or indeed stimulatory, to algae. For these cases, where algal growth could be described by an asymptotic curve, a statistical approach was deemed necessary. A modified version of the logistic curve was developed and used to describe algal growth. The relationship between growth and time is given by:

$$y = \frac{1}{\alpha + \beta e^{-\gamma x^\theta}} \quad (1)$$

where

y = log cell count

x = time, in days

$\alpha, \beta, \gamma, \theta$ = parameters

Taking the reciprocal of y , equation (1) becomes:

$$\frac{1}{y} = \alpha + \beta e^{-\gamma x^\theta} \quad (2)$$

in which α represents the asymptotic value of $\frac{1}{y}$. β represents the change in $\frac{1}{y}$ as x goes from zero to infinity. The term $e^{-\gamma x^\theta}$ represents the factor by which the deviation of $\frac{1}{y}$ from its asymptote is changed as x changes, and γ represents the sign of the change. If $\gamma > 0$, then the deviation of $\frac{1}{y}$ is reduced; if $\gamma < 0$, then $\frac{1}{y}$ is increased. θ represents the rate at which $\frac{1}{y}$ approaches its asymptotic value. The greater the value of θ , the faster $\frac{1}{y}$ reaches its asymptotic value.

Since at time $x = 0$, $y = 4.0$ (i.e., the starting concentration of cells was 10,000 cells/ml), equation (1) reduces to:

$$y = \frac{1}{.250 + \beta(e^{-\gamma x^\theta} - 1)} \quad (3)$$

where $\alpha = .250 - \beta$. In order to obtain a least-squares fit of the data to equation (3), a computer program entitled BMDP3R Nonlinear Regression (8) was employed. The values for β , γ , and θ and their respective standard deviations from the iteration which had the smallest residual sum of squares appear in Table 7.

To determine whether various dilutions of leachate were stimulatory or inhibitory to algae in those cases where growth was not obviously different from growth in the controls, the following procedure was used. First, the equality of the estimates for the three parameters in the growth curves of the controls were tested. Confidence intervals at the 95 percent confidence limits were constructed. If the confidence

Table 7. Estimates of β , γ , and θ , Together with Their Standard Deviations

Run/Vessel/Dilution	Degrees of Freedom	$\hat{\beta}/s(\hat{\beta})$	$\hat{\gamma}/s(\hat{\gamma})$	$\hat{\theta}/s(\hat{\theta})$
2/6/(1/500)	57	0.06343/0.007418	0.01331/0.007765	3.447/0.4915
2/7/(1/10 ³)	32	0.08164/0.0006171	0.08140/0.01075	2.090/0.1067
2/8/(1/10 ⁴)	42	0.08654/0.002037	0.2253/0.01833	1.100/0.07730
2/9	52	0.07968/0.0005218	0.1772/0.02429	1.560/0.1062
2/10 Biological	52	0.08192/0.001160	0.2786/0.03649	1.125/0.1030
2/11 Controls	42	0.08453/0.0007972	0.3052/0.01939	1.102/0.0565
3/10/(1/500)	57	0.07987/0.0003521	0.09400/0.01126	1.938/0.08948
3/11/(1/10 ³)	52	0.08069/0.0004278	0.06620/0.008546	2.045/0.09130
3/12/(2x10 ³)	38	0.3997/—	0.07562/0.004827	0.4953/0.03140
3/13	37	0.08352/0.0005036	0.08277/0.008023	1.944/0.07349
3/14 Biological	37	0.08415/0.006700	0.1079/0.01074	1.739/0.07716
3/15 Controls	37	0.08420/0.0006425	0.1021/0.009374	1.710/0.06769
4/15/(1/10 ³)	37	0.08486/0.0007174	0.1692/0.01091	1.371/0.05112
4/16/(1/10 ⁴)	37	0.08374/0.0007512	0.1919/0.01772	1.407/0.07414
4/17	32	0.08565/0.001041	0.1377/0.01837	1.631/0.1083
4/18 Biological	32	0.08632/0.0006775	0.1567/0.009180	1.500/0.05029
4/19 Controls	47	0.08368/0.0006940	0.1760/0.01900	1.419/0.08046
5/17/(1/100)	57	0.1184/0.05308	0.1626/0.06451	0.5826/0.1031
5/18/(1/500)	37	0.08153/0.0004094	0.1227/0.01118	1.829/0.07377
5/19/(1/10 ³)	47	0.08291/0.0002746	0.1360/0.007493	1.621/0.04116
5/20/(2x10 ³)	52	0.08155/0.0007547	0.1400/0.02062	1.543/0.1062
5/21	52	0.07900/0.001198	0.2756/0.04125	1.143/0.1172
5/22 Biological	52	0.08257/0.001228	0.1984/0.02752	1.252/0.1031
5/23 Controls	42	0.08305/0.0005752	0.07957/0.009454	1.906/0.08610
6/21/(1/100)	57	0.07570/0.0006052	0.05389/0.01020	2.042/0.1261
6/22(1/500)	57	0.07930/0.001394	0.1973/0.03042	1.203/0.1113
6/23(1/10 ³)	52	0.08631/0.003237	0.2555/0.02703	0.9423/0.09847
6/24(2x10 ³)	52	0.08184/0.0007339	0.1563/0.01158	1.308/0.05337
6/25 Biological	47	0.08510/0.0008244	0.1444/0.01420	1.410/0.06990
6/26 Controls	52	0.08117/0.0005188	0.1083/0.01314	1.716/0.08660

*Since β was redundant (not pivoted on) or lay on a boundary, the computer assigned a standard deviation of 0.0.

interval contained a zero, the growth curves were the same. Second, for those controls whose growth curves were the same, predicted estimates of log cell count on day 14 were pooled, as were their variances. Such a procedure for projecting log cell count on day 14 was repeated for test vessels. Finally, for determining whether various dilutions of leachate were stimulatory or inhibitory to algal growth, equality of estimates of log cell count on day 14 for each test vessel and pooled estimates, where applicable, of log cell count on day 14 for the controls were tested. Again, confidence intervals at the 95 percent confidence limits were constructed. For the purposes of this investigation, maximum standing crop was defined as log cell count on day 14. The results of testing for equality are presented in Table 8.

Overall, dilution factors of 1/100 (or less) proved inhibitory and even toxic, to algal growth, while leachate dilutions of 1/500 were inhibitory. One exception to this rule was Run #5 in which maximum standing crop of 1/500 leachate was statistically the same as that of the controls. For most of the test runs, a dilution factor of 1/1000 resulted in the same maximum standing crop in both test vessels and controls. In Run #3, however, 1/1000 leachate was inhibitory to algae, while in Run #5, it was stimulatory, and growth in greater dilutions of leachate (specifically, 1/2000 and 1/10,000 leachate) followed even more dissimilar trends.

To provide some basis for design of an algal lagoon for leachate treatment, a dilution factor based upon one or more constituent materials of leachate seemed extremely helpful. As a means to that end, a regression analysis was performed with 1/100 leachates, 1/500 leachates, 1/1000 leachates, 1/2000 leachates, and 1/10,000 leachates; i.e., those dilutions of leachate which had yielded the most meaningful results. For determining the prediction equation for log maximum standing crop, a stepwise regression analysis was performed. The dependent variable, log maximum standing crop, was regressed on the starting concentrations of nine constituent materials of leachate analyzed. pH was not included in this regression analysis. F-tests for multiple regression were performed at the 95 percent confidence limits. At these confidence limits, the best regression equation (R squared of 0.9259) was:

Table 8. Results of Testing for Equality of Log Cell Count on Day 14 for Biological Controls and Test Vessels

Test Run	Test Vessel (Leachate Dilution)	$\hat{Y}_p - \hat{Y}_i$	$1.96\sqrt{s^2(\hat{Y}_p) + s^2(\hat{Y}_i)}$	Comments
2	6(1/500)	0.6210	0.05740	Inhibitory
	7(1/1000)	0.04100	0.05614	Same
	8(1/10,000)	-0.08300	0.08439	Same
3	10(1/500)	0.1420	0.03318	Inhibitory
	11(1/1000)	0.1140	0.03726	Inhibitory
	12(1/2000)	-0.5320	0.1553	Stimulatory
4	15(1/1000)	0.01000	0.05124	Same
	16(1/10,000)	0.04600	0.05603	Same
5	17(1/100)	0.5580	0.05826	Inhibitory
	18(1/500)	-0.03580	0.05437	Same
	19(1/1000)	-0.08500	0.05073	Stimulatory
	20(1/2000)	-0.03560	0.06947	Same
6	21(1/100)	0.3190	0.06517	Inhibitory
		0.1860	0.05274	Inhibitory
	22(1/500)	0.2220	0.08323	Inhibitory
		0.08900	0.07390	Inhibitory
	23(1/1000)	0.09300	0.09426	Same
		-0.04000	0.08613	Same
	24(1/2000)	0.1300	0.06452	Inhibitory
	-0.003000	0.05193	Same	

$$\hat{Y} = 8.091 = .014PARAM9 \quad (4)$$

where \hat{Y} is the estimate of log maximum standing crop and PARAM9 is the starting value of specific conductance.

In order to construct the regression line, the independent variable, specific conductance, was plotted against the dependent variable, log maximum standing crop predicted from the regression equation. A plot of specific conductance vs. values of log maximum standing crop from both the growth curve and the regression equation is presented in Figure 7. For those cases where a modified version of the logistic curve was not used to describe algal growth (i.e., growth did not tend asymptotically toward a limit), actual observed values of log maximum standing crop were plotted. The results show that the regression equation was a good fit to the observed values of log maximum standing crop.

Equation 4, with a multiple R squared of 0.9259, implies that a good overall measure of toxicity of leachate to algae is specific conductance. Analysts of leachate have expressed a similar viewpoint (9). Another way of stating this is that specific conductance is a strong indicator of the algal growth potential of leachate. In order, then, to ensure that leachate does not enter treatment lagoons in concentrations that algae are unable to tolerate, the resulting specific conductance should be less than 200 $\mu\text{mho/cm}$. This translates to a dilution factor of greater than 1/100 for the pure leachate generated at the Barre landfill site. This has been substantiated by the preliminary field observations of Lavigne (10), who observed that algae flourished in pools of leachate which had undergone dilutions of 1/100 or greater.

CONCLUSIONS

1. Overall, dilution factors of 1/100 or less proved inhibitory or toxic.
2. Leachate dilutions of 1/500 were inhibitory to algal growth in most cases.
3. The single variable which best predicted the effects of leachate on algal growth was specific conductance.
4. In order to ensure that leachate does not enter a treatment lagoon in concentrations that algae are unable to tolerate, the resulting specific conductance should be less than 200 $\mu\text{mho/cm}$.

ACKNOWLEDGMENTS

We wish to thank Dr. Jinnque Rho for his suggestions in designing the experiment and Dr. Otto L. Stein for photomicrographs of the test algae. We wish to acknowledge the efforts of Ronald Lavigne, Robert Pease, Franklin Tirsch, George Lombardo, Sandra Ferry, and Aaron Jennings who performed physical-chemical analyses. Dr. David and Trina Hosmer helped conduct statistical analyses. We express our appreciation to Miss Dorothy Blasko for her patience in typing the manuscript. The manuscript has been published in the Proceedings of the 32nd Purdue Industrial Wastes Conference and is reproduced by permission (11). It has been reduced from a longer technical report by Walker and Adrian (12). This project was funded by the Massachusetts Division of Water Pollution Control Grant Numbers MDWPC 73-7(2) and 76-10(2) and the U.S. Department of Agriculture, Northeastern Forest Experiment Station through grant 23-591. The cooperation of Leonard Martone, owner of the Barre Landfill, on all phases of the project, is gratefully acknowledged.

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THE ISOLATION OF PATHOGENIC ORGANISMS
IN LANDFILL LEACHATE

Sandra Ferry, Research Assistant
David Ferguson, Research Assistant

Environmental Engineering Program
Department of Civil Engineering
University of Massachusetts, Amherst

INTRODUCTION

There are many possible sources of bacteria in landfills. Those introduced in fecal matter are the primary focus of this work. Animal wastes have long been a part of solid waste in landfills and with the increasing use of disposable diapers, human fecal matter has also become a significant component. Peterson (1974) showed that disposable diapers compose between 0.6 and 2.5 percent of solid waste in landfills. This amounts to between 1,800,000 and 7,500,000 tons annually. Fecal coliform and fecal streptococci counts have been found to be over a million organisms per gram of raw residential solid waste (Gaby 1975). If these bacteria can survive in a landfill environment and be transported in leachate, then bacterial contamination of ground and surface waters by leachate poses a potential health hazard.

LITERATURE REVIEW

Several studies have been done to determine bacterial survival and the attenuating properties of soils. One such study was carried out by Blannon and Peterson (1973) on an experimental landfill to determine the presence of fecal coliforms and fecal streptococci in leachate. The landfill was a trench 149 ft long by 30 ft wide (see Figure 1). The bottom of the trench was lined with a 30 mil Hypalon liner which was covered with 18 inches of clay. It was equipped with two pipes for leachate collection, one above the clay and one below it. Two feet of

soil covered the landfill. The first leachates occurred about six weeks after completion. During the first two months of the study the counts from the upper pipe averaged 1.5 million fecal coliform and 48 million fecal streptococci per 100 ml. The lower pipe leachate averaged 280,000 fecal coliform and 460,000 fecal streptococci per 100 ml (see Figures 2 and 3). In the third month, the fecal coliform population dropped sharply while the number of fecal streptococci remained relatively high. The fecal coliform count declined to only 30 organisms per 100 ml from the upper pipe and 20 organisms per 100 ml from the lower pipe. The fecal streptococci ratio ranged from .01 to 6.2 indicating contamination from both human and animal sources. However, due to the rapid die-off the usefulness of this ratio is questionable. Further biological classification of the strains of streptococci showed that one-third were S. faecalis Biotypes II and III, S. durans, and S. equinus; all indicative of contamination by warm blooded animals. The remaining two-thirds contained strains of S. faecalis liquefaciens and an atypical strain of S. faecalis, both found naturally in the environment. These results suggest that the microbiologic qualities of leachate might be compared to storm water runoff or raw domestic wastewater. Blannon and Peterson concluded that the 18-inch clay liner did not provide sufficient filtering action for bacterial removal.

Soil column studies by Glotzbecker and Novello (1974) on the same leachate contradicted these results. They used columns packed with 10 cm of soil from two different sanitary landfill sites. One consisted of 6 percent sand, 58 percent silt, and 36 percent clay and the other was 20 percent gravel, 54 percent sand, 18 percent silt, and 8 percent clay. A nitrogen atmosphere was maintained over the columns and the temperature held at 10°C. A constant head was maintained over each column with 10 cm of leachate seeded with 10^8 E. coli per 100 ml. Percolates from the clay column contained only 3 organisms per 100 ml while the sandy soil removed over 99 percent of the bacteria. They concluded that uniform soils, particularly with high clay contents, underlying landfills have the capacity to protect groundwater from bacterial contamination.

Glutzbecker and Novello (1974) also did research on the question of bacteria survival time. They found that survival time varied considerably between leachates from the experimental landfill and a municipal landfill. The time required for 99 percent reduction in leachate from the experimental landfill was about 5 hours, but for the municipal landfill it was over 100 days (see Table 1). The survival time was affected by temperature, pH, and other undetermined factors. This may indicate that the bacterial counts from the experimental landfill were low and that the rapid coliform dieoff in the third month may have been due to toxicity of that particular leachate, which is not found in all leachates. This is supported by similar studies of the toxicity of aging leachate on virus (Engelbrecht and Amirhor 1976).

Table 1. Days for 99.9 Percent Reduction of Microorganisms in Leachates at 10° and 20°C (Glutzbecker and Novello 1974)

Microorganism	Experimental Landfill Leachate	Municipal Landfill Leachate	
	10°C	10°C	20°C
<u>E. coli</u>	0.12	56	21
<u>S. faecalis</u>	0.21	100	35

Viral contamination of leachate is another possible result of fecal contamination of landfills. Infants and young children are the most effective carriers of enteric viruses and most have been vaccinated with live trivalent polio vaccine. The vaccine is a mutant form of the virus whose virulence has been reduced. However, it has been found that these viruses, particularly after serial human passage, may revert to strains approaching the virulence of wild strains. Viruses, particularly polio virus 3 have been found in 2.9-16.7 percent of fecally contaminated diapers (Peterson 1974).

Bench scale studies by Cooper *et al.* (1975) were carried out at the University of California to determine the presence and survival of viruses in leachate. Lysimeters filled with ground municipal solid waste were used. Temperature was monitored and water added in quantities

to stimulate natural conditions. The resulting leachate was similar in physical and chemical properties to leachate from other landfills. Viruses were recovered sporadically for up to 20 weeks from the systems seeded with poliovirus 1 and also from systems that had not been seeded. Recovery was low and occurred mainly during the second and third weeks of leachate production.

Cooper et al. (1975) also tested lysimeter leachate for virucidal properties. Initially no viruses could be detected in the seeded leachate. However, the addition of a chelating agent (sodium ethylenediaminetetracetate) to the leachate eliminated the apparent toxicity. They found a masking effect of viable viruses due to components in the leachate rather than toxicity as such, and concluded that the leachates produced were not acutely toxic to poliovirus. These results indicate that viruses do occur in landfills and can be transported in leachates. This is reinforced by Engelbrecht and Amirhor (1976) who concluded that viruses can be recovered from leachates but may be destroyed by heat generated from aerobic digestion in young landfills or by toxic substances that occur in leachates of aging landfills.

METHODS FOR ISOLATION

Sampling at the Martone landfill in Barre, Massachusetts, was done at a ponding area near the current fill site. The leachate in this area is diluted by some surface runoff. At the time of sampling the chelating agent, EDTA, was added to protect the bacteria during transport to the lab.

The MPN method and membrane filtration are the two basic methods available for routine analysis and enumeration of total coliform, fecal coliform, and fecal streptococci. In the case of landfill leachate the membrane filters collect heavy metals, other toxins, and suspended solids. Consequently, the bacterial counts are 10 to 1000 times lower than corresponding MPN values. For this reason the MPN method for coliform and streptococci enumeration was used. The presumptive medium initially used for coliform counts was lactose broth. However, there

were a high number of false positive tubes and lauryl sulfate broth was found to yield better results. There were still problems with this medium for samples of older leachates. In older leachates the organisms appear to be stressed and the more selective medium may in fact be toxic to them. An initial short incubation in lactose broth and then transferal to lauryl sulfate might be effective. With younger leachates this is less of a problem and lauryl sulfate is effective. In transferring cultures from the presumptive medium to the confirmed medium, the use of 1 ml amounts rather than the conventional loopful yielded better results. Since the bacteria were stressed initially, their growth in the presumptive medium may not be as strong as it should have been had they come from a more favorable environment. By using a whole milliliter, a larger number of viable bacteria are transferred to the more selective medium. All other procedures used were standard.

An attempt was also made to isolate Salmonella from leachate. Concentration of the samples is necessary as these organisms are usually present in very small numbers. This was accomplished by filtering 500 ml portions of leachate through a number of membrane filters. The filters were then placed in enrichment media which stimulate the growth of Salmonella while inhibiting other bacteria. The media used for this purpose were Tetrathionate broth and Selenite Cystine broth. After a 24-hour incubation period, plates were streaked on Bismuth Sulfite, XLD, Salmonella Shigella, and Brilliant Green agar. At this time a portion of the broth culture was transferred to fresh broth and reincubated. The new cultures were streaked on the same media after 24 and 48 hours. Selected colonies were restreaked and from these typical Salmonella colonies were put through further biochemical tests.

RESULTS

Samples for coliform and streptococci enumeration were run in triplicate. Reproducibility was good (see Table 2). Average total coliform counts from November, 1977 were 440,000 organisms per 100 ml.

Table 2. Typical Data Set Based on Leachate Sample Taken November 15, 1977.
(All counts x 10^{-5} organisms per 100 ml)

Multiple Tube Test	Replicates			Average
	1	2	3	
Coliform presumptive test	7.0	7.9	3.3	6.1
Confirmed total coliforms	4.9	4.9	3.3	4.4
Confirmed fecal coliforms	3.3	3.3	2.4	3.0
Confirmed fecal streptococci	<u>1.09</u>	<u>0.79</u>	<u>0.49</u>	<u>0.79</u>
fecal coliform/fecal streptococci ratio	3.03	4.18	4.90	4.04

Fecal coliforms averaged 300,000 organisms per 100 ml and fecal streptococci 79,000 organisms per 100 ml. These results are lower than those found by other researchers by a factor of about ten. This may be accounted for by the dilution at the sampling area.

It was found that concentration can play a limiting role. While running fecal streptococci counts from 10^0 to 10^{-5} dilutions, five positive tubes resulted in the 10^{-1} , 10^{-2} , and 10^{-3} dilutions but all of the 10^0 dilutions were negative. This may be due to growth inhibition factors in the leachate which are decreased in subsequent dilutions.

Salmonella are found in landfills only in very small numbers and have been isolated from refuse if first put in an enrichment broth (Gaby 1975). From the Barre leachate the best results for Salmonella isolation were those from the second set of enrichment broth. As described, the samples were concentrated on filters and incubated in enrichment broth for 24 hours. Then a portion of these cultures were transferred to fresh broth and reincubated. It is the plates from this streaking that yielded the best results, particularly on Bismuth Sulfite agar. Colonies from these plates were then selected and re-streaked on one of the other agars. Typical Salmonella colonies were then put through further biochemical tests. These tests included urea agar, triple sugar iron agar, and polyvalent antiserum. Of 75 plates initially streaked, 15 colonies have passed the initial biochemical

tests. At this point the colonies are tentatively identified as *Salmonella*.

CONCLUSIONS

The results of this study agree with the literature findings. High numbers of fecal coliforms and fecal streptococci were isolated from the Barre leachate. *Salmonella* was qualitatively carried to a tentative identification. There are still many bacteria present in leachate that have not been identified. Uniform soils may be effective in removing bacteria but results in the literature are inconclusive. It appears that bacterial contamination of ground and surface waters from leachate is a definite possibility and more research should be done in this area.

ACKNOWLEDGMENT

This research was funded by the Massachusetts Division of Water Pollution Control, Grant Number MDWPC 76-10(2).

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THE APPLICATION OF LANDFILL LEACHATE TO A FOREST AND GRASS BIOME

Dr. H. A. Menser, Plant Physiologist
W. M. Winant, Soil Scientist
O. L. Bennett, Supervisory Soil Scientist

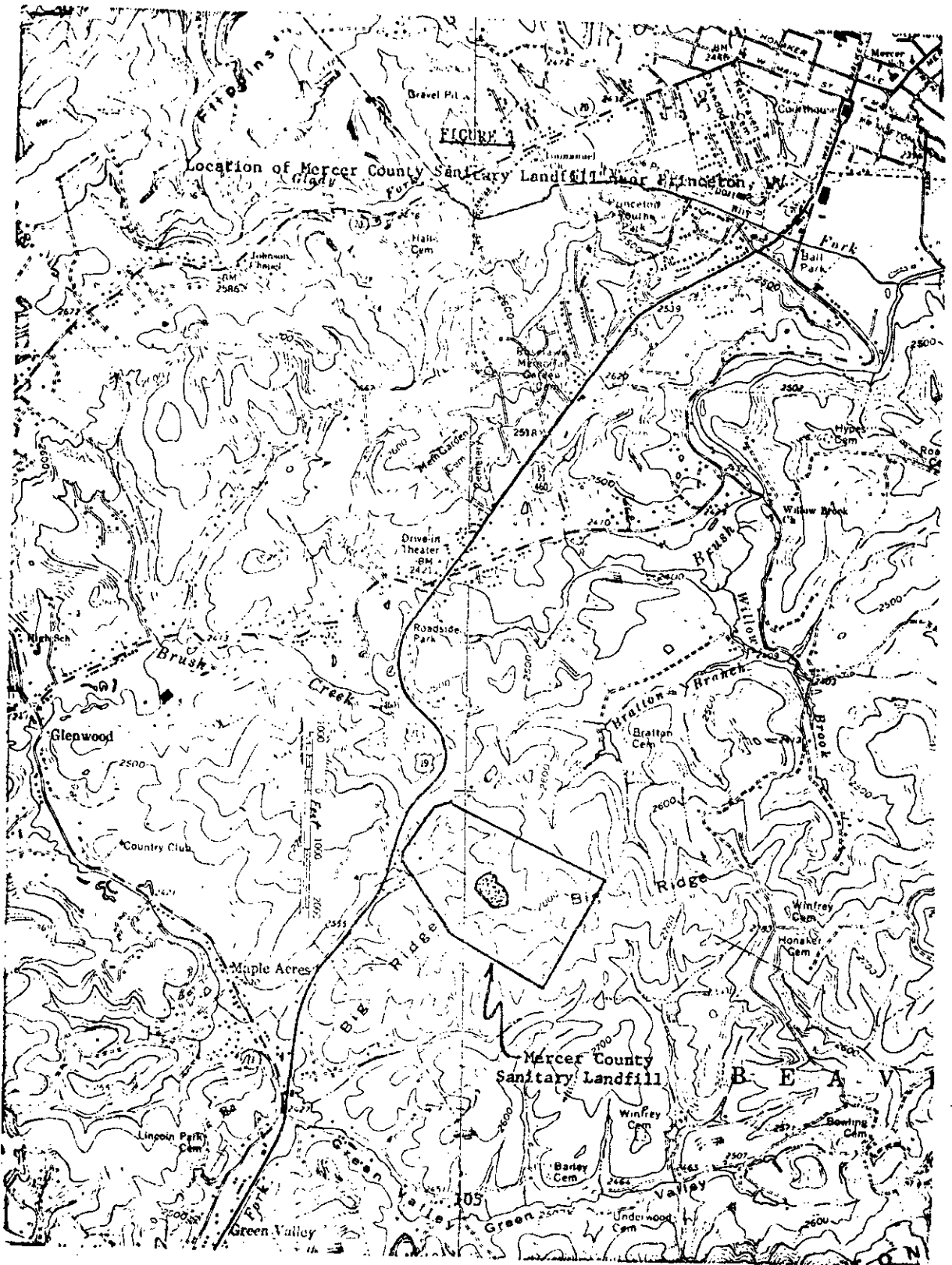
USDA, Agricultural Research Service
Morgantown, West Virginia

INTRODUCTION

The Mercer County Sanitary Landfill was constructed in 1971 in response to the community need for an acceptable means of solid waste disposal. The landfill was built by the Southern West Virginia Regional Health Council with technical assistance from the U. S. Department of Agriculture's Soil Conservation and Agricultural Research Services. About 150 tons of all types of municipal solid waste except sewage sludge and wrecked vehicles are deposited daily by private refuse contractors as well as the general public. User fees compensate for operational costs. The landfill is located along U. S. Route 460 between the cities of Princeton and Bluefield (Figure 1).

Leachate drainage appeared within a few months after the landfill opened. Factors that contributed to the early occurrence of leachate were above average precipitation, highly permeable cover material and the ravine setting of the fill area. Leachate flowed into a small tributary of the Brush Creek Watershed, a source of water for the 10,000 residents of the City of Princeton, West Virginia.

After considering a number of alternatives, Mercer County authorities chose land disposal preceded by leachate collection and primary stabilization as the most practical method of leachate treatment. The ecological impact of land disposal of leachate, however, had never been investigated extensively on a field scale. After consultation with USDA officials, the Regional Health Council decided to apply leachate to a forested ecosystem by using overhead rotary sprinkler irrigation. The Agricultural Research Service initially agreed to act as consultants and



later recognized the opportunity to research the feasibility of spray irrigation as a leachate treatment alternative because no guidelines for field scale use of this method had been established.

Precipitation and temperature vary considerably in West Virginia because of the rugged topography. Precipitation is reasonably distributed throughout the year measuring about 36 inches (90 cm) in the vicinity of the landfill. About 52 inches (130 cm) were recorded in 1972; however amounts have been near normal since then. Temperatures average near 32°F (0°C) and 70°F (21°C) for January and July, respectively.

Soils used for spray irrigation are classified as Typic Hapludults, fine loamy, mixed, mesic, (Gilpin and Shelocta series) Typic Dystrochrepts, loamy-skeletal, mixed, mesic (Berks, Litz), and Lithic Dystrochrepts, loam-skeletal, mixed, mesic (Weikert). These soils developed in situ over a long period of time from Mississippian sandstones and shales. All except the Ernest series are highly permeable exhibiting infiltration capacities of 0.5 to 3.0 inches (1.5 to 7.5 cm) per hour.

Subsurface drains placed beneath the refuse burial area channelled leachate into a catchment basin located at the toe of the landfill (Figure 2). Some surface seepage of leachate also drains into the catchment basin. The retention time of the 405 thousand gallon catchment basin is 19 days. The average leachate generation rate determined from pumping records is about 15 gpm (gallons per minute).

Complaints about noxious odors from community residents led to the installation in 1974 of a 20 HP Peabody-Welles floating surface aerator positioned in the center of the catchment basin. Odors lessened appreciably after the aerator was installed; however, we have not noticed any important changes in leachate quality as a result of aeration.

Other features of the leachate collection and treatment system include stabilization ponds, a spray reservoir, diversion ditches, sediment ponds and the spray irrigation hardware.

The USDA Agricultural Research Service experimental activities began in the fall of 1973 and have continued until the present time. The basic objective of the research has been to develop guidelines for the land disposal of leachate by spray irrigation. Specific objectives were

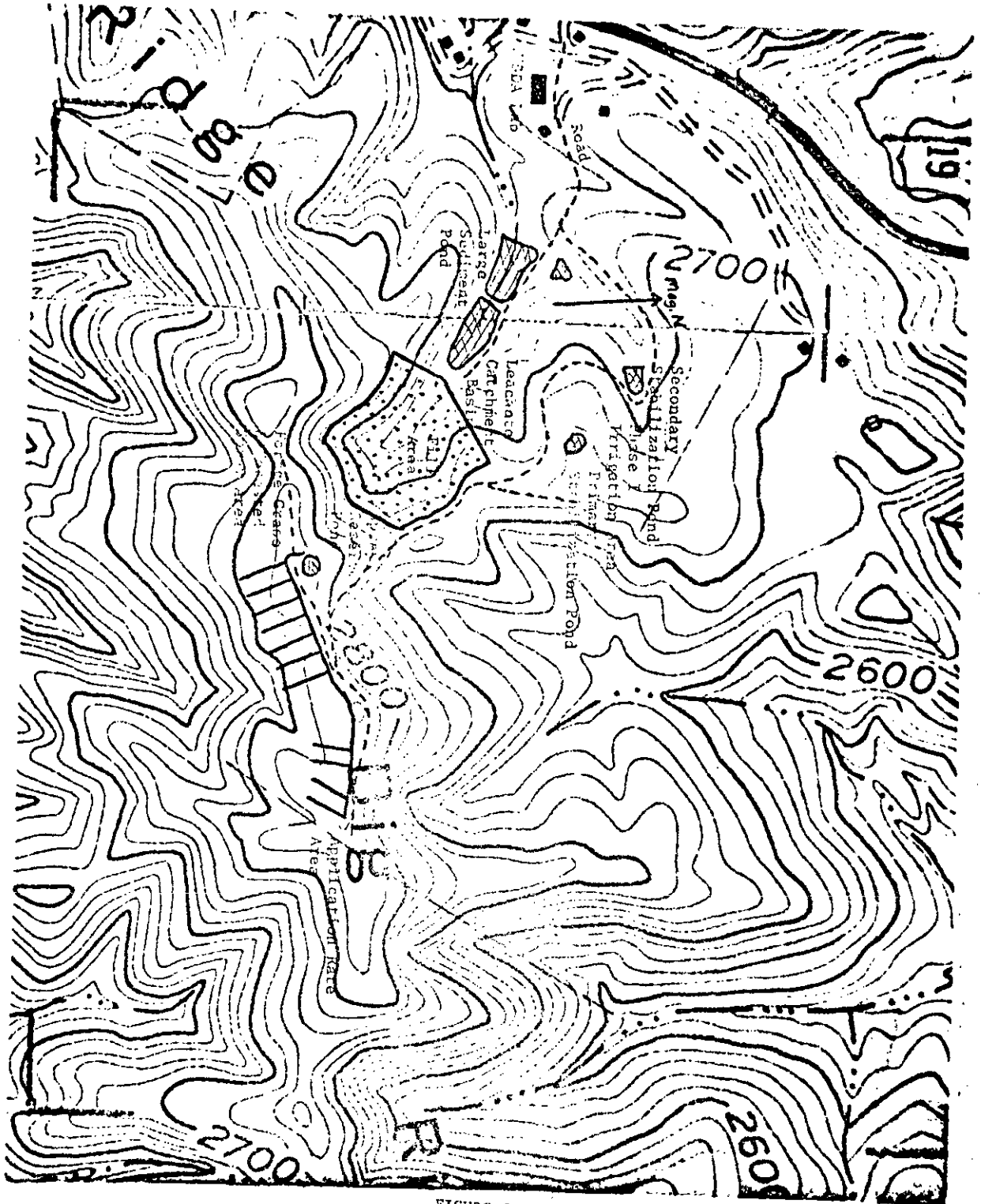


FIGURE 2
Topographic Representation of the Mercer County Sanitary Landfill Showing Principal Features

to determine (1) the capability of soils treated with lime and phosphate fertilizers to complex potentially hazardous heavy metals and to decontaminate wastewater organic pollutants; (2) the tolerance of native and introduced vegetation to spray applications of leachate as reflected by changes in ecosystems; and (3) the effects of leachate irrigation loading rates on soil permeability and associated ecological considerations. Results reported in this paper deal with the first and second objectives.

Three chronological phases of activity have been implemented. The first phase consisted of leachate applications during 1973 and 74 within a small, 5.0 acre (2.2 hectare) self-contained watershed populated with mature hardwoods and understory herbaceous vegetation. Bennett, et al. (1) presented an earlier account of the results of this research phase. Phase II operations, begun in 1974, consist of applying leachate to a native ecosystem and to a selection of six introduced forage grasses. The third phase of research in progress since 1976 consists of applying two rates of leachate to a native ecosystem. Results of this phase are not presented in this report.

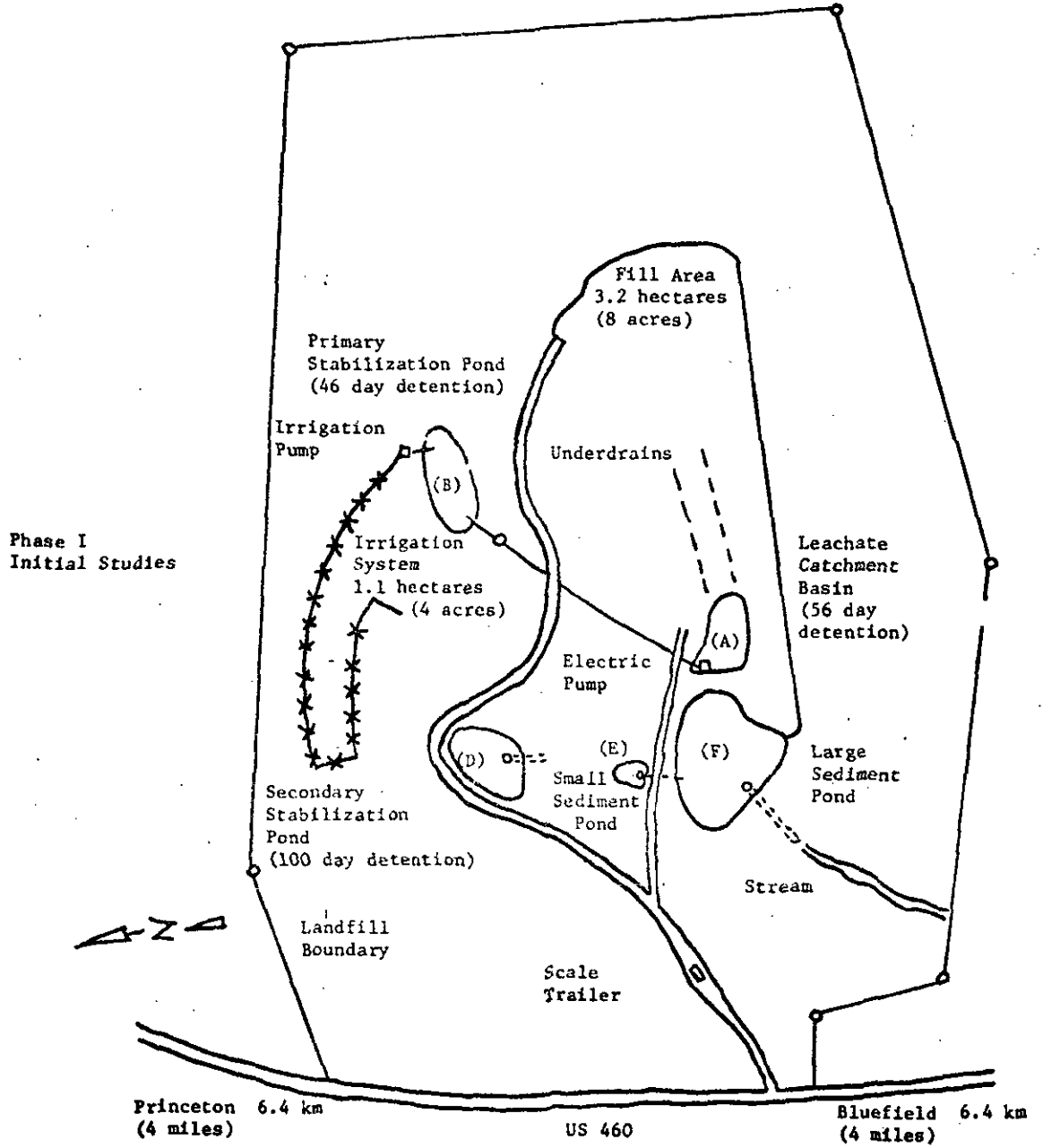
METHODS AND RESULTS

Phase I Studies - Methods

Soil amendments consisting of 5 tons per acre (11.2 MT/ha) each of ground calcitic limestone and raw rock phosphate and 600 pounds per acre (0.67 MT/ha) of superphosphate were broadcast on the surface of replicated plots arranged within 3 randomized blocks situated astride a 4-inch aluminum irrigation lateral installed along the least sloping portion of the watershed (Figure 3). Twenty sprinklers rated at 15.6 gpm and 35 psi (pounds per square inch) nozzle pressure were spaced 60 feet (18.3 m) apart along the irrigation lateral.

Leachate was pumped from the catchment basin to a primary stabilization pond near the spray area. Each week of operation the system applied leachate for about 6 hours at 0.25 inches (.62 cm) per hour. A total of 20 inches (54 cm) of leachate were applied from October 1973 to

FIGURE 3
Mercer County Sanitary Landfill Leachate Treatment System
Princeton, W. Va.



April 1974. Surface runoff was rarely observed during this 7-month interval.

The environmental impact of leachate was determined by collecting samples of soil, soil percolate and vegetation before irrigation, during the 7-month application period and after foliage emerged in late spring. Composite soil samples were collected from each plot with a soil probe inserted to depths of 2, 4, 6, 12, 18, and 24 inches (5, 10, 15, 30, 45, and 60 cm). Soil percolate was collected from suction cup lysimeters installed at depths of 6, 12, and 18 inches (15, 30, and 45 cm). Foliage was taken from multiple numbers of trees, shrubs and ground cover species. Surface water samples were obtained periodically from the several impoundments that were integral parts of the leachate treatment system. All water samples were frozen immediately after collection and preserved frozen until analysis. Soils were air dried, pulverized, sieved through plastic mesh screens, and stored in air-tight plastic containers. Vegetation was dried for 48 hours at 130°F (54.4°C), ground in a Wiley cutting mill to pass through a 40-mesh screen and stored in plastic bags until analysis.

Leachates were analyzed by digestion in 4:1 redistilled nitric: perchloric acid diluted to 50 ml with 0.1 N hydrochloric acid. Atomic absorption spectrophotometry was employed to measure concentrations of aluminum, cadmium, calcium, cobalt, chromium, copper, iron, manganese, magnesium, nickel, potassium, sodium, strontium, and zinc in the digests. Nitrogen was determined by using Technicon Industrial Systems methods of digestion and auto-analysis (12). Chlorides were measured by electrometric titration in a Buchler-Cotlove automatic titrator. Chemical oxygen demand and electroconductivity were determined by US EPA methods for water quality analysis (13). Phosphorous was analyzed by persulfate digestion and colorimetry with ascorbic acid reagent (2). Sulfate determinations by EPA turbidimetric procedures were hindered by interferences. Samples were clarified by passage through a cation exchange resin and processed with a Technicon Auto-Analyzer II (11).

Soil lysimeter's percolate was aspirated directly through the atomic absorption spectrophotometer. The methods and procedures used for analysis of other parameters were the same as those used for leachate analysis.

Plant materials also were digested with 4:1 redistilled nitric: perchloric acid and analyzed by atomic absorption. Elements determined included calcium, magnesium, sodium, potassium, iron, zinc, manganese, copper, lead, cadmium, aluminum, chromium, nickel, strontium, and cobalt. A flameless atomizer accessory was used with the atomic absorption instrument to detect cobalt, lead, cadmium, and nickel. Nitrogen determinations were done according to Technicon automated procedures described by Isaac and Johnson (7). Phosphorous was determined colorimetrically, chloride was measured with a Buchler-Cotlove auto-titrator and sulfur analysis was conducted by combustion in a Leco IR 33 sulfur determinator.

Dilute hydrochloric: sulfuric acid and neutral normal ammonium acetate were used to extract soils. Extracts were analyzed by atomic absorption. The acetate procedure is better suited to calcium, magnesium, potassium, sodium, and manganese, while dilute acid is preferred for iron, zinc, and heavy metals. Phosphorous was determined colorimetrically on a dilute acid solution of hydrochloric and sulfuric acid (10). Total nitrogen was analyzed by Technicon automated procedures (9).

Phase I Studies - Results

The chemical composition of leachate from the Mercer County Sanitary Landfill appears in Table 1. The 1971 data are based on a few grab samples obtained randomly at unspecified times. Data for 1974 and 1976 are averages of many samples collected over a period of one year. COD was higher in 1974 previous to the installation of a surface aerator. Aeration apparently caused the COD reduction observed in 1976 data. Concentrations of calcium, magnesium, iron, manganese, zinc, aluminum and strontium did not differ appreciably between 1974 and 1976. None of the heavy metals including nickel, chromium, lead, cobalt, copper and cadmium occurred abundantly in leachate. Most parameters ranged widely in concentration perhaps because of seasonal effects, length of preservation, and chemical interferences encountered during analysis. Published

Table 1. Leachate quality as determined by analysis of wastewater samples taken from drains beneath the Mercer County Sanitary Landfill, Princeton, WV

Parameter	Year		
	1971	1974	1976
	----- ppm -----		
Kjeldahl N	63	62	101
SO ₄	107	55	--
Cl	274	230	--
Ca	458	605	602
Mg	188	174	156
K	67	45	117
Fe	303	424	562
Mn	182	55	61
PO ₄	--	4.5	1.1
Zn	--	2.5	1.4
Al	--	2.1	1.5
Sr	--	2.6	2.5
Na	257	265	283
	----- ppb -----		
Ni	52	400	225
Cr	34	100	336
Pb	2.5	133	386
Co	--	370	337
Cu	30	19	38
Cd	--	17	58
COD, mg/liter	5757	8973	3371
EC, umhos	2958	4092	4485
pH	5.6	5.3	5.5

data show that leachate parameters vary considerably according to age of the landfill (3). Our data appear to correspond with this report.

Water quality of soil percolate showed that major reductions in wastewater pollutants had resulted from spray irrigation (Table 2). Fall analyses represent free soil water quality shortly before leachate irrigations began. None of the parameters except COD and pH differed significantly according to depth. COD and pH declined as sample depth increased. Soil amendments did not measurably affect water quality of percolate. The data have been averaged to facilitate presentation. Samples obtained 3 months after irrigation began showed substantial increases in free soil water concentrations of calcium, magnesium, sodium, potassium, manganese, COD, and conductivity. Manganese and iron exceeded the Maximum Secondary Concentration Limits for drinking water (14). However, dilution upon entry into ground water reserves should diminish the levels of these elements to safe levels. Copper and zinc did not increase after irrigation. COD and conductivity declined to concentrations well below the levels observed in leachate. Calcium and magnesium moved downward with the percolate. Analyses of soil percolate obtained near the end of the irrigation season showed little change from mid-winter except for slight increases in soil solution aluminum and copper, and further reduction in soil solution acidity. The acidity of leachate is probably due to the presence of organic acids. Soil microbes probably degrade the organic acids which could lead to a gradual neutralization of the soil solution because of the presence of calcium, magnesium, and other cations.

Spray irrigation caused significant increases in soil-extractable amounts of calcium, magnesium, nitrogen, phosphorous and potassium and smaller increases in extractable manganese and sodium (Tables 3, 4). These changes were evident at nearly all soil depths to 2 feet (60 cm); however, the principal effects were evident in the first 6 inches (15 cm). Liming prompted the major increase in calcium at this depth, but the downward movement of calcium and magnesium was accomplished independently of this soil amendment. Leachate may contain unknown components that could promote penetration of the relatively immobile calcium ion.

Table 2. Water quality of soil percolate after one year of spray irrigation of leachate upon a woodland area, Mercer County Sanitary Landfill, Princeton, WV, 1973-74.

Depth, cm	Season											
	Fall				Winter				Spring			
	Ca	Mg	Na	K	Ca	Mg	Na	K	Ca	Mg	Na	K
	ppm											
15	39	11	58	3.5	241	111	158	19.6	157	42	82	14.6
30	50	16	67	3.0	289	88	113	9.2	158	41	91	8.8
45	47	17	56	2.8	414	82	82	4.9	227	42	108	9.3
	Mn	Fe	Al	Zn	Mn	Fe	Al	Zn	Mn	Fe	Al	Zn
	ppm											
15	.80	1.06	.26	.36	17.05	2.61	.24	.58	17.32	4.44	3.86	.30
30	1.45	.98	.16	.20	10.24	2.39	.25	.31	15.62	5.93	2.24	.20
45	1.55	.93	.28	.12	13.78	2.62	.25	.18	13.94	7.62	2.79	.31
	Cu	COD	EC	pH	Cu	COD	EC	pH	Cu	COD	EC	pH
	ppm	mg/l	µmhos		ppm	mg/l	µmhos		ppm	mg/l	µmhos	
15	.046	390	645	6.6	.025	890	1,960	6.5	.067	720	1,780	8.2
30	.029	271	560	5.8	.024	982	1,852	6.5	.062	708	1,165	7.8
45	.039	210	677	6.0	.034	750	1,610	6.4	.098	406	1,850	7.3

Table 3. Changes in extractable elements after one year of spray irrigation applications of leachate to woodland soils, Mercer County Sanitary Landfill, Princeton, WV, 1973-74.

Soil depth, cm	Time of sampling							
	Pre-spray				Post-spray			
	Ca	Mg	K	Na	Ca	Mg	K	Na
	----- ppm -----							
0- 5	1298	118	163	25	3748	200	310	115
5-10	440	53	99	18	1296	88	244	80
10-15	347	38	66	16	932	70	215	95
15-30	317	31	64	14	668	62	134	104
30-45	--	--	--	--	440	66	90	82
45-60	--	--	--	--	460	80	93	80

Table 4. Changes in extractable elements after one year of spray irrigation applications of leachate to woodland soils, Mercer County Sanitary Landfill, Princeton, WV, 1973-74

Soil depth, cm	Time of sampling									
	Pre-spray					Post-spray				
	Mn	Fe	Zn	N	P	Mn	Fe	Zn	N	P
	----- ppm -----									
0- 5	76	4.7	1.5	3290	15.8	112	5.6	3.4	4026	55.0
5-10	41	2.8	.8	1604	13.2	60	1.8	1.0	1986	30.5
10-15	36	2.1	.8	1058	14.2	56	.9	.7	1526	20.0
15-30	20	1.8	.6	837	14.4	40	.6	.8	873	26.4
30-45	--	--	--	--	--	23	.4	.9	608	22.9
45-60	--	--	--	--	--	25	.8	1.0	529	22.8

Organic chelates may provide a mechanism of calcium movement into the soil (6). The increased penetration of calcium and magnesium should benefit trees and other vegetation by extending roots into more favorable zones of nutrient and moisture supply. Farquhar (5) reported column studies that showed desorption of calcium and magnesium when leachate was added to pH 7.1 to 7.4 soils. We do not feel that the calcium-magnesium flush observed in leachate-treated soils in our studies was caused by desorption because initial pH measurements indicated that these soils had a low base saturation.

Manganese was not as effectively sorbed by the soil as iron and zinc. Farquhar (5) suggested that leachate manganese is adsorbed moderately under the probable influence of cation exchange capacity, however Griffin and Shimp (4) concluded that manganese was not adsorbed by clays because the anaerobic action accompanying leachate generation leads to soluble manganese forms. Soils effectively complexed high iron concentrations by attenuation mechanisms proposed by Farquhar (5).

Nitrogen was retained principally in the first 6 inches (15 cm) of soil. Very little of the nitrogen supplied by leachate would be lost from the soil profile because anaerobic conditions would prevent conversion of organic nitrogen to more soluble nitrates and nitrites. Nitrogen from leachate can help to sustain vegetative growth while facilitating wastewater decontamination.

Most of the phosphorous increase observed in soils can be attributed to the phosphate soil amendments. Solid wastes may contain as much as 0.26 percent dry weight phosphorous (15), however leachates show only minor amounts (3). Phosphorous is strongly adsorbed by soil colloids and is subject to major analytical interferences. The problem of accurately measuring phosphorous may conceal the true potential of leachate as a phosphorous source.

Sodium appeared to be rather uniformly distributed through the soil profile following 7 months of leachate applications. The attenuation of sodium compared to other leachate cations is relatively low (4). Most of the sodium is probably flushed through the soil into deep percolation or surface drainage.

Heavy metals including copper, cadmium, cobalt, chromium, nickel, and lead were not detectable in soil extracts at concentrations equal to or greater than were found in leachate. Loading rates for heavy metals were relatively light because leachate did not contain appreciable amounts of these elements. Although heavy metals are soluble in acid soils, the gradual neutralization of soil acidity accomplished by leachate applications can be expected to lead to the formation of soil heavy metal complexes. This process would help to prevent heavy metal contamination of ground water reserves.

Table 5 emphasizes the importance of leachate as a resource for modifying soil acidity. Leachate combined with lime caused a significantly greater increase in soil pH than leachate alone or leachate combined with phosphate amendments. Iron, aluminum, and manganese are more soluble in acid soils and tend to inhibit root development where they are present in toxic amounts. Leachate irrigation appears to promote pH changes that would prevent or diminish the toxicity of these elements and favor the deeper penetration of roots.

Leachate caused major changes in mineral content of woodland foliage. Analysis performed on a wide variety of trees, shrubs, and ground covers showed that elemental accumulation was species dependent. For example, dogwood and wild blackberry contained three to four times the usual amounts (0.20 to 0.25 percent, dry weight) of magnesium found in leaves. Potassium in dog fennel, wild daisy and ragweed was higher than average amounts both before and after irrigation. Iron, zinc, and sodium generally increased and aluminum and strontium decreased after spray irrigation. Manganese content increased in nearly all species but especially in those shown in Table 6. No phytotoxicity was evident in spite of the high concentrations. Hickory accumulated above average amounts of zinc, nickel, and copper.

Spray irrigation caused significant ecological changes in ground cover and understory vegetation of the forest ecosystem. Leaf litter virtually disappeared perhaps because microbial decomposition was aided by the nitrogen and additional moisture supplied by leachate. Such

Table 5. Changes in soil pH after one year spray irrigation of forested plots treated with lime and phosphate fertilizers Mercer County Sanitary Landfill, Princeton, WV, 1973-74.

Soil depth, cm	Pre-spray	Post-spray			
		Soil amendment			
		Lime (11.2 MT/ha)	Rock PO ₄ (11.2 MT/ha)	Super PO ₄ (0.67 MT/ha)	None
5	4.7	6.6	6.0	5.8	5.8
10	4.5	6.4	5.8	5.8	5.8
15	4.6	6.3	6.0	5.9	6.0
30	4.5	6.2	5.7	5.7	5.7
45	—	5.6	5.3	5.2	5.2
60	—	5.5	5.1	4.9	5.0

Table 6. Mn content of woodland foliage spray-irrigated for one year with leachate from the Mercer County Sanitary Landfill, Princeton, WV, 1973-74.

Species	Time of sampling	
	Pre-spray	Post-spray
	----- ppm -----	
Oak	605	1138
Hickory	640	1775
Red maple	386	1275
Wild cherry	464	1100
Wild grape	303	825
Huckleberry	506	1450
Cinquefoil	204	1225

ground cover species as ground pine, cinquefoil, and wild strawberry were eliminated and were replaced by Virginia creeper, poison ivy, wild blackberry and succulent weed species such as pokeweed and fireweed. Herbaceous shrubs and hardwood saplings of sourwood, red maple and poplar died or showed reduced vigor but larger trees showed no adverse effects. Heavy leachate residues covered any obstacle within the radius of sprinklers.

Monitoring of surface water quality in the spray irrigation watershed showed a gradual decline in waste contamination (Table 7). Pollutant concentrations were slightly higher after leachate was pumped from the catchment basin to the primary stabilization pond. Data for the primary stabilization pond represent water quality applied to plots. Water quality improved greatly as shown in samples collected from the secondary stabilization pond. Water obtained from this source was collected from a pipe that drained from the spray irrigation watershed. These samples probably consist of both surface runoff and soil percolate but probably are an accurate representation because lysimeters from the various plots in the watershed have disclosed similar improvement in water quality. Additional pollutant reduction occurred in the secondary stabilization pond as is indicated by the analysis of waters draining from this impoundment into the large sediment pond. Although electroconductivity had decreased, the increase in COD was not expected. However, further declines in all parameters appeared in the analyses of Brush Creek tributary waters downstream from the large sediment pond. Surface runoff leading to leachate dilution and further stabilization in the large sediment pond probably account for the final improvements in water quality before discharge from the landfill property.

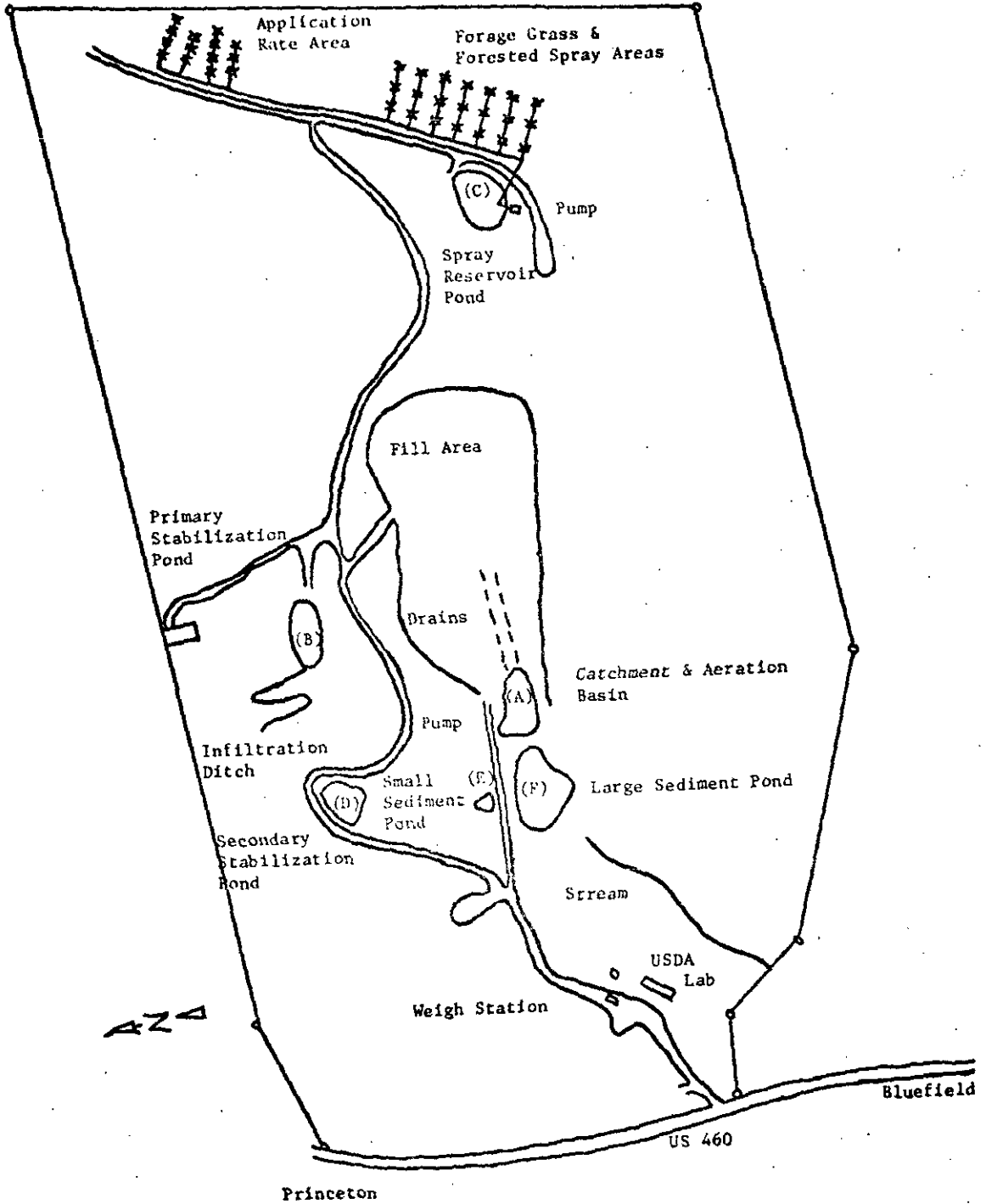
Phase II - Methods

The leachate spray irrigation system was moved in 1974 to a level, hilltop site above the refuse composting area (Figure 4). A 510,000 gallon clay-lined reservoir with a 32-day retention time was built at the site to store leachate pumped from the leachate catchment basin until application could be made through a distribution system of seven,

Table 7. Water quality of impoundments located within the Mercer County Sanitary Landfill spray irrigation leachate treatment system, Princeton, WV, 1973-74.

Parameter ppm	Sample source				
	Leachate catchment basin	Primary stabilization pond	Secondary stabilization pond	Large sediment pond	Brush Creek tributary
Ca	605	694	259	153	162
Fe	464	714	49	62	28
Na	257	348	67	51	31
Mg	174	148	67	20	23
Mn	75	85	39	22	11
K	45	61	10	9	6
Kjeldahl N	37	65	13	3	2
Al	3.1	4.1	.6	1.4	.8
Sr	2.1	2.9	1.1	.6	.2
Zn	6.2	12.9	4.3	3.3	3.6
PO ₄	4.6	3.9	3.5	1.1	.5
COD, mg/liter	8973	10859	3291	3909	1284
EC, μ hos	4092	5237	2235	902	696
pH	5.3	5.2	5.6	6.0	6.1

FIGURE 4
Mercer County Sanitary Landfill Leachate Treatment System
Princeton, W. Va.
Phase II Studies



2-inch laterals spaced 80 feet (24.4 m) apart on a 4-inch main pipe. Each lateral was equipped with three overhead rotary sprinklers spaced 60 feet (18.3 m) apart. Weekly operation and application rates were essentially the same as had been practiced in the Phase I period.

An area 100 feet (31 m) by 150 feet (46.5 m) was cleared of native vegetation consisting of succession Appalachian hardwoods. Six forage grasses selected for adaptability to local climatic conditions were established to determine individual tolerances to leachate applications. The objective of the research was to test the value of forage grasses as complexing agents for heavy metals and other leachate elements.

The cleared area was subdivided into six equal blocks 40 feet (12.2 m) on a side. Each block was further divided into sixteen subplots. Lime and phosphates used at the same rates as in Phase I tests were broadcast on four randomized replicates within each of the six blocks. A fourth set of four subplots was not treated with amendments. Each of the six blocks was planted individually with orchardgrass (*D. glomerata* L.), reed canarygrass (*P. arundinacea* L.), bromegrass (*B. inermis* Leyss.), tall fescue (*F. arundinacea* Schreb.) cv 'Ky 31', and bermudagrass (*C. dactylon* (L.) Pers.) cvs. 'Midland' and 'Tufcote'. The bermudagrasses are considered warm-season grasses while the others are considered cool-season grasses. Soil amendments and plantings were accomplished in May 1974.

Suction cup lysimeters were installed in two replicates of the four soil treatments in the reed canarygrass and Midland bermudagrass plots. Placement depths were 6, 12, 18, and 24 inches (15, 30, 45, and 60 cm).

Pre-spray vegetation was harvested in September 1974. Soils were core-sampled to six depths and lysimeter percolate to four depths before the first irrigation was applied in October 1974. Leachate water quality was about the same as was shown in Table 1 for 1974. The total application from October 22, 1974 to April 28, 1975 averaged about 60 inches (155 cm) while approximately 25 inches (63 cm) of natural precipitation occurred during the same period.

Plots were cleared of surface residue at the conclusion of the irrigation season and uncontaminated forage was permitted to flourish until samples were collected. Reed canarygrass and Midland bermudagrass were harvested in early June and late August while other species were harvested once in July. Forages were prepared for analysis and analyzed by using the methods stated in Phase I studies. Soils and soil percolate also were sampled for changes due to leachate, however the analyses have not been completed.

Phase II Studies - Results

Leachate influenced the accumulation of nearly all macroelements but did not cause significant accumulations of the toxic heavy metals cobalt, nickel, chromium, lead, and cadmium. Table 8 presents results of pre-spray, early and late cutting post-spray analyses of aluminum, sodium, chloride, and Kjeldahl nitrogen in Midland bermudagrass and reed canarygrass. Soil amendments did not appreciably affect the levels of these parameters and for simplicity, have been averaged. Midland bermudagrass declined in aluminum content after irrigation but reed canarygrass increased in late cuttings after irrigation. Both species differed in their tendencies to accumulate the highly mobile sodium ion at early cutting. However, by late cutting sodium probably had passed through the rooting profile. Chloride accumulation tendencies were the same for both species. Although this element is rather mobile, late cuttings disclosed the persistence of chloride in the rooting zone as evidenced by persistently high concentrations. Leachate furnished sufficient nitrogen to cause relatively high concentrations in early cuttings. This recycled nitrogen benefitted early growth of these and other species.

Soil amendments caused significant differences in the elemental contents of manganese and iron in grasses (Tables 9, 10). Species differed in their responses to manganese uptake. Lime clearly suppressed manganese accumulation in early cuttings of Midland bermudagrass. Superphosphate led to five times more manganese in early cuttings of reed canarygrass as compared to other soil treatments. The effect persisted in the second cutting of this grass. Manganese availability as a soluble ion probably is greater in late spring when plant growth resumes. Anaerobic conditions may have developed in leachate-saturated

Table 8. Mineral content of leachate-treated forage grasses cut at early and late growth stages, Mercer County Sanitary Landfill, Princeton, WV, 1974-75.

Cutting stage	Forage grass	
	Bermuda	Reed canary
	Al, ppm	
Pre-spray	83	32
Early	28	20
Late	27	65
	Na, ppm	
Pre-spray	78	48
Early	412	1766
Late	54	58
	Cl, %	
Pre-spray	.18	.19
Early	.62	.83
Late	.62	.89
	Kjeldahl N, %	
Pre-spray	3.39	4.86
Early	4.59	5.68
Late	2.32	2.68

Table 9. Effect of soil amendment and cutting time on the Mn content of forage grasses spray-irrigated with leachate from the Mercer County Sanitary Landfill, Princeton, WV, 1974-75

Forage grass	Soil amendment			
	Lime 11.2 MT/ha	Rock phosphate 11.2 MT/ha	Superphosphate 0.67 MT/ha	None
	----- ppm -----			
Bermuda				
Pre-spray	81	151	164	141
Early cutting	256	778	735	721
Late cutting	32	68	76	44
Reed canary				
Pre-spray	250	376	315	346
Early cutting	572	625	2452	583
Late cutting	58	67	583	86

Table 10. Effect of soil amendment and cutting time on the Fe content of forage grasses spray-irrigated with leachate from the Mercer County Sanitary Landfill, Princeton, WV, 1974-75.

Forage grass	Soil amendment			
	Lime 11.2 MT/ha	Rock phosphate 11.2 MT/ha	Superphosphate 0.67 MT/ha	None
	----- ppm -----			
Bermuda				
Pre-spray	135	101	121	166
Early cutting	336	845	937	637
Late cutting	72	68	78	74
Reed canary				
Pre-spray	96	77	85	98
Early cutting	1142	648	343	550
Late cutting	91	137	96	89

soils causing reduced manganese states. Lime would help to neutralize soil acidity and precipitate manganese. The reason for high manganese contents in foliage grown on superphosphate-treated soils is not clear.

Bermudagrass responded predictably to iron uptake on limed soils. This element becomes complexed as soil pH increases. However, the opposite iron accumulation tendency exhibited by reed canarygrass on limed plots is unexplainable. The accumulation of both elements tends to follow classical understanding in the case of Midland bermudagrass but not reed canarygrass.

The manganese content of orchardgrass, tall fescue, bromegrass, and Tufcote bermudagrass was lower in foliage from limed as compared to unlimed plots (Table 11). These differences are more in accord with pH-dependent manganese availability, however this speculation cannot be presently supported because soil pH data for 1975 are not available. These grass species exhibited significantly different tendencies for manganese accumulation.

Table 12 depicts changes in soil pH after two seasons of leachate irrigation. Only 23.5 inches (59.7 cm) of leachate were applied during 1975 and 76 because fall drought held down leachate generation and cold temperatures prevented system operation much of the time. Lime and phosphate treatments caused more neutralization of soil acidity in the upper 4 inches (10 cm) of soil as compared to plots not treated with soil amendments; however differences between treatments tended to become less below this depth.

Lime clearly benefitted the tolerance and survival of forage grasses to leachate irrigations (Table 13). The impact of leachate was much greater during the first application season than the second. Orchardgrass was nearly eliminated in plots that received no soil amendments. Tall fescue and bromegrass persisted about equally but only slightly better than orchardgrass. Reed canarygrass showed the least tolerance in superphosphate-treated plots. The 8-month interval between the first and second irrigation seasons afforded a significant recovery by each grass. Rootstocks apparently were not as affected by applications as foliage. Leachate applied in 1975-76 was only 40 percent of the amount applied the previous year. Stands showed better persistence in limed and

Table 11. Effect of soil amendment on Mn content of forage grasses spray-irrigated with leachate from the Mercer County Sanitary Landfill, Princeton, WV, 1974-75.

Grass	Soil amendment			
	Lime (11.2 MT/ha)	Rock PO ₄ (11.2 MT/ha)	Super PO ₄ (0.67 MT/ha)	None
	ppm			
Orchard				
Pre-spray	42	58	122	64
Post-spray	159	575	439	322
Brome				
Pre-spray	81	138	121	200
Post-spray	371	459	659	688
Bermuda				
Pre-spray	135	343	206	305
Post-spray	1056	1787	1893	1991
Tall fescue				
Pre-spray	135	206	343	305
Post-spray	549	1303	1365	1217

Table 12. Changes in soil pH after two years irrigation of forage grass plots established with lime and phosphate fertilizers, Mercer County Sanitary Landfill, Princeton, WV

Soil depth, cm	Year				
	1974 Pre-spray	1976			
		Soil amendment			
	Lime (11.2 MT/ha)	Rock PO ₄ (11.2 MT/ha)	Super PO ₄ (0.67 MT/ha)	None	
5	4.7	6.8	6.6	6.6	6.4
10	4.7	6.5	6.6	6.6	5.7
15	4.7	5.7	5.8	5.7	5.3
30	4.5	5.2	5.2	5.2	5.0
45	4.4	4.8	4.7	5.0	4.6
60	4.4	4.8	4.7	4.8	4.6

Table 13. Tolerance of forage grasses to two years of leachate spray irrigation at the Mercer County Sanitary Landfill, Princeton, WV, 1975-76.

Grass	Soil amendment			
	Lime (11.2 MT/ha)	Rock PO ₄ (11.2 MT/ha)	Super PO ₄ (0.67 MT/ha)	None
	Estimated stand, %			
Orchard				
1975	61	38	38	19
1976	86	69	80	29
Reed canary				
1975	89	45	14	24
1976	98	91	90	55
Brome				
1975	80	14	56	15
1976	85	90	91	34
Tall fescue				
1975	89	24	56	32
1976	90	55	82	49

phosphate-treated plots; however vegetation was continuing to recover in plots treated only with leachate.

The bermudagrasses appear rather leachate tolerant, however their resistance to cold weather is poor. Frosts in late spring and early fall force plants to become dormant limiting their value for transpiration. Establishment of this species in northern latitudes may be difficult to achieve as well as impractical for leachate treatment purposes.

SUMMARY AND CONCLUSIONS

Spray irrigation of leachate from a municipal sanitary landfill used for solid waste disposal significantly reduced the contamination of water without causing major environmental impacts.

Twenty inches of leachate applied by overhead rotary sprinklers at a rate of 1.5 inches per week to a dormant Southern Appalachian hardwood ecosystem virtually eliminated ground cover and herbaceous vegetation and stimulated the invasion of a few undesirable species. Leaf litter and many young hardwood saplings perished but mature trees showed no adverse effects. Vegetation accumulated increased amounts of iron, manganese, zinc and sodium while aluminum and strontium levels declined. Elemental uptake appeared to be species-dependent. The principal contaminants of leachate included organic nitrogen, sulfates, chlorides, calcium, magnesium, potassium, sodium, iron, and manganese. Analysis of soil percolate revealed major reductions in concentrations of these elements after movement through 6 inches of soil. Manganese and iron exceeded recommended limits for safe water quality, however ground water dilution should correct this problem. Calcium and magnesium moved downward suggesting possible leachate-induced chelation. Chemical oxygen demand declined from 4000 mg/liter to less than 1000 mg/liter. The pH of the soil solution changed from slightly acid to slightly basic although leachate is moderately acidic. Extractable soil analyses showed soil attenuation of calcium, magnesium, potassium, phosphorous, manganese, iron, zinc, and organic nitrogen. The downward movement of calcium and magnesium led to a significant reduction in soil acidity. This phenomenon

should benefit the growth of roots by preventing toxicity from acid-soluble elements such as aluminum. Heavy metals were not detectable in soil percolate or extractable soil solutions by methods of atomic absorption analysis. Leachate did not contain appreciable amounts of heavy metals including copper, cadmium, cobalt, chromium, nickel, and lead. The gradual neutralization of soil acidity would lead to soil heavy metal complexes and help to prevent heavy metal contamination of ground water. Lime and phosphate fertilizer used as soil amendments decreased soil acidity.

Six forage grasses spray-irrigated with 60 inches and 25 inches of leachate applied during two dormant seasons exhibited differential survival tendencies. Soil amendments beneficially influenced tolerance to leachate, especially with lime applications. Stands of all grasses were moderately-to-severely depleted by the 60-inch leachate application, however rootstocks persisted and excellent recoveries were made. Reed canarygrass and tall fescue showed better tolerance than orchardgrass and brome grass. Bermudagrasses effectively survived leachate treatments but were damaged by early and late summer frosts. Species differentially accumulated many of the mineral contaminants of leachate, especially manganese and iron. Soil amendments influenced the uptake of these elements. Forage grasses appear to be an acceptable concentrating mechanism for leachate pollutants.

ACKNOWLEDGMENTS

This paper was contributed by the USDA, Agricultural Research Service, Morgantown, West Virginia in cooperation with West Virginia University, Morgantown, West Virginia, and the Southern West Virginia Regional Health Council Inc., Bluefield, West Virginia. The research was supported in part by an interagency transfer of funds from the Office of Solid Waste, USEPA, Washington, D. C.

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THE TREATMENT OF LANDFILL LEACHATE
UTILIZING A "LIVING FILTER"

Ronald L. Lavigne, Research Associate
Environmental Engineering Program
Department of Civil Engineering
University of Massachusetts, Amherst

BACKGROUND

"Regulations for Disposal of Solid Waste by Sanitary Landfilling" (1) in the Commonwealth of Massachusetts were officially promulgated in April of 1971. As early as the summer of 1974, concern was being raised by health officials and the Massachusetts Division of Water Pollution Control (MDWPC) regarding groundwater and surface water contamination by landfill leachate.

With grant money from MDWPC, the Environmental Engineering Section of the Department of Civil Engineering at the University of Massachusetts in Amherst initiated a field research program to evaluate the potential leachate pollution problem. In October of 1974, five landfill sites were selected for monitoring. Two of the sites were poorly operated, but the remaining three were well engineered and were run professionally. Included in the study were the towns of Athol, Petersham, Barre, Amherst, and Easthampton, all in Central and Western Massachusetts.

Groundwater and surface water monitoring over a 12-month period clearly indicated that all of the landfills were producing considerable amounts of polluting leachate and that this leachate was leaving the site via surface water streams or groundwater aquifers. During the study many additional leachate problems were investigated by the research team at other New England landfill sites. Preliminary data after a year of field observation clearly indicated that (a) "sanitary landfilling" was not "sanitary", and (b) that leachate production seemed to be a problem regardless of how well the landfills were operated. It should be noted that several of the sites studied had monitoring data collected from as early as 1970 and that sampling has continued to the present time,

providing more than eight years of chronological surveillance. During that time nothing has occurred to indicate that the preliminary conclusions previously stated were inaccurate.

By Spring of 1975, it had been determined that some type of lined or controlled landfill was needed to investigate the many still unanswered questions regarding leachate production and behavior. Some of the questions remaining were outlined by Lavigne (this volume) and are restated below:

1. Does the quantity of leachate production vary from landfill to landfill?
2. What are the chemical components and their concentrations in undiluted leachate waters?
3. Do human wastes and sewage in a landfill pose a public health problem with respect to possible pathogens in leachate?
4. How do location and method of operation affect leachate parameters in landfills?
5. How effective is the 4 foot separation regulation as a minimum distance between refuse and high water table?
6. How effective are New England soils in attenuating leachate by natural, physical, chemical and biological processes?
7. If present regulations are found to be inadequate, what are the alternatives available to properly control leachate production and movement?

RESEARCH FACILITY

Because of the excellent cooperation received from the private landfill owner in Barre, Massachusetts, it was decided that the proposed research facility would be located at the Martone site. Martone also operates a private collection business serving about 10,000 people. His landfill receives about 100 tons per week of household and commercial refuse. Sludge is not disposed of at the site.

During the Summer of 1975, a three acre fill area was completely lined and diked with on-site clay (Figure 1). Four feet of natural clay were placed on the bottom of the new area, and a 12 foot high dike was placed around the perimeter. Above the bottom sill four more feet of natural

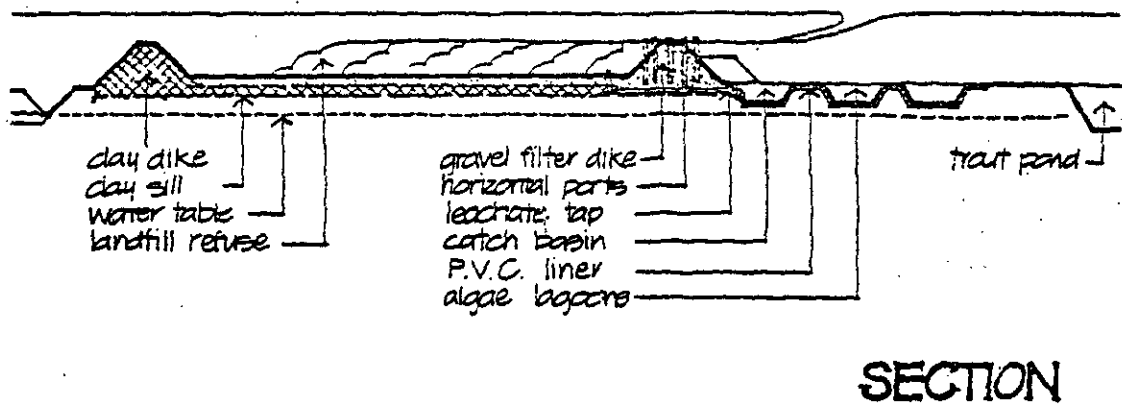
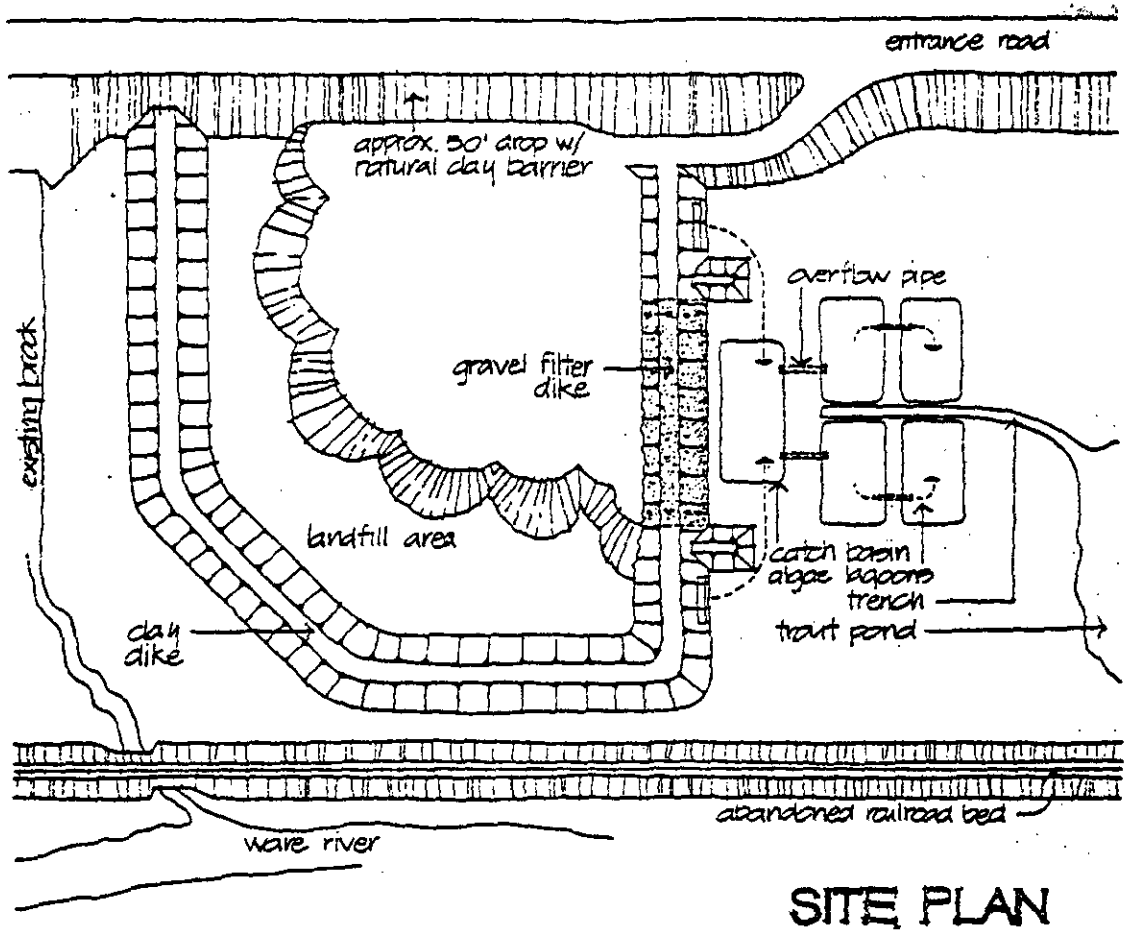


FIGURE 1

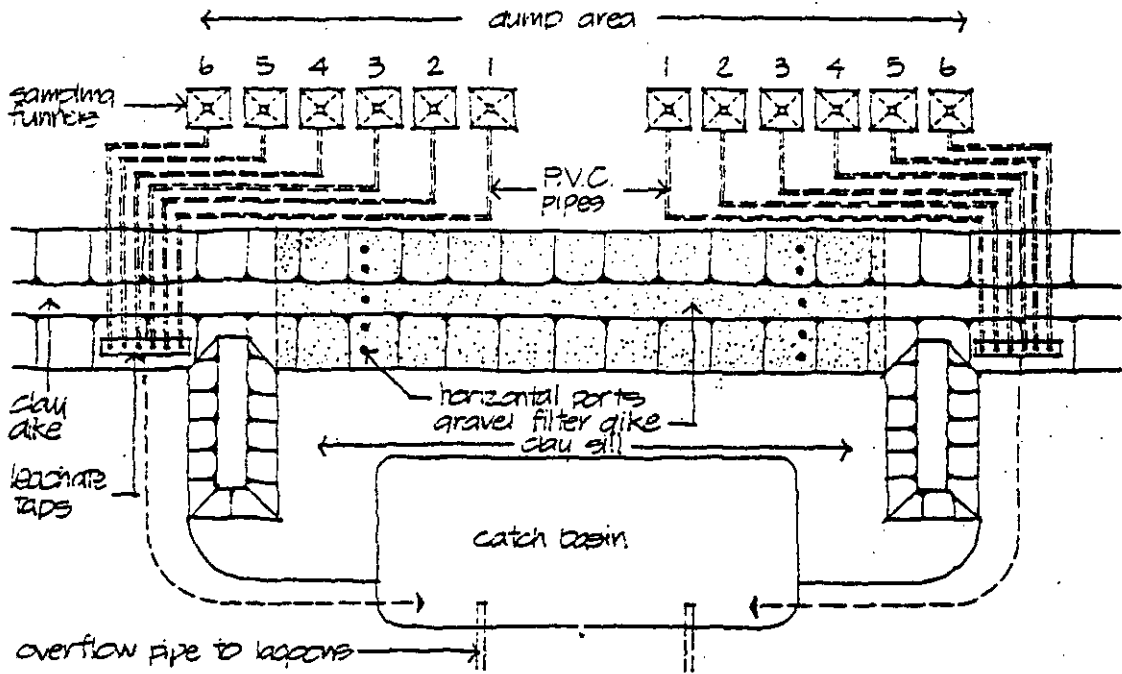
on-site soils were filled in, and a section of the dike was replaced by natural porous soils at the low point of the perimeter. When completed, the facility functioned as a large "bathtub" with a drain at one end and with four feet of porous soil in the bottom.

In the four foot simulated bottom, twelve 100 ft² "super funnels" were placed at different depths. Two were placed on top and immediately under the refuse, two were placed at a depth of one foot below refuse, two at two feet below refuse, two at three and four feet below refuse, and two were buried two feet into the clay. All 12 funnels (two sets of six) were gravity drained to a tap board outside the landfill (Figure 2). The taps permitted sampling of leachate as it percolated down through four feet of natural on-site soils. Additional sampling ports placed at five foot intervals in the porous section of the perimeter dike permitted monitoring of leachate attenuation as it passed through 35 feet of soils horizontally.

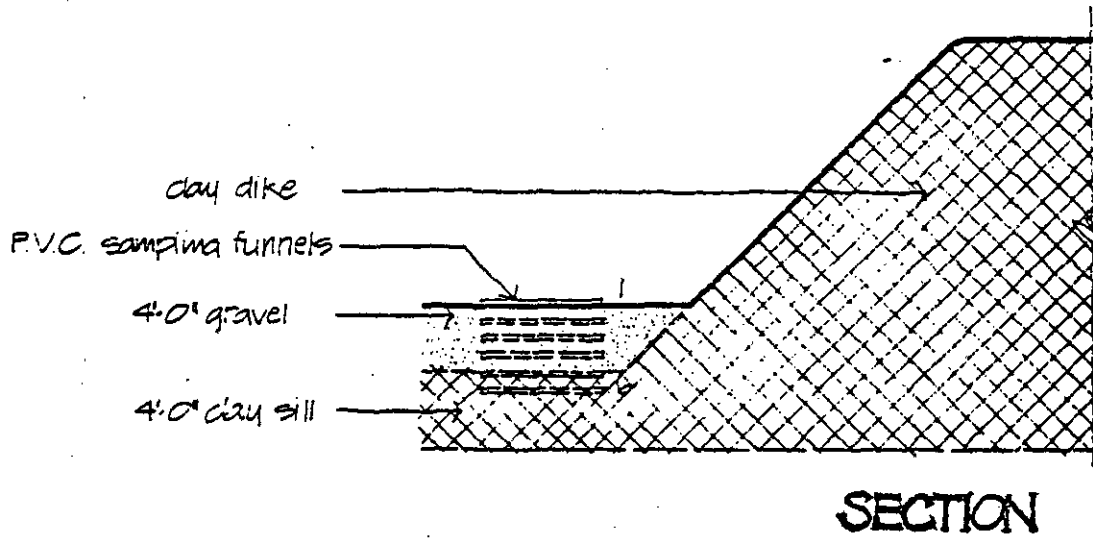
To facilitate experimentation with leachate treatment, four lined lagoons (40' x 80' x 4') were also constructed down gradient of the landfill. When completed the facility monitored movement of leachate through four feet of natural soils, then across the bottom of the fill area through a 35 foot filter dike, and finally collected them in a catch basin with a series of treatment lagoons (Figure 1). The sampling taps to the landfills interior also permitted the collection of anaerobic leachate in an undiluted form. Refuse placement began in June of 1976 and leachate appeared in the catch basin and lagoons during August of the same year.

LEACHATE TREATMENT

Observations in the field during the first year's study suggested that leachate stimulates the growth of several species of aquatic plants, and that they in turn saturate leachate waters with oxygen, facilitating the aerobic oxidation of wastes by bacteria. Because the process functions without an outside energy source (other than daylight) and does not involve chemical or mechanical support systems, it was selected as the mode of treatment for Barre leachate. The lagoons pictured in Figure 1 are usually operated on an alternating batch basis. While the catch basin is overflowing to one side of the trench, treatment on the other side is in process.



LEACHATE FACILITY PLAN



SECTION

FIGURE 2

After discharge of treated leachate to the trout pond from an algal lagoon, it is again filled from the lagoon upgradient of it. At that point the catch basin overflow can be directed to the newly emptied storage lagoon and the process repeated on the alternate side of the trench. Research has shown that a detention time of 90 days is required to effect 99 percent reductions in pollution parameters such as BOD and Suspended Solids (Table 1). During the first 45 days many of the metals are precipitated and natural stripping of the volatile organics occurs. Leachate is quite toxic to most aquatic plants during this treatment phase. During the second half of the process (45-90 days) acclimated representatives of many algal groups along with Lemnaceae "duckweed" thrive in the leachate, and polish it to a quality acceptable for discharge into surface waters or groundwaters.

An anticipated modification to the system is now before the Department of Energy for research funding. The proposed modification would replace the catch basin with a "fluidized bed reactor" designed for methane gas production. Methane would be produced as an alternative energy source and the effluent of the reactor would be polished by the same "filter lagooning" process.

CONCLUSIONS

Since the start of leachate research at the University of Massachusetts in the fall of 1974, considerable knowledge has been gained regarding leachate production, its characterization, its interaction with natural soils, and its treatment using lagoons as "living filters" and "oxidation ponds". This extensive research work has precipitated the following conclusions.

1. With 40 or more inches of rain per year, New England landfills are all producing leachate to some degree. Historically this leachate has been produced in amounts great enough to contaminate surface and groundwater resources (2).
2. The increased use of disposable diapers that ultimately find their way to the sanitary landfill have implicated leachate as a vector of pathogenic organisms (3,4,5).

TABLE 1

LEACHATE TREATMENT - MARTONE LANDFILL, BARRE, MASSACHUSETTS

Parameter mg/l	Quality of Leachate Collected from Interior of Barre Landfill	Quality of Leachate After 30 Days of Treatment	Quality of Leachate After 60 Days of Treatment	Quality of Leachate at Time of Discharge (90 Days)
BOD ₅	21,060	4650	220	10.3
COD	35,680	9500	400	117
PH	5.15	6.50	6.80	7.3
Alkalinity as CaCO ₃	4,600	980	315	32
Sulfate	2,330	450	79	29
Ammonia	437	130	70	3.5
Chloride	372	350	317	200
Total Solids	11,600	4300	2580	1430
Volatile Solids	3,900	1900	1050	320
Iron	1,400	318	120	1.0
Zinc	24	14	12.0	1.0
Manganese	28	11.0	3.3	1.6

Values represent averages for the 1977 sampling year.

3. New England soils are very ineffective in attenuating leachate strength. Barre soils demonstrated 100 percent breakthrough of most parameters in less than 90 days. Column studies of Barre soils at the University substantiated field data and soil analyses indicated Barre soils to be typical of much of New England (6,7,8).
4. The use of aquatic plants has excellent potential for the treatment of landfill leachate if care is exercised during the seeding and acclimation period (9). The energy free nature of the lagooning process is even more attractive given the present national energy situation. With the potential for methane production from leachate, lined landfills will prove to be much more cost effective than conventional refuse-to-energy systems.
5. It is possible to treat leachate with an efficiency in excess of 99 percent using oxidation/"living filter" lagoons if a detention time of 90 days or more is provided. With an average daily leachate flow of approximately 3,000 gpd/acre, long detention times are not difficult to provide.

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SYMPOSIUM PARTICIPANTS

- Adams, Myrna D.
Chairman
Board of Health
Town of Dighton
Dighton, MA 02715
- Adrian, Donald D., Ph. D.
School of Civil Engineering
Marston Hall
University of Massachusetts
Amherst, MA 01003
- Anderson, Paul T.
RFD #2
Winter Street
Middleboro, MA 02346
- Beck, John D.
Baltimore County, Maryland
1423 Putty Hill Avenue
Baltimore, MD 21204
- Berger, Bernard
Director, Water Resources
Research Center
Graduate Research Tower
University of Massachusetts
Amherst, MA 01003
- Blanchette, Dennis
Mill Pond Road
Broadbrook, CT 06016
- Bowers, John
Lower Pioneer Valley Regional
Planning Commission
26 Central Street
West Springfield, MA 01089
- Cahill, Charles
Town Hall
Sylvan Street
Danvers, MA 01923
- Carlson, Robert P.
322 Main Street
Leominster, MA 01453
- Carothers, Peter
Cahn Engineers, Inc.
Alexander Drive
Wallingford, CT 06492
- Clink, Leroy
City Hall Annex
Holyoke, MA 01040
- Clougherty, Thomas F.
86 Forest Street
Clinton, MA 01510
- Cole, Susan G.
Lower Pioneer Valley Regional
Planning Commission
26 Central Street
West Springfield, MA 01089
- Dachos, Deborah C.
Lower Pioneer Valley Regional
Planning Commission
26 Central Street
West Springfield, MA 01089
- Dalsimer, Tom
Box 332
Springfield College
Springfield, MA 01100
- DeCarlo, Peter M., Jr.
Cahn Engineers, Inc.
Alexander Drive
Wallingford, CT 06492
- DeFeo, Fred
Musket Road
Plymouth, MA 02360
- Dratfield, Herbert
Reinhardt Associates
Valley Bank Tower
Springfield, MA 01115
- Dunn, George L.
78 Carter Road
Lynn, MA 01904

- Dym, Clive, Ph. D.
Chairman
Civil Engineering Department
School of Civil Engineering
Marston Hall
University of Massachusetts
Amherst, MA 01003
- England, John
DEP, Solid Waste Unit
State Office Building
122 Washington Street
Hartford, CT 06115
- Fawcett, John
Water Resources Research
Center
Graduate Research Tower
University of Massachusetts
Amherst, MA 01003
- Feng, Thomas H., Ph. D.
School of Civil Engineering
Marston Hall
University of Massachusetts
Amherst, MA 01003
- Ferry, Sandra
285 Ware Road
Belchertown, MA 01007
- Flagg, Tracy I.
Town Hall
Sylvan Street
Danvers, MA 01923
- Frey, John Jr.
26 Windmill Drive
Marlboro, MA 01752
- Fuller, James
Department of Environmental
Quality Engineering
Central Region Office
75 Grove Street
Worcester, MA 01605
- Gagne, Dennis P.
8 Island Street
Kingston, MA 02360
- Galvin, Thomas M.
Old Colony Planning Council
232 Main Street
Brockton, MA 02401
- Goode, Matthew E.
100 Cambridge Street
Boston, MA 02202
- Cropen, Sara
456 Wilbraham Road
Springfield, MA 01109
- Hancuff, William R., Ph. D.
11800 Sunset Valley Drive
Suite 1130
Reston, VA 22091
- Hanks, Marc A.
Lower Pioneer Valley Regional
Planning Commission
26 Central Street
West Springfield, MA 01089
- Harris, Carol
Department of Environmental
Quality Engineering
Central Regional Office
75 Grove Street
Worcester, MA 01605
- Hauser, Robert, Jr.
23 Fairbanks Road
Lexington, MA 02173
- Heil, James
Town of Brookhaven
Patchogue, NY 11772
- Henegar, Warren
RR #4
Bloomington, IN 47401
- Holyoke Sanitary Landfill
142 Casino Avenue
Chicopee, MA 01013
- Hunnewell, Dorothy S.
Department of Environmental
Quality Engineering
Air & Hazardous Materials
600 Washington Street
Boston, MA 02111
- Iantosca, Angelo
Department of Environmental
Quality Engineering
University of Massachusetts
Amherst, MA 01003

Jacques, Robert
Health Department
Shrewsbury, MA 01545

Jennings, Aaron A.
39 South Main Street
South Deerfield, MA 01373

Johnson, Elizabeth
Department of Environmental
Engineering
University of Massachusetts
Amherst, MA 01003

Johnson, Gary A.
Wehran Engineering Corp.
666 East Main Street
Middletown, NY 10940

Joly, Gilbert T.
Department of Environmental
Quality Engineering
75 Grove Street
Worcester, MA 01605

Joubert, Robert H.
The Asphalt Institute
599 North Avenue
Wakefield, MA 01880

Kant, Edward J., Jr.
Loureiro Engineering Associates
20 Tower Lane
Avon, CT 06001

Kanter, Paul D.
18 Frances Drive
Randolph, MA 02368

Karian Victor
Department of Environmental
Quality Engineering
600 Washington Street
Boston, MA 02111

Katz, Paul
100 Malden Street
Springfield, MA 01100

Kiritsy, Paul
11 Browning Road
Shrewsbury, MA 01545

Kozlowski, Paul
Stony Brook Energy Center
P.O. Box 426
Ludlow, MA 01056

Lavigne, Ronald L.
Resource Control, Inc.
Quabbin Office Building
Valley Road
Barre, MA 01005

Leighton, Ira W.
Room 2113 - JFK Building
Boston, MA 02203

Letendre, Nicole
41 Fearing Street
Amherst, MA 01003

Lopes, Antone P.
Board of Health
Town Hall
Fairhaven, MA 02719

Male, James, Ph. D.
School of Civil Engineering
University of Massachusetts
Amherst, MA 01003

Malone, Philip G.
Environmental Engineering
Division
USAE Waterways Exp. Station
Box 631
Vicksburg, MS 39180

Marin, Paul A.
770 North King Street
South Windsor, CT 06074

Marsh, Donald
P.O. Box 489
Montpelier, VT 05602

Martone, Leonard P.
Resource Control, Inc.
Quabbin Office Building
Valley Road
Barre, MA 01005

McCarthy, Joseph G.
Worcester Board of Health
419 Belmont Street
Worcester, MA 01600

McDermott, Richard
100 Hallet Street
Boston, MA 02124

McKendrick, Ryan F.
5100 Suffield Court
Skokie, IL 60015

Menser, Harry
USDA - ARS
West Virginia University
Morgantown, W VA 26505

Mokrzecky, Peter
Department of Environmental
Quality Engineering
University of Massachusetts
Amherst, MA 01003

Moore, William
Browning & Ferris Industries
100 Hallet Street
Boston, MA 02124

Mosher, Dale C.
Wehran Engineering Corp.
666 East Main Street
Middletown, NY 10940

Motts, Ward S.
30 Hills Road
Amherst, MA 01003

Myers, Tom
Environmental Engineering
Division
USAE Waterways Exp. Station
Box 631
Vicksburg, MA 39180

Narbonne, Albert J.
Standard Methods, Inc.
Quabbin Office Building
Valley Road
Barre, MA 01005

Nichols, Robert
1320 Berlin Turnpike
Apt. 324
Wethersfield, CT 06109

Osborne, Arthur H.
138 South Main Street
South Hadley, MA 01075

Pease, Robert
January Hills Road
Leverett, MA 01054

Perlsweig, Paul
Senior Sanitary Engineer
DEP, Solid Waste Unit
State Office Building
122 Washington Street
Hartford, CT 06115

Pharmer, W
39 Mt. Tom Avenue
Holyoke, MA 01040

Potter, Miles M.
22 Longfellow Drive
Cape Elizabeth, ME 04107

Rhyne, Chris W.
Office of Solid Waste
US EPA
401 M Street
Washington D. C. 20460

Rottenberg, Robert
278 Main Street
Room 409
Greenfield, MA 01301

Rundle, David
City Hall Annex
Holyoke, MA 01040

St. Hilaire, William
Department of Environmental
Quality Engineering
Tewksbury State Hospital
Tewksbury, MA 01851

Santos, Edward J.
Chairman
Board of Selectmen
Town of Dighton
Dighton, MA 02715

Small, Maxwell
Building 318
Brookhaven National Laboratory
Upton, NY 11973

Smith, Lawrence R.
8 Chestnut Street
Turners Falls, MA 01376

Stoller, Paul J.
100 East 42nd Street
New York, NY 10017

Tarbell, Kenneth
131 Parker Street
Lowell, MA 01851

Tindilia, Paul
701 Highland Avenue
Princeton, W VA 24740

Tirsch, Frank
50 Meadows Street
#26 Amherst
Amherst, MA 01003

Van Leuven, Nancy
Box 37
Woodbourne, NY 12788

Verrochi, Micheal J.
Browning & Ferris Industries
100 Hallet Street
Boston, MA 02124

Zarette, Francis
Health Department
Shrewsbury, MA 01545