FIRST FINAL DRAFT

GUIDE FOR THE MONITORING AND ENFORCEMENT OF LAND DISPOSAL OF SOILD WASTE

January 1976

PRELIMINARY

ACKNOWLEDGEMENTS.

The monitoring volume of this manual was prepared for the U. S. Environmental Protection Agency, Office of Solid Waste Management Programs, by Wehran Engineering Corporation, Middletown, N. Y. and Geraghty & Miller, Inc., Port Washington, N. Y.

PRELIMINARY

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CHAPTER 1

INTRODUCTION

The land serves as the ultimate repository for over 90% of our Nation's solid waste. Incineration, shredding, and resource recovery processes reduce the amount of solid waste but produce residues requiring disposal.

The main environmental problem of concern at a land disposal site is leachate generation and its resultant potential pollution threat to ground and surface waters. Leachate is liquid which has percolated through solid waste and has extracted dissolved and suspended materials from it. Whenever water comes in direct contact with solid waste it becomes contaminated. In humid areas of the country (where precipitation exceeds evapotranspiration) there will be a net infiltration of water into a land disposal site resulting in leachate generation. In arid and semi-arid areas, precipitation by itself will not be sufficient to result in significant amounts of leachate; however, problems can occur from deficiencies in the site, or its operation and design. The pollution potential of leachate along with the growing concern for the limited assimilative abilities of our Nation's air, water and land resources all point to the importance of monitoring

This manual is primarily concerned with the monitoring of land disposal sites disposing of municipal solid waste (MSW). Emphasis is placed on the monitoring of ground-water quality, with the monitoring well being the key tool in performing this function. The manual is concerned with both existing and new land disposal sites, the former being the more common case.

This manual is primarily addressed to the bureau chiefs of the solid waste regulatory agencies, although its contents can be readily used by operators, researchers and consulting engineers in the field. It is offered as a guide to be used and tailored by the bureau chief, at his discretion, in implementing and directing an effective monitoring and enforcement program in his state and is intended to provide broad general direction and guidance to persons without prior training or experience. It will also bring into one volume information valuable as a reference source for those persons actively engaged in landfill monitoring. It should also prove helpful to the operators and managers of land disposal sites who now will find a need for familiarization and understanding of the fundamental principles involved in ground-water pollution and monitoring. This manual has a companion volume which addresses the enforcement aspects of monitoring. It is intended that, used together, the two volumes will assist the regulatory programs to "bridge the gap" between the monitoring performed and the enforcement data needs.

Generally, this manual includes fundamentals and guidelines to assist the user in,

- establishing the need for monitoring.
- assigning priorities for sites to be monitored.
- implementing and directing a cost-effective, on-going program responsive to the enforcement data needs.

The information, as presented, is offered as guidelines and preferred methods only and site specificity is recognized throughout the manual.

PRELIMINARY

CHAPTER 3

FUNDAMENTALS OF LEACHATE

3.1 INTRODUCTION

As discussed in Chapter 2, it is important to understand and assess the potential for leachate contamination at a land disposal site, in order to properly design, implement and interpret a monitoring program and its data. Here we are referring to leachate production, its quality, quantity and its fate in the hydrogeologic environ. A clear understanding of each of these concepts, their underlying theories, causes and results, should be prerequisite to the design of a monitoring program. Placement of monitoring wells, sampling frequencies, sampling analyses, data interpretation and environmental impact assessment will all benefit from a clear understanding of the above-mentioned concepts.

This chapter presents an overview of the fundamentals of leachate and is keyed into a more detailed presentation in the Appendix of the manual. It is intended that the material presented will be useful to the user of the manual in making an environmental assessment of potential leachate contamination for a particular land disposal site, and utilizing this to properly design and operate a monitoring program and interpret the resultant data. Further, the within information may be useful to regulatory officials in the preparation of background and reference information for enforcement cases.

In approaching the monitoring of a land disposal site, one faces the following

considerations:

- what kind of contamination are we monitoring for?
- how much contamination in terms of concentration and quantity can be expected?
- where, how fast, and how far will the contamination travel?
- how do we best monitor for the contamination?

All of these questions require a clear understanding of leachate production and its fate in the landfill and surrounding environment.

3.2 ORIGIN, COMPOSITION AND FATE OF LEACHATE

In understanding the quality of leachate, and contaminants and the concentrations that may be encountered by monitoring, one must consider its quality as the leachate emanates from the compacted solid waste and its quality as the leachate travels in the subsurface environ. The former would be the quality of pure leachate, while the latter deals with the quality of "leachate enriched ground water."

Precipitation percolates into materials deposited in a solid waste landfill and by lixiviation (dissolving of soluble components) produces a solution called leachate. The landfill leachate under conditions where infiltration is greater than runoff and evapotranspiration combined, moves downward through refuse, and through underlying soil and sediment until it reaches an impermeable layer or ground water. In its journey, leachate traverses three zones of geochemical activity with certain characteristics which are shared and others which are unique to each. The ensuing discussion will describe some of the characteristics in each of the zones and ways in which they interact with the constituents of leachate. The general principles

will be presented here, and a more complete discussion appears in the Appendix.

3.2.1 REFUSE ZONE

Solid waste deposited in municipal landfills is a heterogeneous mixture of organic and inorganic materials and living organisms. Upon deposition, and frequently before, microbial activity begins the degradative process on organic matter. The microbial decomposition of organic matter is encouraged by moisture and warm temperatures.

Microbial activity soon uses up the supply of oxygen and causes the refuse beyond the zone of rapid air diffusion to go anaerobic. Anaerobic conditions cause the end products of decomposition to be somewhat different from carbon dioxide and water which are the products of complete oxidation. Notable among the products of anaerobic decomposition is methane gas. Other organic anaerobic decomposition products such as alcohols, aldehydes, and thiols tend to be more odoriferous than their aerobic counterparts. Of particular importance with regard to leachate, are the anaerobic forms of sulfur, nitrogen, iron, and manganese.

The percolate flows downward through the refuse which is in progressively advanced stages of decomposition, and it passes through layers of buried cover material. Percolate shows a net gain in dissolved constituents as it progresses downward, but may lose some individual ions from cation exchange or other reactions encountered en route.

Nitrogen present in refuse organic matter is released in soluble form with microbial decomposition. In organic substances, nitrogen is in a chemically reduced state. With aerobic decomposition, the nitrogen is oxidized to nitrate ion. Under anaerobic conditions, nitorgen is released as ammonium

ion. Anaerobic conditions are predominant in landfills. Thus most nitrogen in leachate is present as ammonium. The relatively small amount of nitrate produced coupled with its probable denitrification explains the typically low nitrate concentration in leachate.

Organic decomposition releases carbon dioxide in large amounts under aerobic conditions, and in smaller amounts under anaerobic conditions. The enrichment of the interstitial gas in refuse by carbon dioxide results in production of bicarbonate ion. Bicarbonate is frequently a major anion in leachate. Because of the reversibility of the reaction producing bicarbonate, it acts as a pH buffer.

Heavy metals in landfills are primarily in their metallic state and are not soluble. The exception is with deposition of soluble heavy metal salts either as solids or in solution. These may come from certain industrial activities such as electroplating or metal pickling. Most heavy metals occur in solution as cations, but a few are usually present as anions. Anionic heavy metals include vanadium, chromium, and molybdenum.

Phosphorus is released to percolating water by decomposition of organic matter. As discussed below, soils have a high capacity for phosphate attenuation, where as the refuse material does not. Phosphate can be and frequently is produced in substantial amounts in leachate. Were leachate to enter ground water directly, it would almost certainly contribute more phosphate than percolate which has passed through soil and an unsaturated zone.

Water quality parameters which do not measure individual chemical species include biochemical oxygen demand (BOD), chemical oxygen demand (COD), total organic carbon (TOC), color, conductance, and turbidity. The refuse zone provides little, if any, attenuation of these parameters, instead it usually contributes to them.

Fecal coliform and fecal streptococci have been observed in leachate, and poliovirus was reported in leachate from a simulated landfill. The recent trend to use of disposable diapers has increased the source of enteric bacteria and viruses in solid waste. Sewage sludge and septage are also frequently disposed of in municipal landfills.

Movement of bacteria and viruses within the landfill and through the unsaturated zone is dependent upon the porosity of refuse and underlying geologic formations. Refuse may offer many paths through which water can travel relatively unimpeded. If coarse sand and gravel or fractured rock underlie the refuse, percolating water may carry microorganisms with little or no attentuation except for natural die off. These conditions judging from locations which have been studied, are the exception rather than the rule.

Much data on pure leachate quality has been reported in the literature which is worthy of note. The U. S. Environmental Protection Agency has prepared Table 3.1, which illustrates some of the chemical and biological characteristics found in pure leachate and compares fresh leachate to a typical domestic waste water. The quality of leachate depends upon many variables which are specific to each land disposal site. Therefore, a recent EPA report emphasizes the cautious interpretation of reported leachate data:

> "The compositions of leachates reported in the literature are quite diverse..... The breadth of reported data are also typical for individual studies [17] over a long period of time. The many factors that contribute to the spread of data are time since deposition of the solid waste; the moisture regimen, such as total volume, distribution, intensity, and duration; solid waste characteristics; temperature; and sampling and analytical methods. Other factors such as landfill geometry and interaction of leachate with its environment prior to sample collection also contribute to the spread of data. Most of these

TABLE 3-1

CHARACTERISTICS OF LEACHATE AND DOMESTIC WASTE WATERS

Constituent	Range* (mg/l)	Range + (mg/l)	Range‡ (mg/l)	Leachate§ Fresh	Leachate§ old	Waste water§	Ratio§
Chloride (Cl)	34-2,800	100-2,400	600-800	742	197	50	15
Iron (Fe)	0.2-5,500	200-1,700	210-325	500	1.5	0.1	5,000
Manganese (Mn)	.06-1,400	--	75-125	49.	--	0.1	490
Zinc (Zn)	0-1,000	1-135	10-30	45	0.16	--	--
Magnesium (Mg)	16.5-15,600	--	160-250	277	81	30	9
Calcium (Ca)	5-4,080	--	900-1,700	2,136	254	50	43
Potassium (K)	2.8-3,770	--	295-310	--	--	--	--
Sodium (Na)	0-7,700	100-3,800	450-500	--	--	--	--
Phosphate (P)	0-154	5-130	--	7.35	4.96	10	0.7
Copper (Cu)	0-9.9	--	0.5	0.5	0.1	--	--
Lead (Pb)	0-5.0	--	1.6	--	--	--	--
Cadmium (Cd)	--	--	0.4	--	--	--	--
Sulfate (SO_4)	1-1,826	25-500	400-650	--	--	--	--
Total N	0-1,416	20-500	--	989	7.51	40	25
Conductivity (Mmhos)	--	--	6,000-9,000	9,200	1,400	700	13
TDS	0-42,276	--	10,000-14,000	12,620	1,144	--	--
TSS	6-2,685	--	100-700	327	266	200	1.6
pH	5.7-8.5	4.0-8.5	5.2-6.4	5.2	7.3	8.0	--
Alk as $CaCO_3$	0-20,850	--	800-4,000	--	--	--	--
Hardness tot.	0-22,800	200-5,250	3,500-5,000	--	--	--	--
BOD_5	9-54,610	--	7,500-10,000	14,950	--	200	75
COD	0-89,520	100-51,000	16,000-22,000	22,650	81	500	45

*Office of Solid Waste Management Programs, Hazardous Waste Management Division. An environmental assessment of potential gas and leachate problems at land disposal sites. Environmental Protection Publication SW-110.of. [Cincinnati], U.S. Environmental Protection Agency, 1973. 33 p. [Open-file report, restricted distribution.]

+Steiner, R. C., A. A. Fungaroli, R. J. Schoenberger, and P. W. Purdom. Criteria for sanitary landfill development. Public Works, 102(3): 77-79, Mar. 1971.

‡Gas and leachate from land disposal of municipal solid waste; summary report. Cincinnati, U.S. Environmental Protection Agency, Municipal Environmental Research Laboratory, 1975. (In preparation.)

§Brunner, D. R., and R. A. Carnes. Characteristics of percolate of solid and hazardous waste deposits. Presented at AWWA [American Water Works Association] 54th Annual Conference, June 17, 1974. Boston, Mass. 23 p.

** Office of Solid Waste Management Programs. Use of the Water Balance Method for predicting leachate generation from solid waste disposal sites. Environmental Protection Publication SW-168. Cincinnati, Ohio. October 1975. 40 p.

> factors are rarely defined in the literature, making interpretation and comparison with other studies difficult, if not rather arbitrary."

In this same report, the EPA has prepared a comprehensive summary of quality data for both pure leachate and leachate enriched ground water as has been observed and reported by many researchers. The table along with some narrative discussion of the data has been duplicated and included in the Appendix of this manual.

The significance of microbiological organisms in solid waste has been addressed by EPA and worthy of note for selecting monitoring analyses:

> "There is a dearth of information concerning the microbiology of solid waste stabilization as it occurs in a land disposal site. The organisms responsible for stabilization are ubiquitous in nature and are present in the solid waste as well as in the soil. Therefore, there is an ever-present "seed" of organisms, and microbial stabilization is inevitable. The few studies available [3, 4,5, 6,7] confirm this. However, from the viewpoint of environmental contamination, the lack of specifity with regard to types and numbers of stabilizing bacteria and even pathogenic bacteria and viruses, is very disturbing and in need of additional attention and/or research. Peterson has isolated type 3 poliovirus and ECHO 2 from municipal solid waste. Other investigations have determined that one gram of residential solid waste may contain one million or more fecal coliform and fecal streptococci [10,11]. Soiled diapers were found to comprise 0.2 to 2.5% by wet weight of the total waste stream [12] Gaby [11] has shown comparable density of fecal coliform and fecal streptococci (Figure 1). The preliminary findings indicate there is great public health significance associated with any soild waste management process. Additional work is required to clarify these results and determine the bacteriological efficacy of solid waste management processes".

3.2.2 UNSATURATED ZONE

As used herein, the unsaturated zone is defined as the area in soil or sediments between the bottom of the landfill deposits and the water table. The distance can vary between zero (refuse contacting ground water) to several hundred feet. The zone is below what is usually considered "topsoil"

Note: 1. Average of 6 to 8 Determinations

2. Adapted from W. L. Gaby, U.S. EPA Research Contract 68-03-0128, 1972.

3. Total Coliform

4. Fecal Coliform

5. Fecal Streptococci

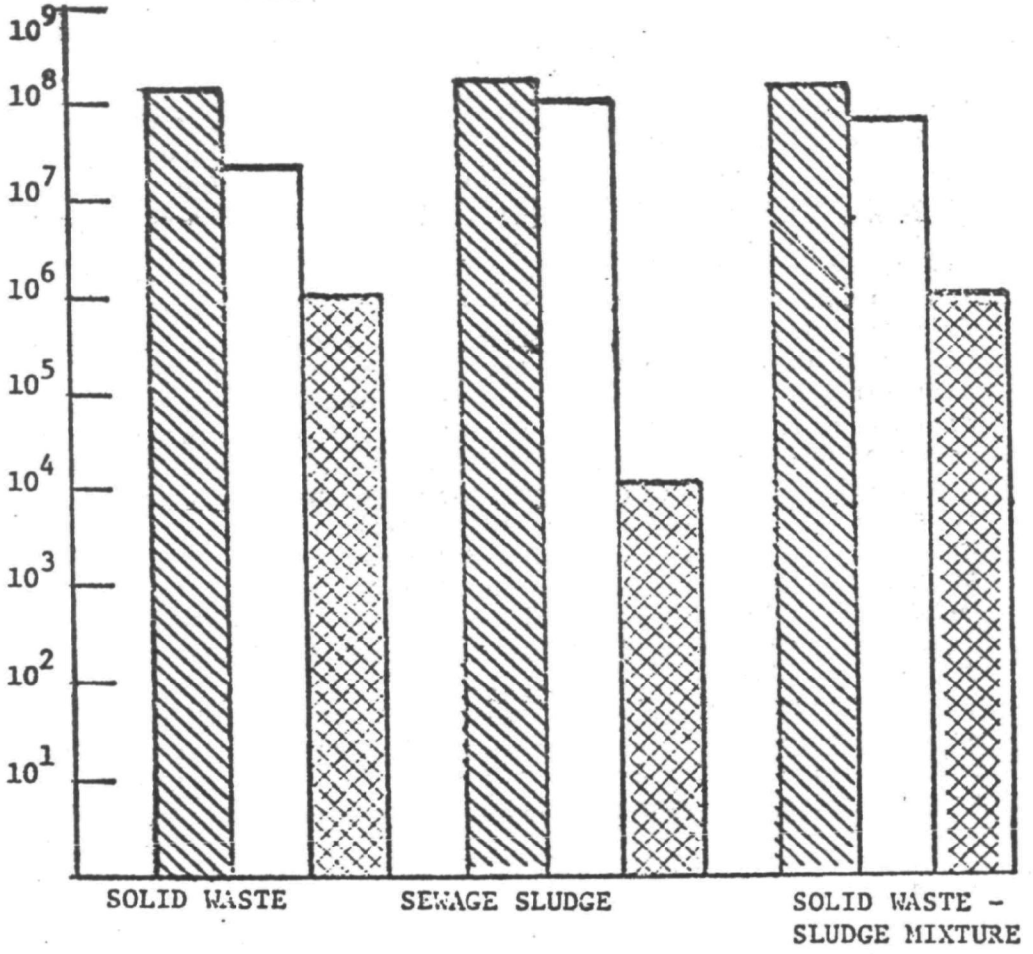

Figure 1.* INDICATOR ORGANISMS FOUND IN WASTE STREAMS.

* REF. 4

or the weathered, organic-matter-rich upper horizons of most soils. At most landfill sites, topsoil has been removed, and sometimes much subsoil also, prior to deposition of refuse. The porous materials comprising the subsoil are likely to be low in organic matter, have a sparse microbial population, and may vary in permeability over a wide range. For purposes of discussion, we will consider the unsaturated zone to be 20 to 200 feet thick. This range allows percolating water an opportunity to react chemically with its environment before reaching ground water. Percolating water has four options in passing through the unsaturated zone. It can move virtually unchanged, can show a net gain of solute, show a net loss of solute, or keep the same total ionic concentration with a net exchange of ions. Since few soils or sediments are chemically inert, changes in transported solute are to be expected.

Chemical activity in the unsaturated zone is primarily located at the surfaces of clay minerals and hydrous oxide coatings. Limited microbial activity may take place either from the indigenous population or that transported from refuse.

Cations will be removed from solution until either the cation exchange capacity is reached, or the limit of displacement reactions is reached. The limit of cation exchange capacity (CEC) can range from nearly zero to probably not more than 60 milliequivalents per 100 grams of soil. Solution concentrations, pH, and percolation rate affect the reactions quantitatively. It should be noted that absorption is not a permanent fixation. Cations may be described with changes in solution composition, pH, or oxidation-reduction (redox) potential.

Divalent and trivalent cations include most of the heavy metals. These are held more strongly than sodium, potassium, or ammonium on the cation exchange complex. Di- and trivalent cations will displace monovalent cations which are adsorbed.

Heavy metals are prone to sorption on hydrous oxide coatings in the soil. The hydrous oxides are frequently cited as so limiting metal solubility that agricultural deficiencies of copper, zinc, and cobalt occur.[5] Attenuation of heavy metals present in leachate is desirable. In locations virtually free of clay minerals, these coatings may be present on sand grains giving the sandy formation some ability to attenuate metallic ions.

Absorption is only one mechanism for removing dissolved ions from solution. Changes in the geochemical environment can also affect solution equilibria. A transition from reducing conditions in the landfill to oxidizing conditions in the unsaturated zone can reduce the concentration of some redox-sensitive species in solution and change the chemical form of others. Iron and manganese will oxidize and precipitate from solution, for example.

If porosity will allow bacterial movement, biochemical reactions involving leachate constituents can proceed. Sulfide and ammonium can be oxidized to sulfate and nitrate. Dissolved organic matter measured in terms of BOD and COD can be reduced through microbial decomposition. Some nutrient elements in the course of these reactions will be incorporated in bacterial cells and thereby be removed from solution until the bacterial cells die off. Conversion of ammonium to nitrate changes nitrogen from a form subject to attenuation to a form which is not. Sulfide to sulfate oxidation is not expected to be as significant. Sulfide can form insoluble precipitates with

many of the heavy metals. For this reason, it may not be present in more than trace amounts in leachate. Microorganisms may also attack the organic ligands associated with chelated and complexed metals. Decomposition or absorption by microorganisms would remove the metals from leachate.

Phosphate reacts with a variety of soil components forming insoluble products. Calcium and phosphate react in solution to form hydroxyapatite, the least soluble phosphate compound known. Iron, aluminum, and manganese can also form virtually insoluble precipitates with phosphate. These reactions lead to a strong attenuation of phosphate when these metal ions are present in the unsaturated-zone.

Carbonate also reacts with calcium, magnesium, and some heavy metals forming relatively insoluble compounds. Calcareous deposits in the unsaturated zone can be valuable in attenuating phosphate and heavy metals from leachate. Because carbonate neutralizes acids, BOD and COD as expressed in organic acid concentration may also be reduced. Carbonate induced alkalinity may change solubilities of heavy metal chelates and lead to a deposition of heavy metals.

The unsaturated zone is influenced by the percolation of leachate into it and influences the leachate which percolates into it. Water of low oxidation potential first infiltrating into the unsaturated zone of high oxidation potential will become more oxidized while simultaneously reducing substances in the unsaturated zone. A continued percolation of reduced water may convert what had been an oxidized system into a reduced one. Or, the percolate may become oxidized if that capacity in the unsaturated zone is greater. The degree of influence of reduced leachate on the oxidized unsatur-

ated zone and vice versa depends upon the reserves of material capable of oxidizing or reducing in the unsaturated zone and leachate. The greater the distance leachate travels between refuse and ground water, the better the chance that the entire path through the unsaturated zone will not become reduced. Raising the oxidation potential of leachate will tend to attenuate some components in solution at the point of exit of the refuse zone.

3.2.3 AQUIFER ZONE

Concepts useful for describing surface water pollution are generally not valid for ground water. Ground-water movement is described by Darcy's Law which states that velocity is directly proportional to the permeability of the aquifer and the hydraulic gradient, and inversely proportional to the porosity. Ground-water flow velocities vary over a wide range with 5 ft/yr to 5 ft/day being a typical range. Highly permeable outwash glacial deposits, fractured basalts and granites, and cavernous limestone aquifers allow very much higher velocities.

The generally slow velocity of ground water allows laminar flow which exhibits different characteristics of mixing than does turbulent flow usually associated with surface streams. A water of different chemical composition from ground water which is injected or percolated into ground water tends to maintain its integrity, and is not diluted with the entire body of ground water. Instead, it moves with the ground-water flow as a plume undergoing minimal mixing.

The plume shape is determined by the physical characteristics of the aquifer.

Porous media give somewhat different shaped plumes from fractured rock or cavernous limestone. Chapter 4 illustrates the paths of ground-water movement in various hydrologic regimes.

Differential attenuation is defined as a reduction in concentration of a dissolved constituent, with distance along the direction of water flow, which is disproportional to changes in concentration of other constituents. Differential attenuation may result from chemical reactions which remove the constituent from solution or from self-destruction. Apparent attenuation occurs from dilution by mixing with water of lower constituent concentration.

Dilution may take place in ground water in two ways. One is hydrodynamic dispersion, and the other is molecular diffusion. Microscopic dispersion describes mixing caused by the tortuous flow of water around individual grains and through pores of various sizes in a porous aquifer. Microscopic dispersion describes mixing as water flows in and around heterogeneous geologic formations. Molecular diffusion operates on a much more restricted scale. It is the diffusion of solute across a concentration gradient from stronger to weaker concentration. Diffusion is seldom possible to measure in the field. There are mathematical formulas which describe dispersion. By measuring enough physical and chemical parameters at a site, over a sufficient length of time, one can calculate an approximate value for dispersion.

Chemical interactions provide the greatest amount of differential attenuation in the aquifer zone. Hydrous oxides of iron, aluminum, and manganese, or clay minerals present in aquifers attenuate cations in the same way that they do in soils or in the unsaturated zone. Because hydrous oxide and clay colloids are in constant contact with water in the aquifer, it can be assumed

that the exchange sites are saturated and essentially in equilibrium with the ambient ground water. Leachate enriched ground water when contacting these colloids will initiate cation exchange which results in desorption of cations which are less strongly held than those replacing them. In this way, hydrogen, sodium, calcium, and magnesium may be released into the aqueous phase by exchange with heavy metals and other cations in leachate. High hardness values associated with leachate plumes may be due in part to this ion exchange phenomenon.

Chemical precipitation in the aquifer is possible if the natural ground water composition includes ions which form insoluble compounds with constituents in leachate. An example would be formation of hydroxyapatite with leachate phosphate and calcium in ground water. Changes in redox potential, buffering reactions, or changes in lithology may produce other attenuation reactions.

The third means of attenuation in aquifers is that termed decay. Oxidation of organic compounds produces carbon dioxide and water, and eliminates the compounds. Radioactive species undergo radioactive decay to stable daughter products, but radioactivity should not be significant in leachate from municipal landfills. Microorganisms carried into the aquifer zone are deprived of a good nutrient supply and are subjected to a generally cooler temperature. This results in a lowering of biochemical activity, frequently to the point of cessation. The inactivation coupled with natural die off tends to reduce bacterial numbers rather rapidly.

There are two additional complications in the interpretation of ground-water quality in leachate plumes. One is the variation in leachate concentration with time, and the other is the discontinuous recharge of leachate which

occurs in most geographical regions. Chapter 6 presents additional discussion on data interpretation.

Leachate production begins as soon as deposited refuse is wetted to field capacity. The lag time depends upon local climatic conditions and rate of refuse deposition. In an active landfill, older organic matter is stabilizing while simultaneously new organic matter is beginning to ferment and produce stronger leachate. The net effect is an increasing leachate concentration from a given area, or an increasing areal contamination, or both as long as the landfill is active.

Leachate produced at the initiation of percolation through the landfill is less concentrated than that produced after several years' refuse accumulation. This leachate will be found at the distal end of the plume of leachate-contaminated ground water. The closer the sampling site to the landfill, the more concentrated should be the contaminated ground water. An increasingly concentrated leachate source in addition to the factors of dilution and attenuation must be considered in interpreting the results of sampling the plume. An erroneously high value for attenuation or dilution may be given if the variation in source strength is ignored.

The intermittent recharge occurring from most landfills also complicates interpretation of leachate-plume configuration. During summer months when evaporation frequently exceeds rainfall, little or no leachate may be produced. Ground water, however, moves under the landfill at a relatively steady rate. Thus, there will be variations in the volume and strength of leachate reaching ground water during the course of time. These variations

will show in the leachate plume as variations in total solute concentration. A sample taken from the plume at any given time may represent a "high" or "low" in the intermittent recharge pattern. One way to visualize this phenomenon would be to watch the response of a conductivity probe in a well screen over time. As leachate-enriched ground water moves past the point, conductivity will vary with changes in dissolved solids concentration. The variations may be noticeable only in time spans of weeks to months. Again, this complicates efforts to calculate values for dispersivity or dilution because concentrations vary from factors other than aquifer characteristics.

A generalized summary of the susceptibility of leachate constituents is provided in Table 3.2. The mechanism of attenuation which affects each constituent is listed for the zones through which leachate may pass. When data are summarized in this fashion. only the principal mechanisms can be cited. For example, no attenuation is listed for all of the constituents in the refuse zone. This is not really true as the previous discussion points out. However, quantification is impossible, and there is a net output of most of the constituents. Sulfate, nitrate, and ammonium are given biochemical conversion alternatives. These ions are subject to oxidation and reduction reactions which may convert or eliminate them. Heavy metals are also prone to one or more of the attenuation mechanisms, and may not be universally present in leachate. Biochemical reactions were not listed for the aquifer zone because biological activity is inhibited. In places, biological activity may be significant in the aquifer, but the amount and type cannot be predicted.

Table 3-2 SUSCEPTIBILITY OF LEACHATE CONSTITUENTS TO DIFFERENTIAL ATTENUATION

Attenuated Constituent	Refuse Zone	Unsaturated Zone	Aquifer
Chloride	0	0	0
Sulfate	0-B	0-B	0
Sulfide	C	C-B	C
Phosphate	0	A-C	A-C
Nitrate	0-B	0	0
Ammonium	0-B	A-B	A
Sodium	0	0	0
Potassium	0	A	A
Calcium	0	A	A
Magnesium	0	A	A
Heavy metal anions (Cr,V.Se,B,As)	0-B	0-B	0
Heavy metal cations (Pb,Cu,Ni,Zn,Cd,Fe,Mn,Hg)	0-A-C	A-C	A-C
Organic nitrogen	0	B	0
COD	0	B	0
BOD	0	B	0
Volatile Acids	0	B	0
Phenols	0	A-B	0-A
MBAS	0	A-B	0-A

0 = no attenuation
A = adsorption
B = biochemical degradation on conversion
C = chemical precipitation

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3.24 Measurement of Attenuation

From the previous discussion, it is evident that attenuation describes two phenomena associated with the way solute is transported. One is dilution which results from dispersion and diffusion, and the other is dilution resulting from chemical or biochemical removal of solute from ground water. The former type of dilution is referred to as apparent attenuation because no active chemical processes are operating to reduce the concentration of dissolved constituents.

In the field, it is important to distinguish between apparent and active attenuation because prediction of future conditions depend upon the extent of active attenuation. To accomplish this, several samples of leachate-enriched ground water must be collected along the path of travel of the leachate plume. Chemical constituents measured in these samples are then chosen on the basis of their susceptibility to attenuation, and relative changes in concentration with distance from the source are noted.

Chloride is the best constituent to measure as a indicator of dilution. Because it carries a negative charge and does not form precipitates with the common cations in water, chloride is unaffected by ambient conditions. Reductions in chloride concentration can then be attributed to the result of dispersion and diffusion. If ground-water equations were used in an attempt to calculate dispersion coefficients for leachate-enriched ground water, chloride concentration data would be first choice for use in the calculations. Nitrate reacts in virtually the same way, but nitrate is less frequently present in leachate in comparable concentrations.

Concentrations of other constituents sampled simultaneously with chloride should represent equal dilution. If they are observed in lower than expected concentrations, this indicates that active attenuation has taken place. Conversely, if their concentrations are greater than those calculated on the basis of chloride, desorption from ion exchange sites or contributions from other sources may account for the nontheoretical results.

An example which is calculated from data obtained in a landfill study is presented below.[3] The cations calcium, sodium plus potassium, ammonium, and iron in leachate-enriched ground water are plotted in percentage of original concentration vs. distance from the landfill (Figure 3-1). Were all of the cations diluted equally, they would plot on the same curve. Reference points for chloride are included to facilitate a comparison of the theoretical dilution-only curve with the actual cation concentration curves.

The plume of leachate-enriched ground water represented by Figure 3-1 is produced by a landfill that has been active for 28 years. The leachate plume can be traced about 10,600 ft. downgradient from the landfill. The plume extends vertically throughout the thickness of the aquifer (about 80 ft) with the most concentrated contamination near the bottom.

Calcium remains above the chloride curve throughout the length of the plume. This is in agreement with other reports which have indicated that calcium is desorbed from clays as a result of cation exchange with leachate components.[3] In this specific situation, there may also be a contribution of calcium from septic tank effluent.

Ammonium remains above the chloride curve for about half the length of the

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plume. It also may be desorbed and is also a component of septic tank effluent. The loss of ammonium at more distant points of the plume may be due in part to generally more aerobic ground-water conditions that allow nitrification of ammonium.

Sodium and potassium generally plot below the chloride points, and iron is even more attenuated. Probably the iron is removed largely through solubility changes resulting from increases in Eh at longer distances from the landfill.

The geohydrologic environment is characterized by soft, rather acid native ground water in a highly silicaceous unconsolidated sand and gravel aquifer. Uncontaminated ground water contains iron in concentrations which frequently exceed the recommended drinking water limit of 0.05 mg/l. No significant amounts of clay or silt are present in the path of the leachate plume. Sand grains are coated with iron oxide which probably exhibits a small amount of cation exchange capacity. Septic tanks in use in the area and intermittent recharge of leachate as governed by climate complicate the interpretation of chemical data along purely theoretical principles.

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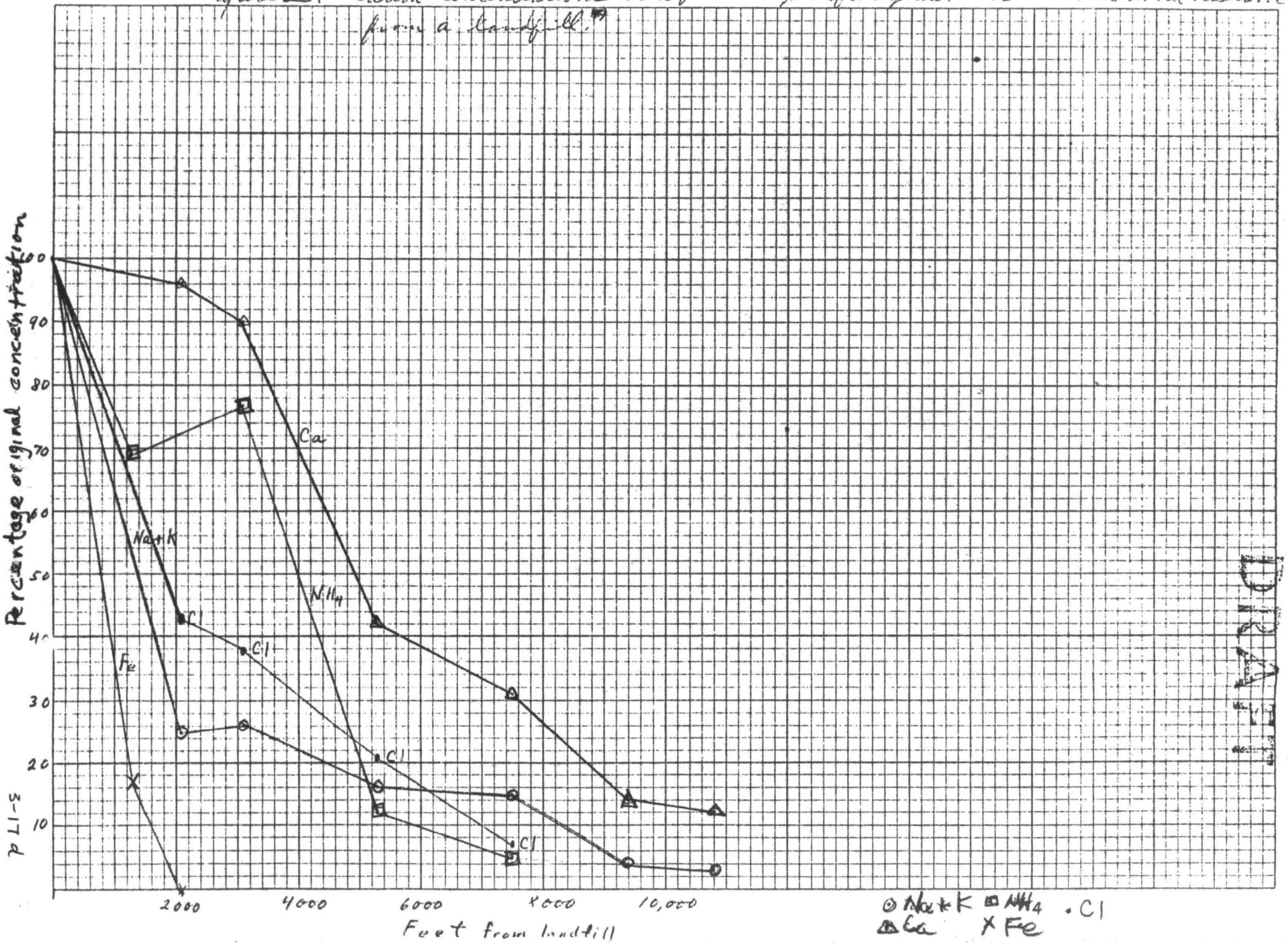
Figure 3-1. Cation concentrations as a percentage of original concentration with distance from a landfill.



3.3 LEACHATE QUANTITY

In a recent EPA report on leachate [6], leachate production was phased into perspective. It stated:

> "It becomes quite evident that the main parameter affecting leachate quality and quantity is purely and simply the quantity of water flowing through the solid wastes. Generally, the more water that flows through the solid waste, the more pollutants will be leached out. Therefore, the proper sanitary landfill design and operational approach is to eliminate or minimize percolation through the solid waste. With the smaller amounts of percolation, the pollutants tend to be more concentrated, but the rate at which they are transmitted to the surrounding environment is not so apt to exceed the capability of the natural surroundings to accept and attenuate most of them to some degree."

Therefore, you can see that the volume of leachate generated is influenced in both the extent of a leachate contamination problem and the relative strength of the leachate and its concentration in the ground water being monitored.

The water balance method has been presented as a useful tool in estimating average leachate quantities at a land disposal site.

> "The sanitary landfill site is a part of the classical hydrologic cycle. The governing criteria for determining leachate volume are those describing the phenomena occurring at the cover material surface. A water balance can be written:

$$W_R + W_{SR} + W_{GW} + W_{IR} = I + R + E \ldots\ldots (1)$$

where W_R = input water from precipitation

W_{SR} = input water from surrounding surface runoff

W_{GW} = input water from groundwater

W_{IR} = input water from irrigation

I = Infiltration

R = Surface Runoff

E = Evapotranspiration

Infiltration can be defined:

$$I = \Delta S_s = \Delta S_R + L + W_D \quad \ldots\ldots(2)$$

where ΔS_s = change in moisture storage in soil

ΔS_R = change in moisture storage in solid waste

L = leachate

W_D = water contributed by solid waste decomposition

Proper design and operation can eliminate input water from surrounding surface runoff, groundwater and irrigation. Some control can be exerted over infiltration, evaporation, surface runoff, and moisture storage capacity of soils and solid waste. The volume of water produced during solid waste decomposition is generally considered negligible." (4)

Figure 3-1 conceptually depicts the above water balance equations. In addition, the above-referenced report presents very useful information and data on the following:

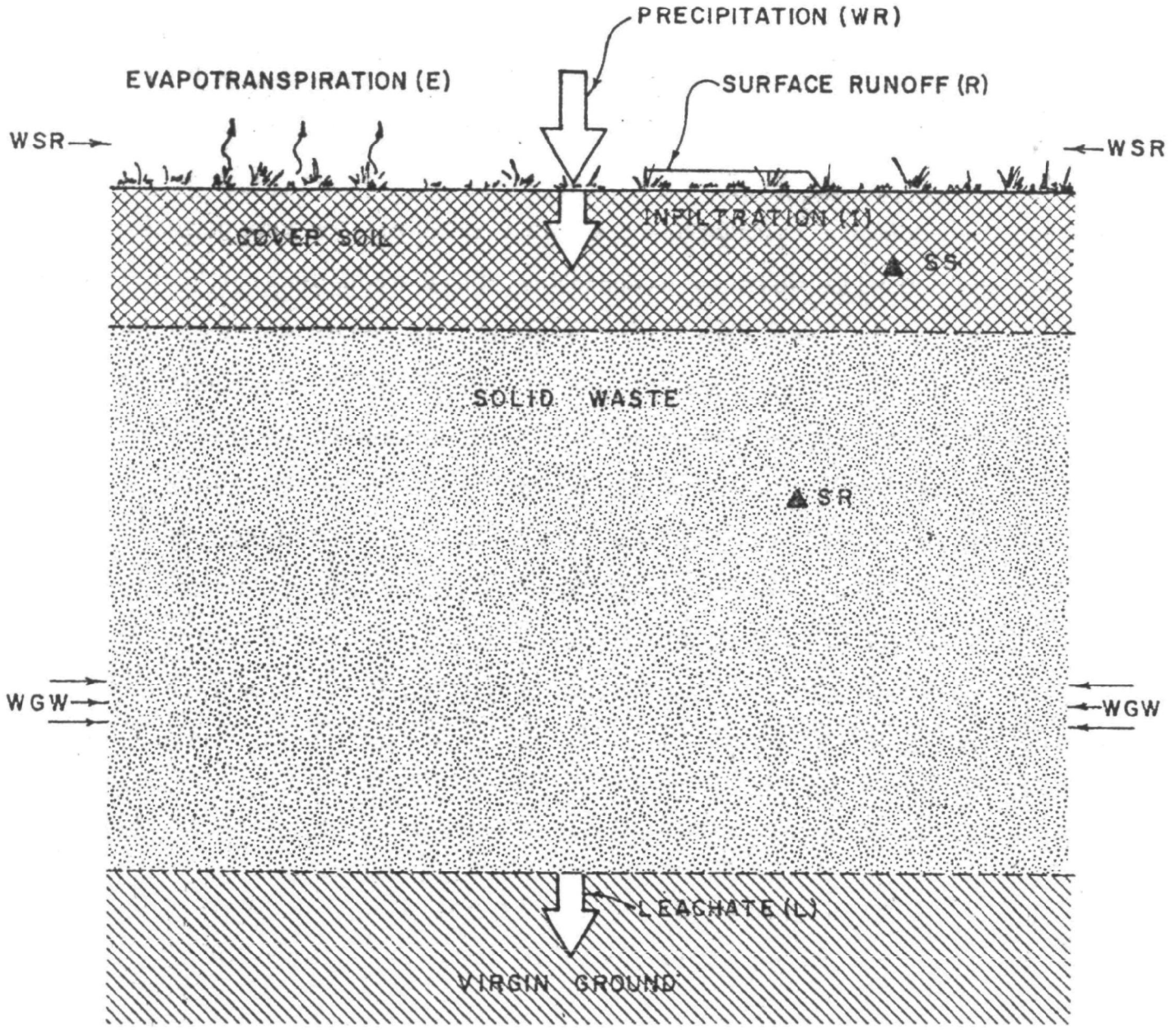
1. the influence of slope, surface condition, and soil type on the quantity of runoff and the potential for leachate production.
2. the dependency of infiltration on the storm frequency, duration, intensity and soil moisture conditions.
3. the influence of vegetation on evapotranspiration and infiltration
4. the relationship of soil permeabilities to infiltration rates and volumes.
5. the moisture retention capabilities of various types of soils as well as compacted municipal solid waste.

All of the above-referenced information provides useful input in assessing leachate at a land disposal site. For the convenience of the users of this

manual, sections of the EPA summary report on leachate have been reproduced and included in the Appendix.

In another EPA report, [3] the water balance method was applied to three hypothetical landfills. These examples are worthy of note and have been included in the Appendix of this manual. They demonstrate how to determine time of first appearance of leachate as well as subsequent average leachate generation and volumes.

Caution must be exercised in applying the water balance method to sanitary landfills. Review of the above-referenced information clearly shows the extreme sensitivity of leachate quantity estimates to the many variables used in the water balance calculations. For example, slight changes in runoff coefficients, evapotranspiration or moisture retention figures can result in a significant change in the leachate quantity estimate. In addition, unless extensive on site measurements are performed, the many parameters in the water balance calculations are purely theoretical and empirical estimates. Therefore, this manual presents the water balance methods as a useful tool for planning, design and assessment purposes and the leachate quantity estimates generated should be viewed with this qualification in mind.



SANITARY LANDFILL WATER BALANCE

FIGURE 3-1

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4.1 MONITORING NETWORKS

In the context of this report the ultimate goal of any monitoring program is to gauge and evaluate ground-water degradation, ~~if any,~~ caused by ~~due to~~ landfill leachate. A presence/absence system is the minimum acceptable approach - is there leachate in the ground water? This approach will work in situations where ground-water contamination did not pre-exist the monitoring network, in other words, a network installed prior to any landfilling operation. As is often the case, landfill operation and subsequent leachate generation have been going on for some time prior to installation of a monitoring network. If contamination already exists, the monitoring program must provide the requisite data for the management program. Here, the maximum feasible approach is a quantitative evaluation of total contaminant accumulation in the aquifer, rates of accumulation and attenuation, and the contaminant dispersion pattern and its controls. To implement this approach, a time sequence of three-dimensional data on the contaminant body is required; ~~and~~ with this information, the proper management scheme can be devised. For any given sanitary landfill, the correct monitoring system will be between or even include, these extremes.

There are two basic approaches to monitoring: active and passive. The former has a measurable, continuing impact on the ground-water regime, considerable altering the flow system in which the contaminant source is located. An active monitoring system is essentially a pumping well which intercepts ground-water from the area that might potentially be affected by the contaminant (Figure 1). Theoretically, any

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contaminant entering the zone of intercepted ground-water flow would eventually be detectable in the monitoring well discharge. This approach is most suited for point source, "one-shot" contaminants introduced into the ground water from such sources as a tank leak or underground nuclear explosion. Unfortunately, this type of monitoring scheme has several drawbacks preventing its application to sanitary landfills: 1) the larger the contaminant source, the greater the number of pumping wells required to intercept ground-water flow; 2) contaminant concentrations will be greatly reduced by the volume of water withdrawn, perhaps below detection limits; 3) disposal of the pumped water can be a problem, especially if contaminated; 4) pumping costs over a period of years will be astronomical; and 5) pumping will accelerate the spread of leachate through the aquifer and eventually the monitoring system will become a pumped withdrawal system.

Passive monitoring, on the other hand, is ideally suited to monitoring landfill leachate. In this type of scheme, wells or other monitoring devices, strategically located in reference to ground-water flow directions, are sampled at regular intervals to determine chemical constituents in the ground water at that point and time. Flow-pattern disruptions are kept to a minimum. This is a system that can be used to monitor continuous, long-term contaminant input from a point source - the situation that would exist at a landfill. To monitor the area which might be affected by the contaminant, a "picket fence" of non-pumping wells is required (Figure 2). The main drawback, compared to the

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active monitoring approach, is that more than one well is required. However, these can be small-diameter wells and sampling procedure costs can be kept to a minimum and should cost less in the long run than a major pumping installation.

Background Data Requisite for Monitoring Network Design

Prior to monitoring well construction, some thought should be given to their placement. A certain amount of information is required: 1) ground-water flow direction; 2) distribution of permeable and impermeable materials; 3) type of aquifer porosity; 4) effect of pumping, present or future, on the flow system; and 5) background water quality. ~~This~~ These data can be obtained by installing a series of low cost wells, collecting sediment samples during drilling, and measuring water levels in the completed wells. Background water quality can be determined from chemical analysis of water samples from these wells. A hydrogeologist or engineer familiar with ground water, may be able to "best guess" the information without actual field work. However, unless such personnel are involved in designing the monitoring network, every effort should be made to collect this information at the site. With this information, monitoring wells can be placed to most effectively intercept any contaminant ~~bulb~~ plume spreading from the landfill.

Monitoring Networks for Sanitary Landfills

The minimum acceptable monitoring well network will consist of one line of three "picket fence" wells downgradient of the

landfill and perpendicular to ground-water flow, penetrating the entire saturated thickness of the aquifer; one well within the confines of the landfill, screened so that it intercepts the water table; and a well completed in an area upgradient from the landfill that will not be affected by potential leachate migration. The actual number of wells will be dictated by the size of the landfill, the hydrogeologic environment, and budgetary restrictions, but there should be a minimum of five at each landfill and ideally one "picket fence" well for every 250 feet of landfill frontage perpendicular to flow direction.

Even if wells are sited according to the background information described above, there is a high probability of one or more of them not intercepting the ~~bulb~~ plume of leachate-enriched ground water because of inhomogenities in aquifer material, ~~etc~~. Sequence of land-filling operations also has a significant effect on the shape of the leachate plume and the possibility of a well not detecting leachate, (Figure 3). For these reasons, it is better to err on the side of too many monitoring wells rather than too few.

Once contamination is detected, several "picket fence" lines of these wells can be constructed and used to gauge downgradient dispersion and attenuation of the leachate, providing the information necessary for predicting the ultimate fate of the plume. If, however, the contaminant exists in a complicated hydrogeologic regime, information on its vertical distribution is required to predict plume behavior and assess its impact. This is the maximum feasible approach and will be a time consuming and expensive process, the necessity for which will depend on the gravity

of the threatened environmental impact or regulatory requirements.

The single well in the landfill, provided it is properly constructed, will give an indication of whether or not leachate is reaching the ground water before it is detected in the downgradient "picket fence" monitor wells. Once detected here, it may be only a matter of time before leachate-enriched ground water reaches the downgradient monitor wells. For this reason, it serves as an important early warning system of potential large-scale aquifer degradation. Detection of leachate in this well should trigger a response from the landfill operator, either implementation of remedial action or wait and see what the "picket fence" wells show. The actual course of action is dependent on federal, state, and local statutes and enforcement agencies governing ground water contamination or an evaluation by the operator (or operator's consultant) of the consequences of aquifer degradation.

Some problems can result from relying entirely on this well or similar wells within the landfill to monitor leachate infiltration to the ground water. First, it is located within the landfill proper and does not provide information on the downgradient extent of leachate-contaminated ground water. Then too, it skims water from only the surface of the aquifer. If any density stratification is occuring within the contaminant bulb, the well would give an unrealistic picture of actual leachate concentration in the ground water.

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The major problem, however, is the potential for artificially elevated leachate concentrations in water samples due to improper monitor well construction. Since the well is constructed within the landfill itself, an improperly backfilled annulus can act as a conduit for downward movement of leachate, introducing it into the aquifer sooner than might have occurred naturally, if at all. Proper construction requires some type of impermeable seal in the annular space between the well casing and the borehole wall, either bentonite or neat cement grout. Because grout can shrink and bentonite can dry and crack, their placement is not a complete guarantee of plugging the annular area and stopping downward movement of leachate. However, neglecting to place this seal during monitor well construction is almost certain to speed and promote ground water contamination.

An upgradient monitor well provides water samples indicative of background water quality, or in other words, the chemical character of "naturally" occuring ground water. This well should be sampled at regular intervals, and the analytical results used as a baseline for comparison with results from the landfill and "picket fence" monitor wells. The background well can also provide information on outside interferences, that is contaminants in the ground water, naturally occuring or otherwise, not due to landfill leachate. When a constituent's concentration rises above acceptable levels in all the wells, an outside

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interference is indicated. A naturally-occuring example of this is iron-rich ground water and artificially-induced examples of contamination are nitrate or sulfate from agricultural operations or low pH due to acid mine drainage. Therefore, proper water-quality baseline date is necessary for the correct interpretation of the chemical analysis of monitor well samples both by the operator and enforcement agencies receiving water quality reports.

Aquifer-Specific Monitoring Networks

Introduction-Aquifers can be subdivided on the basis of porosity when considering how to design an aquifer-specific monitoring network. Although the general design described above will be applicable to all aquifers, there will be important variations on this scheme dependant on aquifer porosity. These variations include monitor well density, depth, construction, drilling method, probability of successful detection of the contaminant plume ~~bulb~~ and alternative sampling methods. To try and provide a guideline to investigators for setting up their own, site-specific monitoring network, aquifer-specific networks are described for materials with inter-granular fracture, and solution porosity.

~~Table 1 is a summary of the information presented in this section. It must be kept in mind however, that these are not hard and fast designs that can be implemented at any site with 100% probability of intercepting the contaminant bulb.~~ Hydrogeologic conditions at a particular site will require modification of the basic design

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Table 1 - Typical Intergranular Porosities

Material	Porosity, per cent
Clay	45 - 55
Silt	40 - 50
Medium to coarse mixed sand	35 - 40
Uniform sand	30 - 40
Fine to medium mixed sand	30 - 35
Gravel	30 - 40
Gravel and sand	20 - 35
Sandstone	10 - 20

in order for the monitoring network to be effective. These basic designs are presented here as guidelines, not standards, because they are the minimum feasible approach to monitoring and by no means will provide all the answers in all of the various hydrogeologic environments found in the United States. At some landfills, more and different placement of monitoring devices may be required, but at no landfill should there be less than the aquifer-specific design presented here.

In an aquifer with intergranular porosity, water occurs in the void spaces between individual particles of the aquifer material, Figure 4 a,b,c. The voids are created by incomplete grain to grain contact, in other words, the individual particles comprising the aquifer touch primarily at points and not entire surfaces. Therefore, open spaces are left for water to fill. These voids must be interconnected and capable of transmitting water in order for the porous material to be considered an aquifer. Typical aquifers with intergranular porosity are unconsolidated sands and gravels in river valleys, coastal plains, intermontane valleys, alluvial plains, and sandstone formations. Sandstone porosity and permeability is reduced compared to that of unconsolidated sands because of grain cementation; cements such as iron oxide partially or completely fill the pore spaces in order to bind the grains together. (Figure 4 d) Clay and silts, also composed of individual particles, can be highly porous, but the voids are not interconnected and they do not readily transmit water. Typical porosity values are shown in Table 1.

As a result of past structural deformation, consolidated sediments and igneous and metamorphic rocks are fractured (broken) to a greater of lesser degree depending on the itensity and/or frequency of the deformation and the rock type. Water can fill these fractures, and if they are interconnected and capable of transmitting water, can form an aquifer. (Figure 4f). The ability of the aquifer to transmit water depends on fracture density and openness: the greater the density of fractures and the wider they are, the greater their ability to transmit water. In some cases, the rocks can have primary and secondary porosity; primary porosity (intergranular porosity) is produced during sediment deposition or rock formation and secondary porosity is caused by fracturing or solution activity after this. Sandstone is an example of both types of porosity since voids exist between the sand grains and sandstone formations are usually fractured. However, intergranular porosity dominates unless the voids are filled with cement.

Carbonate rocks are susceptible to solution by water moving through fractures. With time, these fractures are enlarged into cavities, creating large open spaces in the rock. Figure 4e). If the cavities are connected, ground water can move very rapidly through the aquifer. In fact, carbonate aquifers can have transmissivities of several million gpd/ft. Solution openings and sinkholes provide

open pathways for leachate to reach the ground water, and once there, it will move rapidly through the aquifer. Carbonate rocks can have both intergranular and fracture porosity, but where solution cavities exist, ground water will preferentially move through them.

Type I Network (Intergranular Porosity Aquifers)

Placement, both areal and vertical, of monitor wells in any hydrogeologic environment should be done in reference to ground water flow paths. Rather than discuss this at length simplified diagrams are used to represent typical flow patterns in the type of aquifer discussed and the positioning of monitor wells to effectively intercept any contaminant plume ~~bulb~~. Figure 5 A and 5 B represent vertical and areal flow distribution respectively, in a homogeneous, isotropic sand aquifer. Monitor wells A, B, and C are the background, landfill, and "picket fence" wells discussed above. To review, Well A provides baseline water quality data, Well B is an early warning system showing leachate reaching the water table immediately beneath the landfill and Well C is intended to intercept any plume of leachate contaminated ground water.

Areal placement of the upgradient, background monitor well (Well A) is critical if there is a ground water mound associated with the landfill. This mound, produced by increased infiltration due to landfilling, causes a certain amount of upgradient flow from the landfill (Figure 4B). If Well A is located in this

zone, sampling will produce an anomalous water-quality baseline. Unfortunately, no rules of thumb can be given as to the separation between this well and the landfill proper as the extent of upgradient flow will depend on a variety of factors including amount of infiltration through the landfill and aquifer characteristics. The best policy would be to locate the well at the most distant upgradient point at the landfill site or on adjacent upgradient property if permission of the owner can be obtained. Well depth is not critical, assuming there are no apparent contamination sources in the vicinity, but better baseline data would be provided if the well were screened through the saturated thickness of the aquifer.

Well B can be located anywhere in the landfill but preferentially in the first section to be filled. As discussed above, great care must be taken during construction. In hydrogeologic environments where the water table is 5 to 10 feet below the landfill, monitoring the zone of aeration is probably not necessary because Well B will detect leachate entering the ground water. However, where the unsaturated zone is 10 feet or greater in thickness, some monitoring device is required to detect to downward percolation of leachate before it reaches the water table. Pressure-vacuum lysimeters can be used to trace downward movement of leachate in the vadose zone and can provide data on the amount of attenuation and the likelihood of leachate reaching the water table.

The "picket fence" wells (C wells, Figure 5B) should be

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immediately downgradient of the landfill in order to intercept the leachate plume as soon as possible. Once leachate enters the ground water, it is difficult to control and the sooner it is detected, the easier it is to evaluate its impact and initiate remedial action, if necessary. Well C is shown screened through the entire saturated thickness of the aquifer. This is recommended because the actual flow path of the leachate plume is not known unless previously defined by head relations in a large number of observation wells. The flow path of leachate-enriched ground water shown in Figure 4A is characteristic if the landfill is located in the aquifer recharge area. If the landfill were closer to the point of discharge, in this case the river, the plume would probably be higher in the aquifer. However, since the physical behavior of contaminant bodies hasn't been completely described yet, there is almost no way of knowing where the ~~bulb~~ plume of contaminant will be within an aquifer. Therefore, the monitoring device has to collect water over the entire saturated thickness of the aquifer and the simplest way to do this is to screen the entire interval.

This type of construction can cause some problems. The primary problem is that the well can contribute to the vertical spread of contaminant by providing a conduit for downward movement of intercepted ground water. If the aquifer is 50 feet or less in thickness, this is not a major problem because natural flow conditions would tend to uniformly distribute the leachate throughout the aquifer, especially in recharge areas. Thicker

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aquifers, 100 to 200 feet of saturated thickness, tend to have more pronounced shallow and deep flow systems and there is a chance the leachate plume would remain the shallow flow system. A well screened over the entire saturated thickness provides a movement path from the shallow to deep flow systems. Of course, this is a gross simplification and is intended as a guideline only. The actual flow pattern will depend on the hydrogeologic environment in which the landfill is located.

This drawback can be overcome by using a well cluster (see Section_____) but the landfill investigator will have to balance the extra cost against the probability of promoting the vertical spread of contaminants. Also, well clusters are not particularly effective in aquifers thicker than one hundred feet. Because the exact vertical location of the plume is not known, overlapping or sequential (0 to 10 ft, 10 to 20 ft, and so on) screens are required. For example, in an aquifer with one hundred feet of saturated sediment, five wells with twenty feet long screens are required to cover this interval. As aquifer thickness increases, screen lengths must increase proportionally and the information on the vertical distribution of contaminant becomes less and less precise. If the screens do not overlap, the contaminant ~~bulb~~ plume could pass between the screened intervals and remain undetected.

Another problem is dilution of leachate below analytical detection limits when a sample is pumped from the well. This

would result in a time lag between first arrival of leachate-enriched ground water at the monitor well and its first detection in the sampled water. The magnitude of this lag is hard to predict but, as the contaminant travels in a defineable ~~bulb~~ zone with a small zone of diffusion to uncontaminated water, it shouldn't take very long for the zone of diffusion to move past the well and leachate contaminated ground water start to enter. Then, if detection is still a problem, concentration or extraction techniques could be used for key leachate tracers to determine their presence or absence in the sample. Once detected, a more sophisticated sampling device is required, perhaps well clusters, sampling during drilling, or others discussed below, since C wells are only designed to show presence or absence of leachate and not vertical distribution.

Type II Monitoring Network (Fracture Porosity Aquifers) -

Ground water flow patterns are not as predictable in fractured rock aquifers as they are in aquifers with intergranular porosity. Unless there is primary porosity, as in a sandstone aquifer, ground water flow patterns will be controlled by the fracture pattern. Again, flow patterns are presented visually in Figures 6A and 6B, rather than attempting to describe them in great detail. The same configuration of A, B, and C monitor wells can be used in this hydrogeologic regime (Figure 6A). However, the wells are not screened but rather are open hole with the exception of casing-off

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surficial materials to prevent them from caving into the open hole.

A major problem in fractured rock terrains is intercepting the fractures that might contain leachate-contaminated ground water with a monitor well. As shown in Figure 6C, a well can fail to intercept any fractures and will be dry, necessitating another well nearby. Worse yet, a monitor well could intercept a set of fractures not connected to the landfill and fail entirely to show leachate entrained in the ground water. Without intensive and expensive geologic analysis, it is not possible to predict flow paths other than general ground water flow direction at the site. Monitor wells cannot be precisely taken into account in planning a monitoring network and evaluating the results. To compensate, a high well density is required, perhaps one monitor well for every 100 feet of landfill frontage perpendicular to ground water flow.

Another problem is specifying well depth. Fractures are squeezed shut with depth because of the weight of overlying material. If shut, fractures are unable to act as conduits for water movement and the rock can no longer be considered an aquifer. Closed fractures therefore provide a downward limit on leachate movement. A general rule of thumb is that fractures tend to close at depths of 300 feet or greater, and monitor wells probably should not be drilled deeper unless there is geologic information to the contrary.

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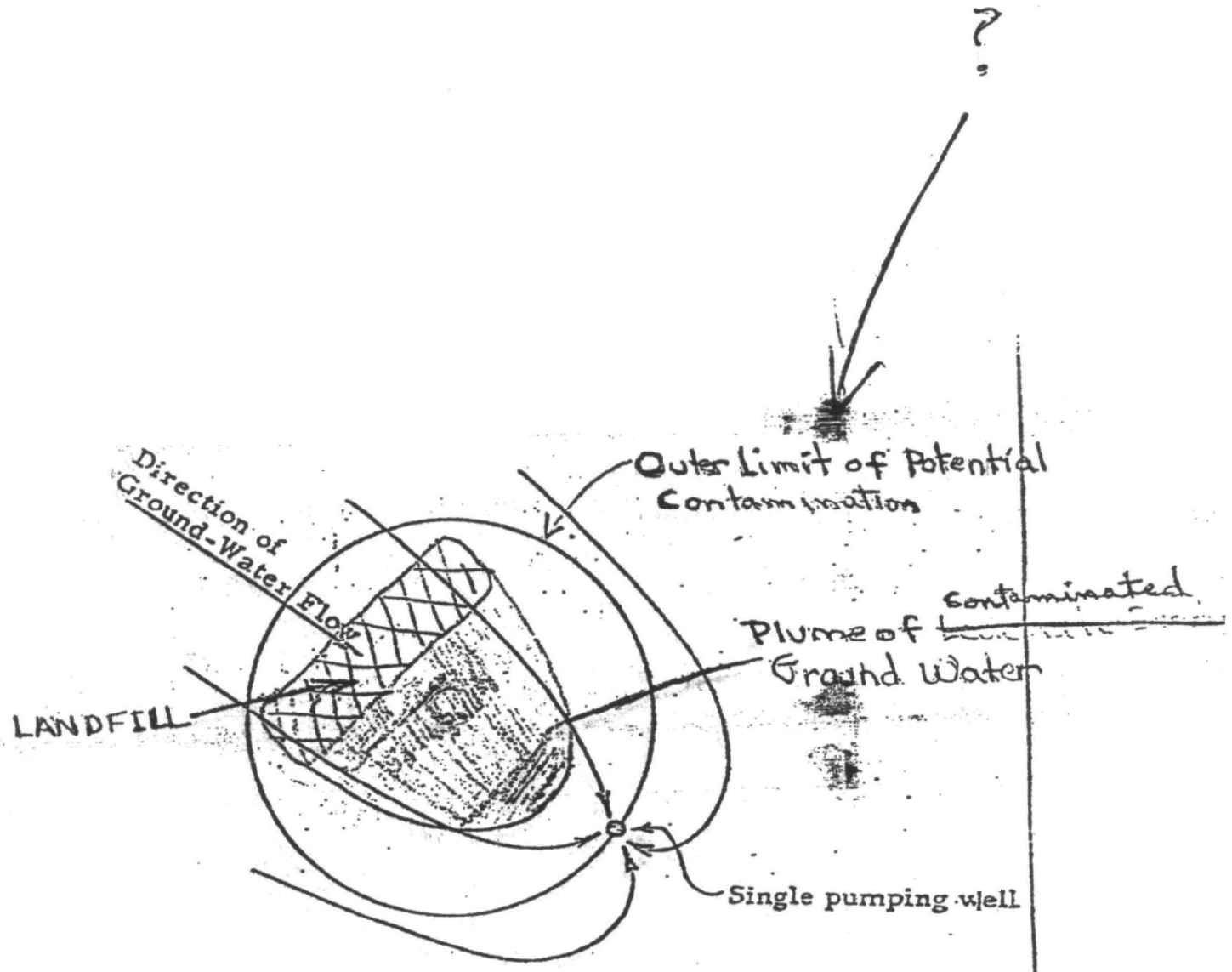
Type III Monitoring Network (Solution Porosity Aquifer) -

Similar to fractured rock aquifers, ground water flow patterns are going to be controlled by solution openings or fractures in carbonate rock aquifers. The positioning of the A, B, and C monitor wells in this type of flow system is shown in Figures 7A and 7B. The monitoring network is the same as that for fractured rock with the same problems: 1) intercepting the solution cavities and 2) well completion depth. Again, increased well density can solve the former but there are no handy rules of thumb for the latter, as in fractured rocks. Sinkholes can be 100 feet or more deep and there is no telling how deep solution cavities will persist. Unless the solution cavities follow a well known regional fracture system, there is no way to predict their position prior to placing a monitor well without geologic analysis. A trial and error approach to placement is mandated by the hydrogeologic environment and there is no assurance that the wells will intercept the contaminant ~~bulb~~.



INTERCEPTION BY PUMPING WELL

Figure 1

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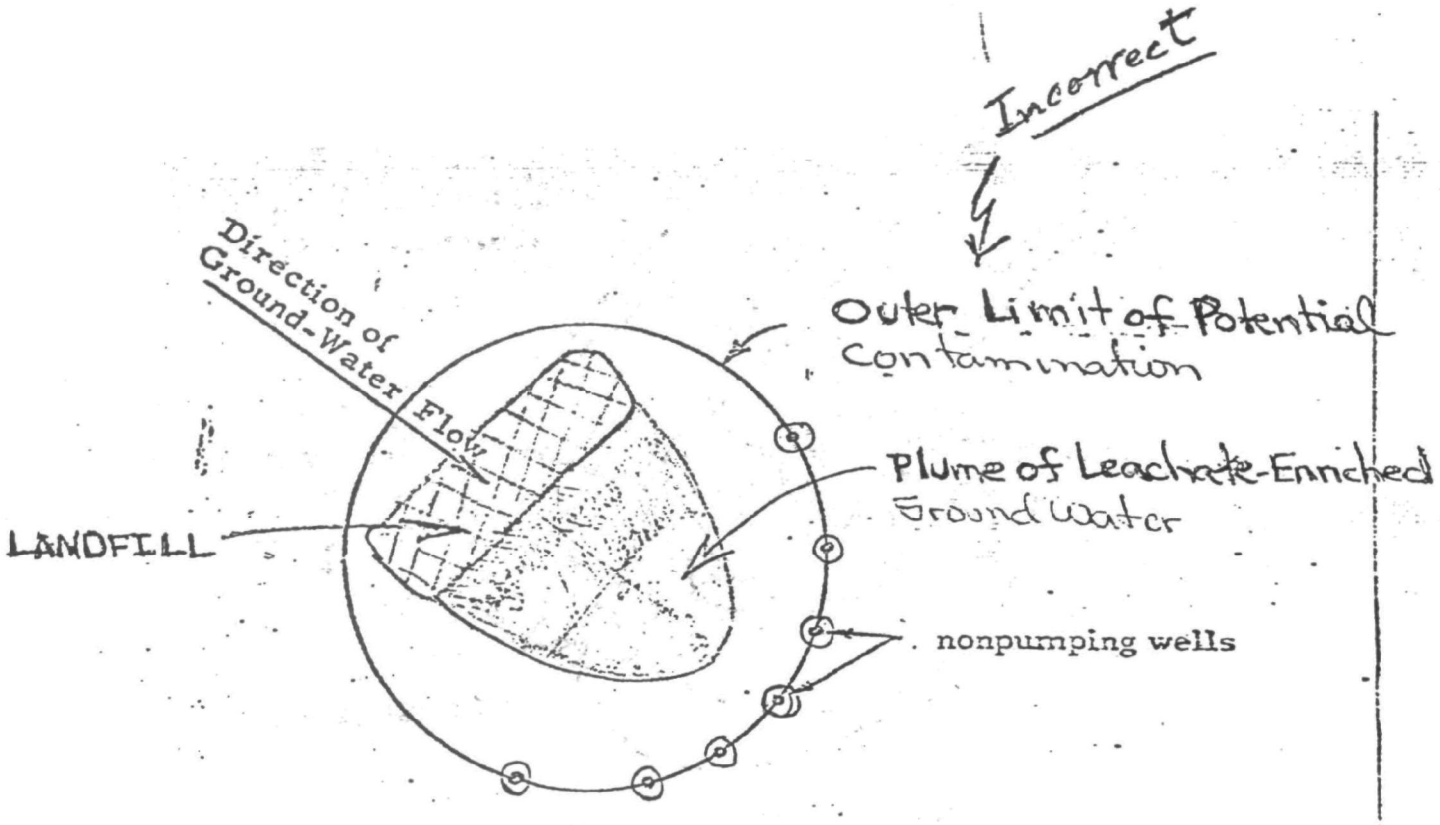
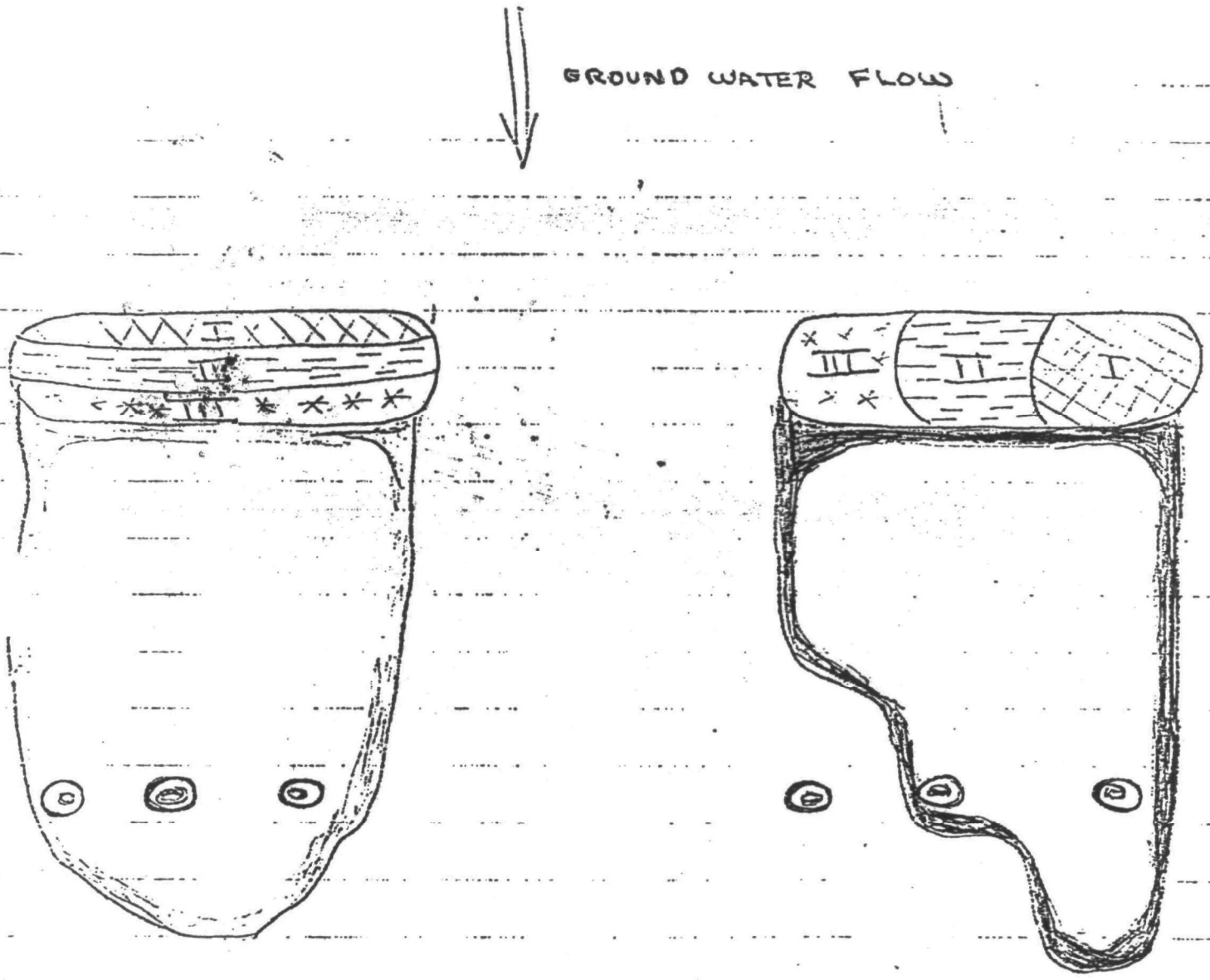


INTERCEPTION BY NONPUMPING WELLS

Figure 2



Legend

I, II, III sequenced landfilling operations

◎ Monitoring Well

Extent of leachate Plumes by landfill operation

Figure 3

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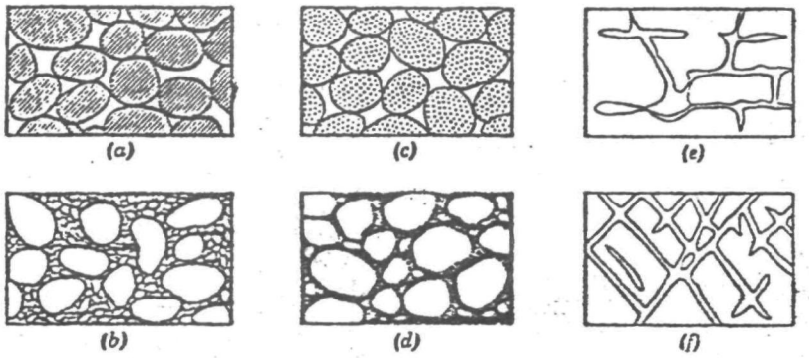


Aquifer Porosity

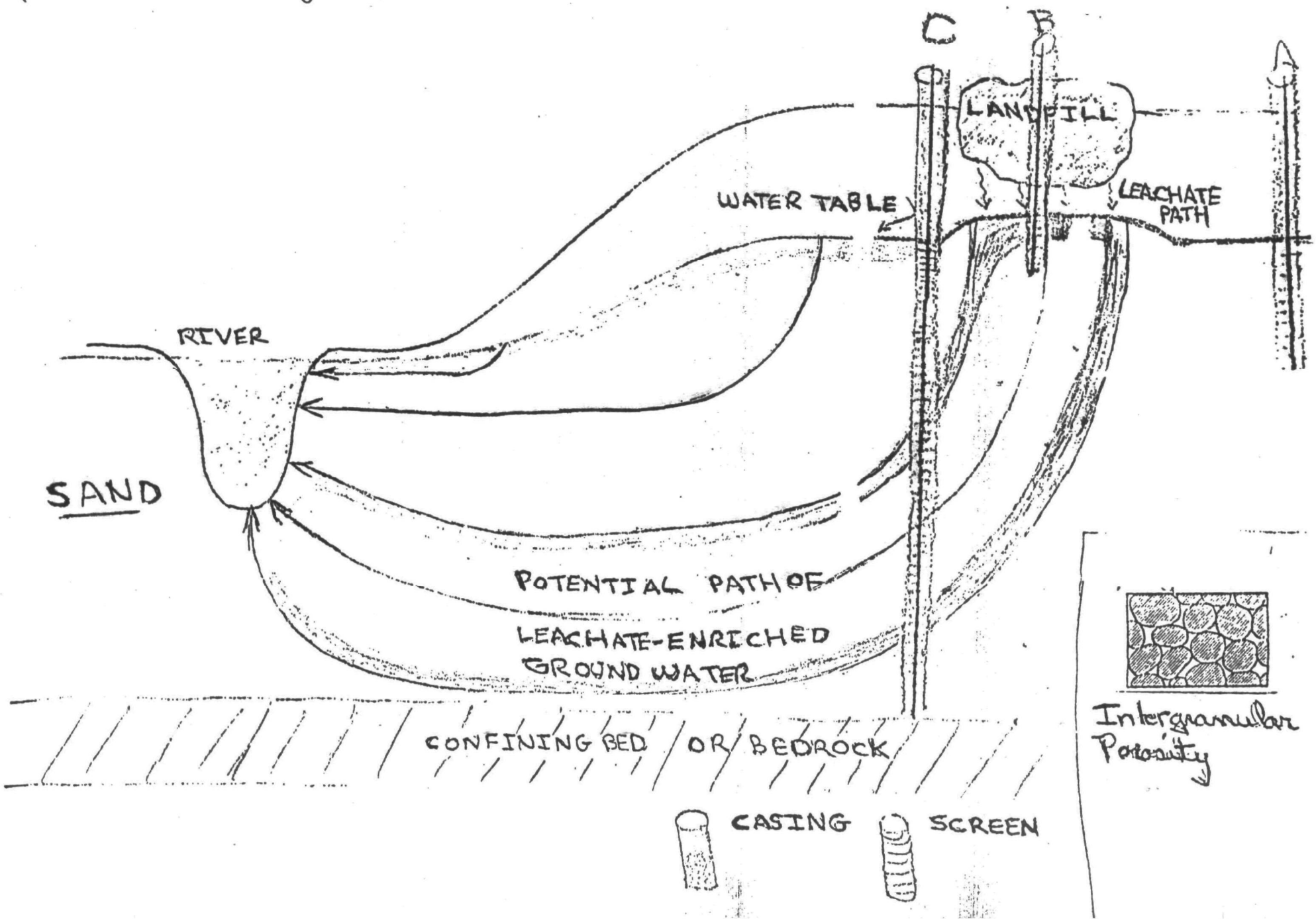
a) sediments with high porosity
b) sediments with low porosity
c) porous grains contributing to overall high Bo porosity
d) low porosity due to grain cementation
e) solution porosity
f) fracture porosity

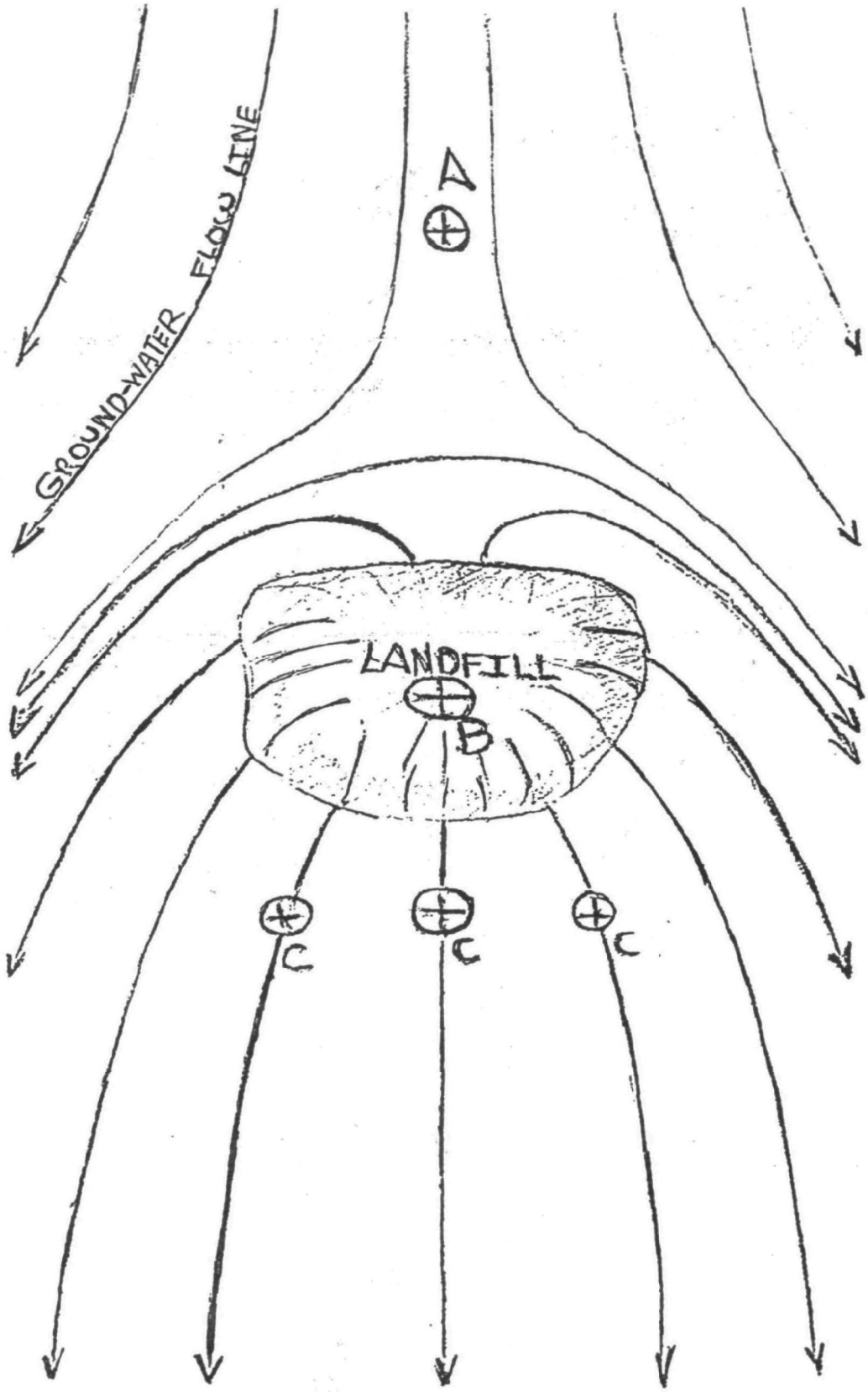
Davis & DeWiest, 1966

Figure 4

Monitoring Network for Aquifers with Intergranular Porosity

Figure 5 A Vertical flow patterns





Monitoring Network for Aquifers with Intergranular Porosity

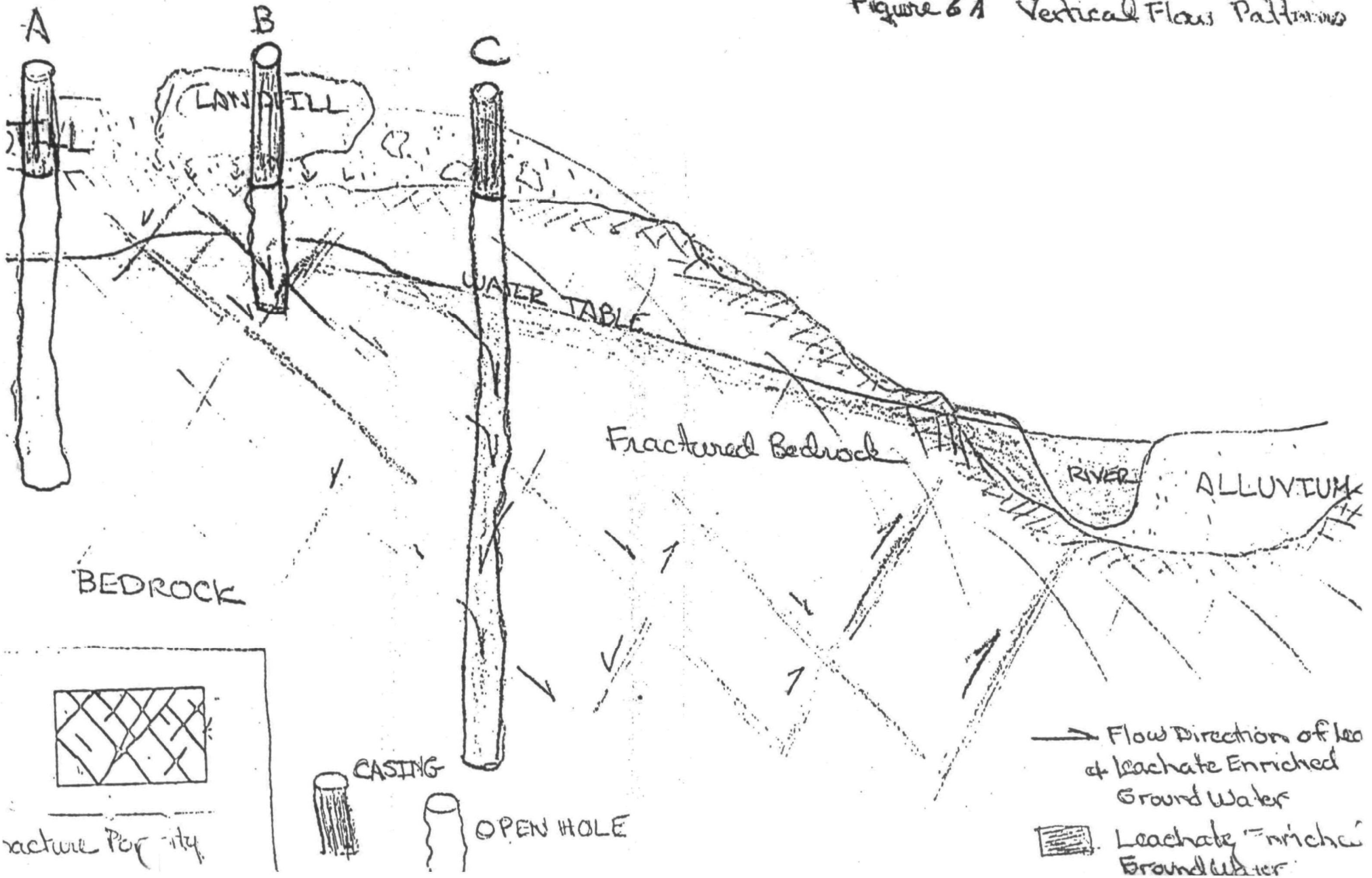
Figure 5B Areal flow patterns

→ Flow direction of leachate-enriched ground water

⊕ Monitor well

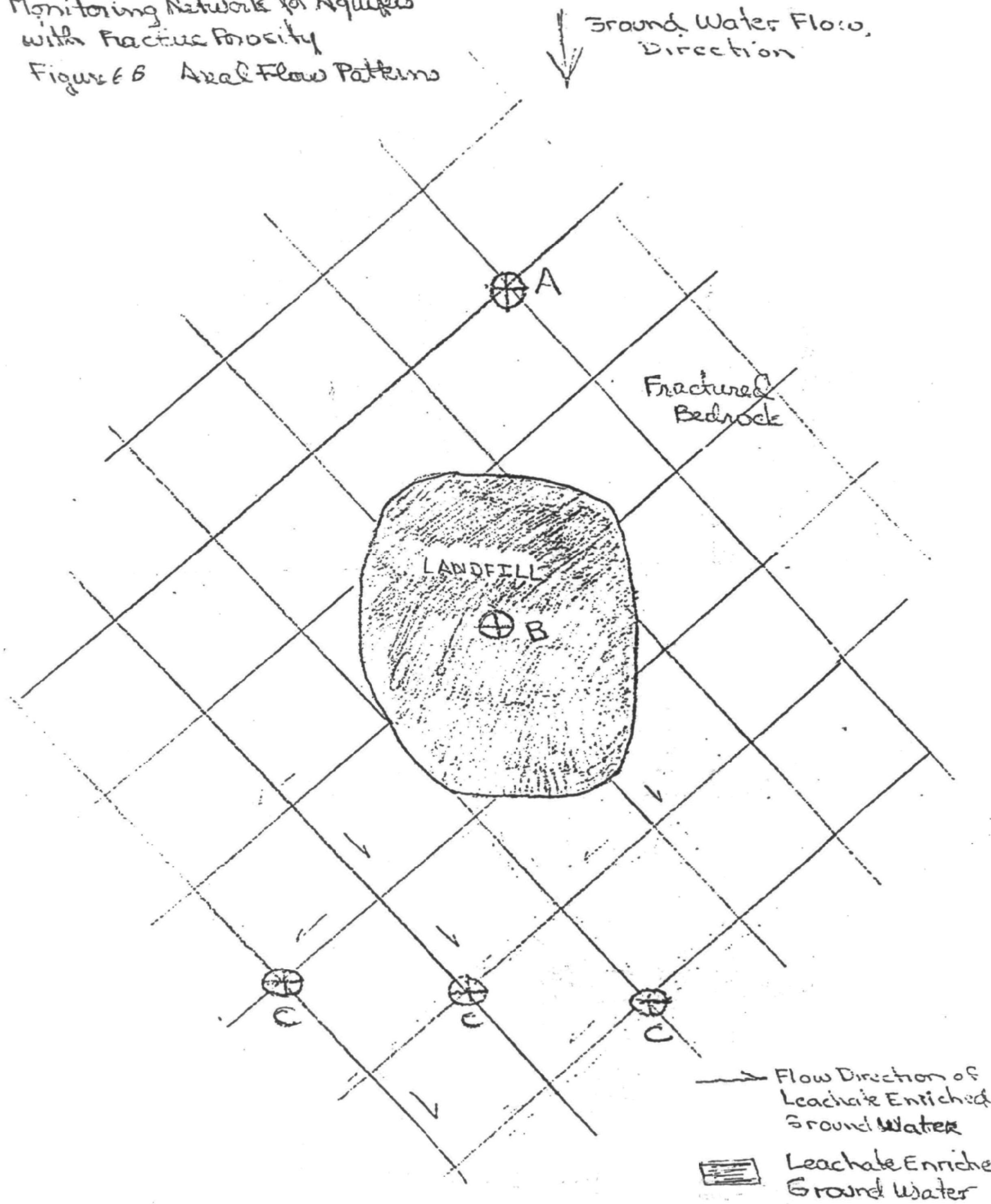
Monitoring Network for Aquifers with Fracture Porosity

Figure 6.1 Vertical Flow Patterns



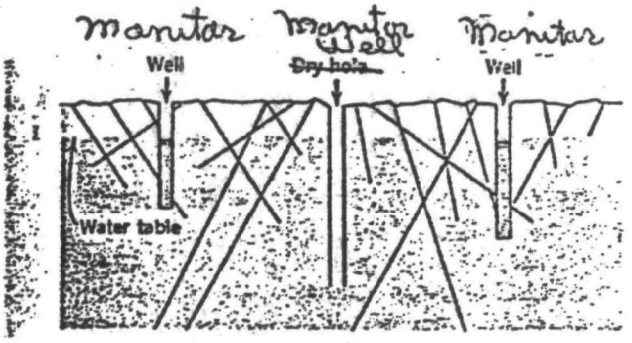
Monitoring Network for Aquifers with Fracture Porosity

Figure 6 Areal Flow Patterns



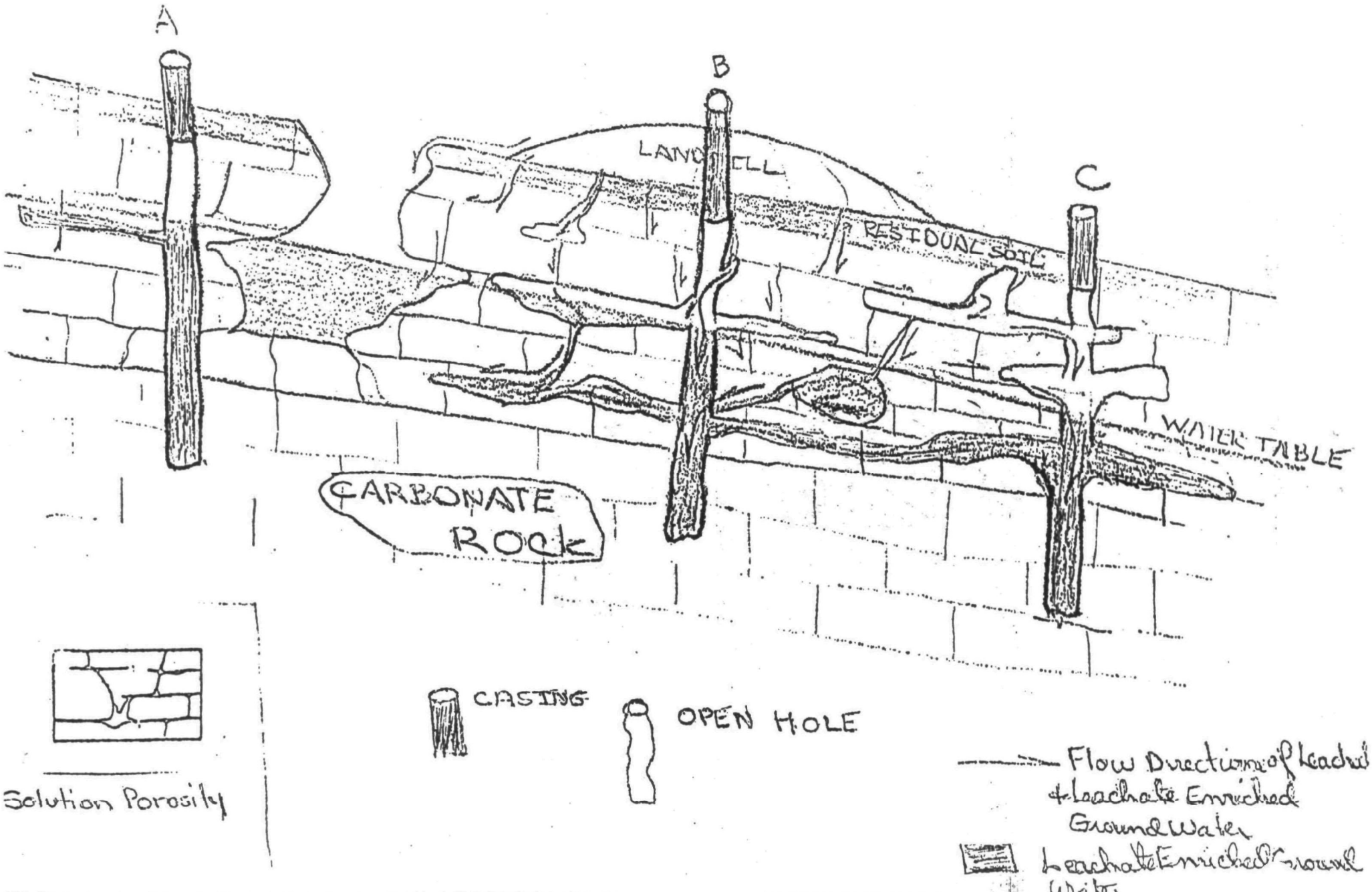
Monitoring Network for Aquifers with Fracture Porosity

Figure 6 C Monitor Well Failure to Intercept a Fracture

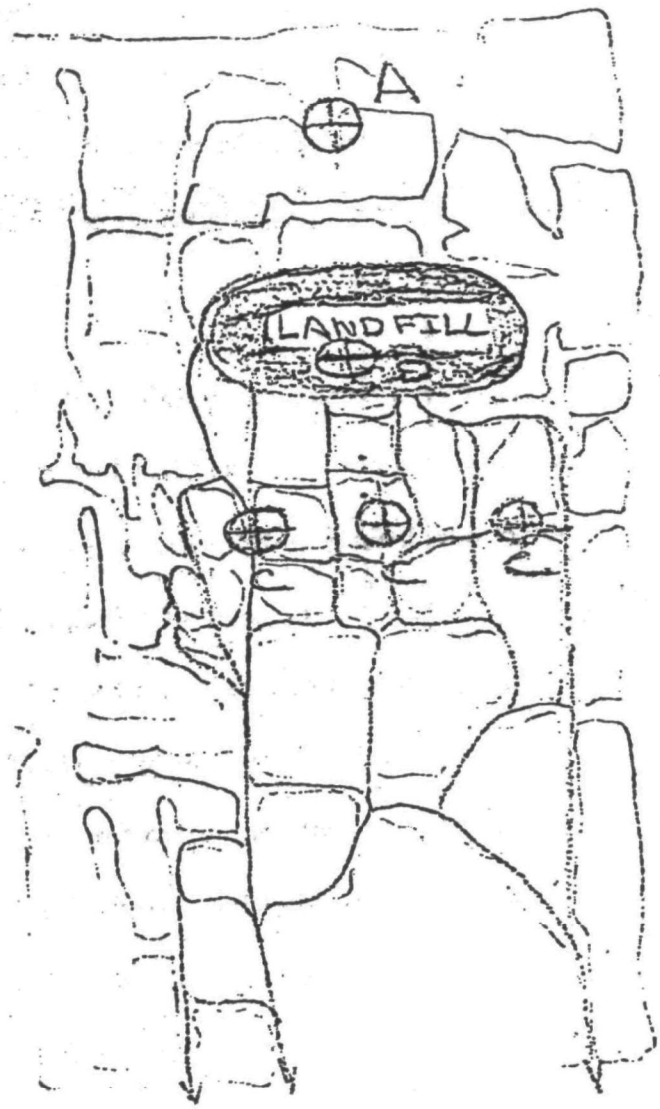


"Picket fence" of monitor wells downgradient of landfill

Monitoring Network for Aquifers with Solution
Figure 7A Vertical flow patterns



Ground Water
Flow Direction



Monitoring Network for Aquifers with Solution Porosity
Figure 7 B Areal Flow Patterns

Flow Direction of Leachate Enriched Ground Water

Leachate Enriched Ground Water

4.2 LEACHATE MOVEMENT IN DIFFERENT HYDROGEOLOGIC SETTINGS

The rate, direction and distance of leachate travel from a landfill to an ultimate discharge point will be largely determined by the hydrogeologic setting. The leachate plume may be confined to the landfill site or it may travel large distances; it may be divided into multiple plumes, move into different aquifers and reverse its direction. It is clear, then, that a landfill monitoring program must account for all possible routes of leachate movement if it is to be effective.

The following series of diagrams illustrate a number of hypothetical hydrogeologic landfill settings. These diagrams are schematic and only intended to illustrate general leachate flow principals. Both the geology and hydrology of the settings are necessarily somewhat simplified over most actual condition, however, the general principals illustrated are still valid. In addition, such complicating factors as differential attenuation of contaminants by subsurface sediments and interference with leachate flow by production wells, have been omitted. Clearly if all factors influencing leachate migration from a landfill were considered, the number of possibilities would be almost limitless. And, this is precisely the reason why each individual landfill should be subjected to a hydrologic investigation prior to the establishment of a pollution abatement or monitoring system.

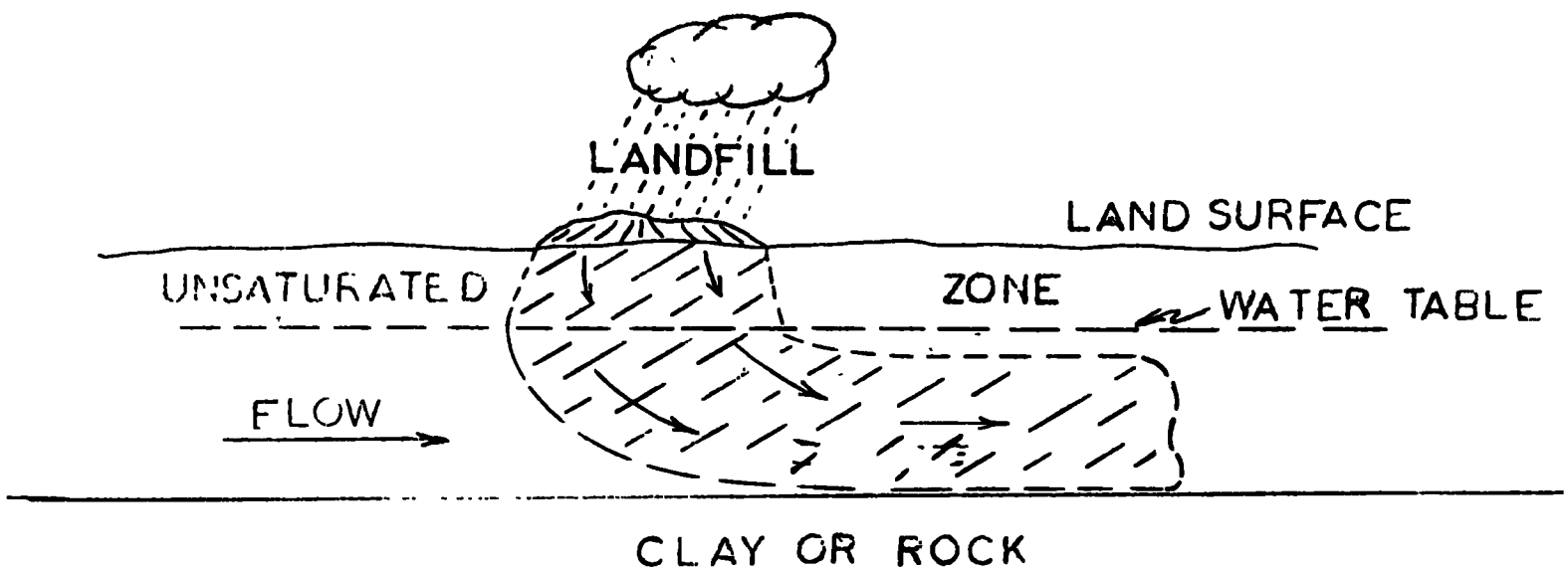


Figure 1. A single aquifer with a deep water table. Leachates percolate vertically downward from the landfill to the underlying aquifer and then moves downgradient as a bulb or plume in the direction of groundwater flow. The mass of leachates may sink to the bottom of the aquifer if of a heavier specific gravity, or float near the top of the water-bearing unit if the leachates are predominately hydrocarbon in nature.

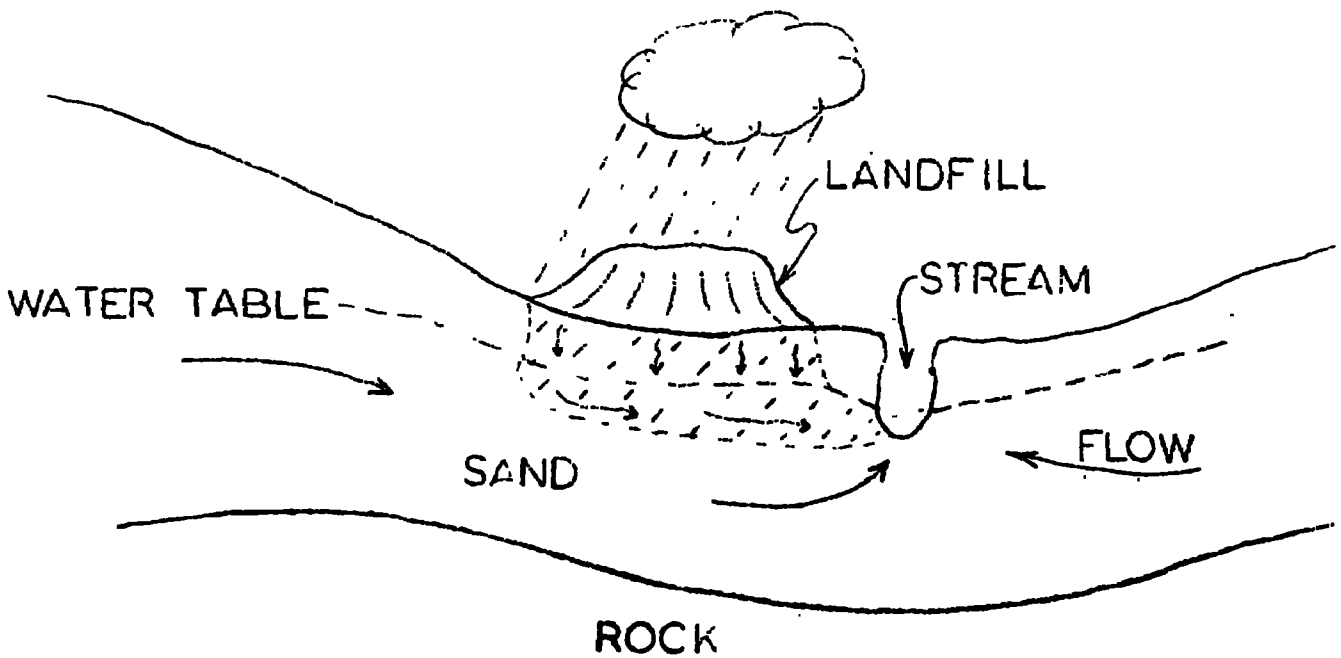


Figure 2. Landfills located in stream-flat ground water discharge areas and within the zone of saturation are always in contact with ground water moving from higher-land recharge areas to the stream discharge point. In such cases, leachates are transported with the ground water to the stream where it becomes diluted with normal stream flow waters.

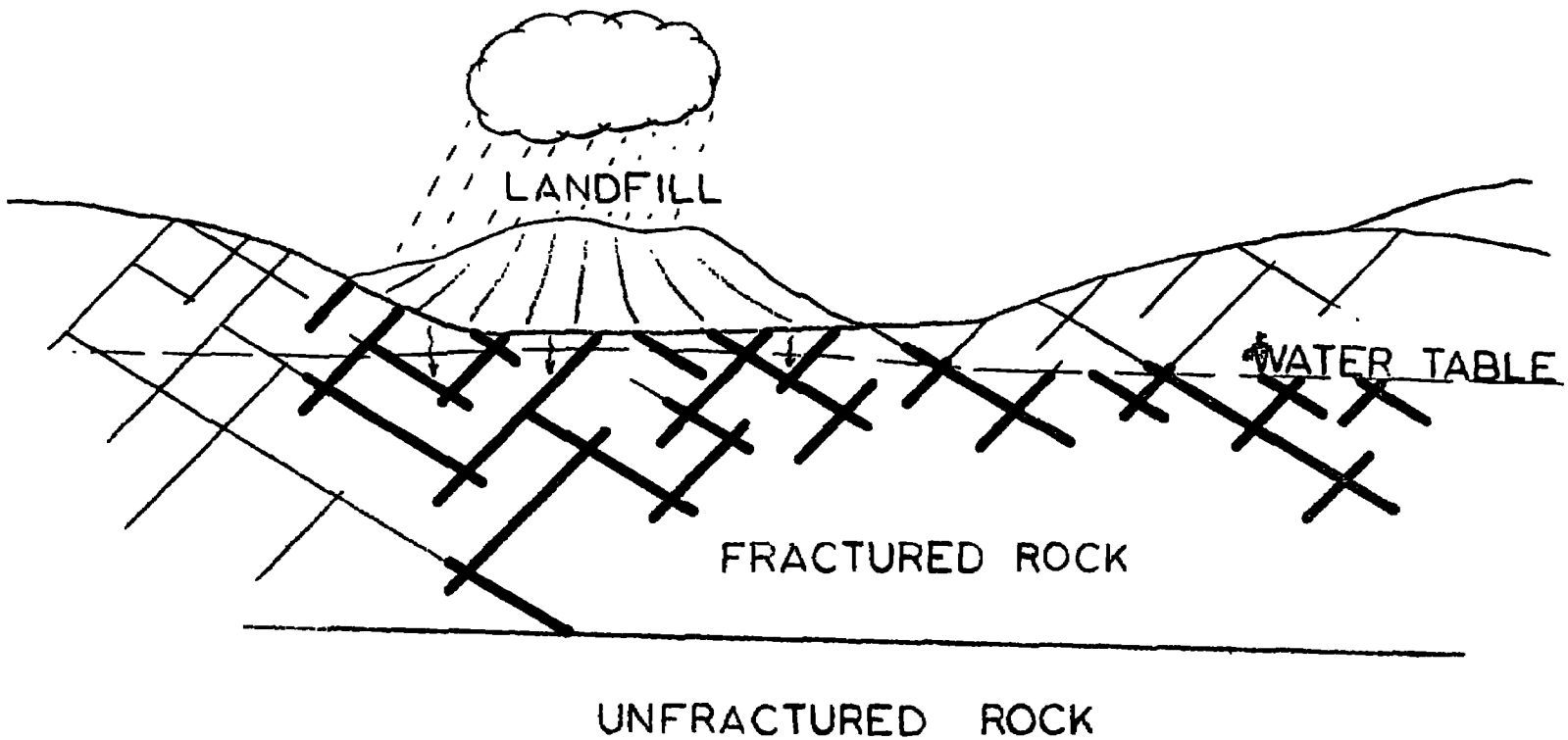


Figure 3. Landfills positioned over fractured rock surface in a high water table area, permit leachates to migrate downgradient along interconnected rock fractures to some lower natural discharge area or a pumping well.

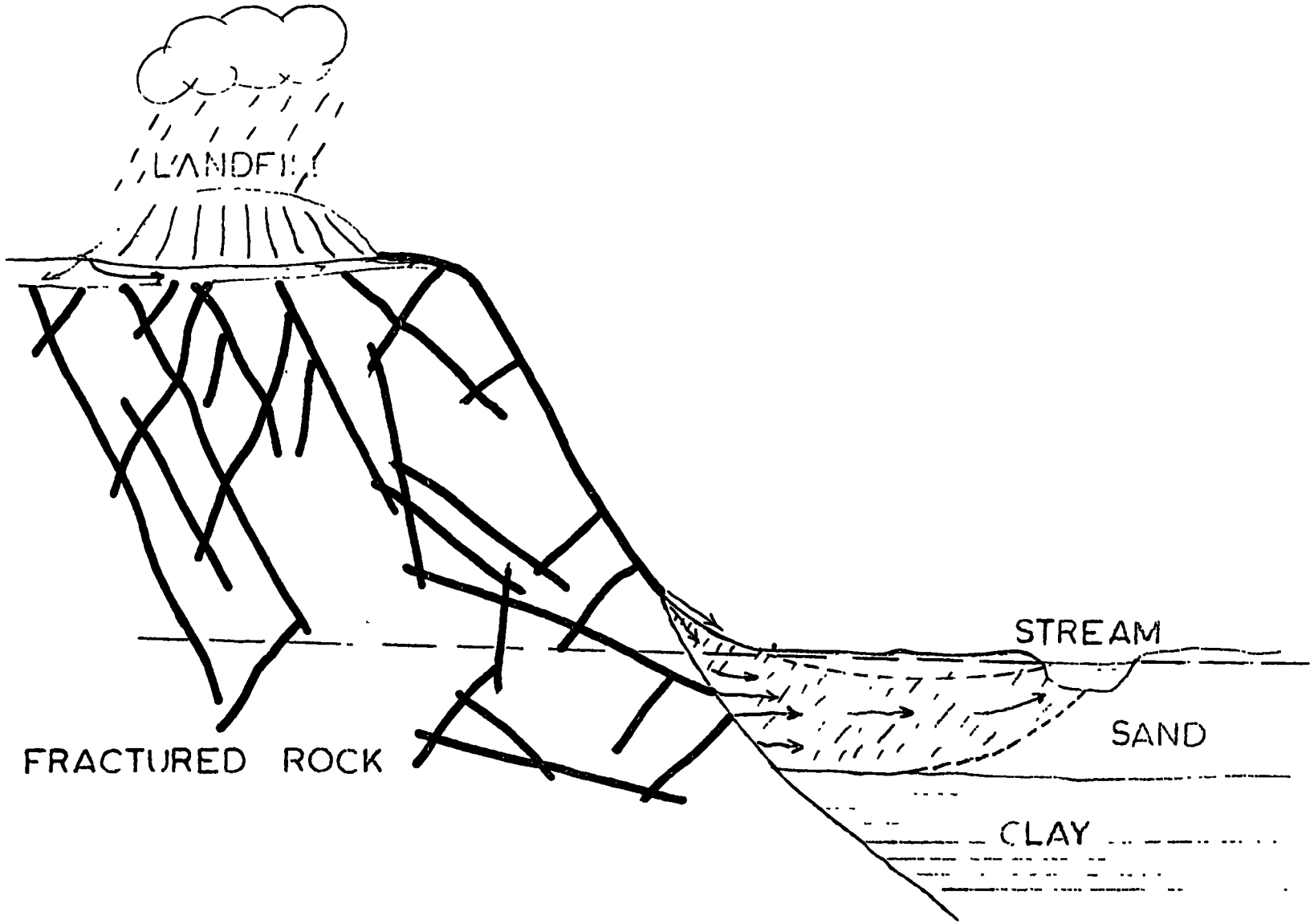


Figure 4 - The landfill rests on a fractured rock surface with a deep water table. Leachate flows into and through interconnecting fractures and discharges either at the surface as springs or into the subsurface where it moves with the ground water to some more distant discharge point.

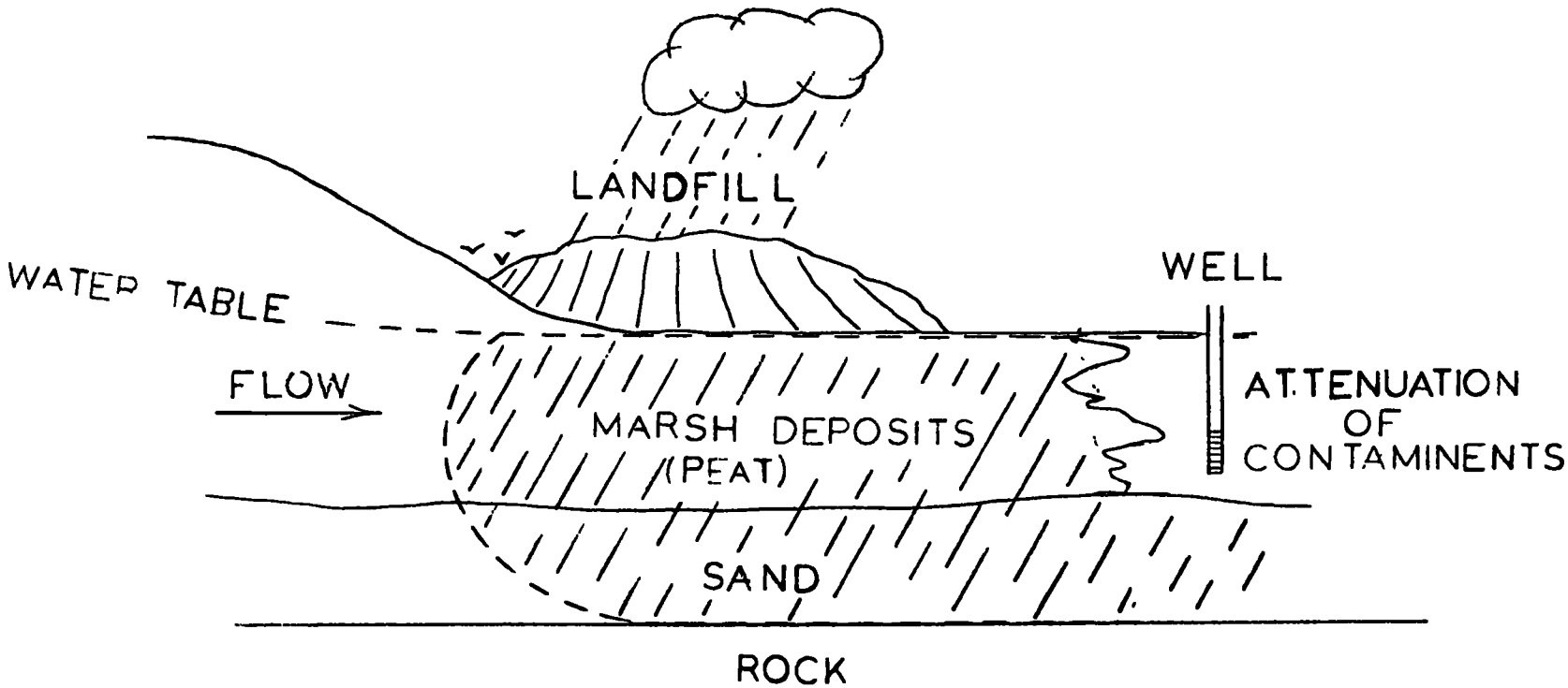


Figure 5 - The landfill rests on a layer of marsh deposits (organic materials) underlain by an aquifer. The water table is high and a mound is formed at the base of the landfill. Leachate migrates downward through the marsh material to the aquifer. Some contaminants may be attenuated within the marsh deposits. That portion reaching the water table moves through the aquifer with the ground water to some surface discharge point.

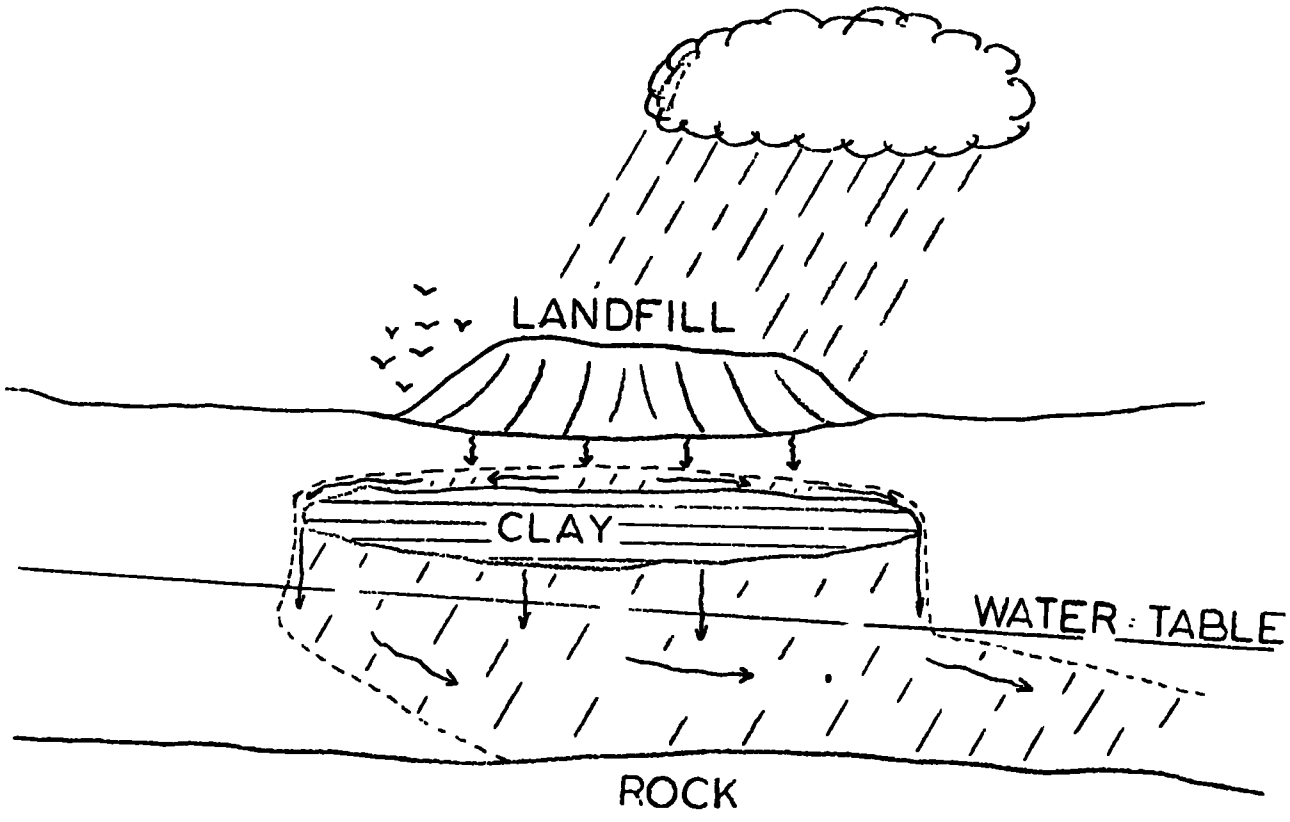


Figure 6 - The landfill rests on a layer of permeable sand interbedded with clay lenses and underlain by a clay layer. The water table is deep. Leachate percolates downward under the landfill, forming perched water tables and finally reaching the actual water table. A series of leachate plumes flow around clay lenses with the ground water.

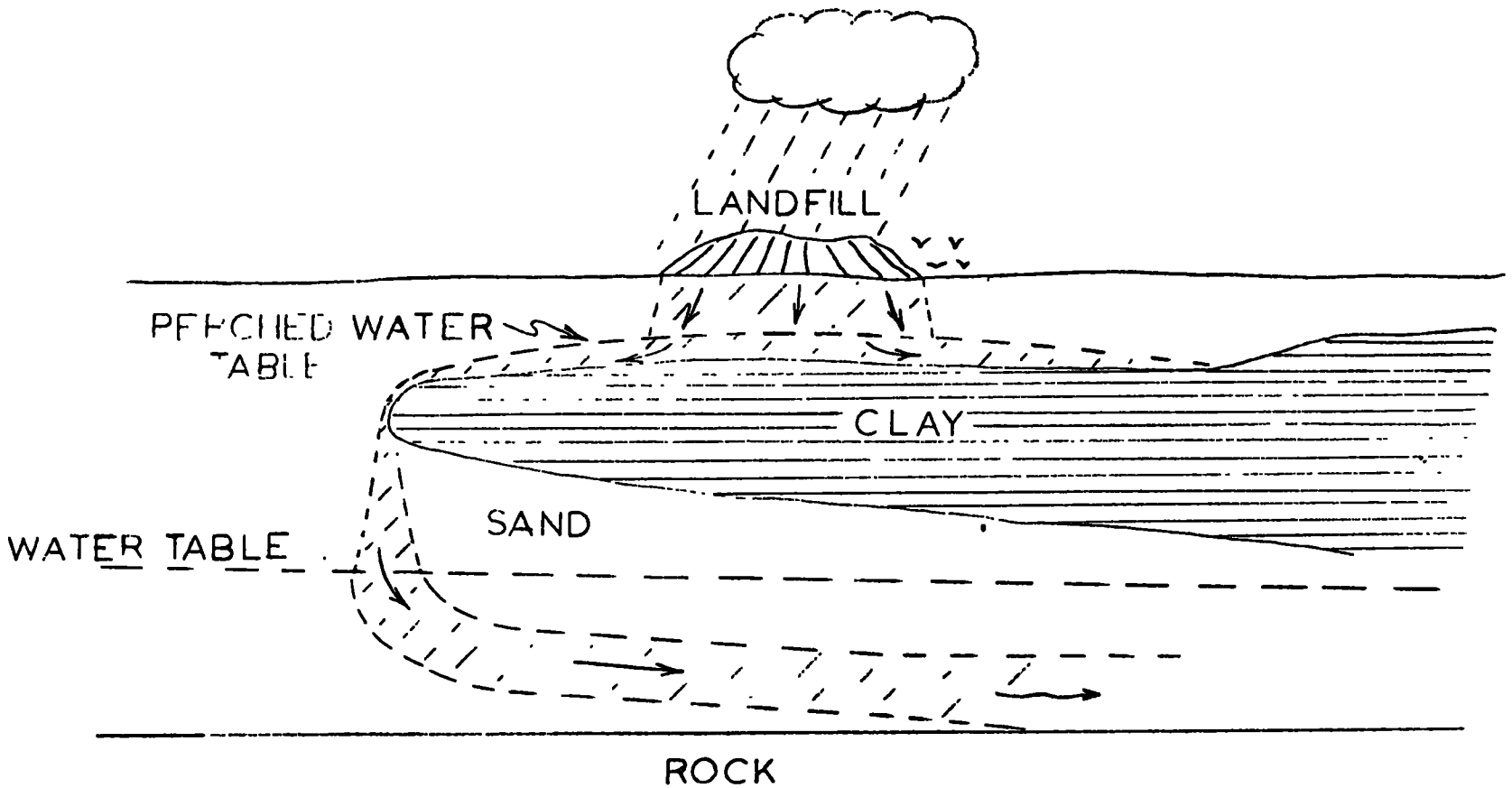


Figure 7. An extensive perched water table is formed under the landfill. Leachate percolates to the perched water table and flows downgradient to the end of the confining layer where it may again move downward to the actual water table.

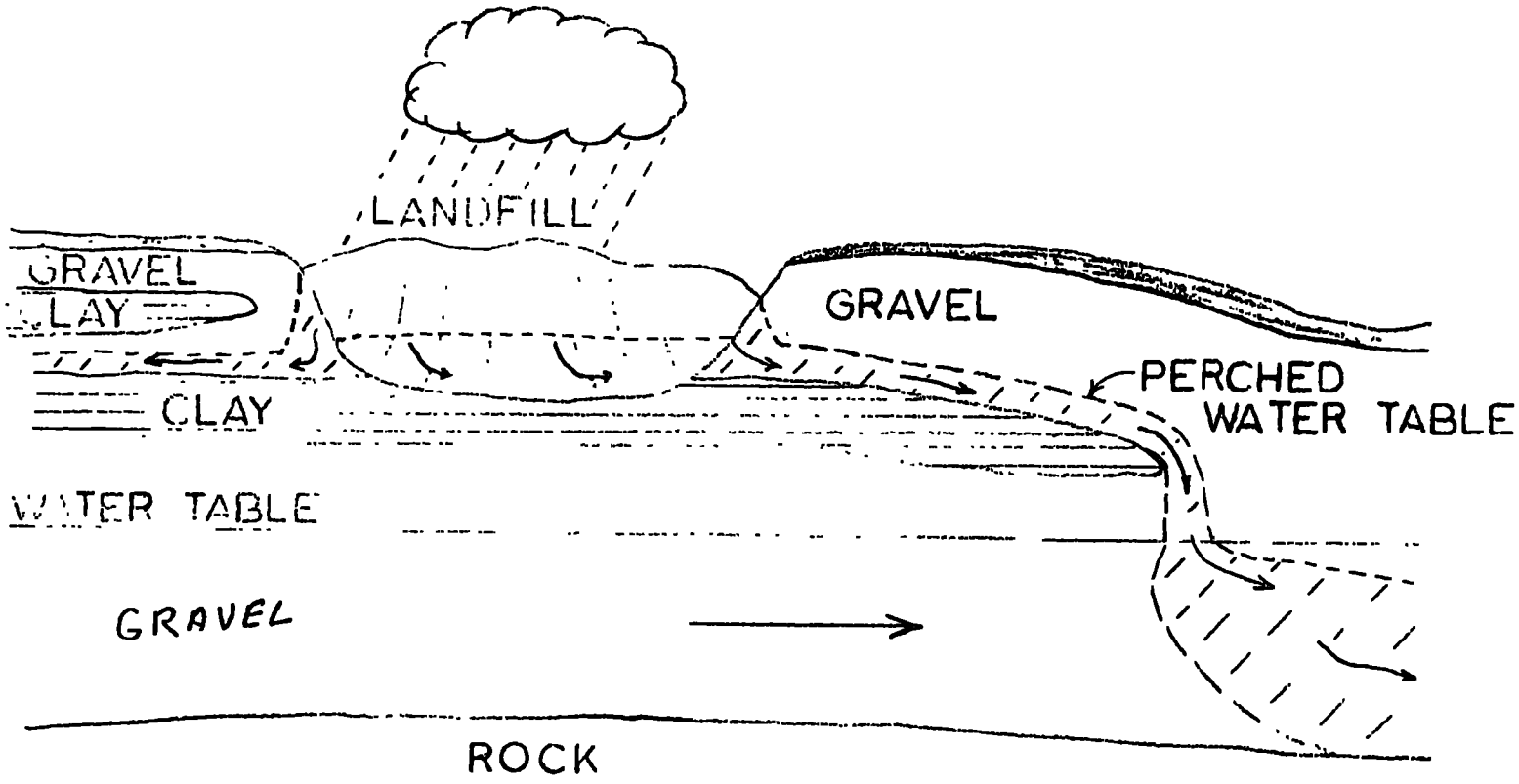


Figure 8 - The landfill occupies an abandoned gravel pit with a clay layer at its base. A perched water table (leachate) will mound up under the landfill and flow laterally through the ground above the clay until it is free to percolate to the actual water table.

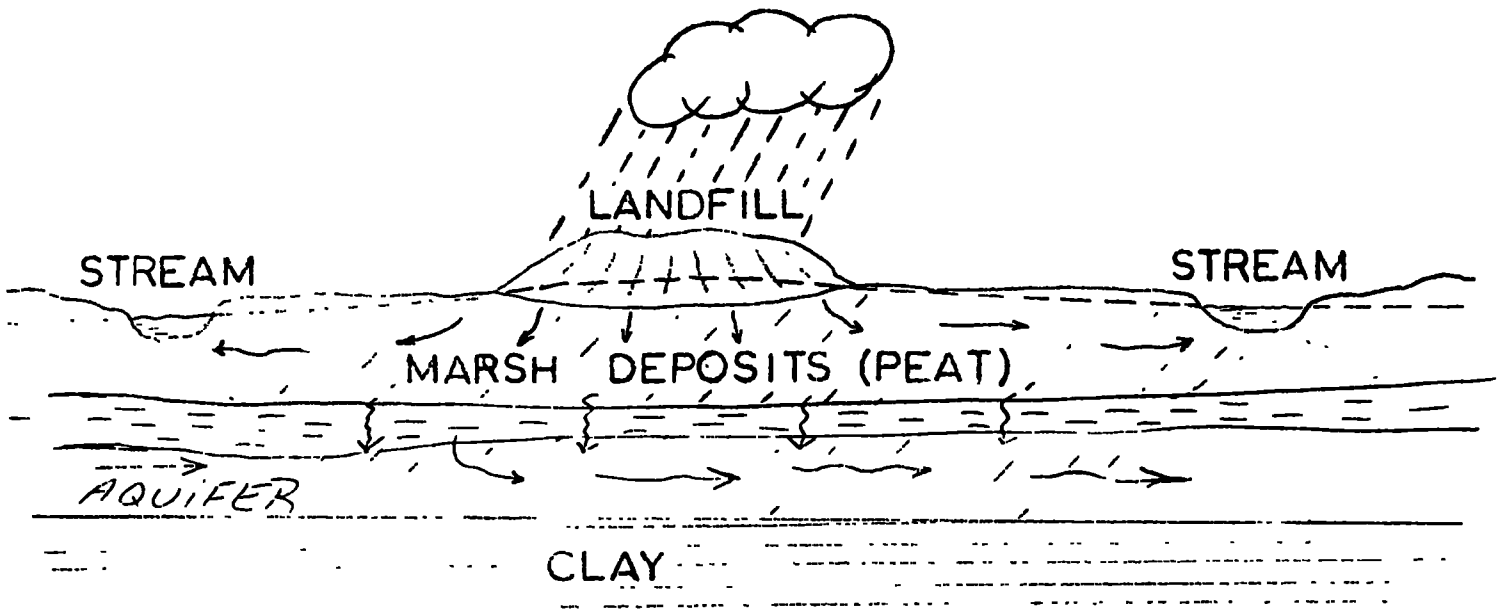


Figure 9. The landfill lies above organic marsh deposits bounded on either side by streams and underlain by a shallow aquifer. Leachate from the landfill may move horizontally through the marsh materials to the stream, or vertically downward with ground-water recharge to the aquifer.

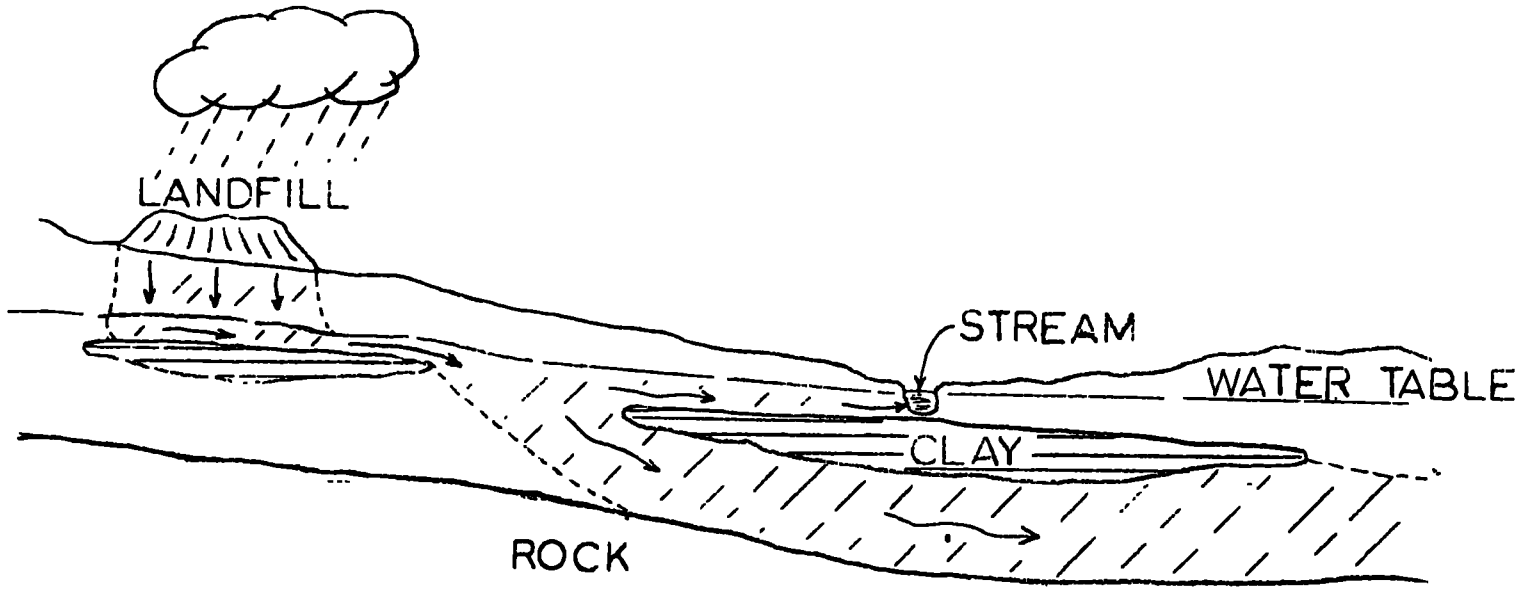


Figure 10. The landfill rests on a single aquifer interbedded with clay lensed. The leachate plume is split into two plumes by a clay lense. One plume discharges near the landfill while the other plume moves deeper into the aquifer and flows to a more distant discharge point.

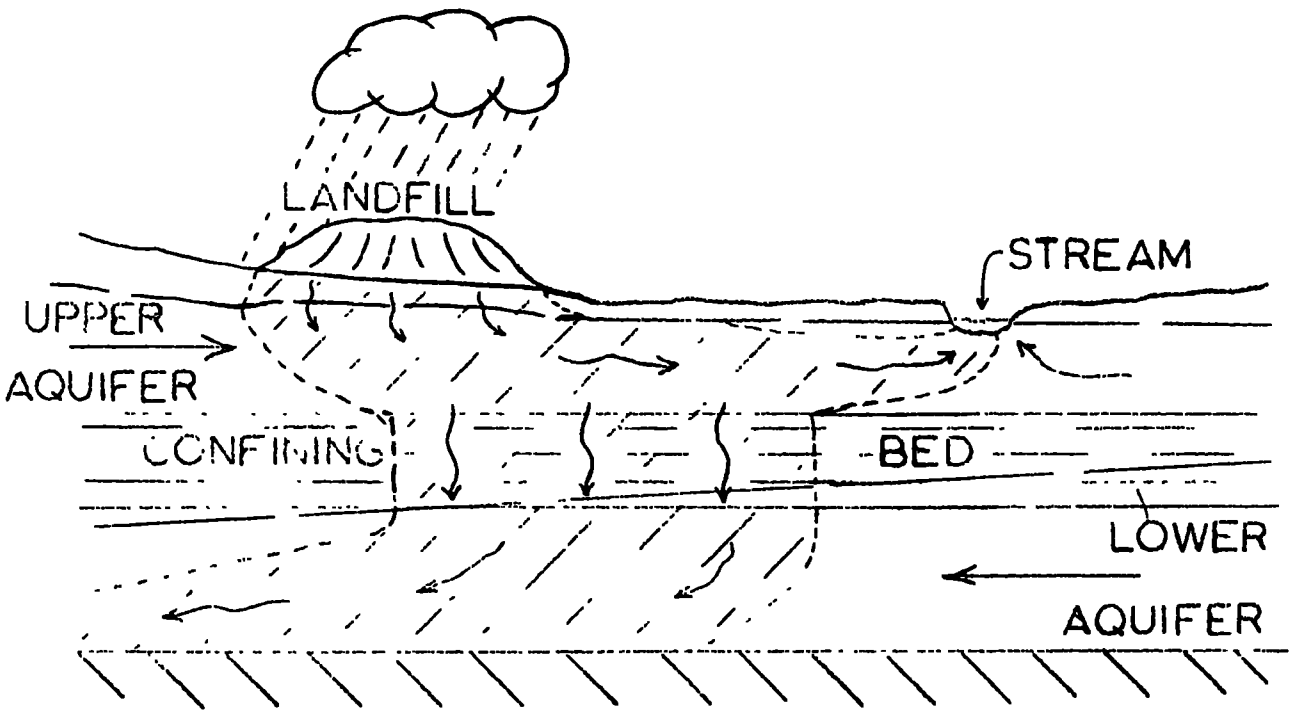


Figure 11. The landfill is situated over a two aquifer system with opposite flow directions. Leachate first moves into and flows with the ground water in the upper aquifer. Some of the leachate eventually moves through the confining layer into the lower aquifer where it flows back beneath the landfill and away in the other direction.

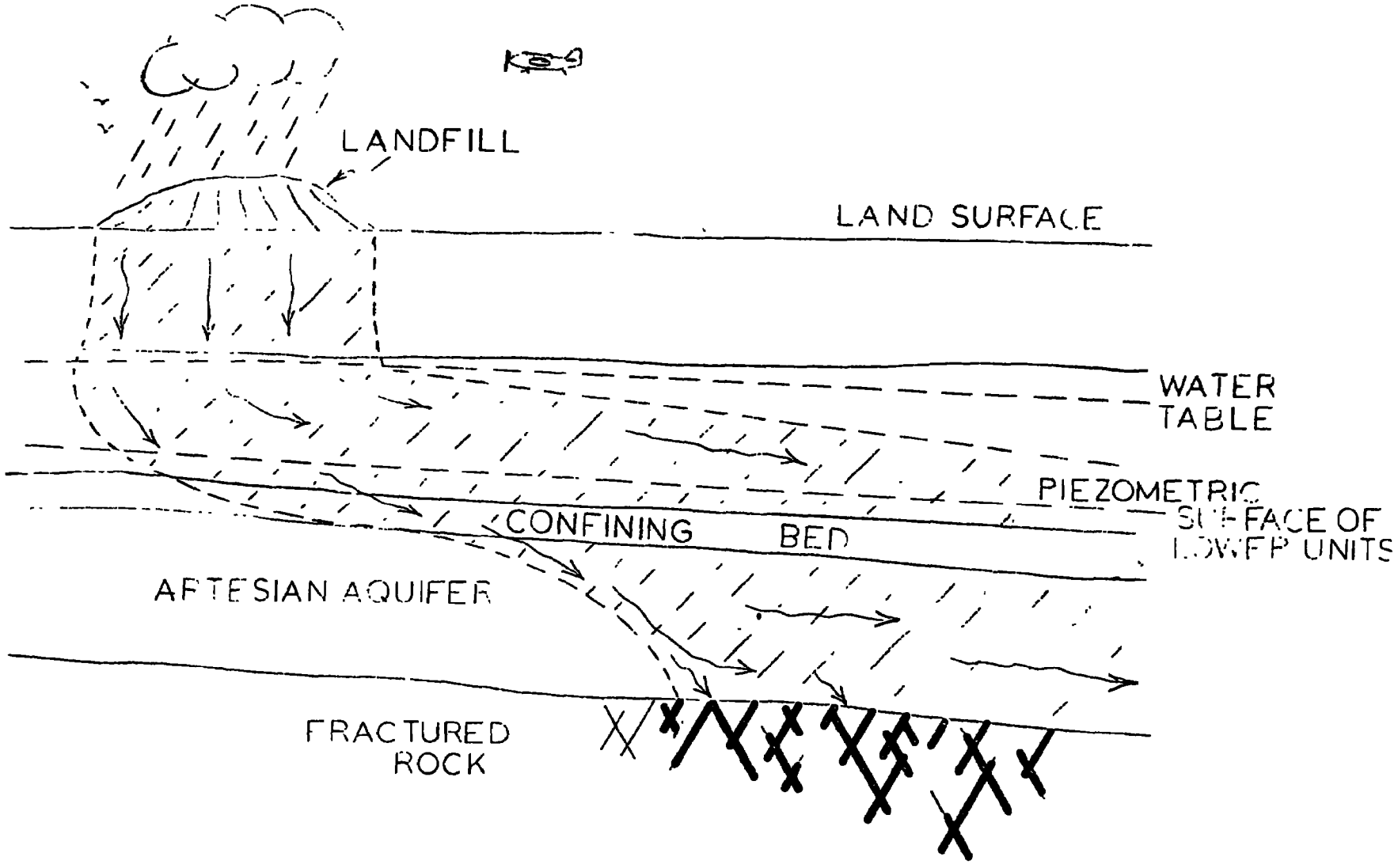


Figure 12 - The landfill is situated over a three aquifer system and a deep water table. Leachate percolates to the upper aquifer where it moves as a plume in the direction of ground water flow. Eventually some of the leachate moves through the confining layer and into the second aquifer that is an interconnected unconsolidated - creviced bedrock water-bearing unit.

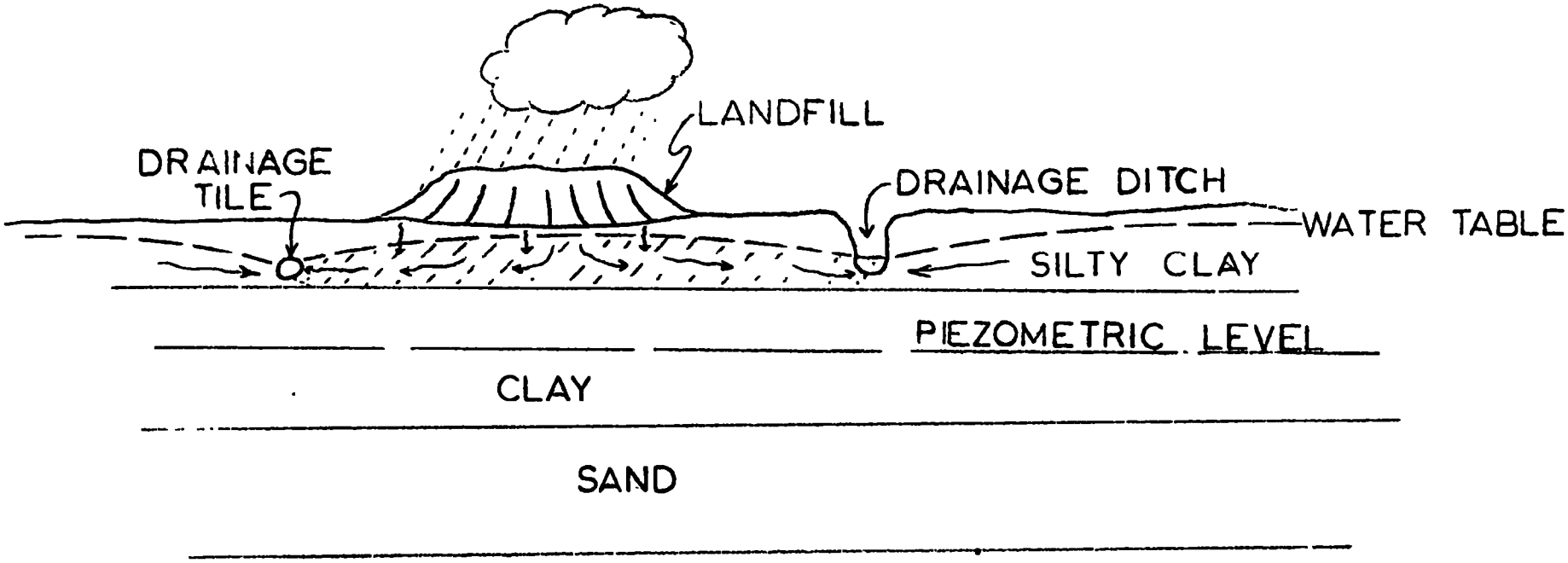


Figure 13. The landfill rests on a thick layer of clay underlain by an aquifer. Leachate is unable to penetrate the clay layer and discharges to the surface tile drainage systems or drainage ditches in the area around the landfill.

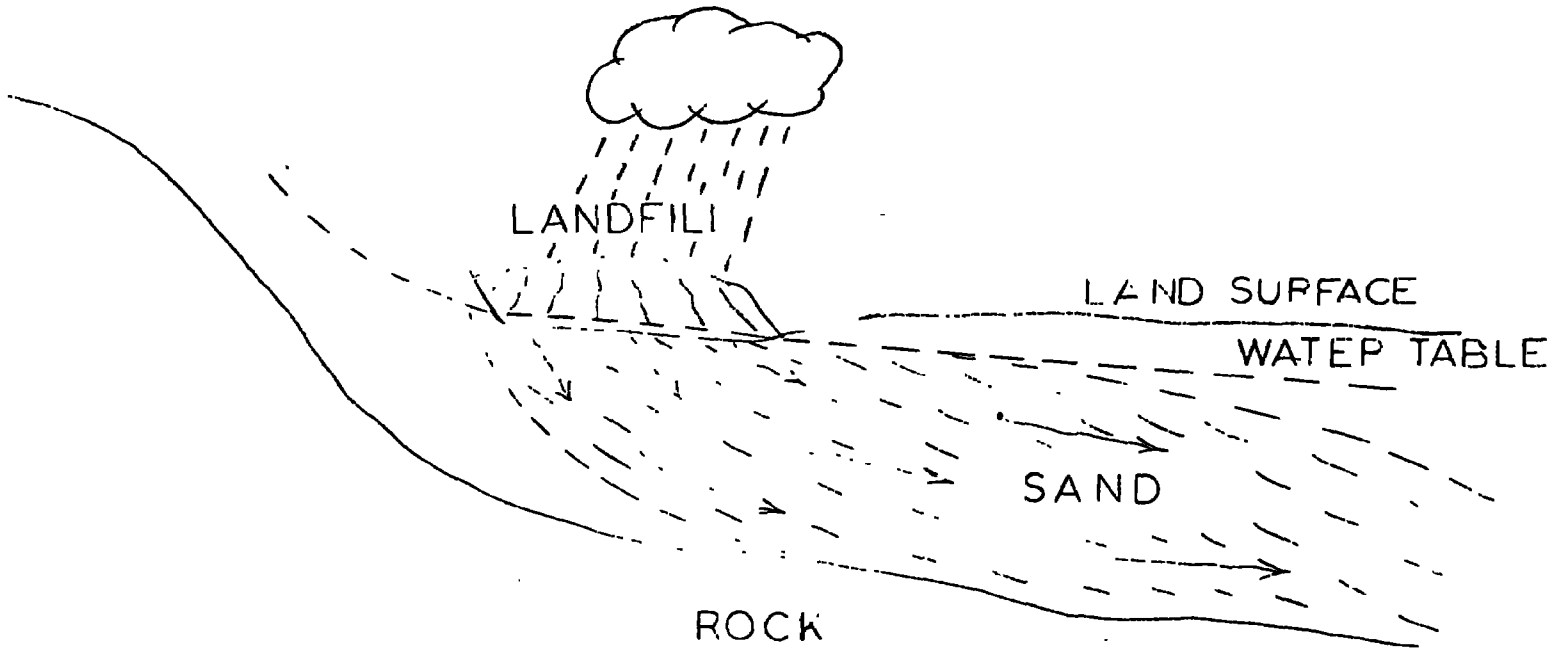


Figure 14. The landfill rests on a single aquifer with a steep, shallow water table which intersects a portion of the landfill. Ground water flows directly into the landfill forming leachate which then flows downward into the aquifer as a plume.

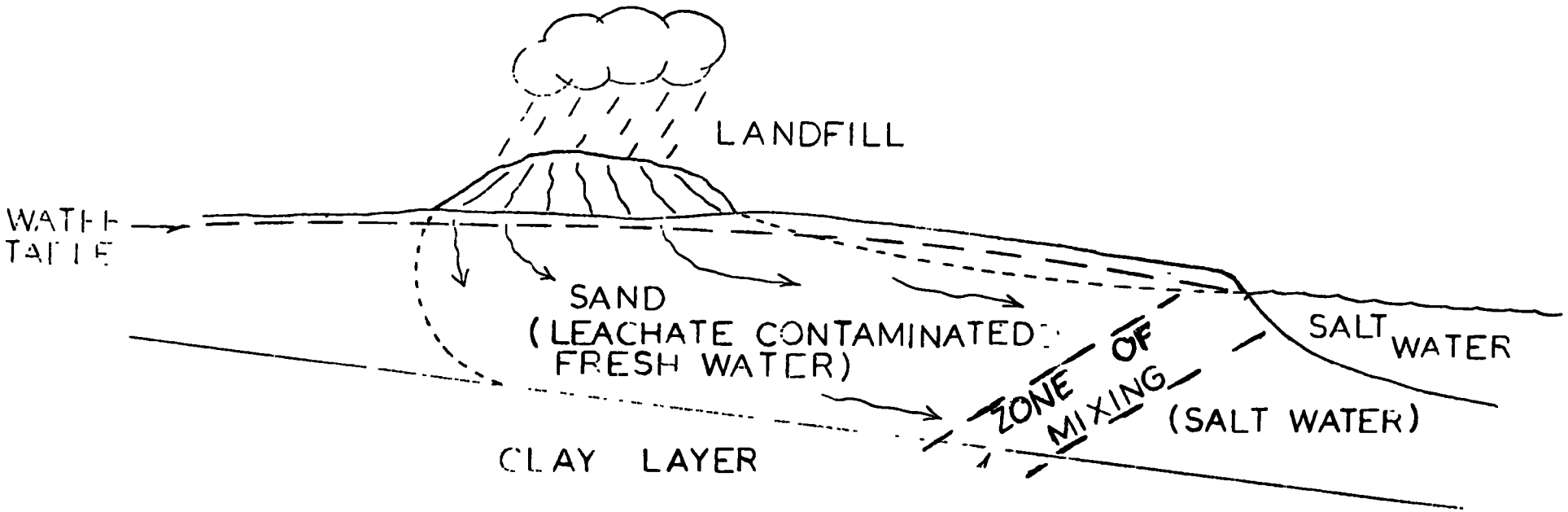


Figure 15 - The landfill is located near a large salt-water body. The leachate plume flows down into the fresh water aquifer and toward the open salt water body. As the leachate plume reaches the fresh-salt interface, it is forced upward along the interface to discharge at or near the edge of the salt-water body.

5 MONITORING AND SAMPLING TECHNIQUES

5.1 ZONE OF AERATION

The zone of aeration is defined as the materials between land surface and the water table. It is through these dry sediments that percolating waters must move on the way to recharging, or contaminating the ground water. In most cases involving landfill contamination, unless 1) scientific research is involved, 2) there are unusual geologic or hydrologic considerations, or 3) extremely toxic chemicals are suspected in the leachate sampling in the zone of aeration would not normally be carried out. Such sampling is difficult and some of the methods are expensive. However, when the decision has been made to monitor water quality in the zone of aeration, the depth to water becomes important. When surface active materials such as clay and silt are present, attenuation will take place. Consequently, the chemical quality of leachate just below the landfill may be many times worse than that sampled at the water table. (See chapter 3.5 for a detailed discussion of attenuation)

5.1.1 Soil Analysis

Soil analysis can be valuable as a monitoring tool for tracing leachate constituents, particularly those prone to cation exchange or other adsorption reactions. Collecting soil cores beneath the landfill can be done as part of a well installation process. Techniques for core collecting are available (Hyorslev, 1965), and methods for soil analysis are also documented (Black, 1965a, b; Hanna, 1964; Soil Science Society of America, 1971).

Soil analysis has had only limited use in leachate monitoring programs for several practical reasons. Probably foremost is the lack of commercial soil testing laboratories which can handle

soil tests outside the scope of agricultural application. For example, testing laboratories are established in each state for soil fertility analysis (nitrogen, phosphorus, potassium, pH), but heavy metals, organic matter, and exchangeable cations are not accomodated. In many places only noncommercial samples are accepted by these labs.

Another restriction in soil analysis is inherent in the methodology. What is actually analyzed is seldom the total soil, but instead, a chemical extract of it. Fundamentally, a separation of the inorganic/organic matrix and chemical species in soil solution or "available" to soil solution must be made. An analysis of the complete soil including the inorganic matrix would be meaningless. To measure the chemical species in solution, exchangeable to solution, available to plants, or accumulated by adsorption or precipitation on the inorganic matrix is the objective of the soil analyst. To meet this objective, soils must be treated with reagents of differing chemical reactivity under a variety of physical conditions. The resulting solutions are then analyzed for the chemical species of interest. Interpretation of results is a function of soil characteristics and analytical methodology. Although methods have been standardized to a degree, the analyst must be able to adapt and interpret according to the dictates of the soil sample. In contrast, water samples are usually analyzed directly or with a minimum of pretreatment. This is not to state that there aren't analytical problems

with water samples, but it takes an additional analytical step to bring a soil sample to the same state that a water sample is in when collected.

Soil samples yield information which cannot be obtained from water samples. Therefore, soil sampling has a place in the leachate monitoring program, and its incorporation should be expanded. Chemical species associated with soil solution as well as those on exchange sites can be traced downward in a soil profile or in the unsaturated zone. Locations of accumulation or leaching can be identified.

Sulfate (SO_4), chloride (Cl), and nitrate (NO_3) are soluble and unaffected by cation exchange reactions in soil. This results in mobility impeded only by the restrictions of water percolation. These anions can be analyzed for in soil samples in addition to cations which are generally more strongly associated with the solid soil matrix. The latter present more difficult analytical problems because they must be released from the soil matrix prior to determination. However, locations of zones of heavy metal or phosphorus accumulations can only be detected through soil analysis.

In addition to the chemical information obtained from soil core sampling, mineralogical information can be gained by as simple a means as visual cont'

observation. Organic matter layers, clays, or silts may be encountered. Knowledge of their locations will aid in interpreting flow patterns and chemical configuration in the plume. If a more sophisticated analysis is desired, a mechanical analysis can be made relatively easily. It is simply a size fractionation of the soil into its respective proportions of sand, silt, and clay. Even more elaborate x-ray crystallographic analysis of clays will identify the clay type. This latter degree of sophistication is beyond the scope required for anything but a research program on leachate production and movement.

Advantages and disadvantages of including soil analysis in a monitoring program are summarized below.

Advantages	Disadvantages
1) Ease of soil sample collection	1) Commercial laboratories capable of non-agricultural soil analyses are scarce
2) Inexpensive sample collection	2) Not a proven standardized sampling method for monitoring programs
3) Accurate vertical and areal sampling locations	3) Cost of analysis likely to be higher per sample than water because of two-step analytical procedure
4) Best method to measure leachate attenuation through adsorption or precipitation mechanisms	4) Applicable mainly in the zone of aeration
5) Long interval between sampling possible because of intermittent leachate production	5) Requires special equipment for each sample collection
6) In situ conditions of sample can be maintained with proper handling	6) Analytical methods not adaptable to high-rate standard procedures as available for water
7) Physical and chemical conditions throughout unsaturated zone can be observed	7) Results help interpretation of water-quality data, but do not replace water-quality data
8) Only part of total sample needs to be consumed in analysis	
9) Samples can be stored for later comparison or further analysis	

Advantages	Disadvantages
10) More representative biological sampling possible than with water	8) Wetting and drying cycles, and changes in redox potental can change chemical reactivity of some soil constituents after collection
	9) State-of-the-art, not documanted in leachate studies

Decisions regarding adoption of soil sampling in a monitoring program will be made on the basis of the following criteria.

1) Relative cost of soil analysis and water analysis from the zone of unsaturation.

2) Availability of analytical facilities.

3) Availability of analytical techniques for the parameters of interest.

4) Applicability of information derived to the monitoring program.

5) Compliance with governmental regulations governing monitoring programs.

Hanna, W. J. 1964. Methods for chemical analysis of soils. Pages 474-502 in Firman E. Bear ed. Chemistry of the soil. American Chemical Society Monograph Series No. 160. Reinhold Publishing Co., New York.

Soil Science Society of America. 1971. Instrumental methods for analysis of soils and plant tissue. Madison, Wisconsin. 222 pp.

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Black, C. A. ed. 1965b. Methods of soil analysis, part 2. Chemical and microbiological properties. Soil Science Society of America, Madison, Wisconsin. 802 pp.

Hvorslev, M.S. 1969 Subsurface Exploration and Sampling of Soils for Civil Engineering Purposes. Engineering Foundation. United Engineering Center, New York, N.Y. —— pp.

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2. Black, C. A. ed. 1965b. Methods of soil analysis, part 2. Chemical and microbiological properties. Soil Science Society of America, Madison, Wisconsin. 802 pp.

3. Hanna, W. J. 1964. Methods for chemical analysis of soils. Pages 474-502 in Firman E. Bear ed. Chemistry of the soil. American Chemical Society Mongraph Series No. 160. Reinhold Publishing Co., New York.

4. Hvorslev, M. S. 1965. Subsurface exploration and sampling of soils for civil engineering purposes. Engineering Foundation, United Engineering Center, New York.

5. Soil Science Society of America. 1971. Instrumental methods for analysis of soils and plant tissue. Madison, Wisconsin. 222 pp.

5.1.2. Pressure Vacuum Lysimeters

Methodology

Suction lysimeters have been used by a variety of investigators, including engineers, soil scientists, and hydrogeologists, to obtain samples of in-situ soil moisture. They are used predominantly in the zone of aeration, but can easily be used to sample ground water. This device, in its most improved form, consists of a porous ceramic cup capable of holding a vacuum, a small-diameter, sample accumulation chamber of PVC-pipe and two sampling tubes leading to the surface. Once the lysimeter is emplaced, a vacuum is applied to the cup. Soil moisture moves into the sampler under this gradient, and a water sample gradually accumulates. Then, the vacuum is released and pressure is applied forcing the accumulated water to the surface through the sampling tube. Construction, installation, and sampling procedures are described by Grover and Lamborn, 1970; Parizek and Lane, 1970; Wagner, 1962; Wengel and Griffen, 1971; and Wood, 1973.

The technology of lysimeter utilization is well established. They have been used to trace pollution from septic tanks (Manbeck, 1975), and cesspools (Nassau-Suffolk Research Task Group, 1969), synthetic detergents (Department of Water Resources, The Resources Agency of California, 1963), and colliery spoil heaps (James, 1974). Apgar and Langmuir (1971) used suction lysimeters, wells, and soil samples to study the movement and chemical characteristics of leachate from a landfill in central

Pennsylvania (Figure 5-1). There the water table is more than 200 feet below ground surface, and monitoring the unsaturated zone is of great importance. To do this, the landfill excavation was graded and lined so that leachate would drain into a percolation trench along one side. The lysimeters were installed underneath this trench. As many as four lysimeters were emplaced at selected depths in a single borehole to a maximum depth of 54.5 ft, each installation separated from the next by a pelletized bentonite seal. Water samples collected from the lysimeter network were analyzed for Eh, pH, temperature, specific conductance, BOD, Cl, SO_4, total alkalinity, NH_3, NO_2, NO_3, PO_4, Ca, Mg, Na, K, and Total Fe. Apgar and Langmuir were able to define differences in leachate concentration from upslope and downslope cells as well as leachate attenuation and rate of movement. Wood (1973) suggested a modification of the lysimeters used by Apgar and Langmuir so that water samples could be recovered from any depth (Figure 5-2). With this modification, deep pressure-vacuum lysimeters appear to be the best method of monitoring the zone of aeration because a check valve prevents pressurization of the porous cups. Pressure exceeding about one atmosphere in the sample chamber would drive accumulated water back through the cup rather than to the surface in deeply placed lysimeters.

Figure 5.1 Typical Vacuum Lysimeter Emplacement Around a Sanitary Landfill

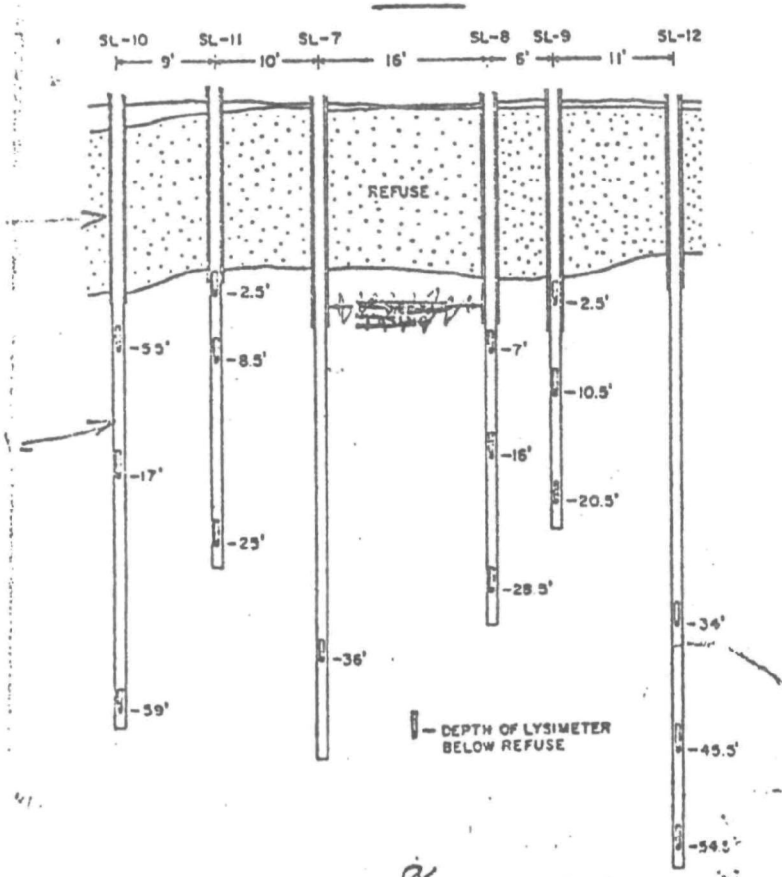


Fig. 6. Cross section of a cell lysimeter network.
DATA FROM ROBERTS AND LUNDQUIST, 1971

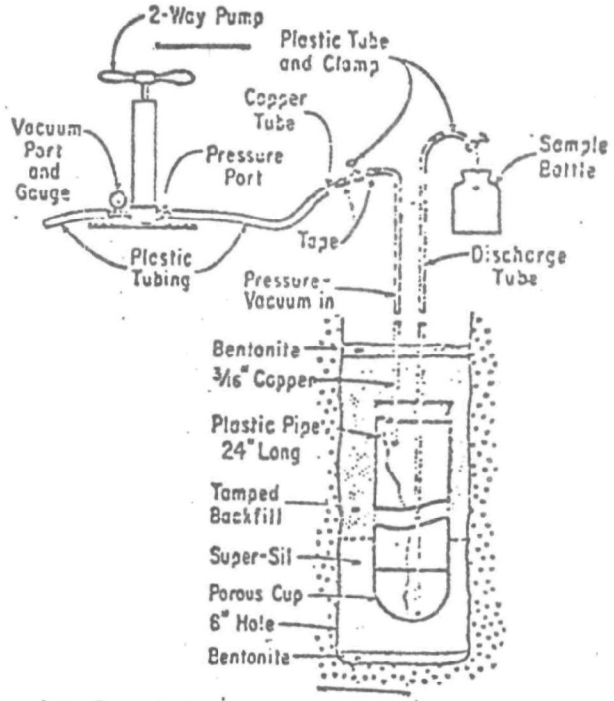
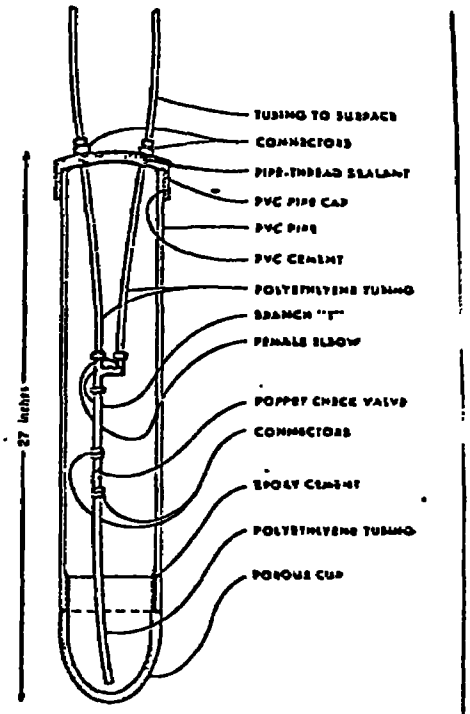


Fig. 8. Cross-section of a typical pressure-vacuum lysimeter installation (compare fig. 7). With the pressure-vacuum line attached to the pump, the apparatus is ready to collect a water sample. Not to scale.
DATA FROM PONIZEK AND LANE, 1970

Installed Lysimeter + Sampling Apparatus



~~Fig. 1. Details of the porous cup assembly.~~

FIGURE 1 Deep Porous - Vacuum Lysimeter

After Wood (1973)

Implementation

Parizek and Lane (1970) have described in detail pressure-vacuum lysimeter installation and sampling procedures. The following is excerpted from their report:

"A typical pressure-vacuum lysimeter installation is shown in Figure 5-1. Placement holes are first drilled to the desired depth. They may be 4 to 6 inches in diameter depending upon the number of lysimeters to be placed in each hole. A plug of wet bentonite clay is placed in the bottom of the hole to isolate the lysimeter from the undisturbed soil below it. This plug is optional. A layer of "Super Sil" at least six inches deep, is placed on top of the bentonite. "Super-Sil" is the trade name for a commercially available, crushed, pure silica-sand of almost talcum powder consistency. This is used to provide a clean transmission medium for soil moisture moving under capillary pressure, to insure hydraulic contact of the adjacent soil medium with the porous tip, to fill uneven voids created during drilling, as well as to discourage clogging of the ceramic tip by colloids, organic matter, or soil particles. The lysimeter is placed in the hole to the desired depth, and "Super-Sil" is placed around it until the lysimeter is about half-buried. Native soil, free of pebbles and rocks, is backfilled and tamped with long metal rods. After the lysimeter is covered with about six inches of soil, a second plug of bentonite is deposited to further isolate the lysimeter and to guard against possible channeling of water down the drill

hole. Backfilling is continued with native soil to the depth where it is desired to set the next lysimeter, at which point the above procedure is repeated.

It was found that three lysimeters were the most that could be conveniently placed in any one six-inch diameter hole. If more than three were installed this led to difficulties in proper depth placement, prevented proper tamping of backfill material, added to the danger of crimping or tangling the copper tubing and to the risk of channeling soil water down the incompletely filled hole. Care was taken to accurately measure the depth of placement of each lysimeter. It was possible to set the lysimeters to within six inches of the desired depth even in 30-foot deep holes.

After the lysimeters are placed, a short section of flexible tygon plastic tubing is secured over the end of each copper access tube with PVC electrical tape to allow thumb-screw pinch clamps to be used to seal the lysimeter between sampling periods, thereby maintaining the vacuum within the lysimeter.

The pump used in conjunction with these pressure-vacuum lysimeters is a two-way hand pump that can either deliver a back pressure or pull a vacuum. This pump can be purchased from any laboratory equipment supply house. The pump is similar to a tire pump. It has a base on which the operator

may stand while working the pump. A small vacuum gauge may be installed on the vacuum part of the pump by means of a tee-union. This enables the operator to consistently apply a desired vacuum to all lysimeters (about 18 inches of mercury). A length of tygon tubing is secured to each of the pump's pressure and vacuum parts to allow the pump to be coupled to the access tubes of the lysimeters. The free ends of the pump's tubing are slipped over a short length of copper tubing that is secured to the pressure-vacuum tube of the lysimeter and is held securely by a small spring-loaded clamp. A typical pressure-vacuum lysimeter sampling sequence is as follows:

1. The lysimeter's discharge tube is clamped shut and the vacuum side of the two-way pump is attached to the "in" tube.
2. A vacuum of approximately 18 inches of mercury is drawn and the "in" tubing is clamped shut.
3. To recover soil water samples, the pinch clamps are removed and the pressure side of the two-way pump is attached to the lysimeter's "in" tube. A few strokes of the hand pump generates enough pressure to force the water out of the lysimeter and into a collection bottle placed under the discharge tube.
4. After emptying the lysimeter the discharge tubing is clamped, the vacuum side of the pump is attached to the "in" tube and the lysimeter is evacuated again to gather another sample."

Advantages and disadvantages of the pressure-vacuum lysimeter are given below:

Advantages	Disadvantages
1. Inexpensive sampling device of great reliability.	1. Moderately complicated sampling procedure and equipment.
2. Inexpensive installation	2. Sampling device failure is irrepairable
3. Standard water analysis can be made	3. Small volume of sample
5. Samples can be collected at a central point	4. Surface tubing subject to tampering unless adequately protected
	5. Use at depth greater than 108 ft, not documented
	6. Sample contamination by porous cup if material is not properly prepared
	7. Possible plugging of cup by colloidal materials, and cup might exclude large molecules

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5.1.3 Trench Lysimeters

Methodology

Several investigators have used trench lysimeters to sample gravity water from irrigation or rainfall in the near-surface zone of aeration. In normal practice, a wood-reinforced trench or concrete-ring caisson is installed to a depth of 10 to 30 feet below land surface. Pans (Parizek and Lane, 1970), troughs (Olin Braids, personal communication, August 1975), or, open end pipes (Nassau-Suffolk Research Task Group, 1969) are forced out of the trench (caisson), through access ports, into the subsoil. These collecting devices intercept percolating gravity water and conduct it to sample bottles inside the trench. Only after irrigation or precipitation is there enough water infiltrating the subsoil to collect a sample. Figure 5-3 shows an installation used to collect cesspool effluent.

Due to the potential accumulation of hazardous gases generated by decomposition of landfilled material, the use of an open trench or caisson to sample leachate in or under a landfill can be risky. Artificial ventilation and gas monitoring devices are required to prevent injury to personnel collecting samples inside the trench.

Implementation

A description of a typical trench lysimeter is excerpted from Parizek and Lane (1970). "A 4-foot wide, 12-foot long trench

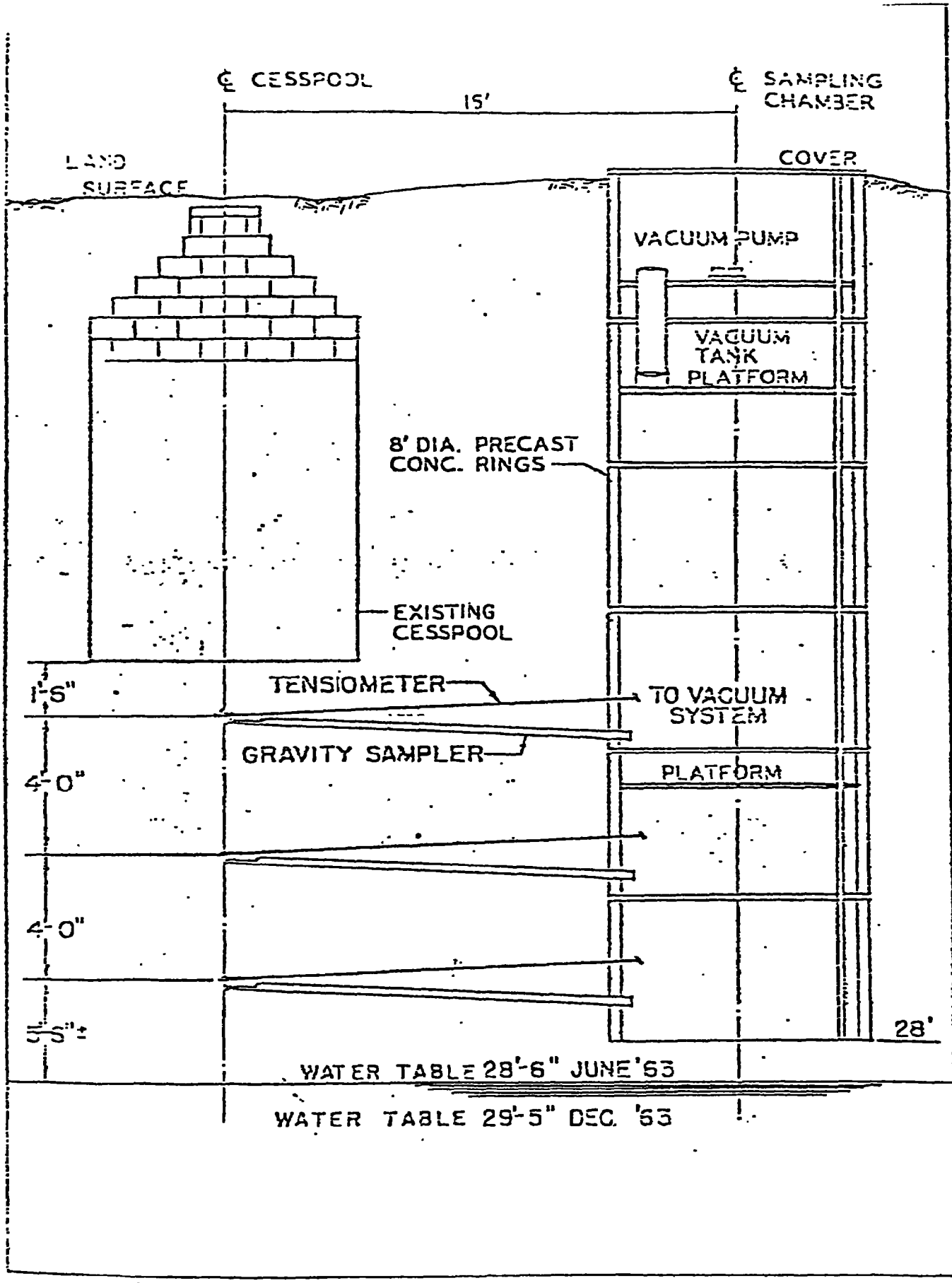
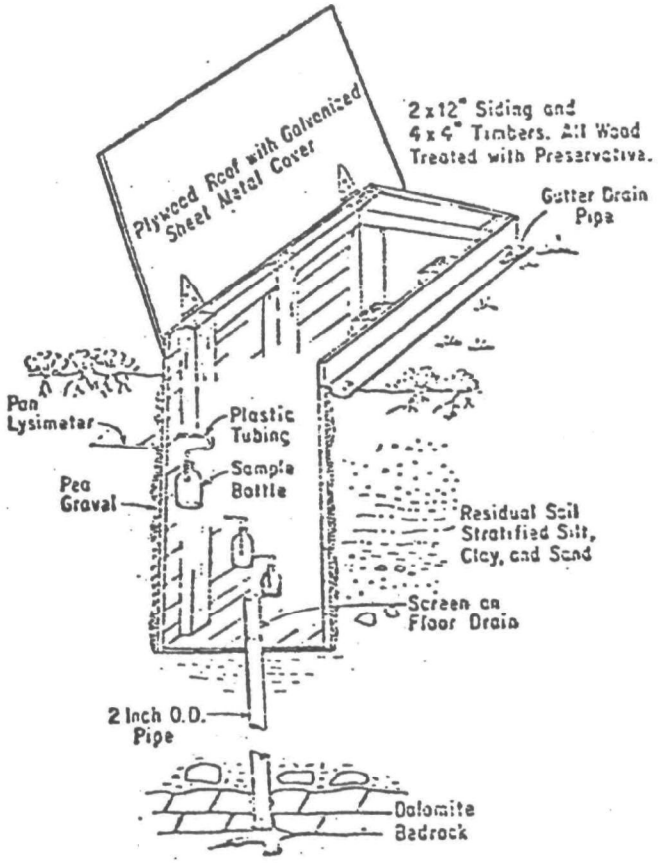


Figure 2 – Section through a cesspool and nearby sampling chamber showing the location of tensiometers and gravity samplers used to sample cesspool effluent in Long Island, New York (after Nassau-Suffolk Research Task Group, 1969).

Fig. 3

was excavated to a depth of 10 feet using a back hoe. The hole was braced with timbers and siding to allow safe access to the trench. The trench was then hand-dug to a 17 foot depth and braced. The entire seepage face was inclined 1 to 5 degrees from the vertical and sloped toward a hill down which soil water and interflow was expected. The residual soil contained resistant chert and quartzite cobbles and boulders and was reasonably well "cemented" with iron-oxide and clay. As a result, the pans could not be inserted into the soil profile without first providing an opening. A sheet metal blade 4 inches wide and 2 feet long was hammered into the overhanging bank with a sledge hammer to provide access for the pans. Pan lysimeters were tapped into these openings and allowed to slope gently toward the trench. Voids above and below the pans were back-filled with soil and were tamped into place. As siding was added to the trench walls, holes were cut to allow the copper tubings to project into the sampling pit. Spaces between the original trench faces and siding were filled with native soil and washed pea-gravel to allow water to flow freely toward the pit floor (Figure 5-4). After the walls and braces were emplaced, plastic tubing was connected to the copper tubing and inserted into plastic sampling bottles. The sampling pit was covered with a sloping roof and a half-round drain pipe was used to divert roof water away from the installation. A ladder was placed at one end of the house to allow access."

An alternative method of construction is to place concrete manhole rings in an open excavation or to sink them to depth using caisson construction techniques, depending on soil stability. This



~~Fig. 2. Schematic diagram of trench lysimeter.~~

Fig. 1 Schematic Diagram of a Trench Lysimeter

Parizek and Lane, 1970

type of construction is shown in Figure 3.

Trench Lysimeter Comparison Lysimeter

Advantages	Disadvantages
1.None	1. Dangerous because of possible flammable gas accumulation
	2. Water will flow to samples only after rainfall
	3. Considerable expense involved in constructing trench or caisson
	4. No documentation of application to landfills

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1. Nassau-Suffolk Research Task Group. 1969. Final report of the Long Island ground water pollution study. New York State Department of Health, Albany, New York.

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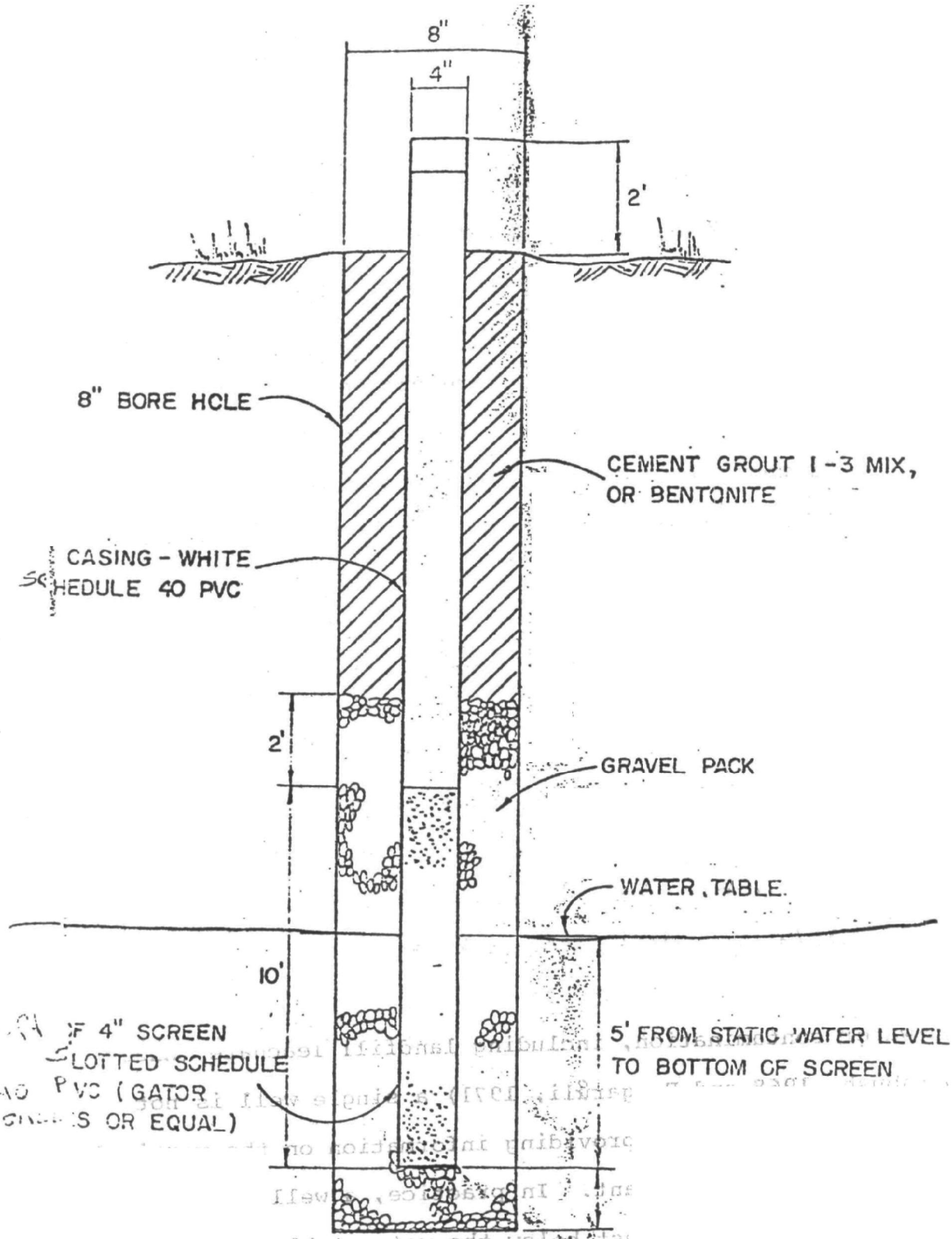
5.2 ZONE OF SATURATION

In the zone of saturation, leachate movement from a landfill will be controlled by a combination of ground-water flow patterns and soil-leachate interactions. Under shallow water-table conditions, only a small zone exists where unsaturated soil-leachate interactions can reduce leachate concentrations. Therefore, careful collection of representative ground water samples from properly constructed wells is necessary to trace leachate movement or determine its presence in the ground-water environment in which the landfill is located.

5.2.1 Well Screened or Open Over a Single Vertical Interval

Methodology

Wells screened over a single vertical section of an aquifer, are the most common construction used to obtain ground water samples from unconsolidated sediments or semiconsolidated rocks. Uncased wells (open hole) in consolidated rock can be used for the same purpose. Although this type of well is routinely used in monitoring ground-water contamination, including landfill leachate (Anderson and Dornbush, 1968 and Fungaroli, 1971) a single well is not particularly effective in providing information on the vertical distribution of a contaminant. In practice, a well is drilled to an arbitrary depth, usually just below the water table in landfill studies, and the screen is set so that it intersects the water table (Figure 5-5). The rationale for this type of construction is that if leachate reaches the ground water, it will be detected in water



OBSERVATION WELL

NOT TO SCALE

Figure 5-5

samples from this type of well. However, this construction is often used when leachate has already reached the ground water. The drawback of the construction is immediately apparent; only a portion of the aquifer is sampled and only the most recently infiltrated leachate can be collected. In most cases, leachate will be denser than water and sink into the ground water under partial control of a gravity gradient. This denser fluid body, "sinking" into fresher water cannot be sampled with a well that skims only the top of the water body. However, in the experience of the writer, great reliance has been placed on this type of well construction to trace the extent of leachate movement into an aquifer.

Even if the well is completed below the water table, it may not provide water samples representative of leachate concentration at that point. For example, the well casing may entirely seal off the contaminated acquifer or the screen may penetrate into another aquifer system (if little is known about site geology), thus providing misleading water samples. This drawback can be partly counteracted if the well is screened over the entire aquifer thickness; however, if the aquifer is thick and the contaminated plume is thin, the composite ground water sample that is obtained provides no information on the vertical distribution of leachate.

Taking everything into consideration, using a single-screen well appears to be justified under two situation: 1) obtaining composite ground water samples from wells in which the entire saturated thickness of the aquifer is screened and 2) areas where depth to water is great and the majority of the sampling program is aimed

at the zone of aeration and the top of the zone of saturation. The latter case is probably the best use of this type of well. The wells, completed in the upper zone of the water body, would serve as an early warning system if any leachate is able to percolate to the ground water. Once detected, other sampling techniques would be required to trace leachate extent and movement in the aquifer.

Methodology Implementation

This type of well can be drilled by a variety of techniques, including mud-rotary, reverse-rotary, air-rotary, jetting, augering, and drive points; ~~with~~ diameters rang~~ing~~e from 1 1/4 inches to greater than but rarely exceeding 36 inches. The drilling method chosen depends on: 1)nature of material to be penetrated, 2) diameter and depth of well desired, 3) site accessibility, 4) availability of drilling water, 5) budget constraints, 6) time constraints and a variety of other factors resulting from individual site conditions. Drilling methods per se are discussed in a later section, but a summary is presented in Table 1. With the possible exception of hand augering and drive points, a drilling contractor should be used to install this type of well unless the investigator has access to a power auger, soil boring, or jetting rig.

Wells Screened Over a Single Vertical Interval

To be able to compare costs of the various techniques described in Chapter 5.2., A hypothetical aquifer is required. For the purposes of illustration, a water-table aquifer composed of unconsolidated sand with a depth to water of 10 feet and total saturated thickness of 100 feet will be used. However, it should be realized that these cost estimates are based on prevailing rates in the northeast and consequently actual costs will be lower or higher, depending on conditions in the investigations ~~local~~ area. Also, drilling and materials prices have been climbing recently and the costs presented here (Fall 1975) will no longer be representative ~~of actual costs~~ in a very short period of time. In spite of this, these estimates will provide an idea of relative cost that should remain relatively unchanged by inflation, etc. Table 1 is a summary of these costs.

As mentioned in Chapter 4.2, the recommended type of construction for monitoring purposes is to screen the well over the entire saturated thickness of the aquifer, which in our example is 100 feet. The quickness and least expensive way to complete this type of installation would be to drill a 6-to 8-inch diameter borehole with a hydraulic rotary rig to the bottom of the aquifer; set 4-inch diameter

Table 1. Cost Estimates for Various Sampling Methods

Sampling Method	Price Per Installation Well Diameter 2-inch	4-inch	6-inch
Screened over a single interval (plastic screen and casing)			
1. Entire aquifer	$1,600 - $3,700	$2,300 - $4,500	$6,400 - $7,500
2. Top 10 feet of aquifer	650 - 1,050	700 - 1,150	-
3. Top 5 feet of aquifer with drive point	100 - 200	-	-
Piezometers (plastic screen and casing)			
1. Entire aquifer screened			
a. Cement grout	2,100 - 4,700	2,800 - 5,500	6,900 - 8,500
b. Bentonite seal	1,850 - 4,150	2,350 - 4,950	6,650 - 7,950
2. Top 10 feet of aquifer screened			
a. Cement grout	1,150 - 2,050	1,200 - 2,150	-
b. Bentonite seal	900 - 1,500	950 - 1,600	-
Well clusters			
1. Jet-percussion			
a. Five-well cluster, each well with a 20-foot long plastic screen	2,500 - 3,800	-	-
b. Five-well cluster, each well with only a 5-foot long plastic screen	1,700 - 2,300	-	-
2. Augers			
a. Five-well cluster, each well with a 20-foot long stainless-steel wire-wrapped screen	4,600 - 5,300	-	-
b. Five-well cluster, each well with only a 5-foot long gauze wrapped drive points	1,800 - 2,600	-	-

Table 1 (continued). Cost Estimates for Various Sampling Methods

Sampling Method	Price Per Installation Well Diameter 2-inch	4-inch	6-inch
3. Cable tool			
a. Five-well cluster, each well with a 20-foot long stainless-steel, wire wrapped screen	-	-	$ 9,850 - $14,150
4. Hydraulic rotary			
a. Five-well cluster, each well with a 20-foot long plastic screen, casing grouted in place	-	$9,050 - $14,900	13,800 - 19,400
b. Five-well cluster, completed in a single large diameter borehole, 15-foot long plastic screens, 5-foot seal between screens	$4,240 - $5,880	8,250 - 11,000	-
Single well/multiple sampling point			
a. 110-foot deep well with one-foot long screens separated by 4 feet of casing starting at 10 feet below ground surface	-	-	3,000 - 4,700
Sampling during drilling	-	3,000 - 4,700	3,300 - 5,200

plotted PVC well screen and PVC casing; backfill with a gravel pack or formation stabilizer; and place a concrete collar around the well casing at ground surface to prevent downward leakage of rainwater or other fluids. Total cost of this installation is in the range of $2,300 to $4,500 for drilling, materials, installation and development. Screen is second only to drilling in terms of cost, running from $1,000 to $1,500 for 100 feet of 4-inch slotted PVC. Cosntruction cost could be reduced to a total of $1600 to $3700 if 2-inch casing and screen are used, but sampling can be more difficult in a well of this diameter. On the other hand, using 6-inch casing and screen facilitates development and water sampling but elevates the cost to the range of $6400 to $7500 per well. In wells of this size, wire-wound metal well screens are more commonly used than PVC, resulting in a substancial cost increase per installation as compared to the 4-inch well.

If the investigator is interested in sampling only the top of the aquifer with a well constructed so that a 10-foot long 4-inch diameter screen intersected the water table, price per installation would range from $700 to $1200. This is a substantial reduction in expenditure required to monitor the ground water, but as discussed above, it is

not a completely reliable technique for assessing ground water contamination by leachate. Even greater reductions in expenditure per installation can be obtained by installing a 2-inch diameter, 5-foot long drive point by hand, total cost of which would be less than $200 per well including labor, materials, and development with a pitcher pump.

Table 1 - Summary Table of Drilling Methods

	Depth			Diameter			Complexity of Operation		
	Shallow 0 -200ft.	Moderate 200-1000ft	Deep 1,000ft	Small 1-4in.	Moderate 4-12in.	Large 12 in.	Low	Moderate	High
Rotary									
Mud	X	X	X	X	X	X		X to	X
Air	X	X			X			X to	X
Reverse	X	X				X		X to	X
Cable Tool	X	X		X	X	X	X		
Jetting	X			X			X		
Augering									
Flight	X			X			X		
Bucket	X					X	X		
Drive Point	X			X			X		

Advantages

1. Inexpensive
2. Small diameter, shallow wells quick and easy to install
3. Can provide composite ground water samples if screen covers saturated thickness of aquifer
4. Can be drilled by a variety of methods

Disadvantages

1. No information on vertical distribution of contaminant
2. Improper completion depth can give incorrect picture of leachate distribution
3. Construction method can contribute to vertical movement of contaminant

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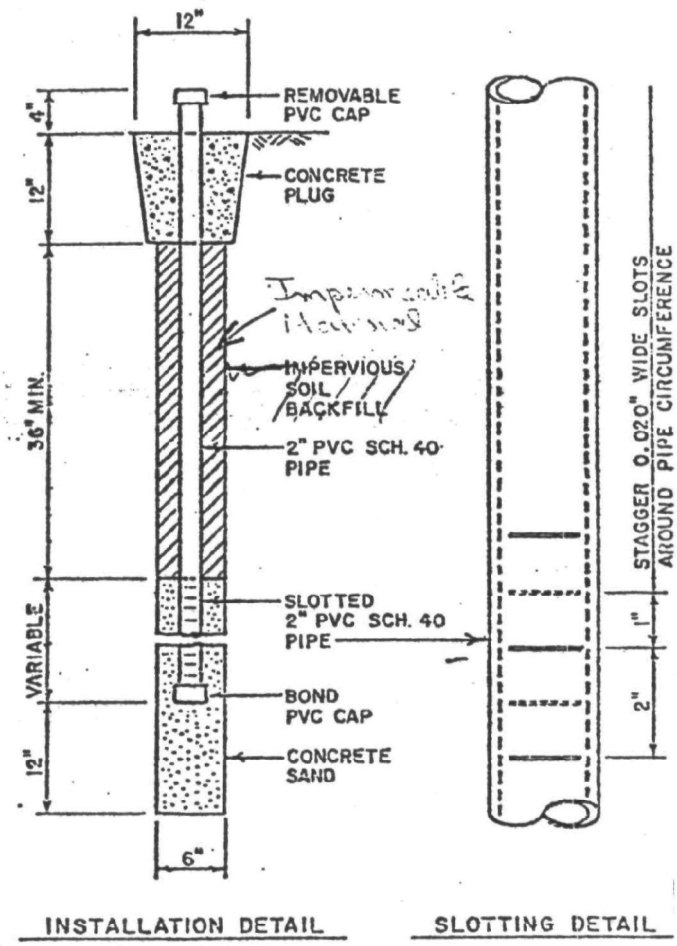
5.2.2 Piezometers

Methodology

Although the terms piezometer and observation well are commonly used interchangable, there is a significant difference between them. As implied by its name, a piezometer is a pressure measuring device, frequently used for monitoring: 1) water pressure in earthen dams or under foundations, and 2) artesian pressure in confined aquifers. The piezometer, a porous tube or plate in the former and a screened well or open hole in the latter, is isolated from other pressure environments by an impermeable seal of either clay or cement. Water samples representative of a specific horizon can be obtained from well-type piezometers, a highly desirable factor in designing a monitoring program (Figure 5-6). Piezometers can also be used to measure vertical head differences under unconfined conditions if the well screen is properly isolated by an impermeable seal immediately above the screen. Any well constructed without this seal cannot be considered a piezometer. However, there is a significant difference in application to landfill leachate monitoring between a piezometer and a well screened over a single vertical interval. The relatively impermeable annular seal will prevent downward movement of leachate into uncontaminated zones of the aquifer.

A low-cost modification of a typical engineering piezometer will allow collection of in-situ ground water samples throughout the saturated thickness of an aquifer. The piezometer, on modification, resembles the deep pressure-vacuum lysimeter described above (Figure 5-7). However, porous PVC is used instead of a ceramic cup, which is not necessary and would, in fact, decrease the effectiveness of



~~Fig. 1.~~ Piezometer installation for shallow ground-water monitoring around sanitary landfills.

Clark, 1975

Figure 5-6

Geraghty & Miller, Inc.

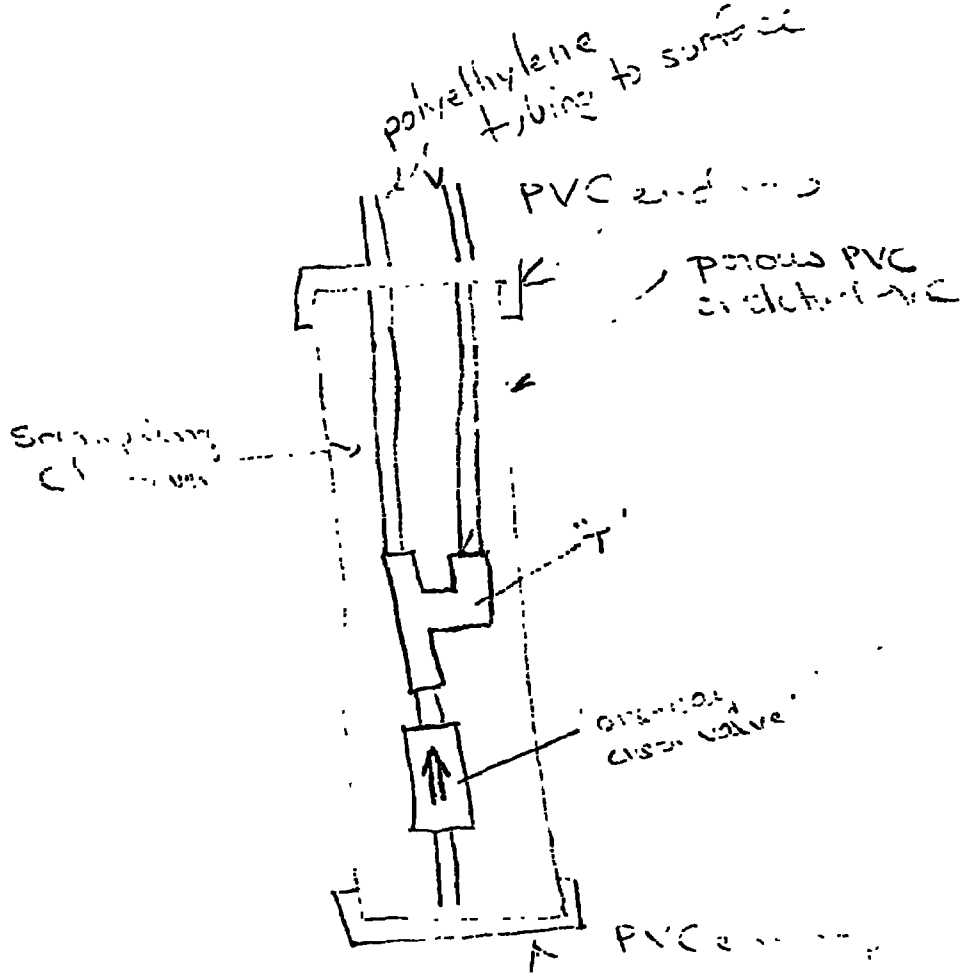


Figure 7

the sampler. With porous PVC, a vacuum applied to the sampling chamber is immediately transmitted to the aquifer, drawing water into it. A porous ceramic cup holds a vacuum and water slowly moves through it, a characteristic necessary to collect soil-water samples. Using porous PVC (or even slotted PVC) should substantially reduce the cost of the sampler below the approximately $30 charge for commercially available deep pressure-vacuum lysimeters. The surface sampling procedure is the same as that for a pressure-vacuum lysimeter.

Implementation

In order to place a grout or bentonite seal around a well casing as required in piezometer construction, there must be an annulus between the casing and the borehole wall. This limits drilling methods to (1) cable tool, (2) one of the rotary techniques or (3) hollow stem augering and a drilling contractor will be required. After casing and screen have been installed, a gravel pack is placed around the screen. To seal the well casing, a neat cement grout or bentonite slurry is poured or pumped into the annulus, thus preventing the vertical leakage that might occur in the annulus if the well is merely backfilled with cuttings or fill. This seal is vital from a sampling standpoint, because the sample withdrawn from the well is from a known vertical interval of the aquifer. Without the seal, rainwater would infiltrate backfill, potentially diluting samples collected from the well or leachate could move downward, causing samples to be non-representative. Another

consideration is that the seal tends to prevent downward movement of leachate in the annular material which may act as a conduit to uncontaminated zones of an aquifer. Constructing a monitoring well that contributes to or hastens the spread of contamination is not a recommended procedure.

Piezometers.

Since piezometers and wells screened over a single interval are the same except for an impermeable seal between the casing and borehole wall, the only price difference will be that incurred for placing the impermeable seal and purchase of necessary sealing materials. In the hypothetical aquifer, only a 10 foot seal is required and a two-man crew should be able to place it in half a day to a day. This would increase installation cost about 500 to 1000 ~~dollars~~ if grout ~~was~~ were pumped into place and approximately $250 to $450 if bentonite ~~was~~ were placed by hand. Total cost per installation is given in Table 1.

Advantages	Disadvantages
1. Point sample is collected from a known vertical section of an aquifer	1. Restricted number of drilling methods
2. Construction prevents downward migration of leachate in borehole	2. No information obtained on ~~artificial~~ vertical distribution of contaminant
3. Can be installed inexpensively and rapidly if casing diameter is small	3. Improper completion depth can give incorrect picture of leachate distribution
4. Can provide composite ground-water samples if screen covers saturated thickness of aquifer	
5. Modification of an engineering piezometer will allow vertical sampling of contaminant	

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5.2.3 Well Clusters

Methodology

The major drawback in using individual wells screened over a short vertical distance of the aquifer is that they provide no information on the vertical distribution of contaminant and only rudimentary information on its areal distribution. To overcome this, investigators (Pitt, 1974; Weist and Pettijohn 1975; Aulenback and Toffenmore, 1975; Parlmquist and Sendelein, 1975; Fryberger, 1972; and Kimmel and Braids, 1975) have used well clusters to define the vertical distribution of a contaminant. Each cluster consists of a group of closely spaced, small-diameter wells completed at different depths in an aquifer from which water samples representative of different horizons within the aquifer can be collected. Careful placement of well clusters at the landfill site and its vicinity will allow reliable delineation of both vertical and areal leachate distribution.

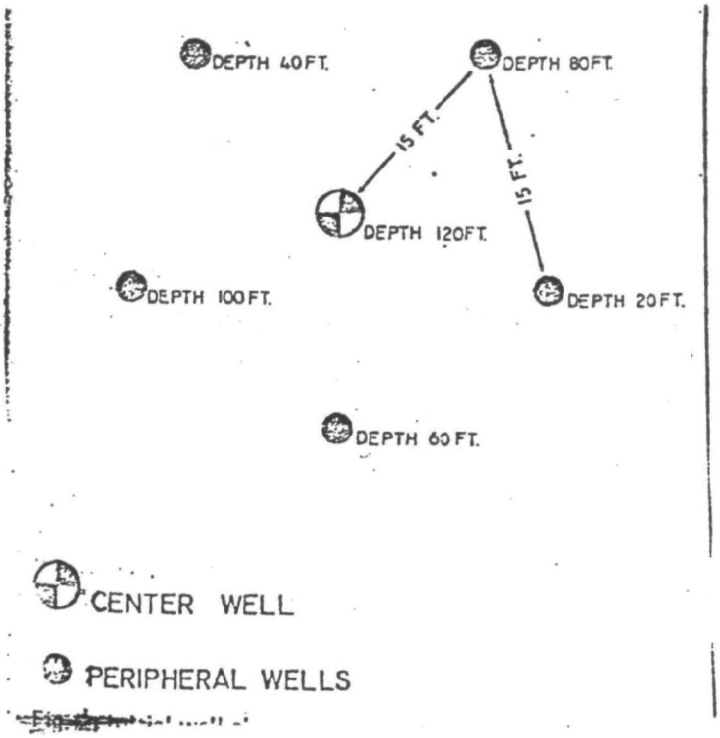
Well clusters are by far the most common and successful technique, to date, for delineating ground water contamination. One shortcoming, however, is selection of the completion depth of each well in the cluster. Several approaches to selecting this depth have been made; 1) a pair of wells, one screened at the top, the other at the bottom of the aquifer (Burt, 1972, and Geraghty & Miller, 1975); a three-well cluster, with screens set on the top, middle, and bottom of the aquifer under investigation (Weist and Pettijohn, 1975); and 3) clusters in which the screened intervals are separated

by preselected intervals, such as the 10, 20, 30, 40, and 60 foot screen depths used by Pitt (1974); the 20 foot separation from 20 to 100 feet used by Yare (1975) (Figure 8), or terminating 2 to 3 wells at 10-15 feet intervals as recommended by Palmquist and Sendelein (1975). The fixed sampling depth, whatever the screen placement selected, limits to some degree the usefullness of the well cluster.

As pointed out by Yare (1975), large vertical zones of an aquifer would not be sampled, dependent on saturated thickness, even if up to five wells are constructed in each cluster. Some uncertainity will always exist as to the actual vertical distribution of contaminant. Construction of more wells per cluster is not the answer; only so many wells can be constructed close enough together to represent vertical contaminant distribution at one point. In addition, construction cost as well as the time required to complete the cluster would become prohibitive. The only way to get a "true to life" as possible picture of leachate distribution is to collect ground water samples during drilling, a technique described below.

Implementation

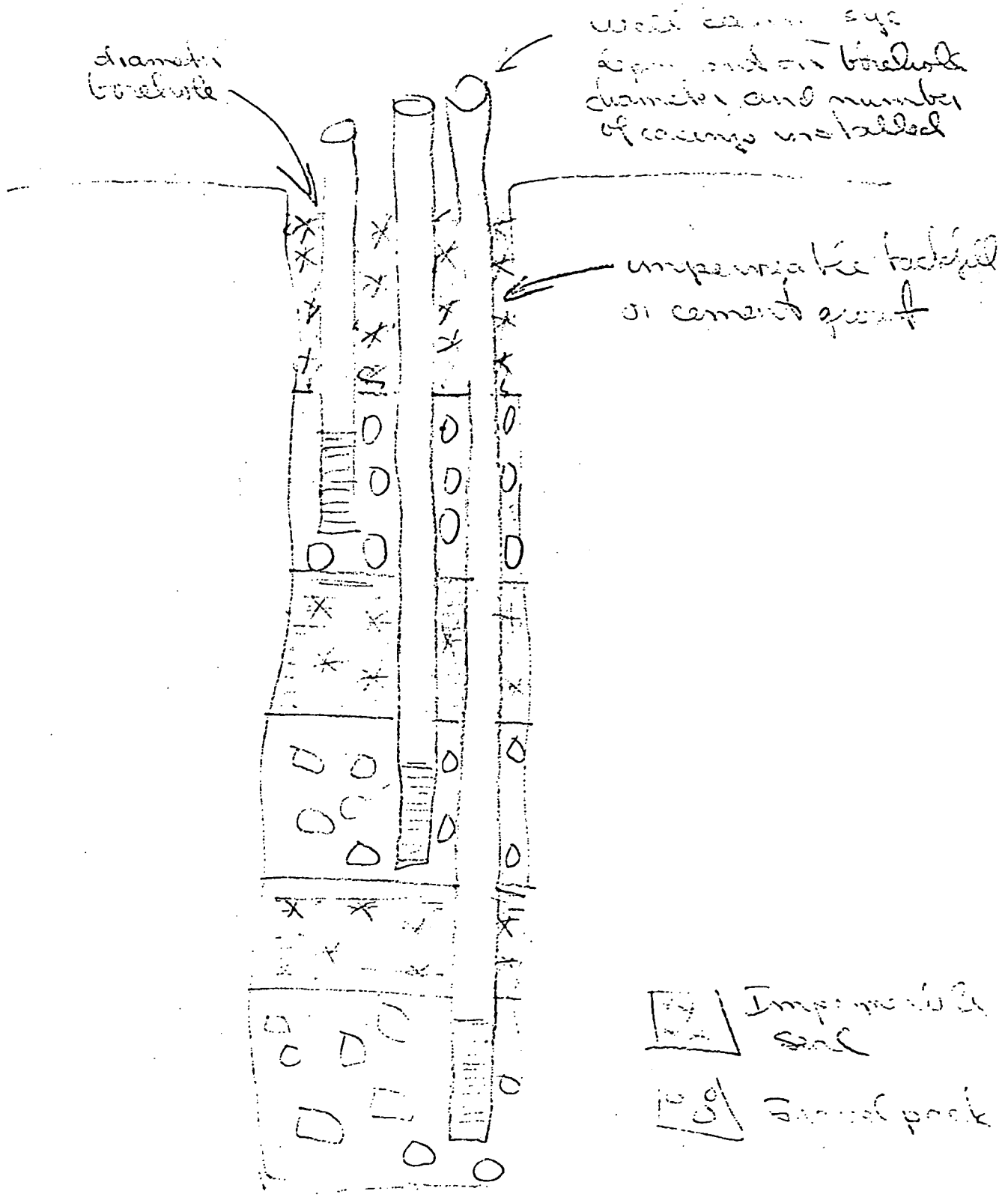
Well clusters are easily installed, a major factor to be considered when designing a leachate monitoring system. Normally, in unconsolidated sediments, small-diameter steel casing (2-2 1/2 inches) is driven by the jet drilling method to the desired depth and the screen is set by the casing pull-back method or by augering a hole



Well Cluster Configuration

and forcing a well point to the desired depth. Alternatively, a hole can be drilled or augered to a predetermined depth and a common well point driven out the bottom of the hole into undisturbed sediments. Either installation technique is relatively rapid and inexpensive. For shallow aquifers (20 - 30 ft), 1 1/4 inch well points can be driven by hand to construct a cluster (Aulenback and Tofflemire, 1975).

Another approach to well cluster construction is multiple well completions in a single borehole. This involves drilling a large-diameter hole, either by a rotary technique or bucket auger, and installing small-diameter wells to selected depths with each screened zone isolated from the others by an impermeable seal. Meyer (1973) completed as many as 3, four-inch PVC wells in a single 22 inch borehole and Hughes and others (1971) install up to six 1 1/4-inch to 2-inch observation wells in one boring. This technique for constructing wells clusters seems feasible provided the cost of drilling large diameter boreholes is not prohibitive and care is taken in placing the impermeable seals between screened zones (Figure 5-9). A great advantage is being able to construct the wells close enough together to get samples actually representative of a single point (areally) in the aquifer, thus increasing the value of the data obtained on water quality. If care is taken in constructing the seals between the individual wells, such as using a shrinkage-inhibitor in the cement grout, reliable samples of in-situ ground water can be obtained. Of course, the greater the number of casings in the borehole, the greater the likelihood of imperfect seals between the casings. To insure that the seals are effective, water levels


diameter
borehole
unpermeable backfill
of cement grout
Impermeable
seal
Gravel pack

in wells not being sampled should be checked for sharp drops. An abrupt drop would tend to indicate a vertical connection between the screens.

Well Clusters

A variety of drilling methods can be used to install a well cluster in the hypothetical aquifer, however, the best method is jet-percussion because of the tight seal between casing and formation and relatively low drilling changes. This installation, consisting of a cluster of five wells 2-inch diameter with screens from 10 to 30, 30 to 50, 50 to 70, 70 to 90, and 90 to 110 feet below ground surface would require an expenditure of $2500 to $3800. If only 5-foot long screens were centered around depths of 30, 50 70, 90 and 110 feet, the well cluster would cost $1700 to $2,300 to install. However, as discussed above, this type of construction is not ideal because of the vertical distances between the screened intervals through which the plume of leachate enriched ground water might pass undetected.

Another way to construct well clusters in this aquifer would be to use a power auger. In this method, a sediment is loosened by a flight of augers and then a well point and two-inch casing is pushed through the loosened formation material to the desired completion depth. One problem with this construction is the potential for vertical leakage of water through the column of loosened soil around the well casing. Another is that sturdy, screens must be used in order to withstand the stress of being driven through the loosened sediments in the borehole. In normal practice,

relatively inexpensive drive point, five feet or less in length are used. However, to monitor the entire saturated thickness of the hypothetical aquifer, five 20-foot long, stainless steel wire wound screens with a drive point will be required, considerably increasing total cost of the installation. An augered five well cluster with 5-foot long, inexpensive drive points should cost $1800 to $2600 while the most effective installation from a sampling standpoint with five 20-foot long, 2-inch diameter stainless steel screens would cost $4600 to $5300, a significant difference.

More expensive alternatives are drilling with either the cable tool or hydraulic rotary method; the former costing about $9850 to $14,150 per cluster of 6-inch diameter wells and the lather from $13,800 to $19,400. The substantial difference in these figures is due to the necessity of grouting the annulus between casing and borehole wall in the rotary drilled holes. Grout is necessary to prevent vertical leakage of water through the annular material, into the screen with leakage, samples would not be representative of formation water in the screened zone. Because of the seal between casing and borehole wall in a cable tool well, grouting is not necessary. No substantial economies could be obtained by switching to 4-inch diameter cable tool holes. Drilling costs will be about the same as for 6-inch

and the main savings would come from using 4-inch stainless steel wire wound screens rather than 6 inch.

An alternative to five individual well completions to make a cluster is to install multiple casings in a single borehole. This type of installation would cost $8,200 to $11,000 for five 4-inch diameter wells installed in a 24-inch diameter borehole and in the range of $4240 to $5880 for five, 2-inch wells installed in a 12-inch hole. Because of the necessity of forming a good seal between each screen, 15-feet long screens will have to be used in the hypothetical aquifer, which allows for a five foot seal between each screened interval. As discussed in Chapter 5.2.3, this seal is critical and if not properly constructed, anomalous water-quality samples will result.

Advantages

1. Simple installation which does not always require a drilling contractor.
2. Excellent vertical sampling if enough wells are constructed
3. Tried and true methodology, accepted, used in most contamination studies where vertical sampling is required
4. Low cost if only a few wells per cluster are involved, and drilling contractor set-up for small diameter wells can be found.

Disadvantages

1. Large vertical sections of the aquifer are unsampled. Artificial constraint on data by completion depths - what's happening in unsampled zones?
2. If jetting rigs or augers are used, installations are limited to 125 to 150 feet total depth and installation is slow.
3. Small diameter wells can be used only for monitoring, cannot be used in abatement schemes.
4. Difficult to develop and sample if water level is below suction lift in small diameter wells.

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5.2.4 Single Well - Multiple Sample Points

Methodology

In order to sample multiple horizons in a single well, screens or casing perforations must be constructed at regular intervals in the well. Spacing will depend on the sample density required and construction expense; the greater the number of open zones, the higher the well costs. The California Department of Water Resources (1963) successfully obtained closely-spaced ground-water samples by perforating steel casing with a mechanical perforator at set intervals in the well isolating each set of perforations with inflatable packers, and pumping the isolated casing segment with a submersible pump (Figure 5-10). The attractiveness of this type of sampling operation is apparent. However, there are some pitfalls. Care must be taken to insure that the packers are isolating the sampled section of screen and that no water from above or below is leaking past the packers, contaminating the sample. Also pumping rates must be kept low to insure that formation water is drawn from only opposite the isolated section. Higher pumping rates will induce flow from horizons above and below the level of the aquifer being sampled, resulting in an unrepresentative sample. If the annulus between the casing and the borehole ~~was~~ are backfilled, the possibility of vertical movement of water in the annular area exists. Therefore, there is no guarantee that a sample does not contain water from a lower or higher horizons which has moved through the annular material under the influence of the pumping gradient. To adequately protect against this type of sample contamination, an impermeable seal of either bentonite or cement grout should be

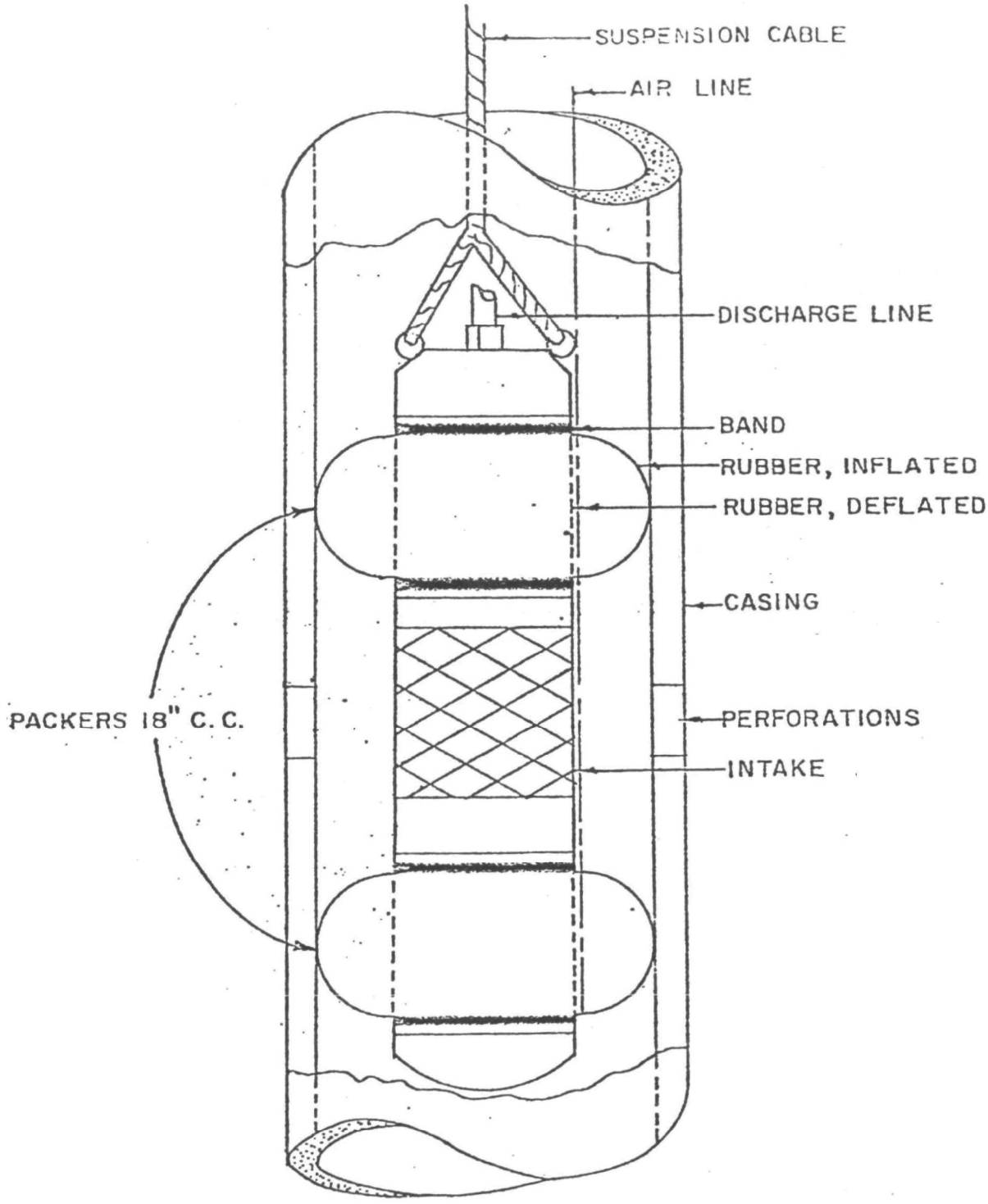
SUSPENSION CABLE
AIR LINE
DISCHARGE LINE
BAND
RUBBER, INFLATED
RUBBER, DEFLATED
CASING
PERFORATIONS
INTAKE
PACKERS 18" C. C.

Figure 5-12

placed between every screen or slotted interval. This may not be possible with closely-spaced screens or casing perforations. A well constructed and sampled ~~taking into account the above~~ according to these specifications will provide an excellent opportunity to samples ~~of~~ the vertical distribution of a contaminant.

Another approach for shallow vertical sampling using a single well is that described by Hansen and Harris (1974). They isolated fiberglass probes at regular spacings inside an 1 1/4 diameter well point (Figure 11). Samples were drawn to the surface through a tube attached to the fiberglass probe after the well point was driven to the desired depth. This type of construction is inexpensive and can be "homemade" with little difficulty, but will only allow collection of samples from depths less than section limit about 30 ft, at sea level.

Implementation

Installing a multiple sampling point well will require the services of a drilling contractor unless the device described by Hansen and Harris (1974) is used. This is essentially a drive point which could be driven by hand to depths of about 30 feet. On the other hand, multiple screen or slotted casing installations require the skills of a well driller. For one thing, a large-diameter open borehole in which casing can be set is needed, necessitating a cable tool or rotary rig. Six-inch diameter or larger casing should be used in order to accomodate the packer pump unit which in turn requires a tripod winch,

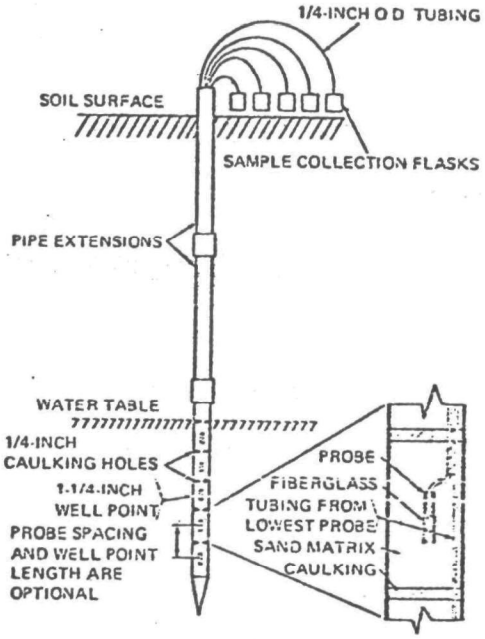

Fig. 5 Construction details of the groundwater profile sampler.

Hansen + Mavis, 1974

Figs. 5-11

power and air supply. If steel casing is not slotted before installation, a special down hole tool is required to make the slots. Skill and equipment requirements for this type of installation therefore necessitate use of a drilling contractor.

The packer/pump is not quite so formidable. Once a good quality submersible pump has been obtained, local investigators or a machine shop can equip it with packers. Cherry (1965), has described the design and operation of a rather elaborate packer pump (Figure F-12).

—> indent for quotation

"This sampler collects a pumped sample of water from a specific zone in an uncased or multi-screened well. Minor modification of the sampler permits remote measurement of several chemical and physical characteristics of the water in the zone being sampled. The sampler can be used in wells with diameters of 8 to 16 inches, inclusive, which do not contain pumps, pipes or other obstructions. It is suspended on a cable from an A-frame, and is raised and lowered by an electric motor that is powered by a 110 volt a-c portable generator. This generator also runs the electric pump which is part of the sampler.

The sampler consists of two inflatable packers or boots - one mounted above the submersible pump and other below it. When the boots are inflated, the zone between them is isolated from the remainder of the well and water can be pumped from this isolated zone.

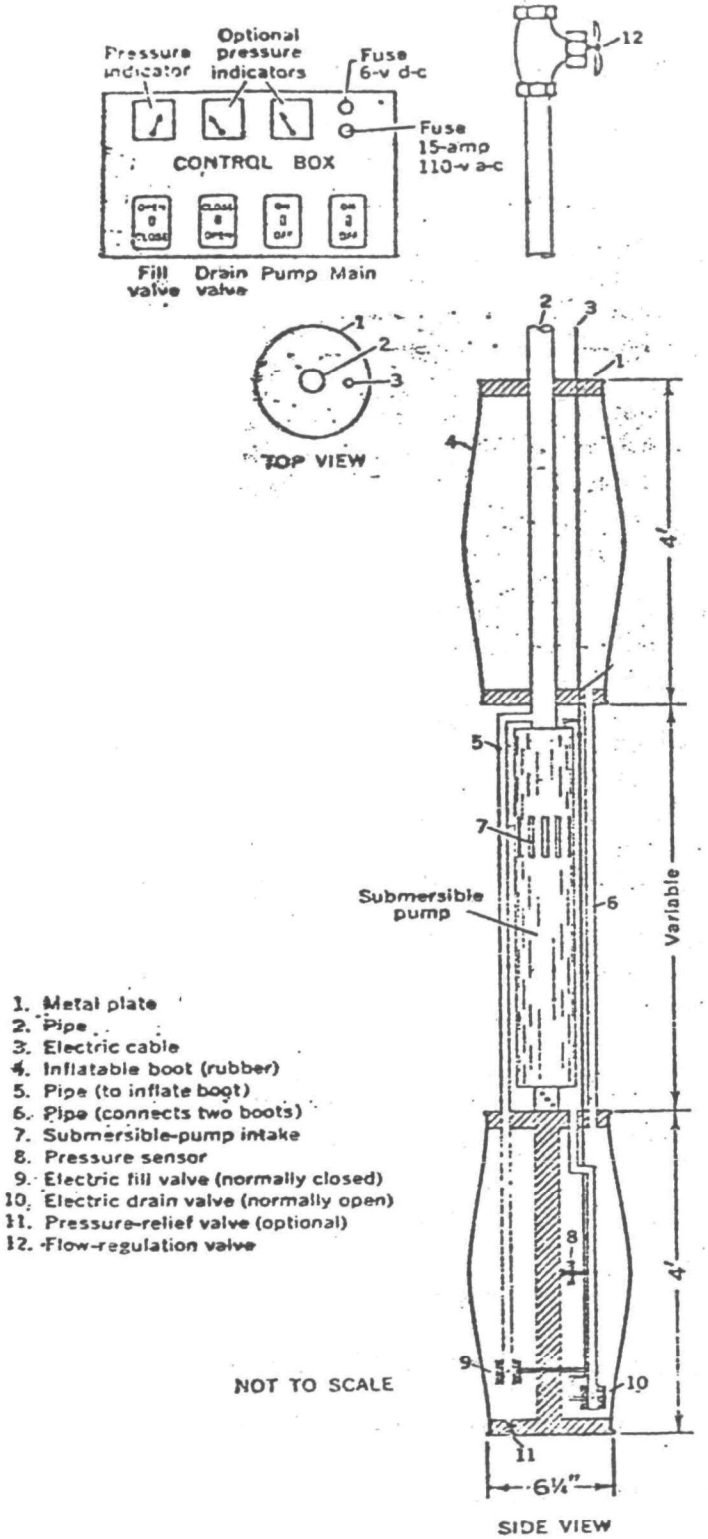

FIGURE 1.—The Casee sampler.

INSTRUCTIONS FOR OPERATING INFLATABLE BOOT SHOWN IN FIGURE 1

To inflate boots:

1. Close flow-regulation valve (12).
2. Open fill valve (9), which is normally closed.
3. Close drain valve (10), which is normally open.
4. Start pump.

Water is pumped from well through valve (9), filling the lower boot. The water then passes through pipe (6), filling the upper boot; water is forced into the boot until the pressure inside is 15 lbs per sq in greater than the pressure outside, and the fill valve (9) is then closed. The pressure is measured electrically (S) and is read on a meter at the control box (above). Other pressure-measuring devices (optional) may be used below the sampler and inside the pump section.

To pump sample:

1. Inflate boots.
2. Close fill valve (9).
3. Open flow-regulation valve (12).

To deflate boots:

1. Open flow-regulation valve (12).
2. Close fill valve (9).
3. Open drain valve (10).
4. Activation of pump will generally reduce the deflation time.

Figure 5-12

The capacity of the pump is about 15 gallons per minute. The spacing of the boots can be varied by using different lengths of connecting pipe between them. The minimum spacing of the boots is 5 feet (length of pump). The boots are inflated by pumping water into them, from the well, through an electrically controlled valve; they are deflated by pumping the water out of them through another electrically controlled valve. Advantages of this sampler over other packer-type samplers are its portability, the ease with which it can be repositioned without removing it from the well, and the fact that it is relatively inexpensive.

Instruments to measure temperature, specific conductance, or other chemical or physical characteristics of the water in the well can be placed in the space between the boots. Experience has shown, that continuous measurement of specific conductance, in this way, is very useful in determining the proper time to collect the water samples. It is desirable to pump from the well a volume of water equal to at least 3 times the capacity of the discharge line and isolated section before collecting samples for analysis, in order to ensure the collection of representative samples."

The packer/pump shown in Figure 10 is less elaborate and more amenable to fabrication without the facilites of a machine shop. Although actual construction is not described in the California Department of Water Resources report, it seems that two rubber diaphragms[9/] possibly cut from tire inner tubes, were clamped, probably with stainless steel hose clamps, to the exterior of the pump. An air line to the surface allows inflating or deflating the packers. If only shallow sampling depths are involved, "home made " diaphragms[9/] and valves should

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take the inflation pressures required, greatly reducing fabrication costs of the packer/pump.

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Implementation

Single Well/Multiple Sampling Points

Perhaps the easiest way to construct a well capable of being sampled at set intervals within the casing is to use 6-inch PVC casing, slotted PVC screen, and glued-joint couplings. The screen sections of set length can be separated by the appropriate lengths of blank casing using only a handsaw to cut the lengths and PVC cement to join them together. This construction can be done rapidly and easily by hand, with simple tools and little skill. A well in the hypothetical aquifer, with one-foot long screen sections separated by four feet of casing would cost $3000 to $4700 to install. Steel casing and screen would be considerably more expensive to assemble in this manner, and therefore casing perforation is necessary. The cost of casing perforation will depend on how familiar a driller is with performing this operation and whether or not the equipment is readily available.

The pump/packer assembly necessary for sampling could be fabricated for $1000 to 2000 or more depending on the pump used and how elaborate the packer system is. Portable generators capable of supplying the power necessary for the pump can be purchased for several hundred dollars. Although these prices seem steep, they are one-time cost and with proper care and maintenance the pump/packer system should last years.

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Advantages

1. Excellent information on vertical distribution of contaminant.
2. Well diameter is large enough to use in a pumped withdrawal program, if necessary
3. Water samples representative of specific horizons within an aquifer can be collected
4. Sampling depths only limited by size of sampling pump
5. Rapid installation possible

Disadvantages

1. Expensive
2. Proper well construction and sampling critical to successful application
3. Complicated sampling procedure involving a a great deal of equipment

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5.2.5
- Sampling During Drilling

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Methodology

A major disadvantage of the sampling techniques described above is that a constraint is placed on the data obtained from the ground water samples by the fixed or arbitrary point at which water samples are collected. In other words, "blind" placement of wells can result in a false representation of contaminant distribution if it is not uniformly dispersed throughout the aquifer at the sampling point. Contaminants are often stratified; underlain, overlain, or interfingering with uncontaminated ground water. To define these relationships adequately, information on the vertical distribution of contaminant must be obtained prior to installation of the monitoring well. This information can be obtained by formation water sampling during drilling.

Several researchers have successfully obtained in-situ ground water samples during drilling using three basic techniques: 1) driving a casing (Fryberger, 1962), or well point (Childs and others, 1974) to the desired depth and bailing or pumping a water sample from that depth, repeating the process to completion depth or refusal; 2) drilling a mud rotary hole to the sampling depth, pulling the drilling string, setting and gravel packing a temporary well screen, and pumping a formation water sample. Yare, 1975); and 3)drilling a borehold to the desired horizon, setting a cone packer and riser pipe into the smaller hole and

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pumping a sample (Harden, personal communication, 1974).

If proper precautions are taken, formation water samples collected using these techniques will be representative of water quality at a known vertical interval of the aquifer. The critical factor in successful application is developing the temporary well to the point where all traces of drilling fluid have disappeared from the pumped water before a sample is collected. Dilution of the sample by the drilling fluid and contributions of chemical constituents by clay particles in the mud will produce erroneous and erratic data, and little information will be gained on the actual vertical distribution of contaminant.

The main advantage of this type of sampling is that the top, bottom, and internal stratification of the contaminated slug can be defined with reasonable accuracy prior to setting a permanent casing and screen. With this information, the well can be designed for the most advantageous sampling and/or withdrawal of the contaminant at that point in the aquifer. Then, changes in the vertical distribution can be monitored very closely.

Implementation

Yare (1975) used a ground-water sampling during drilling techniques in the course of investigating a hexavelent chromium contamination problem. A borehole was drilled with a hydraulic rotary rig and drilling mud was made with an organic

base drilling fluid additive. In order to minimize the effect of drilling fluid on the formation water. A slotted PVC screen attached to a riser pipe was lowered to the bottom of the borehole packed with fine gravel. This, in essence, formed a well in the borehole.

After the gravel had time to settle through the drilling mud and around the screen, the well was pumped until no further traces of mud could be detected. By this time, an effective filter cake had formed on top of the gravel pack, isolating the screen from the fluid in the borehole and the filter cake between the gravel and the borehole wall broke down because of the pumping gradient. To insure that formation water was collected, an additional 100 gallons of water were pumped before sampling. In this formation, the effective radius of the well was estimated at 2 feet.

The drilling and in-situ water sampling procedure which evolved from this investigation consists of the following steps as shown diagrammatically in Figure 5-13:

1. drilling an eight-inch diameter borehole to the sampling horizon;
2. pulling the drill string and replacing the bit with a five-foot long, four-inch diameter wire-wound well screen;
3. lowering the screen and drill string to the bottom of the hole and gravel-packing with number 2 gravel to cover the screen;
4. attaching a gasoline-powered centrifugal pump to the drill string and pumping until the drilling fluid level

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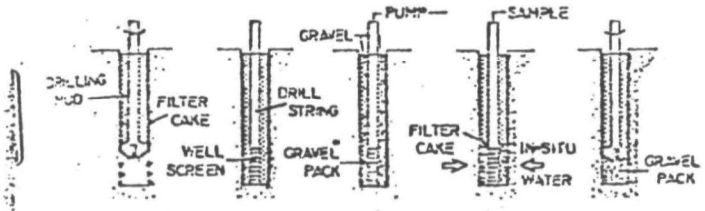

~~Fig. 1.~~ In-situ ground-water sampling procedure.

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Figure 5-13

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stabilizes in the hole and the discharge clears of drilling fluid (in this case, a centrifugal pump could be used because static water levels ranged from 6 to 12 feet below ground surface);

5. pumping at least 100 gallons of formation water before collecting the sample; and
6. pulling and removing the screen, then lowering the bit and drill string, and drilling to the next sampling horizon.

Harden (personal communication, 1974) described a sampling during drilling technique useful in the deep holes in consolidated sediment or rock. In this method, the original hole drilled is 6 3/4 inches in diameter. When the hole penetrates about 15 feet to 30 feet into a sand from which a water sample is desired, drilling is stopped (step 1 in Figure 14), and the hole is reamed to a diameter of 9-7/8 inches down to a point just above the zone selected for water sampling. Then the original 6-3/4 inch hole is washed out to its original depth, (step 2 in Figure 14) and a string of pipe with packer and screen is set in the hole, as shown in step 3 of Figure 14. The pipe is usually 4 inches in diameter, and the packer is a commercial rubber cone type, with typical dimensions of 6 by 9 by 14 inches. Often a canvas "shirt tail" is wrapped by the packer to assist in sealing. The packer is set on the shoulder between the 6-3/4 inch and the 9-7/8 inch portion of the hole. Below the packer a commercial 4-inch water well screen 10 to 20 feet long is attached to the 4-inch pipe. After the packer is seated, the temporary well is developed by airlift for several hours until the water becom s clear. After

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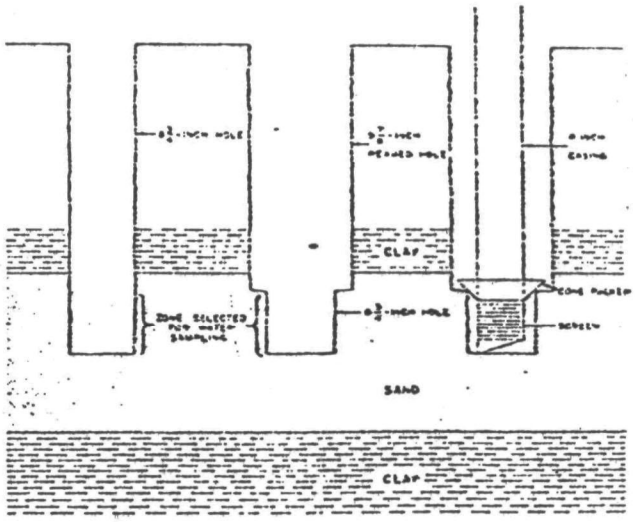
Step 1 Step 2 Step 3



~~Figure 24.~~—Procedure for Water Sampling From Test Hole.

Figure 5-14

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this the airline is removed from the 4-inch pipe, and a small diameter turbine or hi-lift pump is installed and the temporary well is again pumped until the water becomes clear, after which the final samples are taken.

At the end of the pumping, the casing and screen are then pulled from the hole, and drilling of the 6-3/4 inch hole is resumed until a second water-bearing zone is encountered from which a water sample is desired, at which time the entire water-sampling process is repeated.

INSERT 5-39A →

Advantages

1. The best technique currently available for defining vertical distribution of contaminant.
2. Completed well can be used for water quality monitoring and/or pumped withdrawal of contaminant.

Disadvantages

1. Considerable expense per well.
2. Requires a knowledgeable drilling contractor and careful supervision of the drilling and sampling.

5.2.b

Pore Water Extraction from Core Samples

An indirect method of sampling in-situ ground water during drilling is to extract pore water from core samples collected as the borehole is advanced. This is accomplished by placing a segment of the core in a commercially available filter press, hydraulic ram, or centrifuge and forcing interstitial water out of the core. Standard analytical techniques can then be used on the extracted water, typically the titriametric analysis for chloride and the electrical conductivity measurements. (Swarzenski, 1959, Luszynski, 1961;

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Sampling During Drilling

The most efficient technique for sampling the hypothetical aquifer described in 5.2.1 is to use the technique described by Yare (1975). If this method is used, 10 samples can be obtained at depths of 20, 30, 40, 50, 60, 70, 80, 90, 100 and 110 feet below ground surface and five of the most contaminated intervals of the aquifer can be screened with 5-feet long, 4-inch plastic screens for a total cost of $3000 to $4700. Additional sample points would cost $125 to $200 each. These screen segments could be sampled with a packer/pump or by installing and isolating a deep pressure/vacuum lysimeter in each screened interval. If a packer/pump is used, 60 inch casing is necessary and total cost per installation would be in the range of $3300 to $5200. Lysimeters could be placed for approximately $100 each.

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and Gill and others, 1963). A major problem with this technique, is determining the amount of drilling fluid invasion into the core during the process of driving the coring device and bringing it to the surface; the greater the invasion, the less reliable the water-quality data obtained. Sand and gravels are more readily invaded than finer grained sediments.

Luscynski (1961) overcame this problem by putting fluorescein dye (green color) in the drilling mud. Any penetration of drilling mud into a core sample would be shown by the green dye, and the uninvaded core sections could be selected for extraction. Thus, quantitative chemical analysis could be assured because dilution of pore water by drilling fluids would not be a factor. Unfortunately, on completion of drilling, disposal of the bright green drilling mud is a problem (John Isbister, personal communication, 1975). Normally, drilling mud is just dumped on the ground and is eventually eroded away. Because of its natural gray or brown color it is not very obvious. The bright green mud would be a definite eyesore, it is not toxic and would probably have to be disposed of by burial at the site - no real solution.

Low permeability, porous, saturated sediments will retain most or all of their interstitial water during core sampling. Walker and others (1972) used this sediment characteristic to advantage in tracing a bulb of nitrate-contaminated ground water at an Illinois farm underlain by loess (Figure 15). The cores were air dried, leached with water, and the solution was analyzed for nitrate-nitrogen. Because of the nature of the sediment, Walker and his co-investigators were able to trace the contaminant bulb. Under

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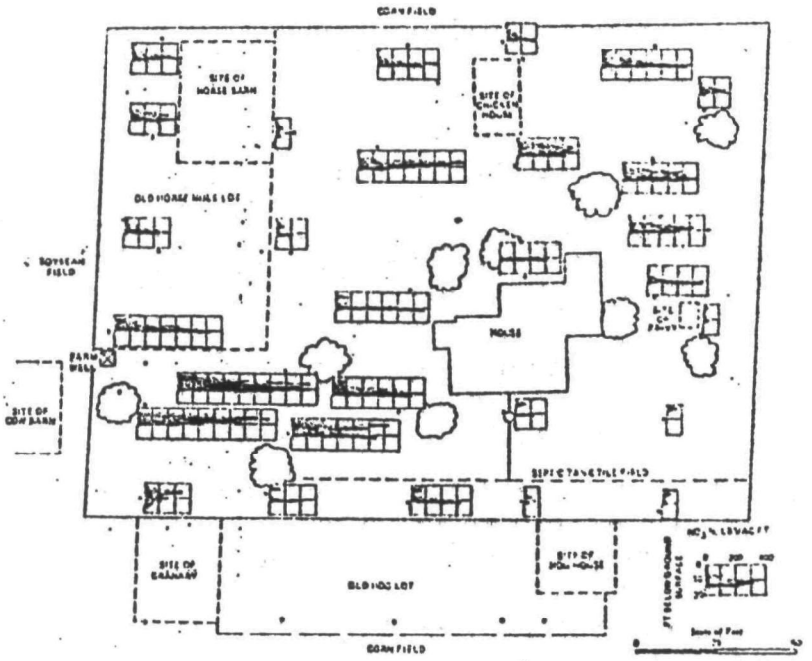

- FARMSTEAD SOIL NITRATE CONCENTRATION AND DISTRIBUTION

different soil conditions, such as sand and gravel, this technique would be less applicable because little interstitial water is trapped in the core sample. In this type of sediment the solution collected from leaching the air dried samples would primarily represent adsorbed constituents whose concentration would depend on the chemical activity of the soil.

Implementation

indent quote

"The filter press used Luscynski (1961) describes in detail using a filter press to extract pore water, is one of several types sold commercially for use in determining filtration properties of drilling muds. The unit consists of a chamber, a filtering medium, a graduated tube for catching and measuring the filtrate, and a pressure-source unit. A cell, base cap. screen, rubber gaskets, and top cap make up the chamber. The cell is 3½ inches high and has inside and outside diameters of 3 and 3½ inches, respectively. The filtering medium is a sheet of filter paper which fits on the screen over the base cap; it has a filtering area of about 7 square inches. The filter paper used is specially hardened to withstand the pressure in the chamber. A graduated cylinder is used to catch the filtrate.

Uninvaded cores consisting of loose material such as sand and gravel are transferred to the filter-press chamber by spatula or spoon. Usually it is not practical to remove much more than 25 to 50 percent of the uninvaded material from the core barrel by this method. Enough material is transferred to fill the chamber about a quarter to half full. It is then tamped lightly until an integrated unit is formed in the chamber and a film of water is

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formed on the surface of the material and along the cylindrical wall.

Usually less than 10 to 20 percent of a solid or silty-clay sample is invaded by the drilling fluid. Plugs of the uninvaded material 1 to 2 inches long are placed in the chamber to fill it about a quarter full. Then they are molded and tamped into an integraded unit. Difficulty can be experienced in molding and tamping a relatively dry solid clay-one having a water content of less than 15 to 20 percent of the dry weight of the clay.

The total time between the opening of the split-spoon core barrel and the placing of the chamber in the frame is usually 1 to 2 minutes for the loose material and 2 to 5 minutes for the tight material. There is thus, very little opportunity for evaporation.

After a sample is properly prepared for filtration, the chamber is fully assembled, placed in the frame, and made airtight by the T-screw (Figure 15). The gas pressure is then applied.

The pressure of the carbon dioxide gas in the chamber moves some of the interstitial water through the filter screen and filter tube into the graduated cylinder. Pressures of 5 to 30 psi suffice for the gravel, sand, and silt samples. Pressures of about 100 psi are usually sufficient for silty and solid clay samples. The carbon dioxide gas does not alter the chloride concentration of water forced from the material into the filtrate tube.

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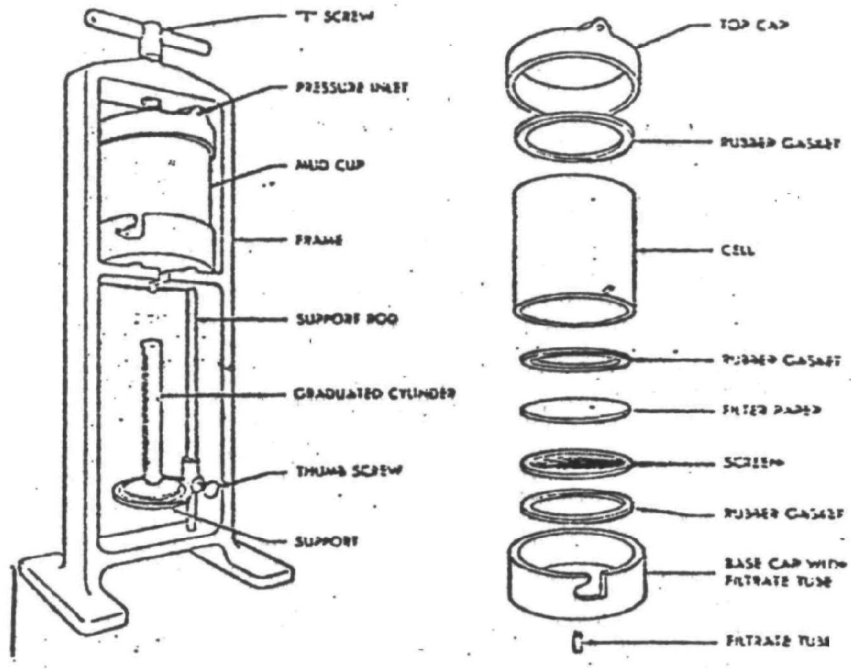

FIGURE 1.—Filter press and chamber assembly. Courtesy of Baciod Division, National Lead Co.

Iwaszenko, 1961

Figure 5-16

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Chloride determinations of the filtrate are made in the field by the standard titration method using silver nitrate solution. Relatively large amounts of filtrate (25 to 50 ml) are needed when fresh water is to be tested, and relatively small amounts (1 to 10 ml) when salt water or diffused water is to be tested.

Usually enough filtrate can be obtained from the uncontaminated material of only one core if its taken in a diffused-water or salt-water zone. However, more than one core may be necessary to obtain the required amount of filtrate if the core is taken in a fresh-water zone; this is particularly true for solid-clay samples, which yield only small amounts of interstitial water."

An alternative to the mud filter-press is to fabricate a core sample squeezer which utilizes a hydraulic ram, as described by Manheim (1965):

"The squeezer utilizes a commercially available cylinder and ram (made by the Carver Co., Summit, N. J.) to which a machined base with a filtering element and fluid outlet is fitted. Construction details are shown in Figure 5-17. The filter unit consists of a stainless steel screen and a perforated steel plate contained in a circular recess in the steel filter holder. Alternatively, a porous (sintered) metal plate may be used to replace both screen and perforated plate. The top surface of the filter should be flush with the other rim of the holder so that it may support one or more paper filter disks. Finegrained, hardened laboratory filters give a visually clear effluent, but membrane or microfilters may be used to assure maximum freedom from suspended matter. The lower part of the filtering assembly, fitted with a rubber washer, protrudes into a recess in the steel base; when pressure is applied the gasket is squeezed against the cylinder and prevents leakage of water around the filter unit. The space in the recess

5-43A

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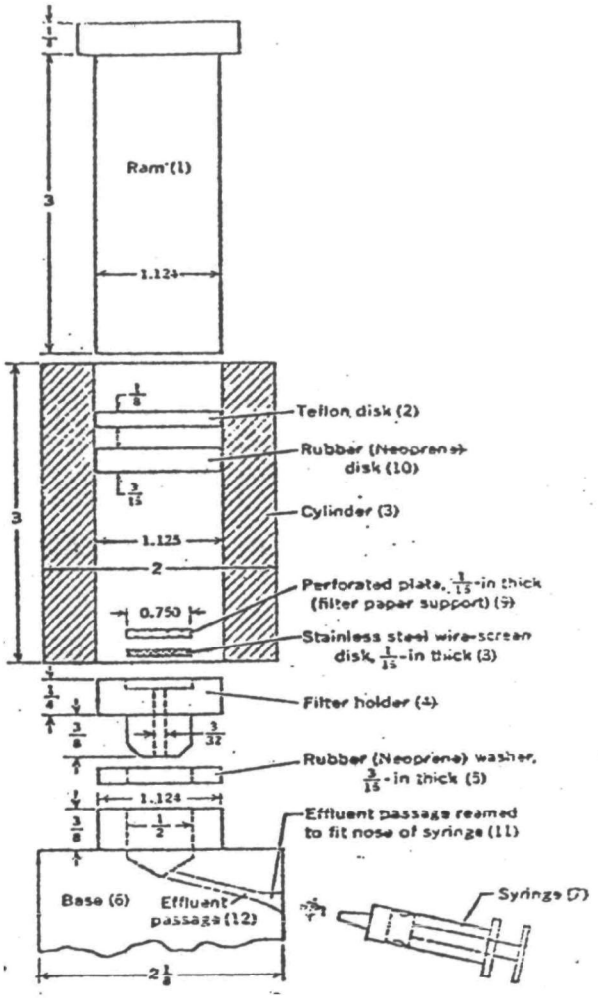

FIGURE 1.—Drawing showing components of hydraulic squeezer. All dimensions in inches. (Base is about 2½ inches in height.)

Figure 17

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and the diameter of the outflow boring are kept small so that little fluid can collect in the squeezer itself. All metal parts in the filter base are made of Iron and Steel Institute No. 303 stainless steel. Rubber and teflon disks just below the piston prevent loss of fluid upward when pressure is applied. The rubber, teflon, and filter-paper disks are punched out with an arbor punch and can be made as needed.

The present design permits insertion of a disposable syringe (preferably plastic) directly into the base of the squeezer to receive fluid. The narrow effluent hole is reamed out to permit fitting of the standard 'Luer' taper of the syringe nose.

A larger squeezer has also been constructed using a $2\frac{1}{2}$ - inch Carver cylinder and piston. The design is similar to that shown in Figure 5-16, except that the filter plate is increased in thickness to give greater strength. The effluent line remains small. Because the cross-sectional area of the cylinder bore of the small squeezer is about 0.88 inch, a 10-ton laboratory press exerting its maximum load of 20,000 pounds will apply a pressure of about 22,000 pounds per square inch to the sediment. However, a 20,000 lb load will apply only about 5,000 psi in the large unit. The large squeezer should therefore be used with a higher capacity press when more compact sediments are to be squeezed.

In sequence, the steps in squeezing a sample are as follows: The filter holder with its gasket is placed in the recess of the filter base. The screen, perforated plate (or porous disk) and 2 or 3 filter-paper disks are positioned. The cylinder is seated over the filter

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unit so that it rests firmly on the base. Sediment is then quickly transferred into the cylinder through the top, followed by the teflon and rubber disks. The teflon and rubber disks can be placed above the sample in either order to obtain a leak-free pressure transfer, but placing the teflon disk below the rubber disk gives a cleaner seal than the reverse order shown in Figure 5-17. The piston is depressed as far as it will go into the cylinder, and the whole unit put in the press for squeezing. Pressure is applied gradually at first, and when the first drop of interstitial fluid is seen the syringe is seated in its hole in the base (effluent passage in Figure 5-17.) The squeezed-out liquid moves the plunger of the syringe back as the liquid is expelled, and there is minimum opportunity for evaporation. When the desired amount of liquid has been obtained, the syringe is removed and capped.

After extraction of the liquid the parts of the apparatus are rinsed with distilled water and (except for rubber parts) with acetone. The acetone helps dry the unit quickly in preparation for the next sample. The squeezing and washing operations together can be completed in 5 to 10 minutes."

Insert 5-45A

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Pore Water Extraction from core samples

The main expenditure in this type of sampling is the filter press. The current price of this piece of equipment can be obtained from Baroid Division, NL Industries, Houston, Texas. Current charges for cores obtained by wire-line, 2-inch diameter split barrel samplers are $30 to $50 per core. A section of core can be taken from the sampler, molded into the filter press, the fluid extracted and analyzed for chloride concentration, and measured for specific conductance in a half hour or less. Therefore, cost will depend on the investigators' salary or hourly billing rate.

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Advantages

1. Inexpensive
2. Pore water extract amenable to field chemical analysis.
3. Excellent vertical sampling if mud invasion into core sample is monitored, quantitative analytical results.
4. Samples can be obtained from almost any depth if wire line coring apparatus is used.
5. Qualitative use of pore water extract allows presence/absence determination.
6. Can be used with consolidated rock as well as unconsolidated sediment samples.

Disadvantages

1. Quantitative analysis require careful control during sample collecti
2. Interstitial water can drain from unconsolidated sand and gravel reducing volume of water sample that can be obtained.
3. Disposal of dyed drilling mud is a problem.
4. Core recovery in coarse sand and gravel can be difficult and time consuming.
5. Small sample volume available for chemical analysis.

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1. Gill, H.E., and others 1963. Evaluation of geologic and hydrologic data from the test drilling program at Island Beach State Park, New Jersey. New Jersey Division of Water Policy and Supply. Water Resources Circular 12.

2. Lusczynski, N.J. 1961. Filter-press method of extracting water samples for chloride analysis. U.S. Geological Survey Water Supply Paper 1544-A.

3. Manheim, F.T. 1966. A hydraulic squeezer for obtaining interstitial water from consolidated and unconsolidated sediment. U.S. Geological Survey Professional Paper 550-C, p. C256-C-261.

4. Swarzenski, W.V. 1959. Determination of chloride in water from core samples. American Association of Petroleum Geologists Bulletin 43 (8):1995-1998.

5. Walker, W.H., T.R. Peck, and W.D. Lembke 1972. Farm ground water nitrate pollution - A case study. American Society of Civil Engineers Annual and National Environmental Engineering Meeting October 16-22, 1972. Meeting Preprint 1842.

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5.3 Field Inspection – Caps.

~~The~~ field inspection is an extremely valuable tool in evaluating landfill sites. Although an inspection in the hands of a trained observer would produce more data, even the most unskilled person can identify the presence of leachate in springs, seeps, and streams by its color and odor. Frequently, vegetation that has been exposed to leachate can be found in a dead or dying state. The appearance, surface configuration, and drainage away from a landfill gives insight into the amount of infiltration of precipitation that might be taking place. A study of surface drainage, topography, and nearby wells enables the inspector to make an estimate of ground-water (and leachate) movement. Field observations increase in value when combined with geohydrologic information and other pertinent basic data contained in published reports and agency files.

In the following pages, it becomes evident that many of the techniques discussed can be combined with the field inspection to provide even more information. The degree of success is strictly within the ability of the inspector to interpret the situation and the amount of time available for the study.

In practice, the inspector should have in his possession at least a sketch map of the landfill site; a detailed map or areal photograph would be better. By walking around the ~~perimeter of the~~ landfill and ~~into the~~ surrounding acreage, and recording what is found, the overall picture is recorded ~~to~~ on the map, where it is more easily interpreted.

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Advantages

1. Can be carried out quickly and inexpensively.
2. Helps place the overall problem in perspective.
3. Establishes the extent of additional investigations which may be required.
4. When combined with a literature and available data survey, can be used by an experienced hydrologist to roughly establish the overall situation.
5. Provides an opportunity for first hand discussion with landfill operator and other personnel.

Disadvantages

1. Untrained inspector may overlook subtle but valuable data.
2. Findings are not always conclusive in detecting ground-water contaminated by leachate.
3. Provides no indication of changes in condition with time.
4. Provides little hard data.
5. Untrained inspector may be misled by visually impressive ~~by~~ but environmentally insignificant features. This could occur in either a positive or negative direction.

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The cost of a field inspection of a landfill can be quite variable, depending on the size of the operation and the complexity of the surrounding terraine. In this and each of the following sections, an estimate of the cost of carrying out the task described is given. This cost is based on an estimated daily rate for the personnel required to perform the task, an estimated length of time for the tasks based on an average situation and other related expenses such as lab fees and living expenses. In all cases, the estimated cost should be considered accurate only to about plus or minus fifty percent.

For a field inspection of an average (50 acre) landfill, one man at the hydrogeologist or engineer level would be required for 2 to 3 days at $200. per day. The estimated total cost is $600.

Hughes, G.M., R.A. Landon, and R.N. Farvolden. 1971. Hydrogeology of solid waste disposal sites in Northeastern Illinois. U.S. Environmental Protection Agency publication No SW-12d. 154 P.

5.31 Seeps

Small springs of discolored, malodorous leachate which are frequently found along the lower edges of many landfills are referred to as seeps. These may be the only visible indication of landfill leachate and, therefore, receive more than their share of attention. In fact, however, they represent only a very small fraction of the total leachate being generated by the landfill. The few gallons per minute visible in seeps are insignificant when compared with the hundreds and perhaps thousands of gallons of leachate seeping down unseen to the water table. However, as they are indicators of leachate, they deserve some consideration.

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Seeps may represent the intersection of the water table and the land surface, or they may be discharge of a small perched water body within the landfill. Sometimes a distinction between these two situations can be made by inspection. For example, if the land surrounding the landfill is dry, a seep discharging along the face of the refuse is not likely to represent the water table. A further and more definite way of distinguishing between the two situations is by installing a well point nearby and establishing the true water table position near the seep. This well point has the added advantage of permitting a sample of ground water to be collected and tested for leachate.

One value of seeps is in the collection of concentrated leachate samples. However, it should be kept in mind that it is possible that the seep may not be representative of the large volume of leachate generated in that particular landfill. In fact, the chemical characteristics of any leachate sample, regardless of its source, should not be considered representative of the total volume of leachate. Typically, landfill leachate has proven to be highly variable, both from location to location in a landfill, as well as from time to time at the same point. A second value of seeps is that substantial changes in seep locations or flow rates, or the sudden appearance of new ones, indicates a changing flow system within the landfill. The exact nature and causes of the change, however, must be investigated by other means.

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Advantages

1. Where present, definite indication of leachate generation.
2. Convenient point of collection for leachate sample.
3. Changes in flow rates or locations of seeps is indicative of interval landfill changes.

Disadvantages

1. May not indicate presence of contaminated ground water.
2. Chemical quality not necessarily representative of bulk of leachate in landfill.

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Examination of seeps would typically be included as a part of the field inspection and would not represent an additional expense.

5.3.2 Vegetation Stress

A significant impact produced by a landfill on the surrounding area is stress and possible destruction of vegetation. Stressed species may include agricultural crops, stands of trees, and marsh or meadow vegetation.

In marsh environments subject to leachate discharge, the vegetation is an excellent ~~environmental~~ monitor to assess ecological stress on the total system. In addition to being stationary and sensitive, marsh vegetation can be studied for signs of stress using aerial remote sensing techniques as well as directly by the botanist in the field. Crops and trees growing in areas of deeper water table than is associated with the marsh environment are more likely to be stressed by landfill generated gasses than by leachate. Various types of agricultural crops as well as fruit orchards, have been destroyed by migrating gasses generated within a nearby landfill. Preliminary stresses placed on these species, prior to their actual destruction, are often detectable by the botanist and by aerial remote sensing.

While identifying the precise cause and mechanisms of stress may be prohibitively costly, it may be possible to relate the stress to a general cause which may in turn be related

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to the presence of the landfill. Mapping the extent of stressed vegetation is an excellent indication of the extent of the total impact of a landfill on its surrounding environment. Also, early detection of stress sometimes permits the opportunity to institute corrective measures in time to prevent irreparable damage.

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A cursory look at vegetation stress would be included in the field inspection taks and would not represent an additional expense. A detailed survey of vegetation stress, including an assessment of probable cause would require 1 to 2 days of field work by a botanist plus some laboratory work. The estimated cost of such a survey is $1,000.

If vegetation stress is to be used for monitoring, or if specific recommendations regarding the saving or replacement of stressed species, the required program might cost between $10,000 and $100,000, depending on the extent, complexity and goals of the programs.

Geraghty & Miller, Inc. 1973. Environmental Feasibility, Proposed Silver Sands State Park, Milford, Conn. Project Bi-T-55A. Report to State of Conn., Public Works Dept. Dept. of Environmental Protection.

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Advantages

1. When found, good indicator of contamination by leachate or gas.
2. Mapping extent of stressed vegetation gives good indication of the limits and source of contamination.
3. Stressed vegetation can be mapped remotely i.e. aerial photographic methods, thus allowing wide coverage in a short period of time.
4. Stress changes provide a good monitoring device. This effect may be enhanced by actually planting selected species in by areas and watching the results.

Disadvantages

1. Evidence of stressed vegetation, especially in early stages not always evident except to trained botanist.
2. Some species of plants are more resistant to to effects of contamination than others. This may be an advantage in multi-special area as an indicator of increasing, or decreasing contamination or by producing a clue as to stress cause.
3. Stress may be caused by many factors, some unrelated to the presence of the landfill. Determination of the responsible factor or factors is usually extremely difficult even for the botonist.

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Disadvantages, continued

4. Stress will not occur unless physical or chemical change occurs at the surface or within the ~~vast~~ vadose zone. Therefore, provides no indication of sub-surface problems.

5.3.3 Specific Conductance and Temperature Probes

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Two physical characteristics of ground water which can be readily measured in the field are specific conductance and temperature. Since landfill leachate generally has substantially higher temperature and specific conductance than natural fresh ground water, the presence of leachate often can be determined using these two characteristics.

Typically, in situ measurements of ground-water characteristics would be made by lowering a remote-sensing probe into a well and recording the results from instrumentation on the surface. In areas of high water table, however, the measurements can be made without installing a well. The method involves the use of a self-contained conductance-temperature probe. Construction details of such a device are shown in Figure 5-18.

The probe can be pushed directly into the ground where sediments are soft, or inserted into a small-diameter hand augured hole where the ground is harder. When the probe is below the water table, the outside tube, which has protected the perforations from clogging during insertion, is retracted, allowing the ground water to flow into the tube. Specific conductance and temperature of the ground water can then be recorded. After removal from the ground the perforated end of the probe is washed in clean water.

Under good conditions, a two man crew can carry all necessary equipment into the field and make a series of probe measurements over a typical landfill site in 2 or 3 days. In addition, measurements can be made easily in swampy areas not accessable to drilling rigs or resistivy survey crews.

Figure 5-18. Specific Conductance - Temperature Probe.

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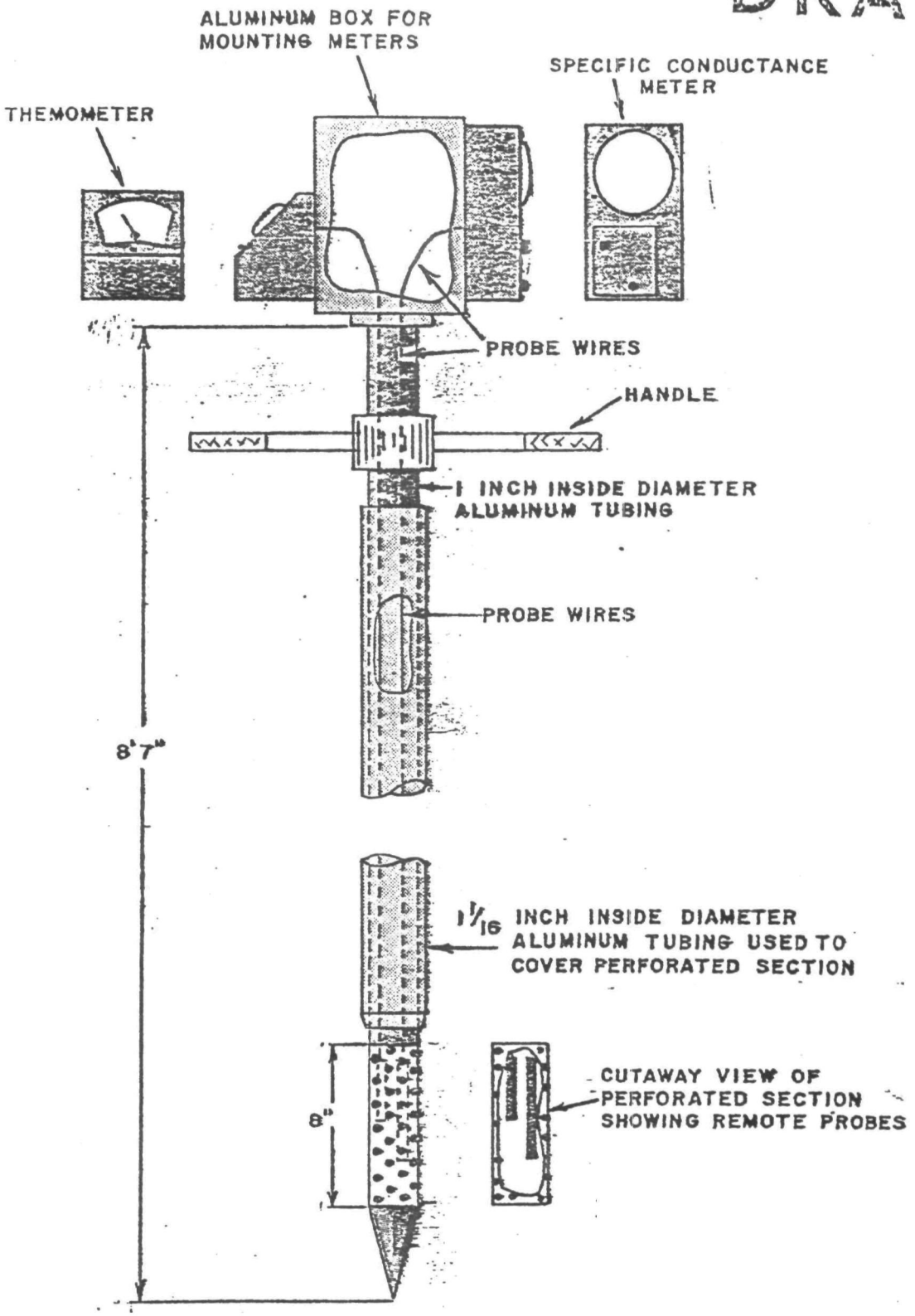

The cost of a ground-water conductance and temperature survey using a probe such as the one described above, and assuming the surface conditions were such that this type of survey would be practical, would be about $900. This estimate is based on the cost of a hydrogeologist or equivalent and a helper ($60. per day) for 3 days. The survey may require a day or two more or less depending on the size of the site to be investigated and its accessibility.

Geraghty & Miller, Inc. 1973. Environmental Feasibility, Proposed Silver Sands State Park, Milford, Conn. Project Bi-T-55A. Report to State of Conn., Public Works Dept. Dept. of Environmental Protection.

5.3.4 Electrical Earth Resistivity

An electrical earth resistivity survey can be used to define subsurface geology and the extent of leachate contamination of ground water. The results of a resistivity survey can be used with a minimal amount of direct sampling as a basis for decisions on the necessity of remedial action, or it can be used as a preliminary investigation from which a detailed drilling and sampling program is designed. Since resistivity is an indirect method, however, and is subject to possible error in interpretation, it would be unwise to base final conclusions on resistivity results alone.

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The earth resistivity method depends upon the conduction of electric current through the subsurface materials. The magnitude and distribution of the current flow is a function of the effective resistivity (or its reciprocal, conductivity) of the subsurface material. The effective resistivity of saturated materials is dependent upon moisture in interstices and pores because the vast majority of the constituent minerals are poor-conductors. The pore spaces that contain water also contain some dissolved salts, and it is these ionic solutions that allow the passage of current from the surface into the underlying material. It has been found that the resistivity of materials such as moist clays and silts is low; but, in dry, loose soils, sand and gravel, or sand and gravel saturated with high-quality water, the resistivity is high. The electrical resistivity of a material is a function of the actual resistance of the material, and the length of the current flow. Because earth materials are not homogeneous, the measured resistivity is actually termed apparent resistivity and is defined as the weighted average of the actual resistivities of the individual subsurface materials or strata within the depth of penetration of the resistivity measurement.

To measure earth resistivity, a known current is introduced into the earth through two current electrodes and the resulting potential drop is measured between a second pair of potential electrodes. If the electrodes are arranged in a straight line and the separations are increased at constant increments,

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it is possible to make inferences about the relations of variations in apparent resistivity, depth of penetration, and electrode spacing. Various procedures have been developed to interpret resistivity data. The procedures are grouped into two basic types: Theoretical and empirical. In using the theoretical method, the field data are plotted, describing a curve which is compared with sets of master curves developed for numbers of resistivity layers with definite ratios of resistivity and thickness. With this method, the values of resistivity for each geologic unit as well as thicknesses and depths can be determined. An example of an empirical interpretation is shown on Figure 5-19. With this method the apparent resistivity and accumulated apparent resistivity values are plotted. The first curve indicates the type of material and the second curve shows the depth of the interface between layers.

Use of the resistivity method to define a Leachate Plume relies as the fact that the conductivity of the ground water is inversely proportional to the resistivity measured in a section of earth containing that ground water. Since the conductivity of landfill leachate is generally much higher than that of natural fresh ground water, a sharp decrease in apparent resistivity will occur where leachate is included in the measured section. Thus, by running series of resistivity measurements at the appropriate depth on a grid over a landfill site, it is possible to define the lateral extent of the leachate plume by contouring the resistivity values obtained. The results of a resistivity survey at a landfill site are shown on Figure 5-20.

Figure 5-19. Interpretation of resistivity measurments using the empirical Cumulative method.

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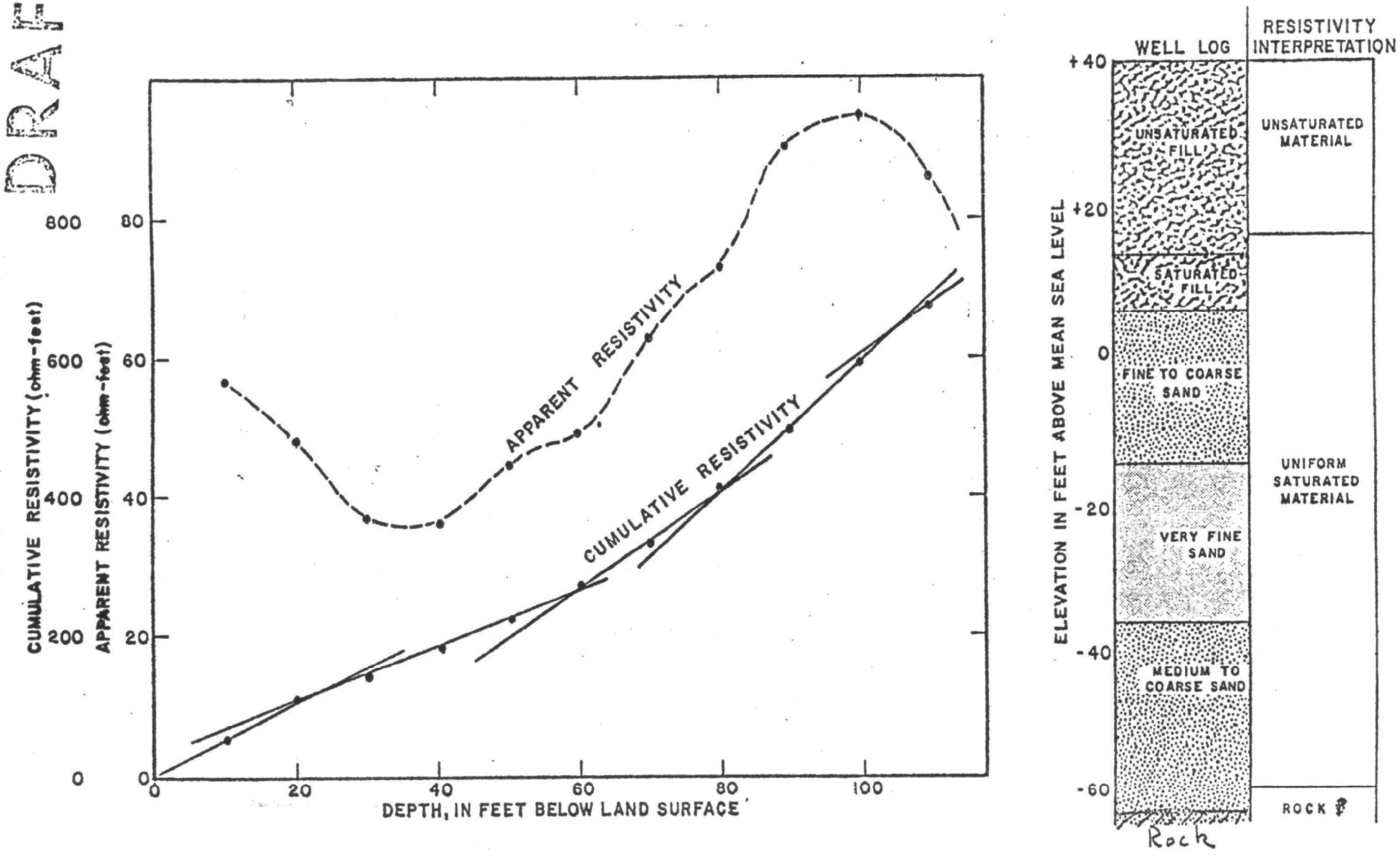

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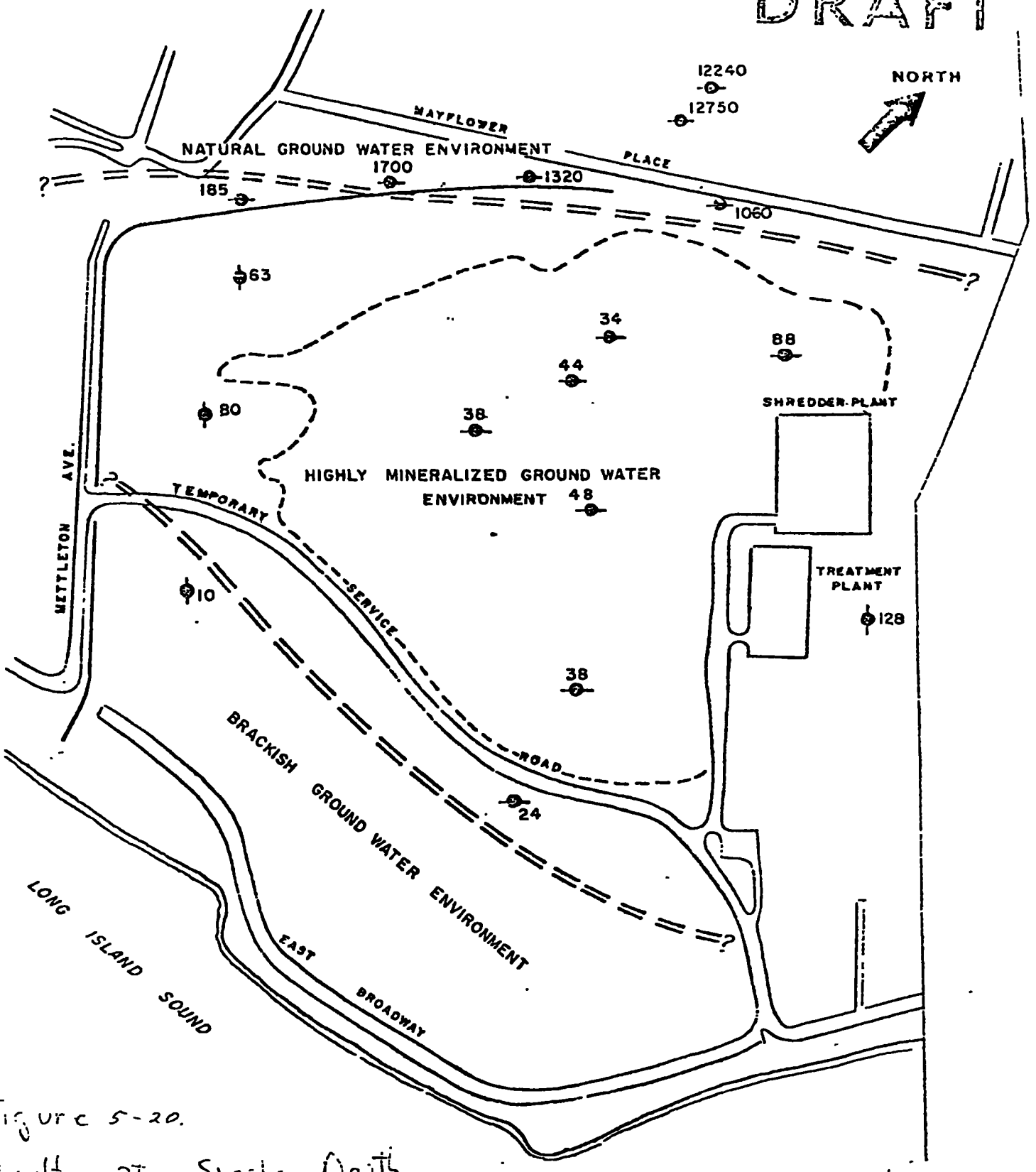

Figure 5-20.

Results of Single Depth Resistivity Survey at a Connecticut Landfill site

Legend

⊖ resistivity Station (showing direction of electrode spread)

38 Resistivity Values in ohm-ft.

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Advantages

1. Can define subsurface geology and contaminated water bodies much faster and cheaper than drilling.
2. Can be used to greatly reduce the number of sampling wells required.
3. Surveys can be duplicated periodically to provide monitoring data.

Disadvantages

1. Indirect method-requires some substantiation by drilling.
2. Experienced operator is usually necessary to obtain useful data.
3. Many natural and man made field conditions preclude resistivity surveys.
4. Data interpretation in complex situation is often questionable.

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The cost of a resistivity survey would be essentially the same as for a seismic survey ($1,800. for a typical landfill site) and the same qualifications would apply.

Cartwright, K., and M.R. McComas. 1968. Geophysical surveys in the vicinity of sanitary landfills. Ground Water. 6 (5):23.

Stollar, R.L. and P. Roux. 1975. Earth resistivity surveys - a method for defining ground-water contamination. Ground Water. 13 (2): 145-150.

Parasnis, D.S. 1962. Principals of Applied Geophysics. John Wiley & Sons, New York.

Geraghty & Miller, Inc. 1973. Environmental Feasibility, Proposed Silver Sands State Park, Milford, Conn. Porject Bi-T-55A. Report to State of Conn., Public Works Dept. Dept. of Environmental Protection.

5.3.5 Seismic Surveys

Seismic surveys are used to determine the depth to bedrock and the thickness of the materials overlying the bedrock. The refraction method of seismic exploration utilizes the principle that energy waves can be propagated through earth materials. The velocity of propagation is governed by the elastic properties of the earth materials through which the waves are travelling. These elastic waves can be timed from their initiation to a known distance from the energy source to determine their velocity. With known velocities and distances, depths to the various geologic interfaces can be calculated.

The seismic reflection method of geophysical surveying may also be used. This system, in which the energy wave is reflected from the different geologic horizons, can usually penetrate greater depths than the refraction method.

Where well data are available, correlations are made between the results of the seismic survey and existing information for more refined interpretations. Where well information is not available, evaluation of seismic data is based on the interpretation of the geologic environment and experience in geophysics.

The techniques of operation in the field depend on the various applications of the seismic refraction method. In order to determine depths and seismic velocities of various materials, the reverse profile method is used. A reverse profile is one in which the most distant energy source and the geophone, which is the receiving unit, are interchanged after recording a profile and a second profile is then recorded. The energy source is a hammer blow on a steel plate or an explosive charge. With a single geophone seismic unit, a seismic profile is conducted by implanting the geophone firmly in the ground and moving the impact point away from the geophone at measured distances. For a multi-geophone unit, the geophones are placed at selected distance intervals along a line, and a single energy source, usually an explosive device, is activated. By observing the energy arrivals for different separations between the impact point and the receiver or receivers, a travel-time curve can be constructed illustrating the energy travel-time with distance.

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A seismic survey requires a trained operator and an experienced geophysicist to interpret the data. The complexity of the data reduction process, generally requires the use of a computer. For these reasons, seismic surveys should be contracted to a firm providing geophysical services.

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Advantages

1. Can provide subsurface geologic information much faster and cheaper than drilling.
2. Can be used to extend geologic data over broad areas on a limited budget.
3. Can be used in certain areas where access for a drilling rig would be difficult.

Disadvantages

1. Being an indirect method, it requires more direct substantiation such as drilling.
2. In complex geologic formations, interpretation is difficult and substantial errors may occur.
3. Requires a trained person and computer access to reduce and interpret data.
4. Subject to noise interference in many field situations.

The estimated cost of a seismic survey for a typical landfill site is $1,800, based on two days of field work for the seismic crew and data reduction and interpretation by a geophysist. This would be a typical survey to define subsurface geology in the area immediately surrounding a 50 acre landfill. For surveys encompassing substantially larger areas, the cost would increase proportionately. If access were difficult, if areas had to be cleared of bush for example, the cost of this task would have to be added to the cost of the actual survey.

Parasnis, D.S. 1962. Principals of Applied Geophysics. John Wiley & Sons, New York.

Anon. 1972. Ground Water and Wells. Pub. by Johnson Division, Universal Oil Products Co., St. Paul, Minn. 440 P.

Geraghty & Miller, Inc. 1973. Environmental Feasibility, Proposed Silver Sands State Park, Milford, Conn. Project Bi-T-55A. Report to State of Conn., Public Works Dept. Dept. of Environmental Protection.

5.4 SURFACE WATER QUALITY MEASUREMENTS

Surface water bodies such as ponds or streams which are in close proximity to landfills often have an orange color and an oily film on their surface. These obviously polluted water bodies are discharge points for contaminated ground water which originate within the landfill. Location of these discharge points on a topographic map of the landfill site will often help provide a reasonable preliminary picture of the ground-water flow patterns.

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Where surface water bodies are large or rapidly flowing, dilution of leachate as it discharges is often sufficient to prevent detection by visual inspection. In such cases, water samples would be taken and analyzed to establish the presence of typical leachate constituents.

Prior to the collection of surface water samples, a specific conductance, ~~Ph~~ pH, Eh or dissolved oxygen survey, using portable instruments to make in site measurements, should be conducted. Such a survey can provide much useful information itself, or at least indicate the locations from which surface water samples should be taken. The importance of an analysis of surface water quality at a landfill site is twofold; first, determination of leachage discharge areas is important in establishing an overall hydrologic picture, and second, surface water quality degredation is an important component of overall environmental degredation and should be carefully examined. Also in a full investigation of surface water bodies near a landfill, the native biota should be studied for leachate effects.

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Advantages

1. Useful in locating leachate discharge points.
2. Can be a quick and inexpensive means of estimating environmental impact of landfill.

Disadvantages

1. Detailed analysis of water samples can be fairly expensive.
2. Surface water may be subject to contamination from other sources not defined.
3. Dilution may be too great to provide useful information.

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The estimated cost of a surface water quality survey as described above, assuming significantly complex surface water bodies exist, is $300. This is based on 1 to 2 days work for a field hydrologist or technician at a daily rate of $150. It is assumed that this program would be part of a more extensive investigation and that analysis of the results of the surface water quality survey would be covered under a more general data analysis phase.

Geraghty & Miller, Inc. 1973. Environmental Feasibility, Proposed Silver Sands State Park, Milford, Conn. Project Bi-T-55A. Report to State of Conn., Public Works Dept. Dept. of Environmental Protection.

Landfill Gas Measurement - Landfill gasses, particularly carbon dioxide (CO_2) and methane (CH_4), can present serious problems at landfill sites and their concentrations and movement should always be investigated. Gas related problems include explosion vegetation destruction, and ground-water pollution. Since generation of carbon dioxide, methane and other gases is the natural result of organic decomposition, all landfills will produce these gases. The questions to be resolved are the direction, distance, and rate of movement of the gases prior to discharge to the atmosphere. The answers to these questions will establish the location and design of gas venting systems should they be necessary.

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At least two methods of gas measurement are available; collection of a gas sample for laboratory analysis and in-situ measurement of the explosive potential of confined gas. Sample collection or explosive potential measurement of gasses in the subsurface sediment is by specially designed gas probes, an example of which is shown in Figure 5-21. A gas sampling bottle or measuring instrument is attached to the upper end of the probe and evacuated. Gas from beneath the landfill then flows through the probe to be collected or measured. Multidepth probes may be installed as shown in Figure 5-23. In addition to landfill installations, probes should be installed in natural sediments around the landfill to establish the lateral migration patterns of the gasses. All enclosed spaces near the landfill, such as basements, manholes, etc, should be tested for accumulation of explosive gas. Examples of a measuring device and sampling bottle are shown in Figure 5-24.

The potential for methane recovery at a major landfill should be explored (Ref 7) The potential revenue from this resource may offset the cost of the investigation and pollution abatement and monitoring systems.

Problems associated with gas migration and buildup at and near a landfill site may be alleviated by the installation of gas vents and/or gas barriers. Typical gas vents employed in landfills are shown in Figure 5-25 Gas barriers would either be put in place prior to landfilling where the landfill will abutt a natural permeable face such as the vertical wall of a gravel pit, or, in some cases, clay slurry trench may be constructed after the landfill is completed if conditions indicated a shallow barrier would be effective.

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Figure 5-11 Probe used to measure the concentration of landfill gasses.

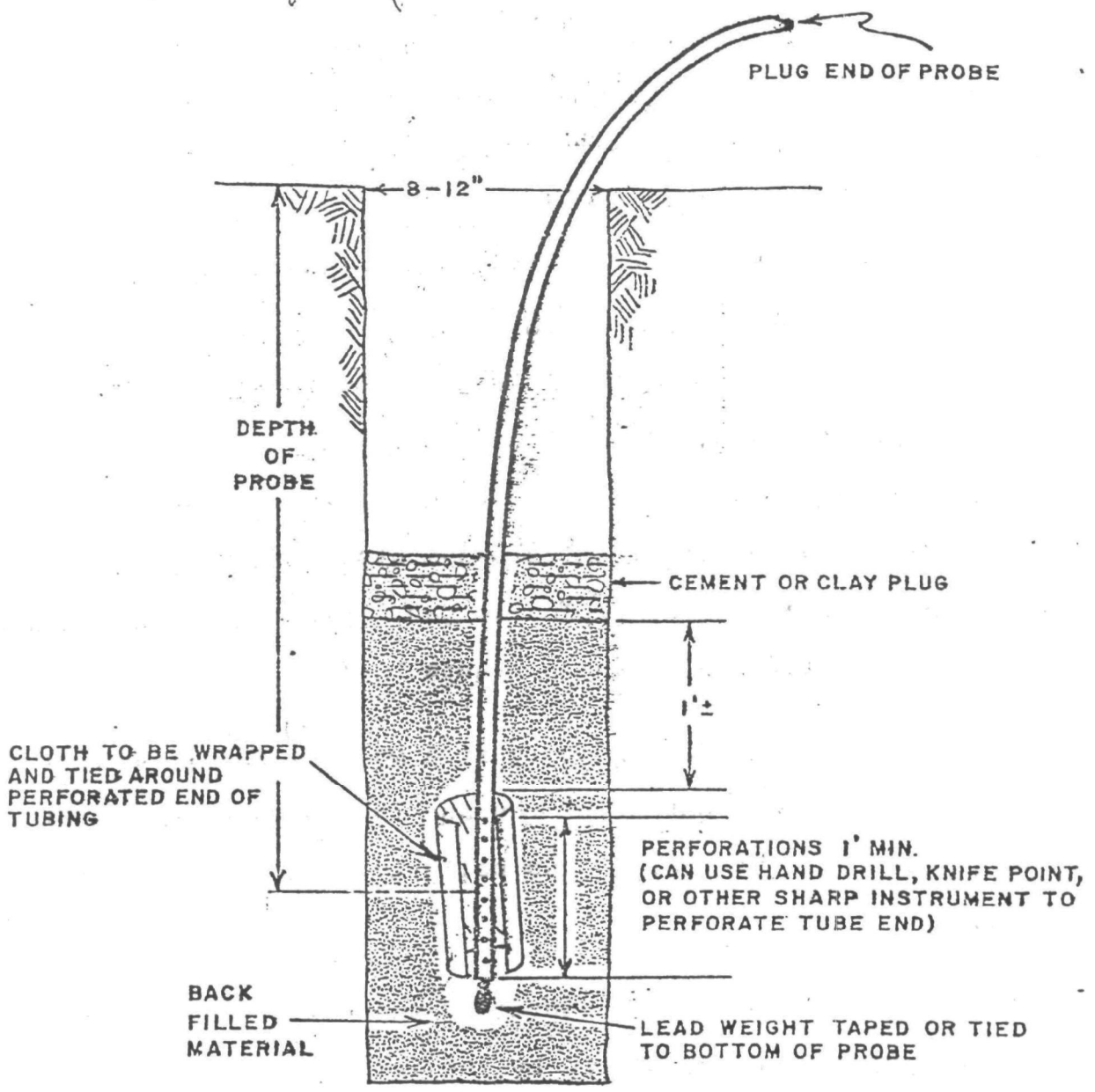

Figure 5-22. Multidepth probe for the measurement of ~~landfill~~ gas concentration in a vertical section of a landfill

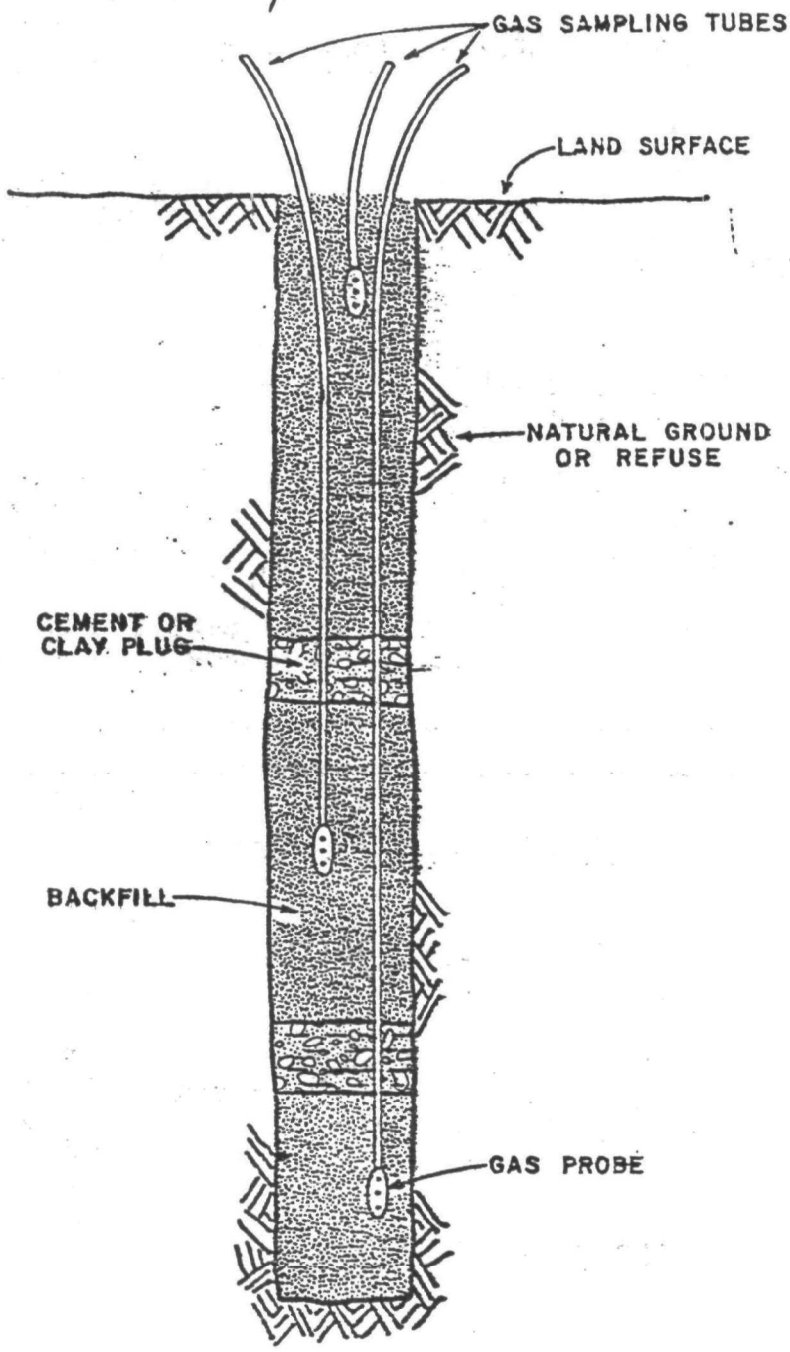

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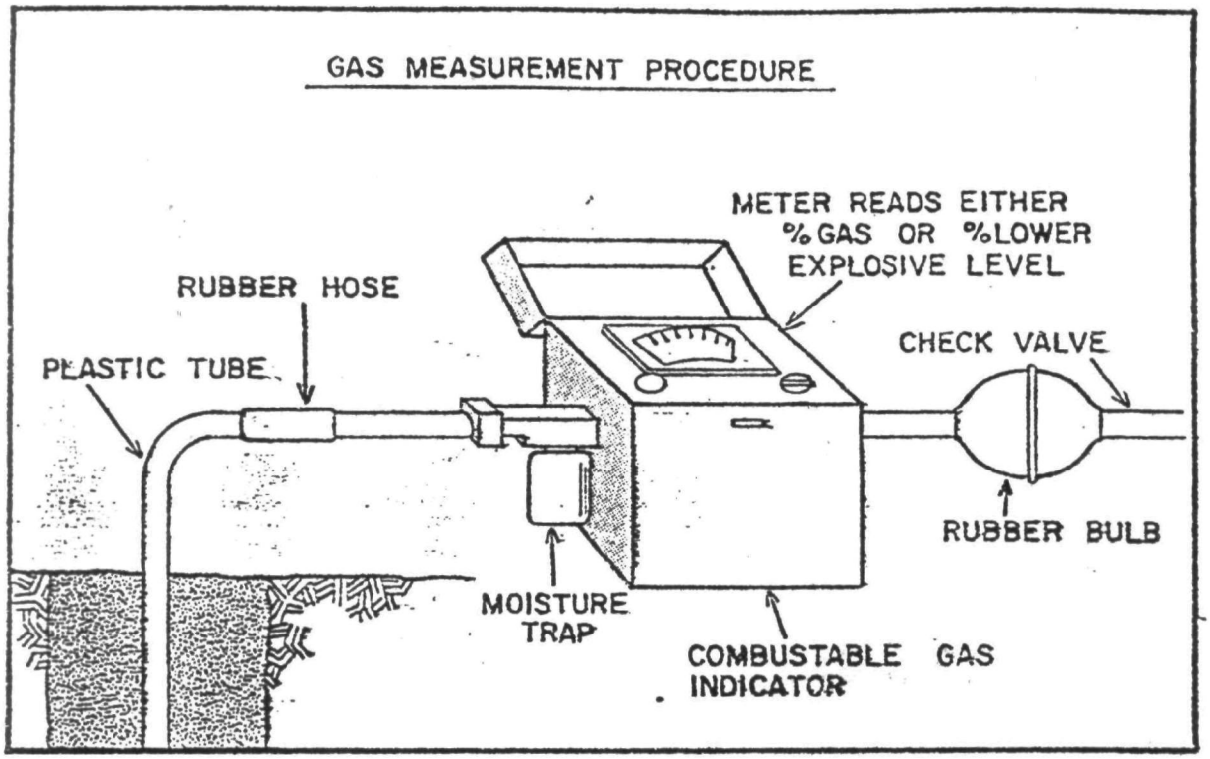
Figure 5-24. Landfill gas measurement and sampling procedures.



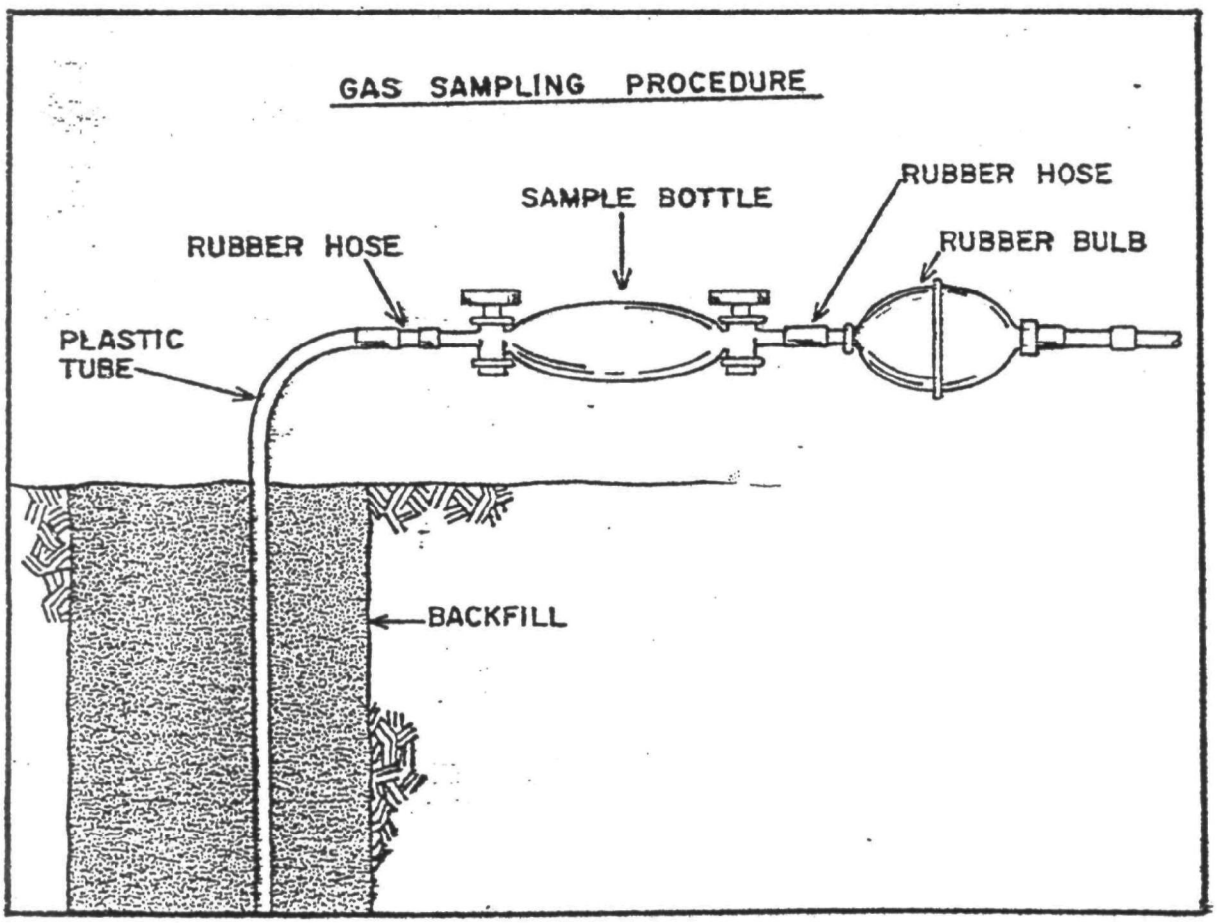


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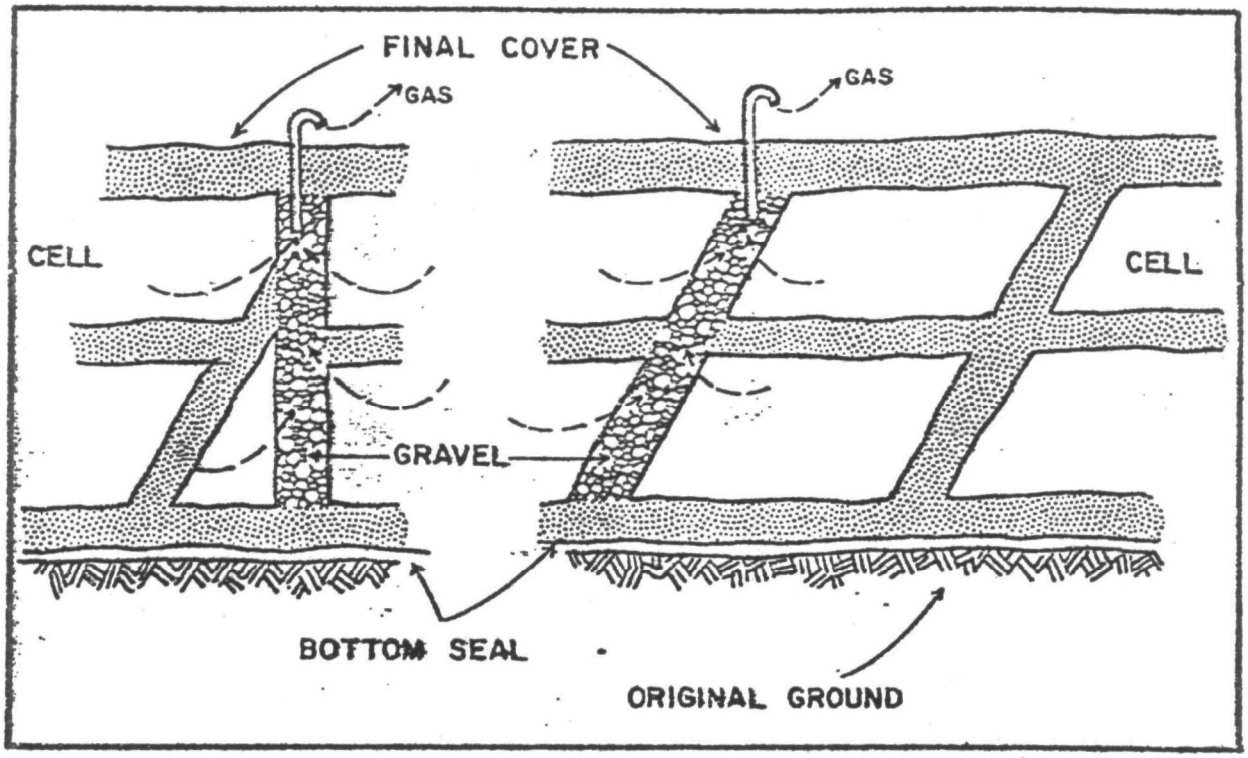
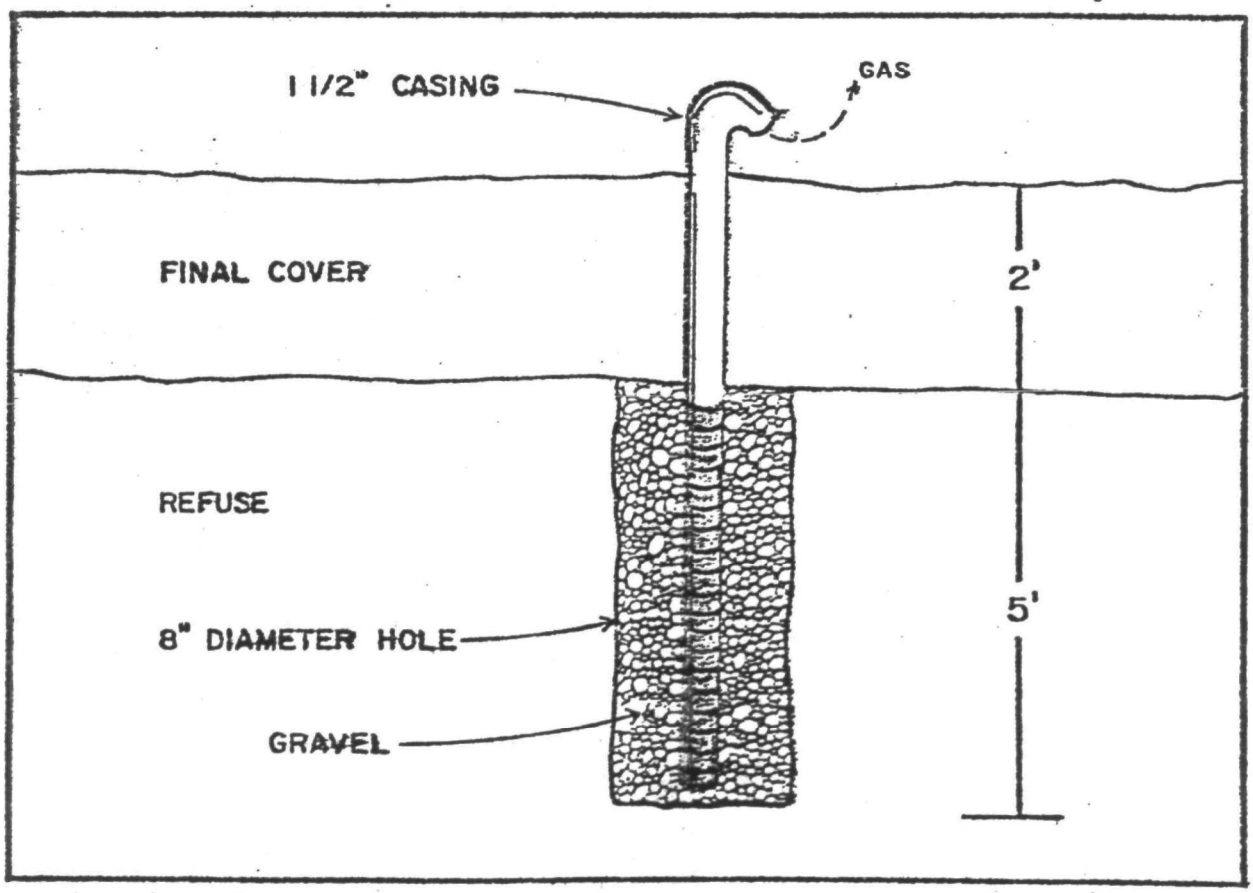
Figure 5-25. Landfill gas venting systems. DRAFT





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Advantages

1. Detection of methane accumulation can prevent explosion hazard to personnel and property.

2. Establishes the need for special gas venting system.

3. Provides a clue as to possible cause of vegetation stress.

Disadvantages

1. Proper analysis of gas measurement data is complex and would require experienced personnel.

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The cost of a landfill gas survey would be about $900, including 2 or 3 days of field measurements by a hydrologist, engineer or equivalent and laboratory analysis of several samples. Detailed analysis of the results of the survey and remedial recommendations are not included and the complexity of such a task and thus its cost would depend on the results of the initial survey.

Merz, R.C. Determination of the Quantity and Quality of Gases Produced During Refuse Decomposition. University of Southern California, Los Angeles. Engineering Center Quarterly Reports. U.S.C.E. Report 83-3, Sept. 30, 1962; 86-6, July 30, 1963; 87-7; Sept. 30, 1963; 89-8, Dec. 31, 1963.

Merz, R.C. and R. Stone. Gas Production in a Sanitary Landfill. Public Works, 95:84 February, 1964.

Engineering-Science, Inc. In-situ investigation of movements of gases produced from decomposing refuse. Final report prepared for California State Water Quality Control Board. Pub. No. 35, April, 1967.

5.4.2 Aerial Photography

Aerial photography has several important uses in landfill studies. In its simplest form, an aerial photograph, whether black and white or color, will show the landfill and drainage away from it. For large areas, remote sensing of vegetation stress using aerial photography map be a justifiable undertaking. Advances stress may be visible on color photography and less advances stress may be enhanced and distinguished using infrared photography. Another

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method which has been used in landfill investigations is multispectral aerial photographs.

Multispectral photography uses special equipment to determine subtle differences in light reflected at different wave lengths for stressed and unstressed species. Photographic filters that will enhance this difference are used, and several images of the same area are made at the same time using a multi-lens camera and the selected filters. Differences between stressed and unstressed vegetation are further enhanced by projecting the images through different color filters and superimposing on a projector screen.

In addition to vegetation stress, aerial photography is frequently useful in constructing contour and location maps of landfill sites. Accurate contour maps of the landfill surface are used in determining hydrologic characteristics of the landfill. Stereo color photography is used to construct and up-date topographic maps of the active landfill sites as the surface changes. Bench marks, wells, and other sampling points.

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Advantages

1. Frequently can detect stressed vegetation evidence of contamination.
2. Can be used to prepare contour maps relatively inexpensively. Also provides certain geologic information.
3. Much less costly than a detailed ground survey of vegetation stress.
4. Yearly photos can provide unbiased and indesputable evidence of surface changes; eg. landfill configuration, vegetation condition, surface water body location.
5. Can be used to precisely locate on a map key points on the landfill site such as wells or Seismic Stations.
6. Enables persons to quickly grasp the situation without visiting the site. (other consultants, veg, people, etc.)

Disadvantages

1. Availability of aerial photographs and photographic services sometimes limited.
2. Indicates little about sub-surface conditions.
3. Indicates little as to precise causes of detected surface changes.
4. Requires trained interpreter to evaluate results.

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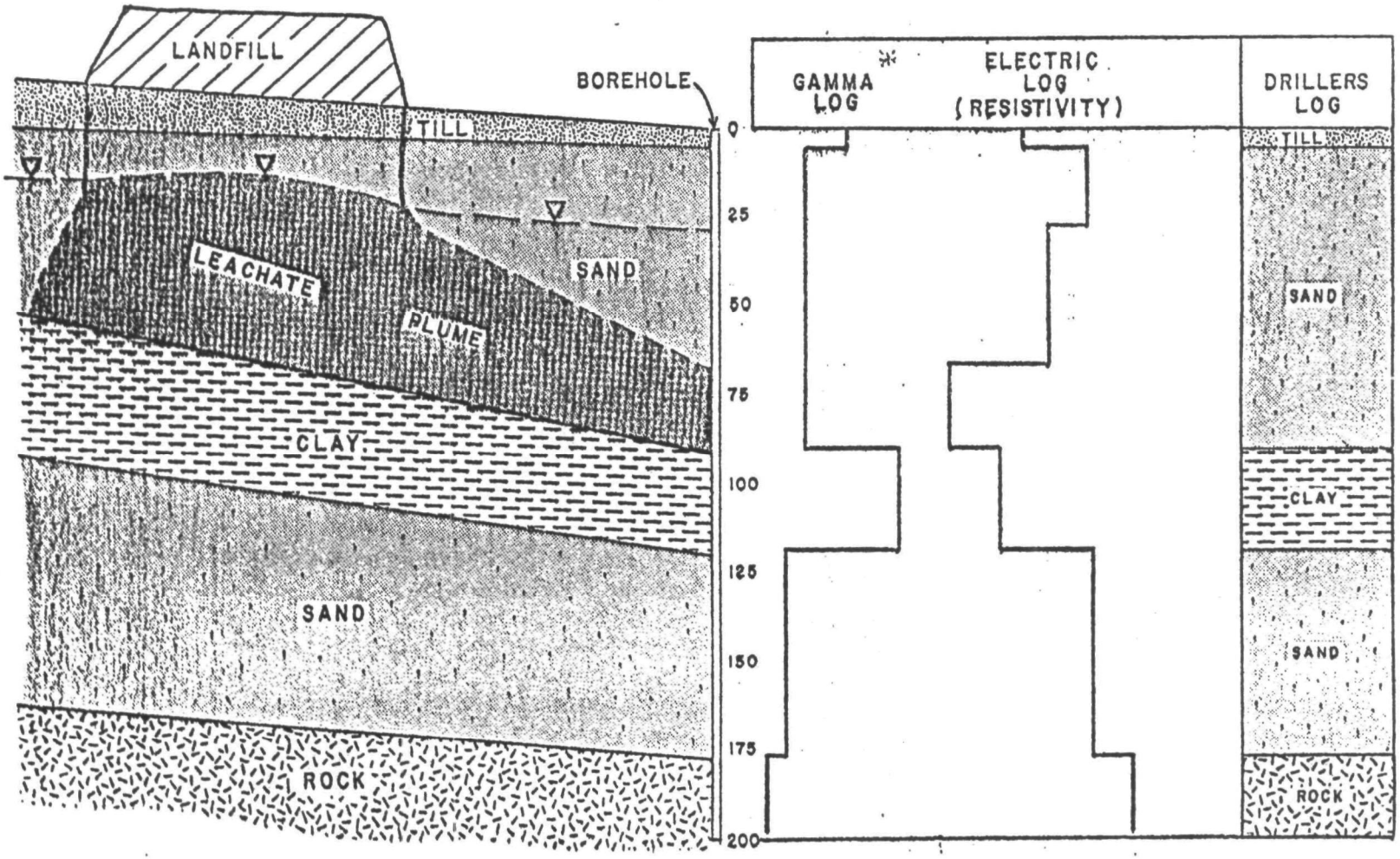
Aerial photographs of a landfill site may be readily available from a local firm or it may be necessary to have the site flown. Available photographs generally cost about $10. to $30. and having black and white photographs taken generally costs about $100. to $300. Special photography, such as color infrared or multispectral photography, with the necessary interpretation will cost up to about $2,000. For this sum, a topographic map and a map showing vegitation stress along with a report of the result of the photo interpretation would be included.

Geraghty & Miller, Inc. 1973. Environmental Feasibility, Proposed Silver Sands State Park, Milford, Conn. Project Bi-T-55A. Report to State of Conn., Public Works Dept. Dept. of Environmental Protection.

5.4.3 Geophysical well logging - this method provides indirect evidence of sub-surface formations that indicate the relative permeabilities as well as the depths of the formations. The most common borehole geophysical operation is electric logging. An electric log consists of a record of the apparent resistivities of the sub-surface formations and the spontaneous potentials generated in the borehole, both plotted in terms of depth below the ground surface. The measurements of apparent resistivity and spontaneous potential are related to the electrical conductivity of the sediments, which is a function of the size of the grains. Thus, fine-grained sediments containing silt and clay will have a lower resistivity than clean, coarse sand and gravel. In addition, a leachate plume may be detectable by an electric log as illustrated schematically in Figure 5-25.

† fig 5-25 Detection of a leachate plume using an electric well log.



* gamma log is included because it indicates the leachate plume is not actually a clay layer.

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Electric well logs can be run only in uncased boreholes.

Gamma-ray logging is a borehole geophysical procedure based on measuring the natural gamma-ray radiation from certain radioactive elements that occur in varying amounts in sub-surface formations. The log is a diagram showing the relative emission of gamma-rays, measured in counts per second, plotted against depth below land surface. Because some formations contain a higher concentration of radioactive elements than others, formation changes with depth can often be accurately determined. For example, clay and shale contain more radioactive elements, such as uranium, potassium and thorium, than does sand or sandstone. In addition to interfaces between two layers of different materials, the relative amount of silt and clay in the formations can be determined by the inflections on the gamma-ray log. Unlike electric logs, gamma-ray logs can be run in cased wells.

Geophysical well logs are used to supplement the drillers' and geologists' logs of the materials penetrated by the borehole. An example of the comparison between a geologic, electric and gamma-ray logs is shown in Figure 5-26. An accurate evaluation of the sub-surface geology at a landfill site is essential to the determination of the direction and rate of movement of leachate from the landfill and the contaminant attenuation capacity of the materials through which the leachate must move.

Geophysical well logging generally is applicable only to those landfill investigations which include test drilling, and is therefore not an independent tool. Gamma-ray logging can be used,

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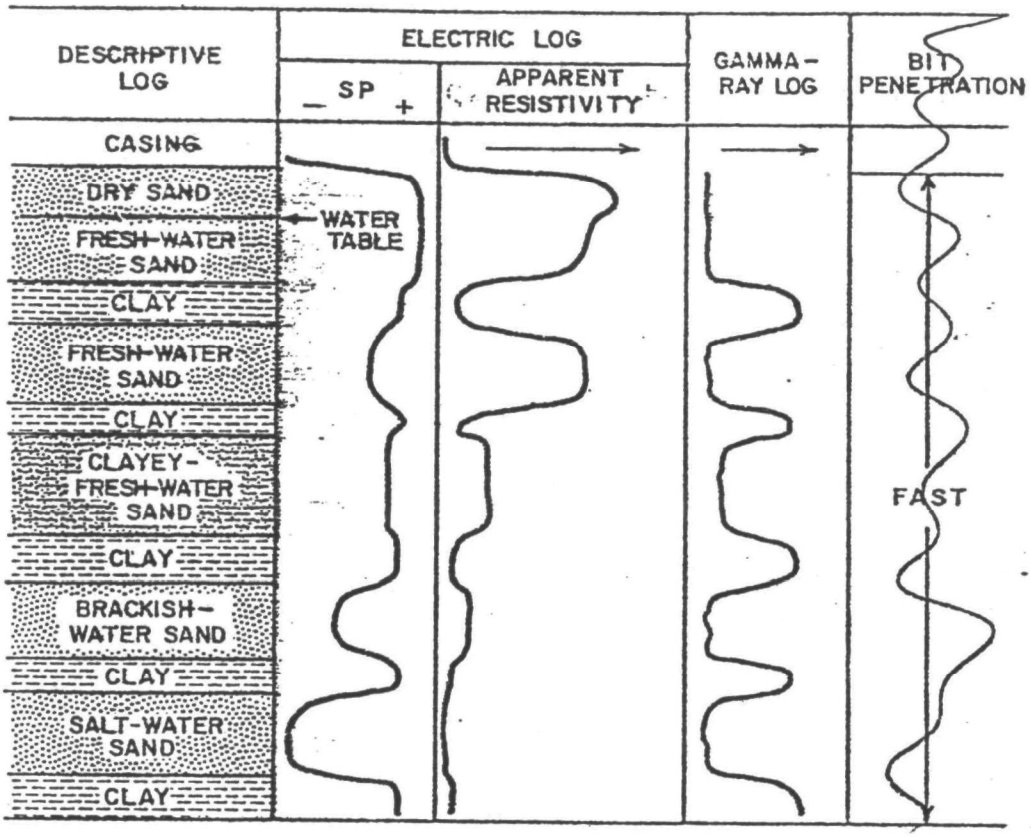
Figure 5.24. Comparison of geologic, electric and gamma-ray logs.



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however, to gain some understanding of the sub-surface geology at a landfill site from existing wells which may be in the vicinity and for which no geologic logs are available.

Since geological well logging requires specialized equipment and trained operators, the task would be preferred by a firm offering geo~~graphical~~ physical services. In some cases, larger well drilling companies are equipped to provide such services, in which case the logging operation can be included as part of the well drilling operation.

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Advantages	Disadvantages
1. Provides back-up data to substantial drillers and geologists log of borehole	1. Requires special equipment and trained operators and thus adds considerable expense.
2. Allows a more accurate determination of depth to formation change than may be achieved with routing sampling.	
3. Allows a geological log to be constructed for an existing well that was not logged when drilled.	
4. May be useful in locating top and bottom of a contaminated ground-water body. Selective log	

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The cost of a geophysical well logging would be $300 to $500 per day depending on the complexity of the equipment and size of the necessary crew. Normally five or six shallow wells or two or three deep wells (several hundred feet) can be logged in a day. Interpretation of the logs by a geophysicist would cost about $400 for a typical landfill situation of six 100 to 200 foot deep wells.

Campbell, M.D., and J.H. Lehr. 1973. Water Well Technology. McGraw-Hill Book Co. New York. 681 P.

Anon. 1972. Ground Water and Wells. Pub. by Johnson Division, Universal Oil Products Co., St. Paul, M inn. 440 P.

Parasnis, D.S. 1962. Principals of Applied Geophysics. John Wiley & Sons, New York.

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Water-Balance Simplified

The water-balance or water-budget method is the measurement of the continuity of flow of water for any given time interval and can be applied to any drainage basin.(1) In this case, the drainage basin being considered is a hypothetical landfill ~~and~~ together with the land immediately surrounding it. The purposes of establishing a water-balance for a landfill are to determine the rate of leachate generation and to establish which of the available pollution abatement procedures would be most effective.

The calculation of the water-balance for a landfill requires the measurement of numerous physical parameters and can be a relatively difficult and expensive task. For most landfill investigation and monitoring work, however, a reasonable approximation of the magnitude of the various water-balance components will be sufficient. Methods of estimating each of these components, using as much available information and as few field measurements as possible, are given below.

The seven principal water-balance components of a hypothetical landfill are shown by arrows, on Figure 5-27. These are; precipitation and irrigation, surface runoff onto the landfill, surface runoff from the landfill, evapotranspiration, underflow, infiltration, and leachate. Also, given on Figure 5-27 are references to the table or figure in this section which can be used to estimate the magnitude of the components and the relationships between them.

Precipitation and Irrigation

Figure 5-28 shows average annual precipitation for various regions across the United States. Such a map, however, can be considered only generally accurate. Significant variations in

Figure 5-29 Landfill Water balance simplified.

COMPONENT VALUES

A - Figure 2
B - Table 1
C - Table 1
D - Table 2
E - Figure 5
F = A+B-(C+D)
G = F+E

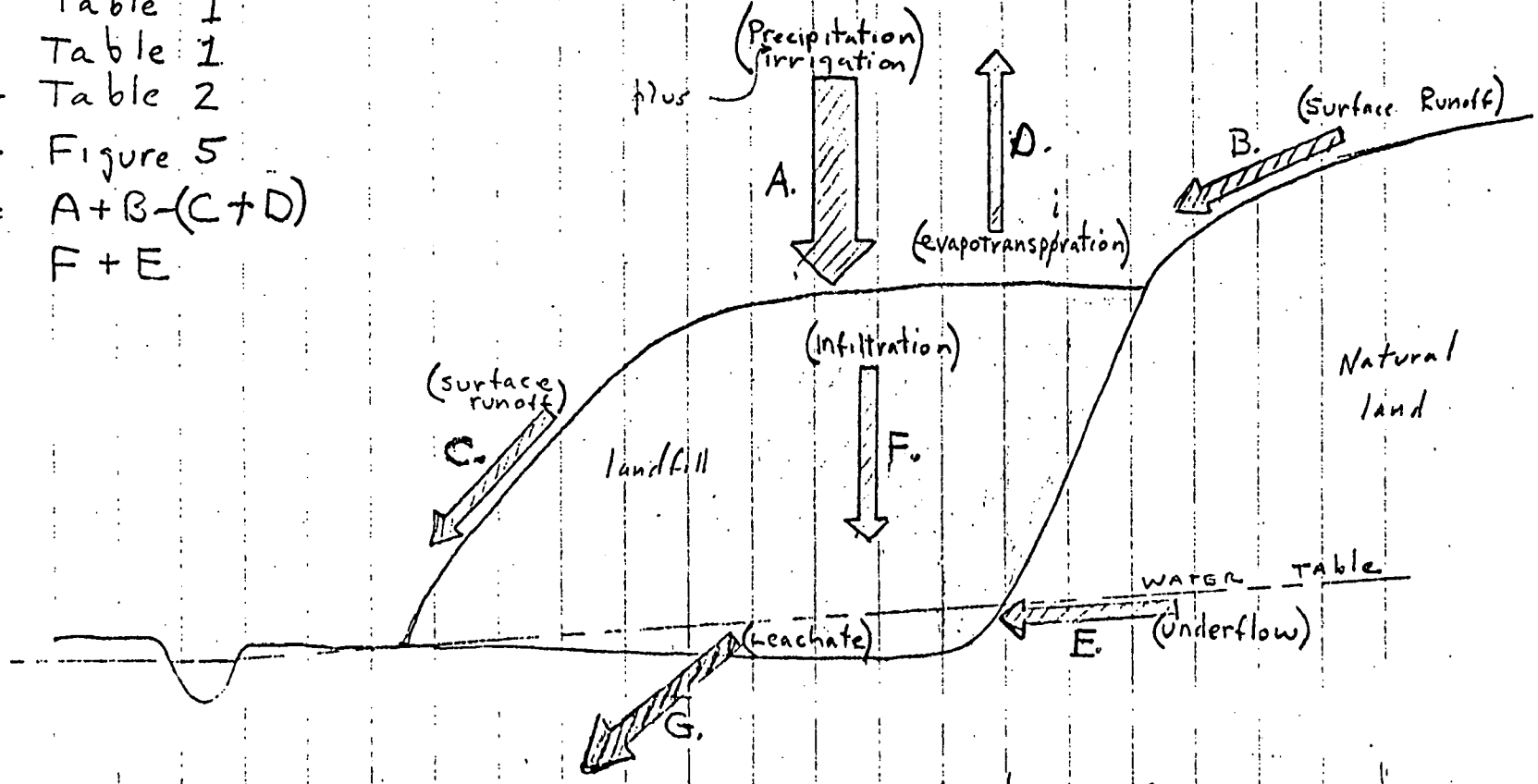

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Figure E-28

Distribution of Precipitation
(Average Annual)
Inches of precipitation in an average year
(Based on 40 years of record)
GULF OF MEXICO
NORTH DAKOTA
SOUTH DAKOTA
NEBRASKA
KANSAS
IOWA
KENTUCKY
NEW MEXICO
ARIZONA
TEXAS

precipitation may occur in certain localized areas, especially in mountainous regions. Significant variations may also occur with time, an abnormally wet year for example, and such abnormalities cannot be shown on a general map. For this reason, it is advisable to seek precipitation data specfic to the landfill site and for the year immediately preceeding the investigation. Historical precipitation records for weather stations ~~near the landfill site~~ can be obtained from the U. S. Department of Commerce, National Oceanic and Atmospheric Administration, Environmental Data Service, Asheville, North Carolina. The locations of weather stations for which data are available are shown on maps obtainable from the above address. Interpolation of the data from two or more stations can be made to more closely approximate the precipitation at the landfill site. For extended investigations or monitoring programs, it may be desirable to determine the precise volume of precipitation reaching the landfill surface. For this purpose, a rain gauge would be installed in a suitable location on or near the landfill. There are many types of rain gauges available and the selection of one would be based on the particular conditions of the monitoring program and the available budget.

Irrigation may be used on the landfill surface to maintain a desired vegetation growth, particularly when the landfill is completed and its top surface is being used as a golf course or other recreational facility. The volume of water used for irrigation should be measured with a flow meter and added to the precipitation.

Surface Runoff

The percentage of precipitation which flows onto the landfill from adjoining higher ground and off the landfill surface to adjoining lower ground can be calculated by the rational runoff

formula described by ~~Ven Te~~ Chow.(1) A reasonable estimation of runoff can also be made from the data presented in Table 1.(2) where the rational runoff formula was applied to a series of typical situations. Areas and slopes are measured by a survey and surface conditions are determined by inspection.

Table 1 - Percentages of Surface Runoff for a 2.5 cm Rainfall (2)

Surface Condition	Percent Slope	Percent Surface Runoff: Sandy Loam	Clay or Silt loam	Clay
Pasture or meadow cover crop	-			
Flat	0 - 5	10	30	40
Rolling	5 -10	16	36	55
Hilly	10 - 30	22	42	60
No vegetation- not compacted	-			
Flat	0 - 5	30	50	60
Rolling	5 - 10	40	60	70
Hilly	10 - 30	52	72	82

Evapotranspiration

Evapotranspiration is the sum of water loss by evaporation and transpiration (plant water consumption). Methods of calculating evapotranspiration are given in the hydrologic literature (see ~~Ven Te~~ Chow) (1) However, the large number of variables that must be measured to perform the calculations make it a difficult process.

Estimation of evapotranspiration from available generalized data, such as potential evapotranspiration maps or annual water consumption figures for different plant species, may be misleading. This approach cannot account for numerous specific variables such as soil type, soil water available and vegetation density. Since evapotranspiration from a landfill surface may be anywhere from insigificant to the single most important mechanism for the removal of water from a landfill surface, an accurate estimate of the actual magnitude of evapotranspiration, from the specific site, and at the specific time of the investigation, should be obtained.

Because of the difficulties, in arriving at an accurate figure for actual evapotranspiration, it is suggested that professional assistance be obtained. If a hydrologic consultant is retained for the landfill study, he will be able to estimate actual evapotranspiration for the specific case involved. If such a consultant is not used, information on evapotranspiration rates for an area will often be available from a local agricultural research station, a nearby United States Geological Survey field office, or possibly the agriculture department of a nearby university.

Underflow

Underflow is defined here as the rate of ground-water flow from adjoining areas directly into the landfill. This condition will occur only if the base of the landfill is below the water table. A second necessary condition, however, is that the landfill adjoins or is situated near an area of elevation substantially higher than the base of the landfill, i.e. that there is a significant water table gradient beneath the landfill. If the landfill is situated on level ground

and substantial percolation of water through the landfill is occurring, leachate being generated by the percolation will move away from the landfill and in directions and underflow, as defined above, will not occur. (See Figure 5-2)

Precise measurement of underflow, it it is occuring, it not feasible. A determination of the occurrence of underflow, and a reasonable approximation of its rate can be made, however, by means of a relatively straightforward hydrologic investigation. Figure 5-3 is a schematic diagram illustrating the method for estimating the rate of underflow. This process requires the drilling and testing of at least two wells and ~~is~~ therefore, considerable expense will be incurred. The drilling, however, would normally be necessary for other determinations ~~anyway~~ (e.g. water quality) and thus would be required in any case.

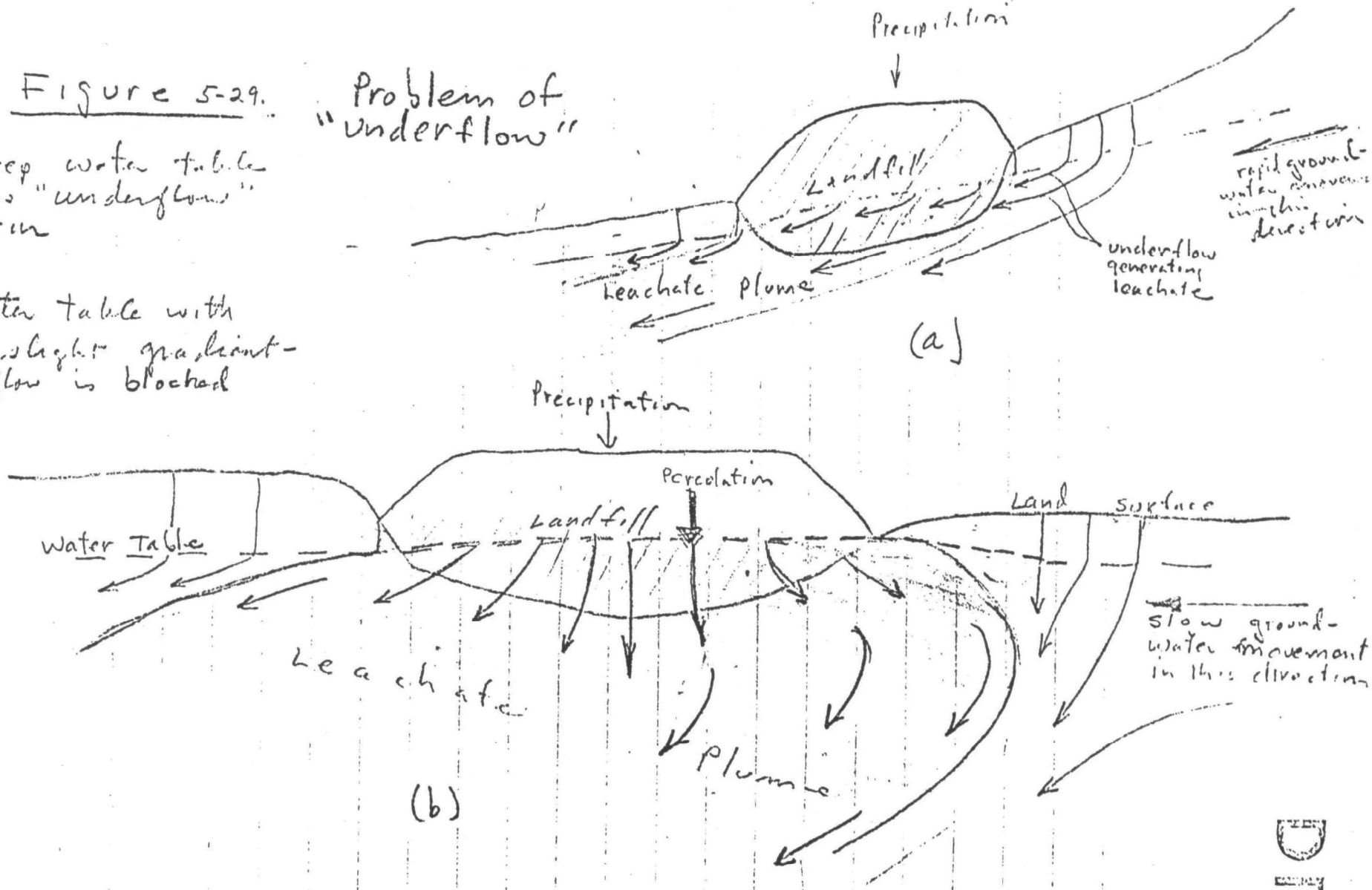
Percolation

Infiltration is generally defined as that portion of surface water which penetrates ~~the~~ land surface. Some of this water may then be taken up by plants and some may be directly evaporated from the first few inches of soil. That portion of water which migrates below the root zone and into the refuse, and thus contributes to the volume of leachate generated is the percolate. One additional point is that refuse, when it is placed in a landfill, is capable of absorbing a certain volume of water and holding it against the force of gravity. The volume of water so absorbed by the refuse is termed its field capacity. Generally, municipal refuse field capacity is about one inch of water for each foot of refuse. Leachate will not be generated in any significant volume

Figure 5-29. "Problem of "underflow"

(a) - Steep water table allows "underflow" to occur

(b) - Water table with only slight gradient - underflow is blocked



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until almost all of the refuse has reached field capacity. For the present discussion, it is assume that the landfill has reached its field capacity.

Calculation of the rate of percolation of precipitation and irrigation into the hypothetica landfill is shown on Figure 5-27. Calculation of the rate of leachate generation follows by adding the value for underflow. Direct measurement of percolation is possible using a sub-surface water trap such as the one shown in Figure 5-31. If infiltration is measured directly, somewhat more confidence can be placed in the calculated values for leachate generation, surface runoff and evapotranspiration.

5.4.5 Pollution Abatement Based on Water-Balance

Based on the results of a hydrologic investigation of the landfill site, it may be determined that reduction of the leachate generation rate is the best course of action (other possible courses of action would include leachate removal, hydrologic barriers, physical barries, etc). In this case, the, existing parameters such as side slopes vegetation type, etc. which control the components shown on Figure 5-27, can be altered to increase C and D and decrease B, E, and F and consequently decrease G. The degree of alteration required for each parameter to achieve the desired reduction in leachate generation can be determined by calculating the effect the proposed modification will have on the various components of the water balance. By this means, various alternatives for modifying the landfill can be compared and the optimal method selected.

The cost of a water balance study by a consultant, for a landfill where underflow is not a problem would be about $1,000, including both the field and office work. In many situations,

Figure 5-30 — Estimation of Landfill underflow.

where:
Underflow = Q
Permeability = P
gradient = I
area = A

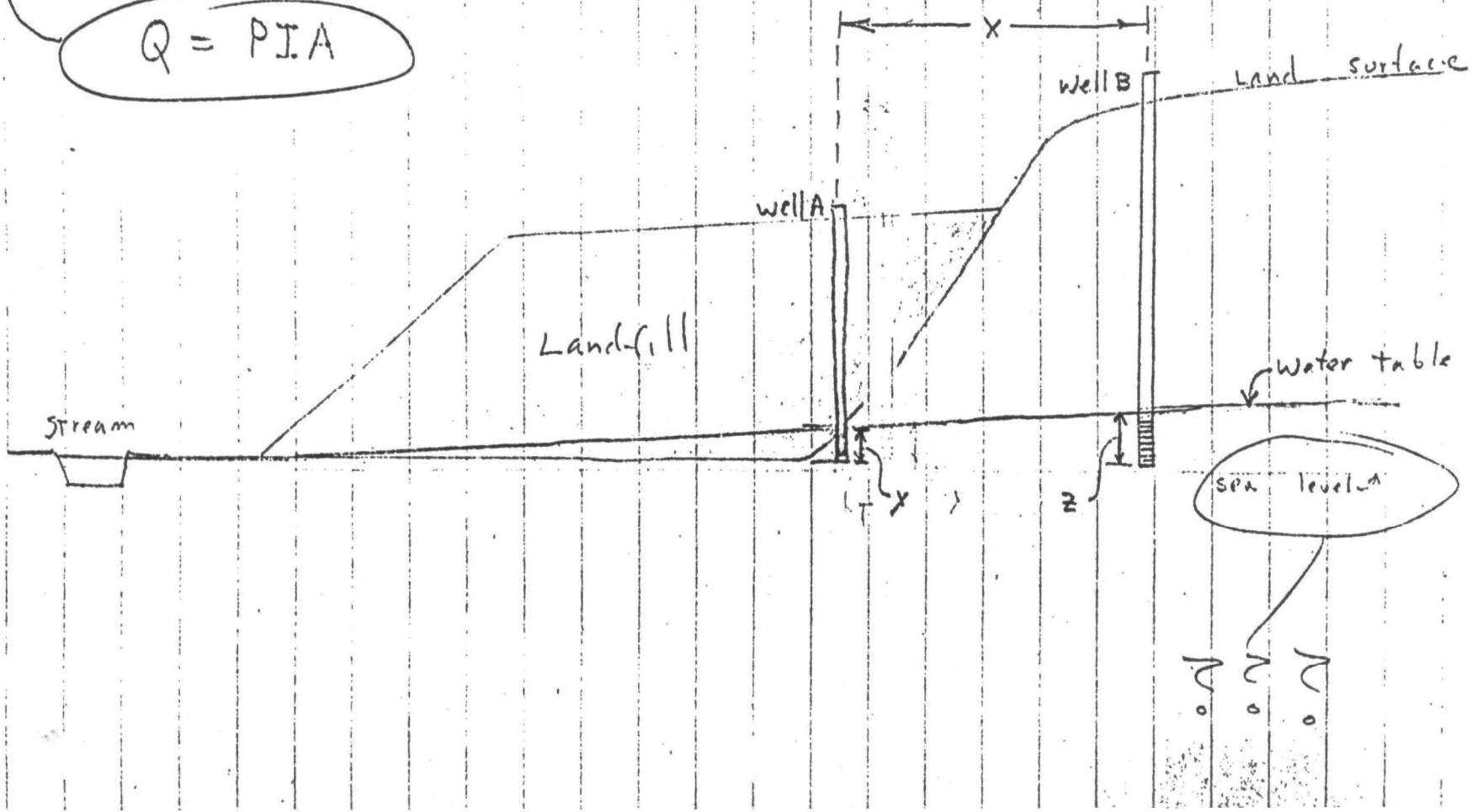
Units ?

$Q = PIA$

P is estimated by pumping tests or injection Tests in Wells A and B, or by examination of geologic samples removed during drilling.

$I = [Z\,(feet) - Y\,(feet)] \div X\,(feet)$

$A = (Y) \times$ (horizontal length of contact between landfill and high ground.)



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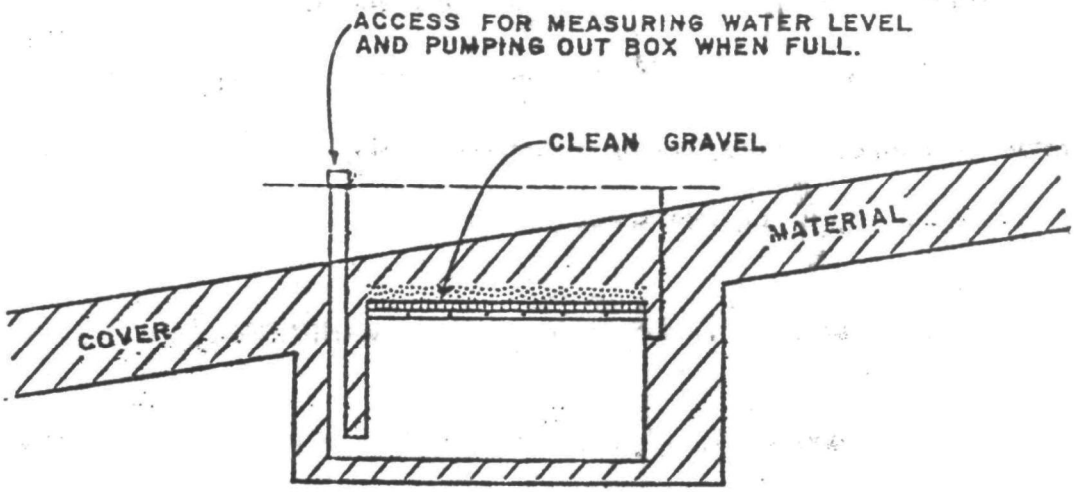

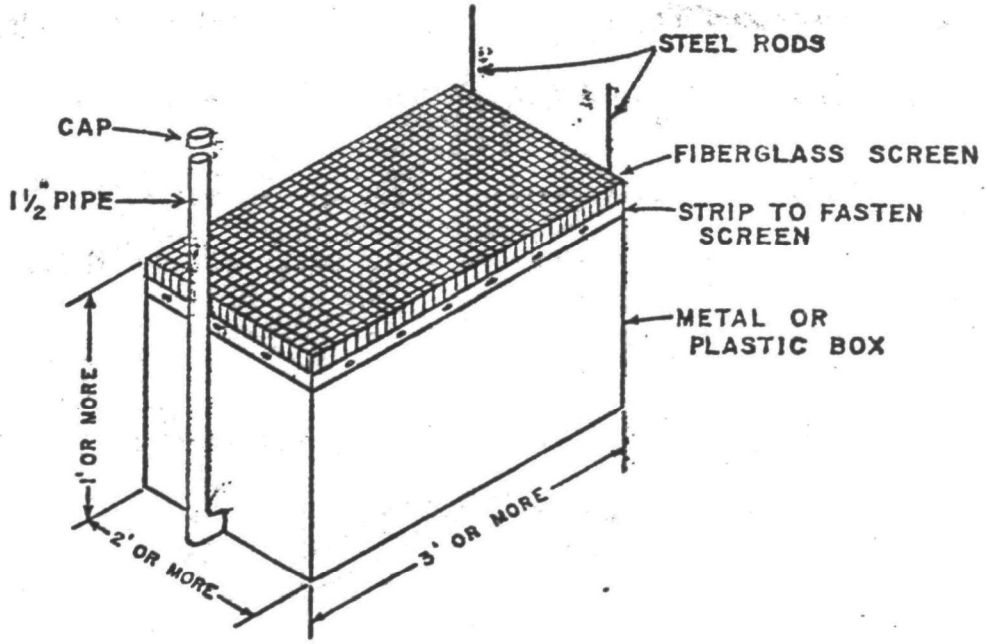

Figure 5-31. Device for direct measurement of landfill infiltration.

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the necessary field work would be accomplished during the course of other tasks, such as the field inspection, and the cost would be reduced to about $400. If underflow were a significant problem, the cost of the water balance study would be closely tied to the drilling program, as multiple-use wells would be installed and the cost spread out over several tasks.

References:

(1) Chow, Ven, 1964. Handbook of Applied Hydrology, McGraw-Hill Book Co., New York.

(2) Hughes, G. M., R. A. Landon, and R. N. Farvolden, 1971. Hydrogeology of Solid waste disposal sites in Northeastern Illinois. U. S. Environmental Protection Agency publication No. SW-12 d 154 p.

(3) Fenn, D. G., and K. J. Hanley, 1972. Use of the water balance method for predicting leachate from sanitary landfills. Office of the solid Waste Mangagement Program, U. S. Environmental Frotection Agency. Unpublished manuscript. 55 pp.

~~(4) Fievet, Schrois, Edmonston, Bass, ... and ... from Engineering ... p. 434.~~

(2) Salvato, J.A., Wilkie, W.G. and Mead, B.E. 1971. Sanitary Landfill Leachate prevention and control. Journal WPCF, 43 (10) 2084.

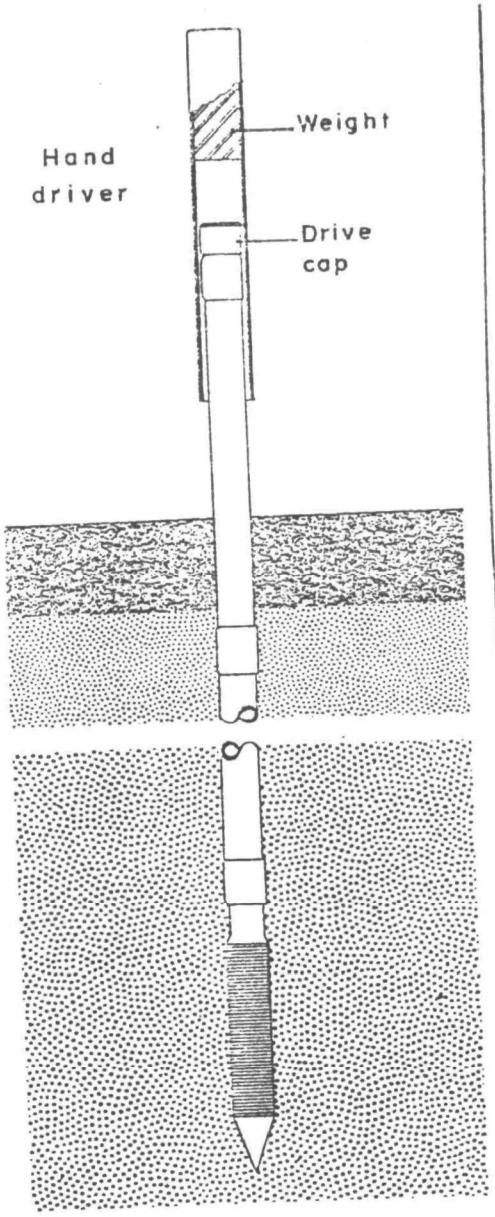
Well Technology

DRILLING METHODS

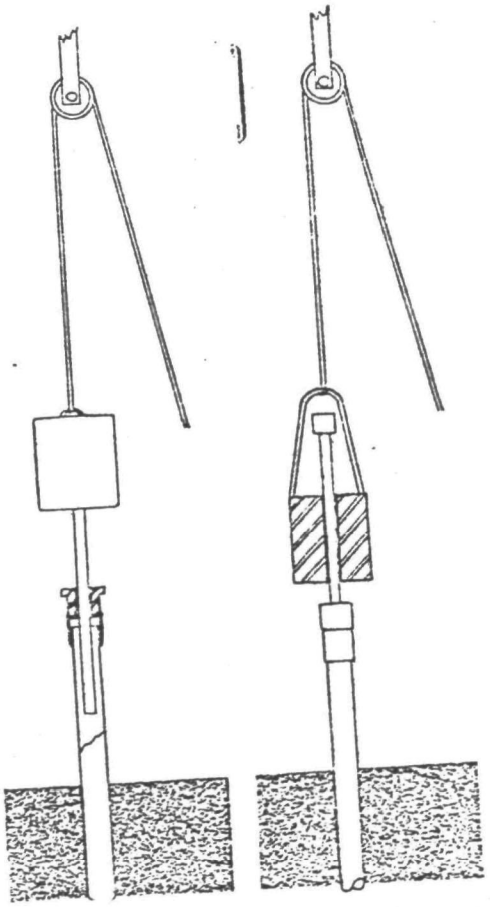
Drive Points - In this method of drilling, a 1¼-or 2-inch diameter drive point is attached to a 2-inch pipe and driven to completion depth with a sledge hammer, drive weight, mechanical vibrator, or pneumatic hammer. The point can be driven to approximately 30 feet by hand, and up to 100 feet if a mechanical drive weight is used, but only if driving is done in sands or finer grained sediments that offer little resistance to penetration. Boulders cannot be overcome. Powell and others (1973) report using a mechanical vibrator to drive points to depths of 65 feet. Drive points, because of their small diameter, are used in areas of high water table (near-surface) from which water can be removed by suction pumps, for example kitchen pitcher pumps or centrifugal pumps.

Reliance on a drilling contractor to install drive points is unnecessary. Local investigators can drive them with a minimal investment in equipment and manpower. The first step is to bore a vertical hole as deeply as possible with a hand auger slightly larger than the well point. (Figure 17). The drive point is attached to a length of ~~user~~ riser pipe (5 foot lengths are preferable) and placed in the augered hole. A drive cap is placed on the top of the casing prior to driving.

Casing can be driven with a tool similar to the type used for driving steel fence posts, or by drive weight suspended from a tripod or derrick. Drilling will be more efficient if there is a source of power to lift the weight, as they can weigh 75 to 450 lbs. ~~A cathead on a portable wash boring rig can be used or one can be improved~~ using a rear axle of a auto and tire rim for cathead. Drive points can also be driven with a sledge hammer, but



~~Figure 100.~~ Simple tool for driving well points to depths of 15 to 30 ft.



~~Figure 101.~~ Heavier drive-block assemblies commonly operated by drilling rig or with tackle.

Figure 17

this is difficult and slow going unless the investigator is a J

driven, it is turned slightly to keep the threaded joint tight.

Advantages

1. Inexpensive.
2. Easily installed by hand, to limited depths.
3. Closely spaced vertical samples can be collected during drilling.
4. Can expect a good seal between casing and formation, little or no vertical leakage.

Disadvantages

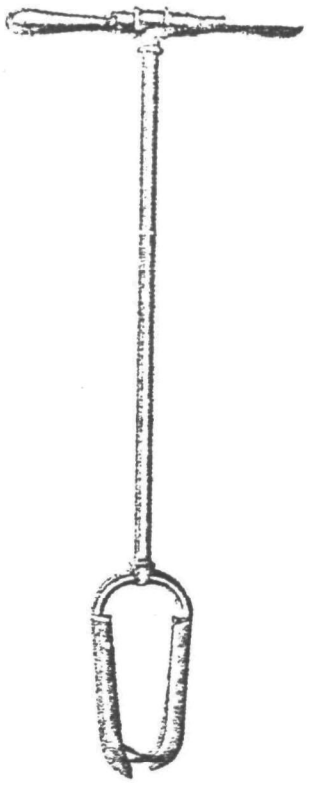
1. Difficult to develop and sample if water table is below 15-20 ft deep.
2. Extreme depth limitations applicable to shallow work primarily less than 30 ft.
3. No formation samples, only information on subsurface material penetration rate (blow counts, et
4. Only certain types of pumping can be used.
5. Drive point screen may become clogged wih clay, if driven through a clay unit.
6. Can be used only in unconsolidate sediments.

Augers - In auger boring, the hole is advanced by rotating and pressing a soil auger into the soil and withdrawing and emptying the auger when it is full of soil. As much as possible, the borehole is kept dry because water tends to prevent accumulation of soil in the auger. Hand augering, as anyone who has dug a post hole knows, can be easy or difficult depending on whether or not clay and sand or gravel, respectively is being, removed. Small diameter helical or posthole augers can be used to advance 2 to 12-inch diameter holes by hand to depths of 20 to 30 feet (Figure 18). If a tripod and pulley are set up to aid in pulling the auger from the hole, depths of 80 feet can be reached. If the hole can be kept open below the water table, usually only in cohesive material, a screen and casing can be set, backfilled, and developed.

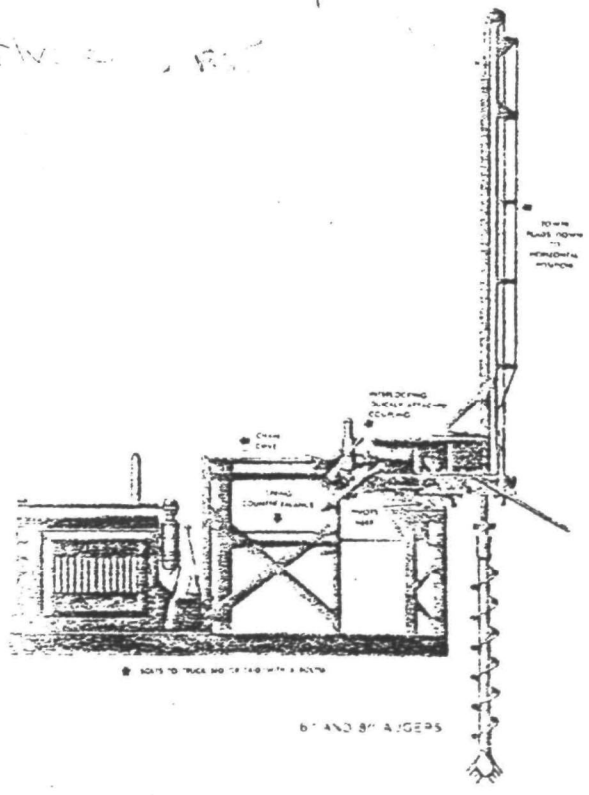
This process becomes much simpler and less time consuming if power augers are used. Here, flights of spiral hollow-stem augers are forced into the ground while being rotated, and the spiral action of the augers conducts cuttings to the surface (Figure 18). On completion of drilling, a small diameter casing and well point are pushed to the desired depth. With bucket augers, a large-diameter barrel fitted with cutting blades, (up to 48 inches in diameter) is rotated into the ground until it is full. The earth-laden bucket is then brought to the surface, pulled to one side, and dumped. This process is repeated to completion depth. Bucket augers would not normally be used in landfill investigations, and they are not evaluated below.

Power augers can be used very effectively in cohesive soils. On the other hand, these augers are not well suited for use in very hard or cemented soils, and they often fail to retain very soft soils and fully saturated cohesionless soils. However, if setting a drive point is the main purpose of the hole, slumps or cave-in of the hole in cohesionless sediment is not a major drawback.

SMALL HELICAL AUGER



POSTHOLE OR IWAN AUGER



BUDA EARTH DRILL WITH CONTINUOUS HELICAL AUGERS

FIG 18

Advantages

1. Inexpensive.
2. Small, high-mobility rigs can get to most sites.
3. Can be used to quickly construct shallow well clusters.
4. If borehole reaches refusal depth too soon, set up time is low and rig can be moved rapidly.
5. No drilling fluids introduced into the borehole, no possibility of diluting formation water.

Disadvantages

1. Limited penetration, normally 100 feet, max. 150 feet.
2. Vertical leakage through sediment left in borehole, through which drive point is forced to completion depth. No way to isolated screened zones of aquifer.
3. Careful attention during drilling is required to get a correct log of formation materials penetrated.
4. Unable to collect ground water samples during drilling.
5. Core sampling is possible only if hollow stems augers flights are used.
6. Can be used only in unconsolidate sediments.
7. Borehole will collapse in cohesion less sediment.

Wash Boring - A wash boring is advanced partly by a chopping and twisting action of a chisel-shaped bit and partly by the jetting action of a stream of water pumped through the drill rod and out the bit. (Figure 19). As the bit penetrates the formations, the casing sinks of its own accord due to the washing action of the bit alone. ~~Cutrings~~ Cuttings are carried to the surface by the water circulating in the annular space between the drill pipe and casing. The drill string is lifted and dropped to get a cutting action with the bit at the same time it is rotated to make the bit cut a round hole. These operations, as well as the pumping, may be performed entirely by hand, but a small motor-driven winch and pump are generally used. A closed system is used to recirculate the drilling water. Water is pumped from a pit into the drill string and out of the bit. This water, after it circulates from bottom to top of the borehole, is conducted back to the pit where the cuttings settle out. Normally, small pits are used to reduce the volume of water required. As a result, cuttings have to be cleaned out of the pit at regular intervals.

The drill rod is generally 1 to 2-inch black iron pipe. Casing is required to keep the hole open in soft clays or sand and gravel, but is often not necessary in stiff clays or similar cohesive sediments. If the borehole stays open by itself, casing and screen are simply lowered and backfilled to construct a well. If casing is required to drill, slip screens are set by the casing pull-back method.

Drilling equipment is simple and readily available to local investigators who wish to do their own borings. The basic units are a tripod, a pump, and a cathead. The only components that need to be purchased from a drilling rig company are the water swivel and the drill bits, although the bits can be easily fabricated in a metal-working shop.

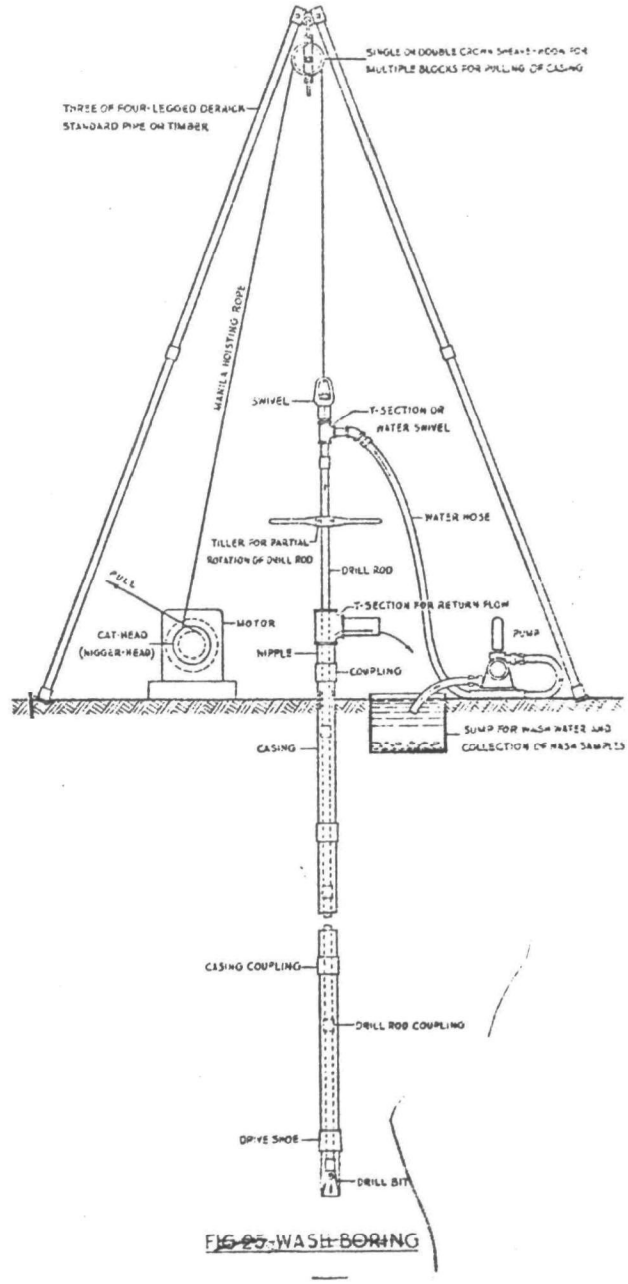


FIG 25. WASH BORING

FIG 19

Advantages

1. Inexpensive and light equipment, drilling contractor not required.
2. Excellent for shallow bore holes in unconsolidated sediments.
3. Can get vertically spaced ground-water samples if drive point is forced ahead of borehole and pumped.
4. Drilling equipment can get to almost any site.
5. Core samples can be collected.

Disadvantages

1. Slow, especially at depth.
2. Maximum depth of 100 to 150 feet
3. Cannot penetrate boulders and wash up gravel.
4. Difficult to develop and sample if water table is deeper than 15 to 20 feet.
5. Can be used only in unconsolidated sediment.
6. Wash water can dilute formation w must be taken into account in vertical sampling.
7. Interpration of geology from wash samples requires skill.
8. Can set only short sections of screen without difficulty.

Jet Percussion - The drill tools and the drilling action of the jet-percussion method are the same as those described for wash boring, however, casing is driven during drilling with a drive weight and not allowed to advance of its own weight. Normally, this method is used to place 2-inch diameter casing in shallow, unconsolidated sand formations, but has been used to install 3 to 4-inch diameter casings to 200 feet. Screens have to be set by the casing pull back method.

Most jet-percussion rigs are moderate-sized pieces of equipment and drilling contractors used to working in unconsolidated sediments will probably be the best source of a rig.

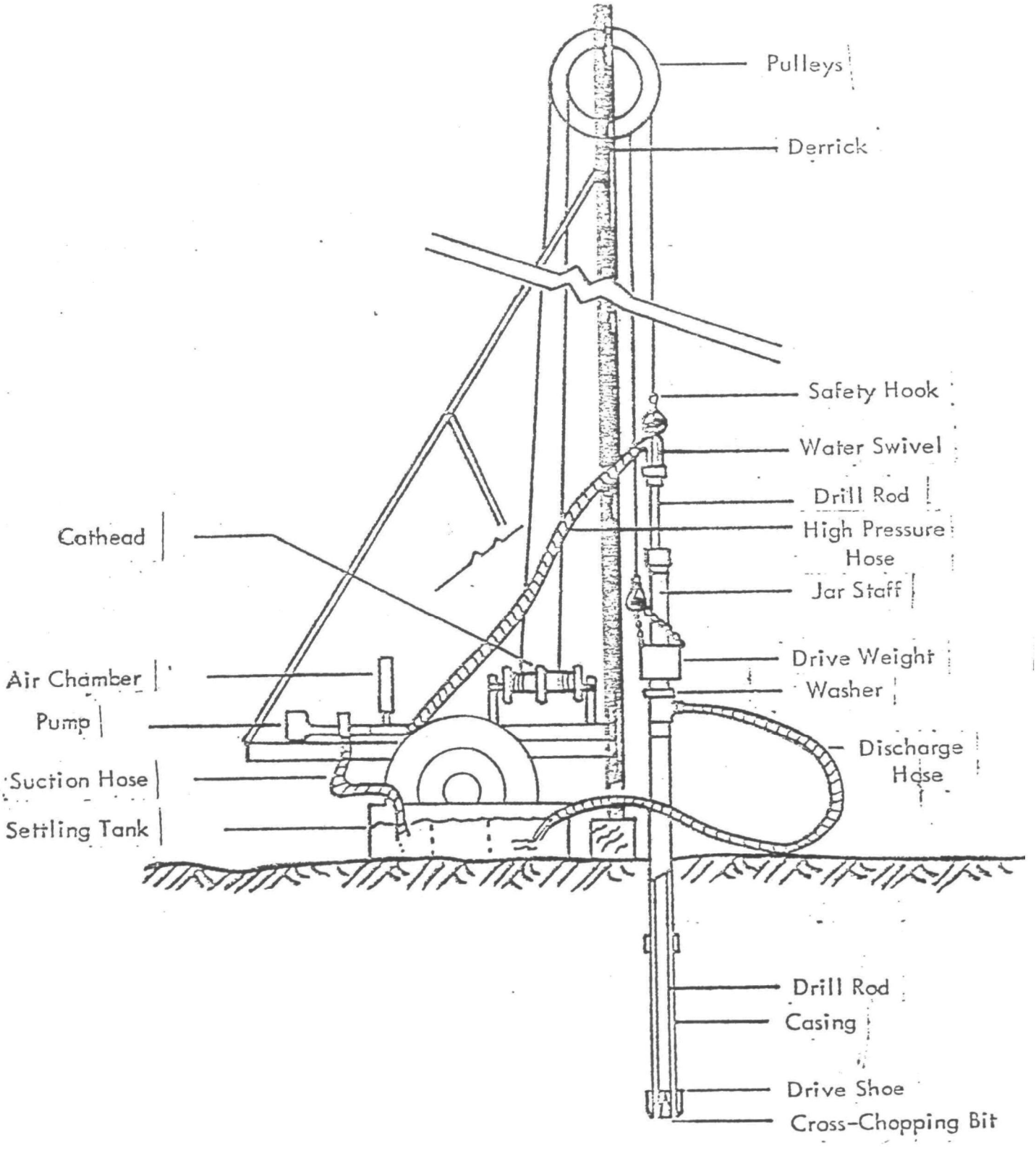

Pulleys
Derrick
Safety Hook
Water Swivel
Drill Rod
High Pressure Hose
Jar Staff
Drive Weight
Washer
Discharge Hose
Cathead
Air Chamber
Pump
Suction Hose
Settling Tank
Drill Rod
Casing
Drive Shoe
Cross-Chopping Bit

FIG. 20

Advantages

1. Inexpensive.
2. Simple equipment and operation.
3. Good seal between casing and formation prevents vertical leakage of formation water.
4. Can obtain a reliable formation water sample at completed depth.

Disadvantages

1. Slow.
2. Use of water during drilling can dilute formation water.
3. No formation water samples can be taken during drilling.
4. Poor soil samples because fines are washed out of sample.
5. Small diameter (2in.) and shallo maximum depth (125 ft.) limits usefulness of this type of well to water sampling at shallow depths
6. Large number of wells requirec at one location to obtain closely spaced samples throughout the contaminated thickness of the aquifer.
7. Can be used in unconsolidated sediments or weathered rock.

Cable-Tool Percussion - In cable-tool percussion drilling, t

regularly lifting and dropping a heavy string of drilling tools in the borehole (Figure 21). The drill bit breaks or crushes hard rock into small fragments and in soft, unconsolidated sediments, loosens the material. The up and down action of the drill string mixes the crushed or loosened particles with water to form a slurry or sludge. If no water is present in the formation being penetrated, the necessary water to form the slurry is put into the borehole. Cuttings are allowed to accumulate until they start to lessen the impact of the bit, and then are removed with a bailer or sand pump.

A cable-tool drill string consists of four units: the drill bit, drill stem, drilling jars and rope socket. The bit provides the cutting edge of the drill string, the action of which is enhanced by the weight of the drill stem. This weight also acts as a stabilizer, keeping the hole straight. The jars are a pair of sliding, linked bars which provide a play in the drill string of 6 to 9 inches. If the tools become stuck, the jars permit successive upward blows in the attempt to free them rather than a steady pull on a cable which might part. The shaking and vibrations produced by the jars helps in freeing a stuck drill string. The rope socket connects the string of tools to the cable and allows the tools to rotate slightly with respect to the cable.

The bailer consists of a section of pipe with a check valve at the bottom, and is filled by an up and down motion in the bottom of the hole. Each time the bailer is dipped, the valve opens, allowing the cuttings slurry to move into it. The up and down motion is con-

Mud drilling ... w/ ...
which reduces ... Permeation.

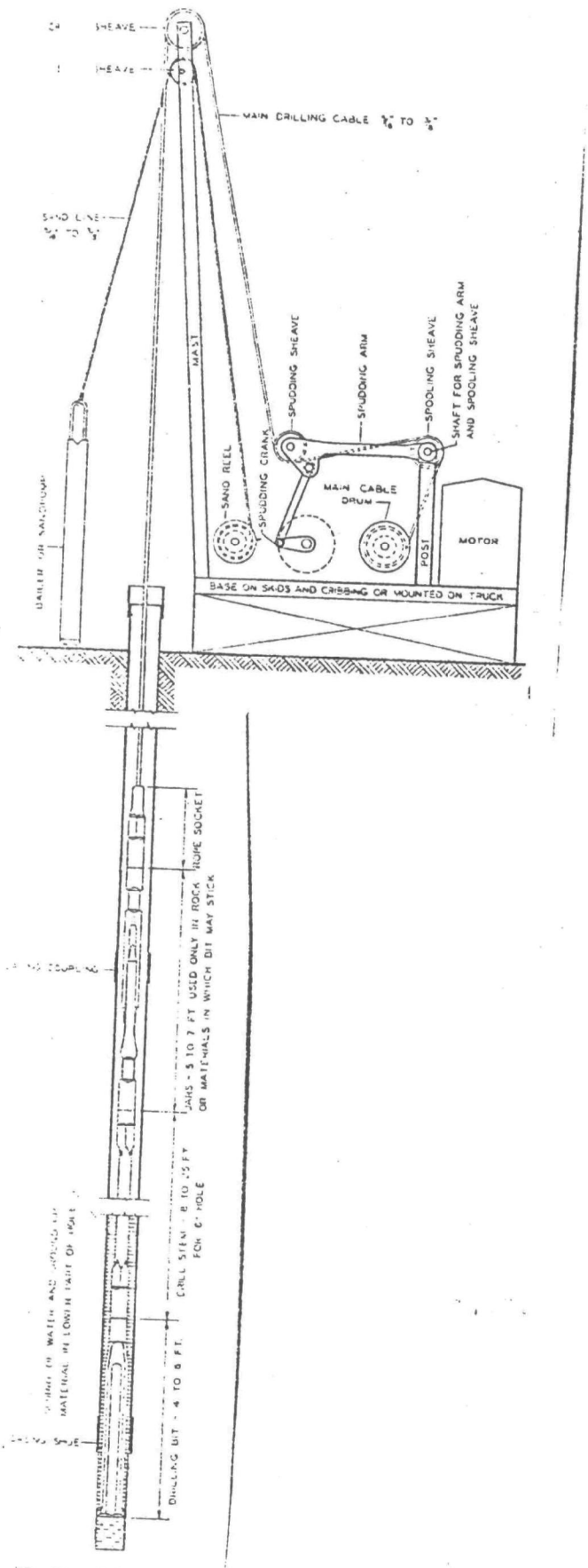


FIG 30 - PERCUSSION DRILLING

FIG 21

tinued until the bailer is full. At this point, it is brought to the surface and the contents dumped on the ground. The sand pump is a bailer that is fitted with a plunger so that an upward pull on the plunger tends to produce a vacuum that opens the valve and sucks sand or slurried cuttings into the tubing.

Casing is driven by attaching a drive clamp to the drill stem and the reciprical action of the rig hammers the casing into the ground as the clamp makes contact with the top of the casing. Operation can be speeded up by drilling ahead of the casing, but only if the hole will stay open by itself. If drilling open hole and there is a cave in, the drill string could be tapped. Cautious drillers, therefore, rarely drill ahead of the casing unless they are going through rock. Normal procedure in unconsolidated sediments is to drive the casing into the formation and then clean out inside the casing with the drill tools. This is slower but safer than drilling ahead of the hole.

GERAGHTY & MILLER, INC.

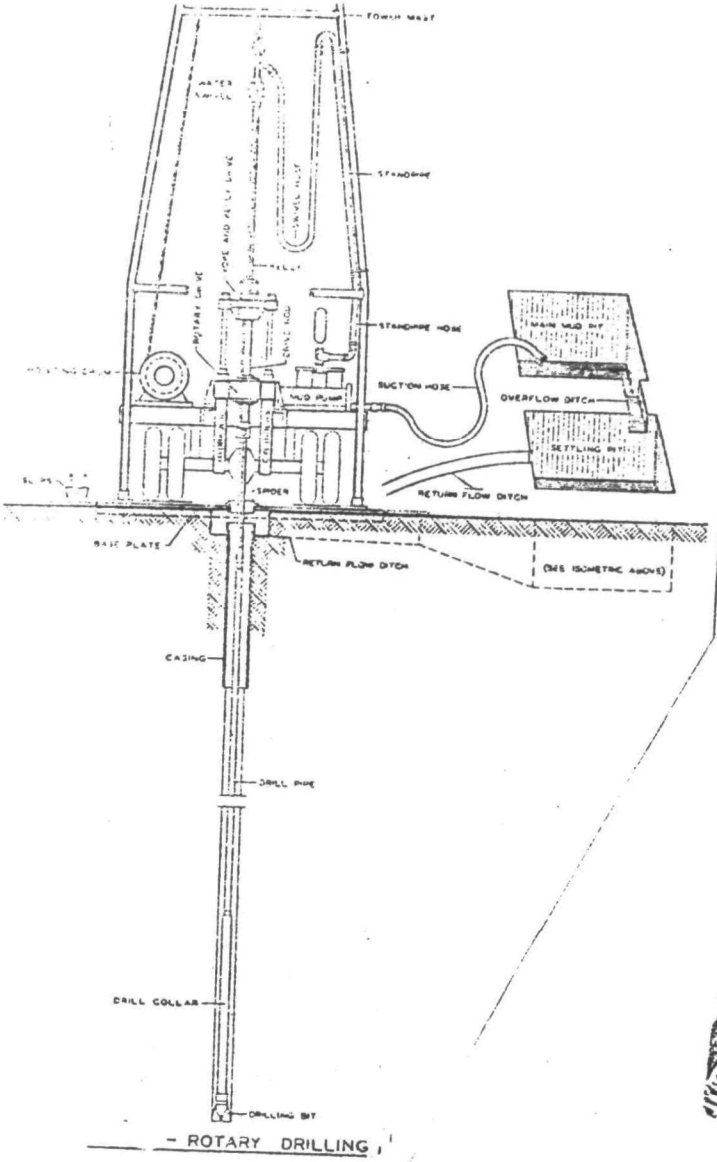
— DRAFT —

Advantages

1. ~~Inexpensive, if non union drillers are involved.~~
1. 2. Simple equipment and operation.
2. 3. Good seal between casing and formation if flush joint, casing is used.
3. 4. Good disturbed soil samples, know depth from which cuttings are bailed.
4. 5. Core samples can be collected.
5. 6. If casing can be bailed dry without sand heaves, a formation water sample at that depth can be collected.
6. 7. Can be used in unconsolidated sediments and consolidated rocks.
7. 8. Only small amounts of water are required for drilling.
9. Once water is encountered, changes in static or potentiometric levels are readily observable.

Disadvantages

1. Slow.
2. Use of water during drilling can dilute formation water.
3. Potential difficulty in pulling casing in order to set screen.
7 4. No formation water samples can be taken during drilling unless open-ended casing is pumped.
5. Heavy steel drive pipe is used and could be subject to corrosion under adverse contaminant characteristics.
6. Cannot run a complete suite of geophysical well logs because of steel casing.



– ROTARY DRILLING

Hyd... 1965

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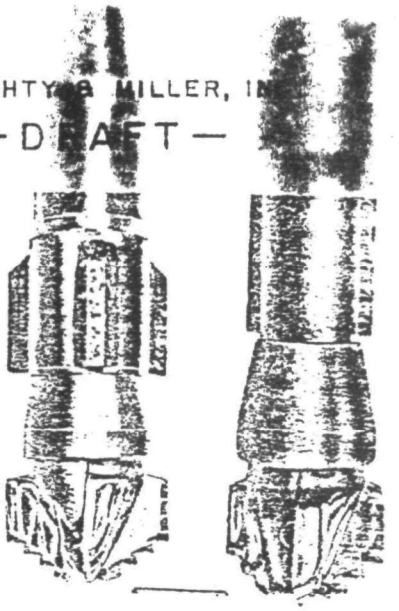
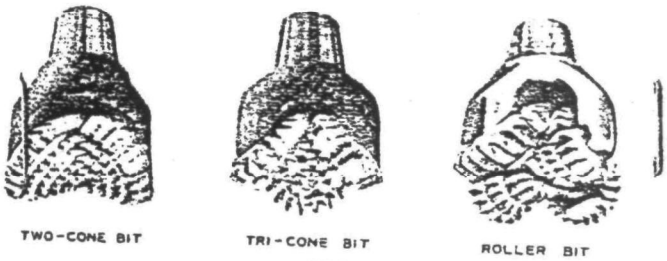


Drag-type bits with replaceable blad...

Johnson, 1972



– ROCK BITS

Hyd... 1965

Hydraulic Rotary Drilling Equipment

Fig 22

Hydraulic Rotary - To drill a hole by the hydraulic rotary method, a roraring bit breaks up the formation and the cuttings are brought to the surface by a recirculating drilling fluid (Figure 22). Drilling mud is pumped from a settling basin, through a water swivel, and down the hollow interior of the drill rod. At the bit, nozzles direct the fluid to ~~efficiently~~ clean cuttings from around the bit and it then flows upward in the annulus, carrying the cuttings to the surface. Here, the fluid is discharged into the mud pit and the cuttings settle out. At the other end of the pit, the water is sucked into the pump to circulate down the drill rod again.

Teh drill string consists of the bit, a stabilizer, and the drill pipe. Two basic types of bits are used: roller bits in rock ~~and consolidated sediments~~ and drag bits in unconsolidated materials. Roller bits have conical rollers with hardened steel teeth of various lengths, spacing, and number, dependent on the type of material to be drilled. Some rollers have inset carbide buttons for drilling in hard, tough rock. As the rollers rotate, they crush and chip the formation material. Drag bits have fixed blades, the cutting edge of which is surfaced with carbide or some other abrasion-resistent material. ~~Drilling action is the result of these blades scraping material off the borehole wall.~~

The bit is attached to a heavy, weighted section of the drill string called a drill collar or stabilizer. This weight just above the bit tends to keep the borehole straight and vertical. The drill rod connects the stabilizer to the kelly, ~~bar~~ and and ranges in outside ~~chamber~~ from 2 3/8 to 4-inches. The kelly is a fluted bar which passes through a rotary table, which imparts a rotary motion to the drill string. When the length of the ~~keey~~ has been drilled, a new section of rod is added and drilling is started again.

Advantages

1. Fast.

2. Dilution of formation water is limited by formation on a filter cake on borehole walls.

3. Formation water sample can be obtained with a special technique.

4. Good disturbed soil samples from known depths if travel time of cuttings up borehole is taken into account.

5. Flexibility in final well construction, such as screen placement.

6. Can run a complete suite of geophysical well logs.

7. Core samples can be collected.

8. Can be used in unconsolidated sediments and consolidated rocks.

1. Expensive.

2. Complex equipment and operation

3. Potential of vertical movement of water in formation stabilizer material placed between casing and borehole wall after completion.

Air Rotary - As in hydraulic rotary drilling, a rotating down-hole hammer is used to break up formation material by percussion, but rather than a liquid carrying cuttings to the surface, high velocity compressed air is used. Down-hole hammers are essentially pneumatic hammers similar in operation to those seen being used to cut up pavement by road repair crews. Normally this type of drilling equipment is used in rock because of fantastic penetration rates compared to cable tool or mud rotary drilling; drilling rates of one to two feet per minute are not unusual. Unfortunately, down-hole hammers larger than 6 inches are not readily available, limiting the size of the borehole that can be drilled. Much of their speed advantage is lost when conventional roller cone bits are used. However, drilling water is not required, eliminating a logistics problem that can become difficult especially in arid regions. Most rigs are equipped with a small mud pump so they can drill a conventional rotary hole through unconsolidated overburden on top of the rocks. When the hole is finished, casing is set slower to rock to prevent caving.

Advantages	Disadvantages
Same as discussed in hydraulic rotary drilling	

A minimum upward air velocity of 3000 feet/minute is required to lift cuttings to the surface. When drilling a four inch diameter hole with 2-3/8 inch rod at least 150 cubic feet/minute (CFM) of air are required to lift cuttings. If a prolific aquifer is penetrated, the hammer may be "drowned out", ~~ie~~ i.e., the compressed air cannot lift the volume of water entering the hole to the surface. At 3000 ft/sec air velocity, this threshold is met at about 50 gpm in a 4 inch hole, and at about 150 gpm in a six inch hole. When this happens, a

larger air compressor is required or drilling must ~~switch~~ to the hydraulic rotary method. Air rotary rigs are available with compressors capable of supplying 1100 CFM at a pressure of 250 psi.

Well Casing & Screen Materials

Landfill leachate can be characterized as a strong electolyte which may be corrosive. Specific characteristics of the leachate will depend on the type of material accepted by the operators. Therefore, some thought must be given to the materials used in monitoring well construction in order to prolong the installation's operating life to at least match that of the landfill. This is not an adequate design criteria, however, the monitoring well should be serviceable for as long as required after the landfill is completed. Review of comparison tables of various pipe materials to chemical attack (Robintech, Inc.) indicates that PVC pipe is resistant to most chemicals, with the exception of ketones, esthers, and aromatics (among the more common chemicals), when compared to the other normally-used well casing, steel pipe. PVC casing (~~pipe~~) is a nonconductor and will not be involved in electrochemical reactions as will, for example, a steel casing and brass or iron well screen. Nor will it normally interact with the leachate as will steel casing.

From a leachate sampling standpoint, PVC is very attractive. Because of its chemical inertness, it will contribute little in the way of chemical constituents to a leachate sample except in the parts per billion range. Steel pipe can be expected to contribute at least iron, and probably other ions to a sample. Of course, this sample contamination can possibly be avoided by proper flushing of the well before collecting a sample in both steel and PVC casing.

A major drawback to PVC casing is its lack of strength. Landfill equipment or vandals can easily snap off a PVC casing projecting above ground surface. Therefore, special well protection measures (described in the Well Security section) must be taken. In spite of all this, PVC casing and screens appear to be the best materials to use in constructing landfill monitoring wells.

Actual well construction, however, will be dictated by a variety of constraints, such as drilling method, aquifer type and formation materials, cost of well consrruction materials, ease of installation, and personal prejudices, among others. Proper construction materials can be best evaluated for each situation by a person familiar with landfill investigations, but a person not familiar can fall back on a drilling method that will allow PVC casing and screen to be installed and be confident that the well will last and not bias the samples.

Well Security

Once a well has been completed, some measures must be taken to protect the installation from: 1) normal landfill operations, especially heavy equipment and, 2) vandals. In areas being actively landfilled, provisions have to be made for extending the well casing and its protection above the active level of the fill. An installation capable of protecting the monitoring well and being added as the depth of the fill increases is shown in Figure 23.

Construction of this protective installation is straightforward and inexpensive, with a reasonable likelihood of remaining undamaged by landfill equipment on vandals. To do this, a 10-foot length of steel casing several inches larger in diameter than the monitoring well is placed

over it. This casing is grouted in place with a cement collar at least four or five feet deep to hold it firmly in position. Although this will not withstand a run-in with a compactor or bulldozer, it will withstand attempted vandalism. The casing should be threaded so that a screw cap can be used to close the well. Two heavy duty, hardened steel hasps welded on opposite sides of the cap and casing will allow the well to be locked. As long as heavy duty hardened steel hasps and padlocks (capable of withstanding a 48-inch bolt cutter) are used, the efforts of even the most determined vandals will be in vain. If this type of installation is broken into, it will be for ~~the purpose of~~ sabotage, not simple vandalism.

Unless this well is highly visible, chances of it being struck by equipment during normal landfill operations are fairly high. To avoid this, a sample tripod constructed of timbers (railroad ties or equivalent) should be constructed over the well and crowned with a brightly colored object, such as a flat or painted tire (Figure 23).

When landfilling threatens to overtop the installation, the tripod is temporarily knocked down, additional casing added to the monitoring well and protective shell, and placement of trash and cover is continued around the well. If this procedure is followed, only a slight interruption in the normal course of landfill operation will be required to protect the monitoring well for future sampling.

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Water Withdrawal Methods

Water can be withdrawn from wells by a variety of methods including: bailers, thief samplers, pumps, or compressed air. The primary consideration in collecting a sample is insuring that all stagnant (standing) water has been removed from the well casing before a sample is collected. On cessation of pumping, water standing in a well begins to stratify, with water in the screen mixing with formation water due to normal ground-water flow, and water above the screen becoming more and more isolated because there will be little or no vertical mixing with the water in the screen. Improper well construction can cause all the water in the casing to be stagnant because of vertical leakage of leachate down along the well casing to the screened zone, or vandals dropping material into the well. Therefore, to obtain ground-water samples representative of chemical quality in the aquifer at the time of sampling, at least one volume of water standing in the casing and discharge pipe must be removed before sampling.

However, removing one volume is no guarantee that the stagnant water has been flushed from the well - four or five volumes are required to ~~be on the safe side~~ for a representative sample. If this seems like ~~a lot of effort just to get~~ obtain a water sample, consider the expense of chemical analyses and the possibility of having to repeat an analysis because stagnant water was sampled. Remember that the stagnant water may contain material introduced from the surface, inadvertently or deliberately, which would result in analytical results ~~elevated beyond actual~~ unrepresentative of aquifer water quality. This might ultimately result in adverse actions by an enforcement agency, all because of an improperly collected sample.

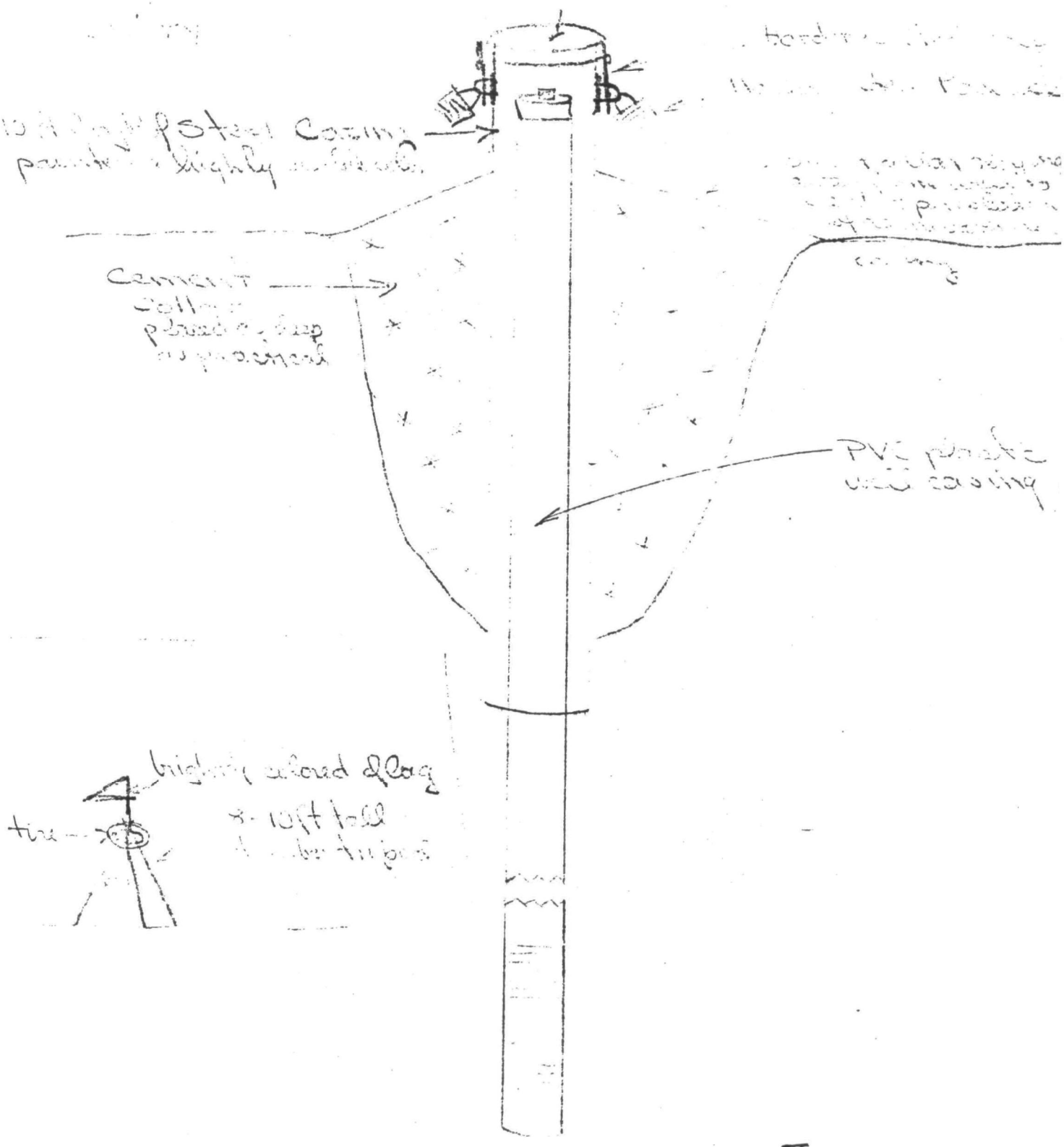

Fig 23

(FIGURE 24 in preparation)

~~IV-2~~ X 20 DRAFT

In light of this discussion, bailing by hand is not a recommended well sampling method unless adequate precautions are taken. Bailing is accomplished in small diameter wells by lowering and raising a weighted bottle or capped length of pipe on a length of rope. Rarely can a sufficient quantity of water be removed to adequately eliminate stagnant water from the sample, unless ~~innumerable~~ many, time consuming trips in and out with the bailer are made. Often, people sampling a well will use the first bailer full of water as the sample because of the ease with which the sample can be collected. The reliability of this sample is nil and this fact must be impressed on the sample collector. Of course, in situations where the well can be bailed dry or there is only several feet of water in the bottom of a shallow well, representative samples can be obtained with a bailer because the casing can readily be emptied.

Where the water-table is within suction lift, small-diameter wells can be sampled with centrifugal, peristaltic or pitcher pumps. Peristaltic pumps have rather low pumping capacities but are attractive because the sample is conducted through inert silicone rubber tubing, reducing the possibility of sample contamination by constituents from the sampling apparatus. Small, highly portable centrifugal pumps are available with pumping rates from 5 to 40 gpm, and removing stagnant water and flushing the discharge set-up clean will pose little difficulty, allowing collection of representative samples. If concentrations of less than 1 ppm are being investigated, extra care must be taken in sampling, and in the extreme ppb range, the peristaltic pump would have to be used.

~~IV-3~~ 21

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Pitcher pumps can be easily carried to a site, screwed onto a well, and used to pump a sample. No power source is required other than the investigator and ~~pump~~ equipment costs are low, ~~about $25~~

As an alternative to these pumps, an inexpensive bailer pump can be constructed from readily available materials (Leonard,). This pump consists of a length of garden hose with a foot valve at its bottom end and fittings at its top end that allow a vacuum to be applied to the hose. A water sample is collected by moving the hose up and down, activating the foot valve, and the partial vacuum assists in bringing water to the surface. Vacuum is obtained from an automobile engine. Keep in mind, however, that this sampler should be used where the well contains only a small volume of water ~~so that~~ in order that clearing stagnant water ~~out of~~ from the casing does not become an inordinately time consuming process.

An inexpensive air lift sampler can be constructed from polyethylene or any reasonably flexible tubing as shown in Figure 24. Because the tubing is flexible, it can be readily coiled and moved conveniently from well to well. Primary limitations on the sampler are the amount of air pressure that can be safely applied to the tubing and a source of compressed air. A high-pressure hand pump ~~wold~~ would serve nicely for a shallow water table but a small air compressor may be required for lift greater than 30 feet. The advantage of this sampler is that it can be ~~reduced~~ designed to fit the monitoring well ~~devices as small as 3/4 inch across could be made~~. Of course, the smaller the tubing used the less that can be pumped, but this can be compensated for by pumping longer to obtain the minimum sample volume required for analysis.

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Somewhat more elaborate pumping equipment is required in small-diameter wells where the water level is below suction lift. The easiest, but not the driest, way to collect a sample is to install an airline in the well and blow the water out. However, trying to adjust airflow so that water flows smoothly over the top of the casing instead of blowing violently into the air is a difficult task. Addition of some simple, relatively inexpensive hardware to cap the well can make sampling a straightforward and easy process. Sommerfelt and Campbell (1975) have described such an installation (Figure 25) and Trescott and Pinder (1970) have pumped water from as deep as 190 feet with this type of installation. Air pressure can be supplied from a gasoline powered air compressor, an engine air pump, or a compressed air cylinder. The source of air pressure selected will depend on well site accessibility and budgetary constraints. If a well can only be reached on foot, low-volume, high-pressure hand pumps (available in stores handling racing bicycles) can be used to supply air pressure of up to 160 psi. (Volume, rather than pressure, is usually the key limit)

The best, but most expensive, method of collecting water samples is with a small submersible pump. McMillion and Keeley (1968) have designed a truck portable submersible pump capable of being set at depths up to 300 feet inside a casing as small as 4½ inches ID. with a capacity of 7 to 14 gpm. A portable generator is used to power the pump and the discharge hose spool. With this equipment, a pump can be set as desired in the well and operated until a representative sample can be obtained. If run long enough, sample contamination from the sampling equipment would probably be eliminated by the flushing action of the discharge. Samples could also be collected for dissolved gas analyses, provided there are no air leaks in the pumping system.

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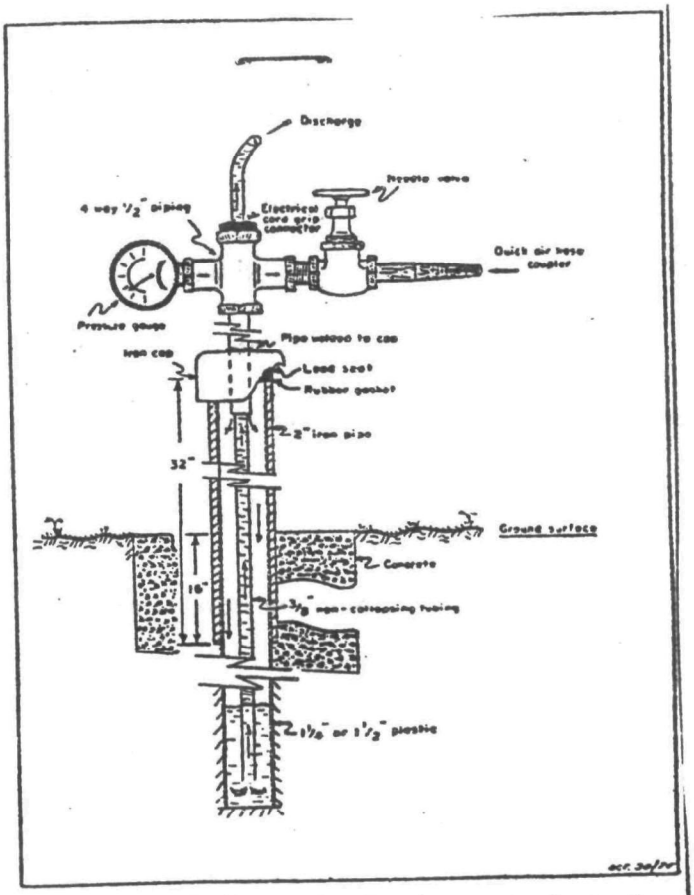

Fig. 2. Schematic diagram showing the construction and mechanism of the pump.

From Sommerfeld & Grimpelli, 1977

Figure 35

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PRELIMINARY

CHAPTER 6

INDICATORS OF LEACHATE

6.1 INTRODUCTION

As can be seen from the composition data presented in Chapter 3, leachate represents an extremely complex system containing soluble, insoluble, organic, inorganic, ionic, nonionic and bacteriological constituents in an aqueous medium. Figure 6-1 schematically depicts an extensive characterization of leachate by means of physical, inorganic, bacteriological, and organic parameters.

In setting up a monitoring program, one must consider all the factors affecting the quality of the pure leachate and "leachate-enriched ground water" and the resultant environmental impact including:

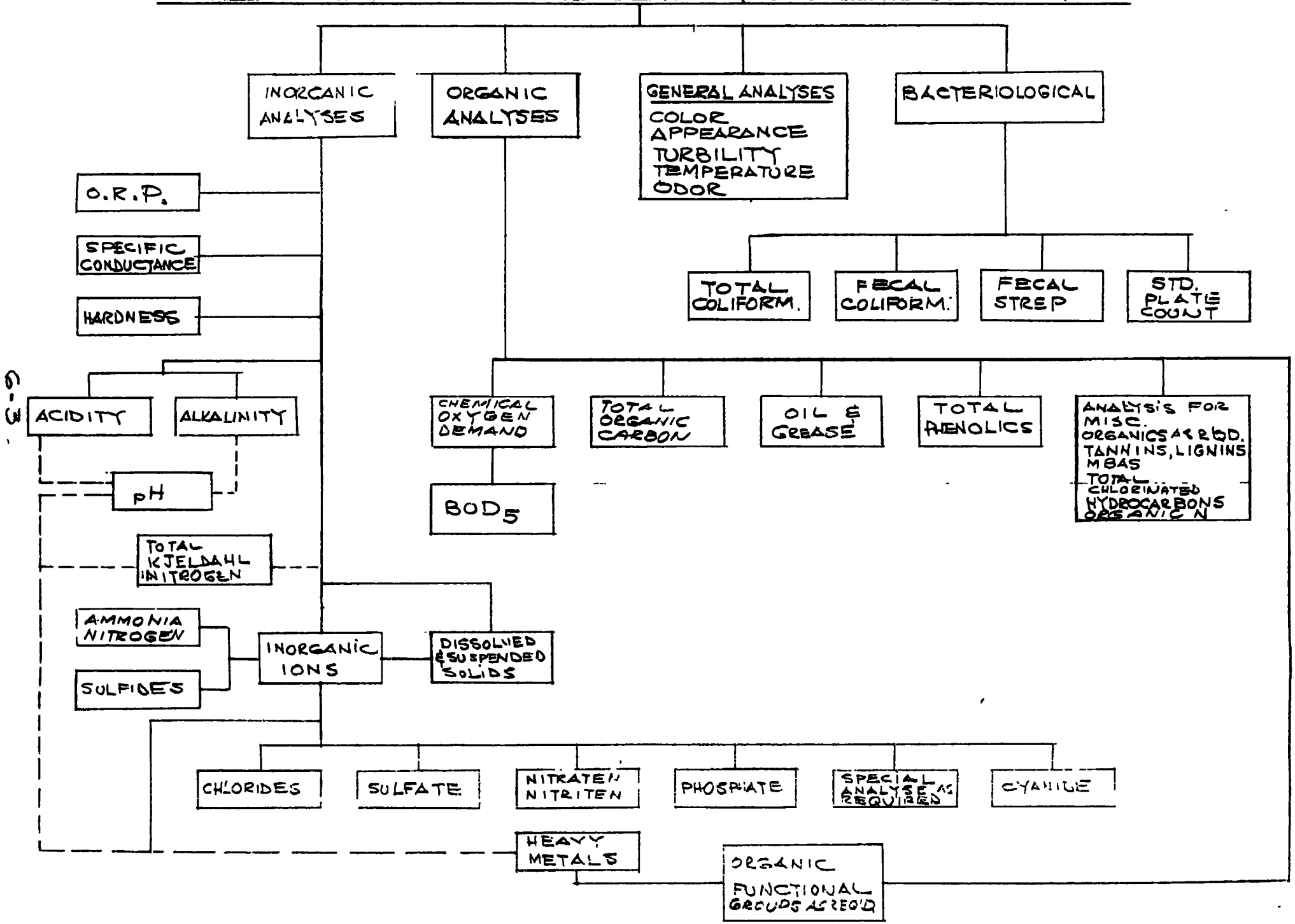
- purpose for monitoring
- background quality of ground water at the site
- other sources of ground-water pollution
- hydrologic conditions of the site and the resultant monitoring network being utilized
- climatologic influences
- costs and availability of manpower and laboratory facilities
- site history

A major item in any monitoring system will be the costs for the analytical measurements. There are several ways these costs may be minimized, yet

meet regulatory requirements, the most important being proper selection of indicator parameters to be monitored.

It will be the function of the regulatory agency to specify monitoring requirements for land disposal sites. Some of the necessary analyses will be time-consuming and relatively expensive. The regulatory agencies should maintain flexibility to consider approval for substitution of a less expensive analytical indicator if the paramenter requiring the more expensive anslysis can be accurately inferred from the simpler, less expensive analytical indicator. An example of this would be the substitution of COD (Chemical Oxygen Demand) analyses for a portion of the BOD_5 (5-day Biochemical Oxygen Demand) analyses if it can be shown that a satisfactory correlation exists between the two parameters. Monitoring land disposal sites can also be looked at in the quality control sense. Here, a regulatory agency can allow, at least for the frequent monitoring, the selection of indicators be subject only to the requirement that a quality control comparison between ground-water quality and permit specifications be made. It is here that a strong effort should be made to utilize inexpensive indicator analyses which can provide quick, accurate and correct information of those indicators requiring more expensive analytical techniques. Promptness of analysis is quite important since having the results for early action will greatly simplify control requirements and sample degradability effects will be minimized. As an example, conductivity can sometimes be used as an indication of total dissolved solids. This is a simple measurement, and one which gives immediate results. It is absolutely necessary, however, to obtain a correlation between the two indicator analyses for the particular land disposal site being monitored.

SCHEMATIC DIAGRAM OF LEACHATE CHARACTERIZATION



This chapter will provide guidelines for selecting indicators as well as scheduling and data management and interpretation resulting in a representative, valid and cost/effective monitoring program. The emphasis of this chapter is on passive monitoring which Chapter 4 defined as the sampling of monitoring devices, strategically located in reference to ground-water flow directions, at regular intervals to determine chemical constituents in the ground water at that point and time. In addition, this chapter assumes that the landfill being monitored includes only normal municipal solid waste. Where special wastes are involved, such as hazardous chemical and liquid wastes, the indicator selection and sample scheduling would be modified accordingly, to be more waste specific.

The presentation in this chapter will be keyed into the fundamentals of leachate and the monitoring networks that were presented in Chapter 3 and 4, respectively.

6.2 BACKGROUND QUALITY OF THE GROUND WATER

The background water quality at a land disposal site must be considered in selecting the indicators for a monitoring program. For example, a ground water with a high background iron content would certainly lessen the value of iron as a leachate indicator, because it will require higher concentrations of iron to differentiate from background. In a given landfill situation, it is necessary to obtain adequate background data in order to draw reliable conclusions regarding possible leachate contamination. Therefore, consideration must be given to both the ground-water quality which occurs in nature, as well as other possible sources of contamination which may affect the background quality.

Reliable data on background quality of ground water can be of critical importance relative to regulatory and legal considerations.

6.2.1 CHEMICAL QUALITY OF NATURAL GROUND WATER

All ground water contains chemical constituents in solution. The kinds and amounts of constituents depend upon the geologic environment, movement, and source of the ground water. Typically, concentrations of dissolved constituents in ground water exceed those in surface waters. This is particularly true in arid regions where recharge rates are low.

Dissolved constituents are primarily derived from minerals in contact with ground water and percolating water going to ground-water recharge. Common chemical constituents of ground water include:

Cations	Anions	Undissociated
Calcium	Carbonate	Silica
Magnesium	Bicarbonate	
Sodium	Sulfate	
Potassium	Chloride	
	Nitrate	

Table 6-1 lists relative abundances of these and other chemical constituents in natural ground water. Minor and trace constituents are present selectively depending upon the mineralogy of the region. Analyses of ground water samples enriched in silica, iron, calcium, and sodium are given in Table 6-2. These elements are frequently enriched in ground water. Brines and thermal spring waters were not included in Table 6-2.

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TABLE 6-1

RELATIVE ABUNDANCE OF DISSOLVED SOLIDS IN POTABLE WATER [7]

Major Constituents (1.0 to 1000 ppm)	
Sodium	Bicarbonate
Calcium	Sulfate
Magnesium	Chloride
Silica	
Secondary Constituents (0.01 to 10.0 ppm)	
Iron	Carbonate
Strontium	Nitrate
Potassium	Fluoride
Boron	
Minor Constituents (0.001 to 0.1 ppm)	
Antimony	Lead
Aluminum	Lithium
Arsenic	Manganese
Barium	Molybdenum
Bromide	Nickel
Cadmium	Phosphate
Chromium	Rubidium
Cobalt	Selenium
Copper	Titanium
Germanium	Uranium
Iodide	Vanadium
	Zinc
Trace Constituents (generally less than 0.0001 ppm)	
Beryllium	
Bismuth	
( )	Silver
Cesium	Thallium
Gallium	Thorium
Gold	Tin
Indium	Tungsten
Lanthanum	
Platinum	Zirconium

Ground-water quality is classified according to domestic and industrial use on a simplified basis for convenience. Salinity, the concentration of total dissolved solids, and hardness, the combined calcium and magnesium concentrations, are classificatory criteria. The classification scheme is shown on Table 6-3. Water with a high concentration of dissolved solids can build up scale in boilers, be harmful to plants when used for irrigation, and interfere with quality of products in manufacturing. Hard water also builds up scale deposits in boilers, and forms scums with soap in laundering.

Within a large body of ground water, the natural chemical composition tends to be relatively consistent. Variation of ground water with time is minor in comparison with surface-water quality changes.

Ground water under natural conditions tends to increase in salinity with depth. Most of the geologic formations in the United States are underlain by brackish to highly saline waters. Density and permeability differences act to maintain a separation between these waters and the overlying fresh ground water.

6.2.2 OTHER SOURCES OF GROUND-WATER CONTAMINATION

It should be evident from this discussion that ground-water composition can vary widely under natural conditions. Man's activities add another dimension to the complexity of ground-water quality. The effect on ground-water quality of point sources of contamination such as waste lagoons, acid mine spoils, and oil well brines are relatively easy to trace. Diffuse sources of contamination such as regions of septic tanks, irrigation, or farm chemical usage may affect bodies of ground water creating a chemical enrichment which is relatively uniform. Detection of point-source contamination

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Table 6-2 ANALYSES OF GROUND WATER IN WHICH THE INDICATED ELEMENT IS A MAJOR CONSTITUENT. CONCENTRATIONS IN MG/L UNLESS NOTED (4) ?

Constituent	Silica	Iron	Calcium	Sodium
Silica (SiO_2)	99	20	29	16
Aluminum (Al)	--	--	--	--
Iron (Fe)	0.04	2.3	--	0.15
Calcium (Ca)	2.4	126	636	3
Magnesium (Mg)	1.4	43	43	7.4
Sodium (Na)	100	13	17	857
Potassium (K)	2.9	2.1		2.4
Carbonate (CO_3)	24	--	--	57
Bicarbonate (HCO_3)	111	440	143	2,080
Sulfate (SO_4)	30	139	1,570	1.6
Chloride (Cl)	10	8.0	24	71
Fluoride (F)	22	0.7	--	2.0
Nitrate (NO_3)	0.5	0.2	18	0.2
Dissolved solids				
Calculated	348	594	2,410	2,060
Residue	--	571	--	--
Hardness as $CaCO_3$	12	490	1,760	38
Noncarbonate	0	131	1,650	0
Specific condustance				
umhos/cm @25	449	885	2,510	2,960
pH units	9.2	7.6	--	8.3
Temperature, °C	50.0	13.3	--	--

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Table 6-3 CLASSIFICATION OF WATER

NAME	Concentration of total dissolved solids ppm mg./L.
Based on salinity	
Fresh	0-1000
Brackish	1000-10,000
Salty	10,000-100,000
Brine [1,7]	over 100,000
Slightly saline	1000-3000
Moderately saline	3,000-10,000
Very saline	10,000-35,000
Briny [4]	over 35,000
Based on hardness	Hardness as $CaCO_3$) ppm
Soft	0-60
Moderately hard	61-120
Hard	121-180
Very hard [4]	over 180

within a region of general contamination will require relatively higher concentrations of contaminants in comparison to comparable uncontaminated areas.

No estimate of background concentrations of leachate indicators typical for a geohydrological setting could be used in lieu of actual measurements. The only way to ascertain the ground-water quality at a given site is to measure it. However, man's activities, as previously mentioned, can influence ground-water quality. A description of contaminants associated with certain of these activities could be helpful as it would point out those indicators which need to be delineated from unnaturally high background concentrations in order to trace leachte-enriched ground water.

The following sources of ground-water contamination are listed in Table 6-4 which summarizes their potential contributions of ground-water contaminants. Their similarities to and distinctions from leachate should be carefully noted so that interferences will be recognizable.

Highway deicing - Over 6,567,000 tons (5,962,000 tonnes) of deicing salts were used nationwide in 1966-7. (1) The most common salt in use is sodium chloride, with calcium chloride use amounting to only 4 percent of that of sodium chloride.

Open storage of salt or salt/sand mixtures may result in leaching of salt with rainwater. The leachate after reaching ground water, will form a plume of salt enriched ground water which could contaminate wells in the vicinity. On the other hand, spreading of salt on the road results in a more diffuse salt-enrichment of ground water. Wells located near major highways have been affected by deicing salt.

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Table. 6-4 CONTRIBUTION OF LANDFILL LEACHATE INDICATORS TO GROUND WATER BY OTHER SOURCES

Indicator	Highway deicing	Leaky sewers	Septic tanks	Mining	Irrigation	Land disposal sludge	Petroleum expl & dev	Feedlots
Phosphate		M	M	P		P		P
Calcium	M				M	L	M	
Magnesium					M	L		
Sodium	H	L	L		M	L	H	
Potassium					M			
Ammonium		M	M					
Chloride	H	L	L		H	L	H	L
Sulfate		M	M	H	M		M	
Nitrate		H	H		L	H	M	H
Bicarbonate		M	M					
Iron		H	M	H		H		
Manganese				H		H		
Boron		L	L					
Selenium								
Zinc				M		M		L
Copper				M		L		
Lead				M		L		
Other h.m.				P		P		P
MBAS		M	L					
Phenols		P				P		
PCB						P		
Org N		M	M			M		L
PAH-HC						P		
TOC		H	H			L		M
BOD		H	M					M
Coliform		P	P			P		P
Virus		P	P			P		P

H = High
M = Moderate
L = Low
P = Potential

Leaky sewers - Sewer pipes which have been in service over a period of years are likely to be leaky. Sewage gases form acids which dissolve concrete and mortar, usually the substance of older sewer pipes. When the pipes are located in the unsaturated zone, raw sewage may leak and percolate to the ground water.

Sewage contains some inorganic salts, sulfur, nitrogen, trace metals, and suspended and dissolved organic compounds. Sulfur and nitrogen are generally present as sulfide and ammonia. After entering the zone of aeration, these ions are oxidized to sulfate and nitrate. Thus, sulfate and nitrate are associated with leaky sewers in Table 6-4. The organic matter exerts a large BOD and COD. Enteric organisims, bacteria and viruses, are present in large numbers creating a potential for biological contamination. It has been estimated that approximately 500 million gallons of sewage is lost annually in the U. S. through leakage. (2)

Septic tanks - Contaminants carried to ground water in percolating septic tank effluent are similar to those from leaky sewers. The major difference is that septic tanks have provided an opportunity for some anaerobic decomposition. Thus, MBAS and BOD levels are reduced from those of raw sewage. Again, percolation of effluent through the zone of aeration converts ammonium and sulfide to nitrate and sulfate. A reduction of dissolved oxygen in the percolate or ground water by a high BOD flow can cause dissolution of iron, hence the iron rating in Table 6-4.

Unless the septic tank is within a couple of feet of the water table, bacterial contamination should not be a problem. However, a septic tank located in coarse-textured soil overlying a fractured rock aquifer might

cause considerable bacterial and viral contamination.

Septic tanks in a density of one per acre or less are not likely to significantly influence regional ground-water quality. As density increases, these point sources of contamination blend together and result in a general degradation of regional ground water.

Mining - A variety of contaminants are generated in leach-mining and ore beneficiation which are specific to the mineral type and mine location. Locations generating these contaminants would be obvious, so it is not necessary to deal with them here.

A more general contamination problem is generated by wastes from strip and shaft mining particularly as these methods pertain to coal. In strip mining, overburden must be removed to expose coal or ore seams. The coal is separated from waste rock and is washed. The wastes produced in these processes are termed spoils and gob piles.

Frequently, the waste rock and mineral contains pyrite (FeS_2), an iron sulfide mineral. When exposed to air, pyrite oxidizes with the help of iron-oxidizing bacteria. The oxidation produces sulfuric acid which keeps iron in solution and frequently dissolves other heavy metals from waste minerals. Drainage water from strip mine spoils or from mine shafts may have a pH of less than 2. This acid mine drainage kills fish in surface waters and produces a red-yellow scum that is unsightly.

The acid character of the spoils and gob piles prevents plants from covering them. The resulting erosion continuously exposes new pyrite to oxidation. Millions of tons of sulfuric acid are introduced into the environment each

year from acid mine drainage. Large regions of Pennsylvania, West Virginia, Ohio, Indiana, and Illinois have been contaminated from coal mining wastes. Colorado has a similar problem from abandoned metal ore mines.

Acid water can contaminate ground water with a variety of heavy metals which it dissolves. Reduction of the ground-water pH also occurs creating a more corrosive medium. Iron and/or manganese accompanying acidic water add a metallic taste to water and cause staining of plumbing fixtures and when it is used in laundering.

Irrrigation - Ground or surface water used for irrigation becomes more mineralized as it percolates through soil and dissolves mineral and fertilizer constituents. Irrigation water going to recharge carries an enrichment of some or all of the following ions: calcium, magnesium, sodium, potassium, chloride, sulfate, and nitrate.

Continued irrigation increasingly mineralizes ground water. This enrichment may become limiting to further ground water use. In California, there are closed basins where ground water has been recycled by irrigation and has become so mineralized it is approaching the limit of usefulness.

Land disposal of sludge - The literature on sludge chemistry has reported all of the indicators listed in Table 6.4 as being present in one sample or another. Concentrations range from parts per billion to percentages.

Sludge is applied to the land surface. Therefore, its influence on ground-water quality is determined by the transport of its constituents through the soil and underlying unsaturated zone. The contribution of landfill leachate

indicators to ground water is calculated on the basis of sludge leachate having undergone reactions in the soil and unsaturated zone.

Ammonium in sludge will nitrify and the portion that leaches will move as nitrate. Most of the heavy metals and phosphate will probably be retained in the soil and be in extremely low concentrations in percolate.

Bacteria have been studied after sludge application to land. Fecal coliforms exhibit a die-off rate which reduces their number to a negligible population in a matter of 2-3 weeks. Movement through soil is usually no more than a few centimeters. Viruses have been shown to be more in soil, but are not likely to be a serious contaminant if digested sludge is used.

Petroleum exploration and development - Brines are almost universally associated with oil deposits. They are sometimes produced in greater quantities than crude oil especially from older fields. Brine pits are the most common waste disposal facilities. In theory, water evaporates leaving salt accumulations. In practice, frequently brine leaks from the pit and carries high concentrations of salt into underlying aquifers. Sodium and calcium are the most common cations, with chloride, sulfate, and nitrate as the most common anions. Some brine contains enough bromide for economic recovery. Brine may also be used as a secondary recovery injectant. As an indication of the extent of the problem, over 400 billion gallons of brine were produced in the U. S. in 1974. (3)

Feedlots - Ground-water contamination from feedlots occurs principally from leaching of nitrate. Some mineralization of infiltrating water may also occur, but not usually to an extent that serious contamination results.

Phosphate is a serious pollution hazard to surface water from feedlot runoff. However, phosphate is retained in soil and doesn't usually move into ground water. Of the heavy metals, zinc is present in the highest concentration in manure. None of the heavy metals are in concentrations as high as those associated with municipal sludges of mixed domestic and industrial origin. No appreciable contribution of manure contained heavy metals to ground water is anticipated.

There are a couple of sources of ground-water contamination for which it is difficult to assign specific chemical constituents. Waste lagoons and oxidation ponds is one of these categories (Table 6.5). Such facilities may contain almost anything of an inorganic or organic type imagined. Therefore, in listing leachate indicators, probabilities are given of their occurrence in lagooned and ponded waste leakage as it enters the ground-water system.

Buried pipelines and tanks are another source of ground-water contamination. Probably the most common contaminants from these sources are petroleum products. Chemical storage tanks have also been documented as contributors of contamininants to ground water. Again, the probabilities shown in Table 6.5 represent the probabilities of the given indicator actually reaching ground-water.

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Table 6.5 PROBABILITIES OF LANDFILL LEACHATE INDICATORS FROM GIVEN SOURCES CONTAMINATING GROUND WATER

Indicator	Waste lagoons and ponds	Buried pipelines and tanks
Phosphate	II	III
Calcium	III	III
Magnesium	III	III
Sodium	I	II
Potassium	III	III
Ammonium	II	III
Chloride	I	II
Sulfate	I	II
Nitrate	I	II
Bicarbonate	III	III
Iron	I	III
Manganese	I	III
Boron	II	III
Selenium	II	III
Zinc	II	III
Copper	II	III
Lead	II	II
Other h.m.	II	II
MBAS	III	III
Phenols	I	I
PCB	II	III
Org N	II	III
PAH-HC	III	I
TOC	II	I
BOD	II	I
Coliform	III	III
Virus	III	III

I = Highly probable
II = Probable
III = Unlikely

TABLE 6-6

LEACHATE INDICATORS

APPEARANCE	
pH	
Oxidation-Reduction Potential	
Conductivity	PHYSICAL
Color	
Turbidity	
Temperature	
Odor	
Phenols	
Chemical Oxygen Demand (COD)	CHEMICAL
Total Organic Carbon (TOC)	
Volatile Acids	Organic
Tannins, Lignins	
Organic-N	
Ether Soluble (oil & grease)	
MBAS	
Organic Functional Groups As Required	
Chlorinated Hydrocarbons	
Total Bicarbonate Solids (TSS, TDS)	
Volatile Solids	
Chloride	
Sulfate	
Phosphate	
Alkalinity and Acidity	
Nitrate-N	
Nitrite-N	Inorganic
Ammonia-N	
Sodium	
Potassium	
Calcium	
Magnesium	
Hardness	
Heavy Metals (Pb, Cu, Ni, Cr, Zn, Cd, Fe, Mn, Si, Hg,As, Se, Ba, Ag)	
Cyanide	
Fluoride	
Biochemical Oxygen Demand (BOD)	BIOLOGICAL
Coliform Bacteria (Total, fecal; fecal streptococcus)	
Standard Plate Count	

6.3 CHEMICAL, PHYSICAL AND BIOLOGICAL INDICATORS

A comprehensive listing of leachate indicator parameters has been prepared and presented in Table 6.6. This listing is based on the composition of leachate which was presented in Chapter 3 and reflects the most widely used leachate indicators by researchers in the field and state regulatory agencies. (4)

The schematic diagram on Figure 6-1 and the list of leachate indicator parameters in Table 6-6 represent the principle undesirable characteristics of leachate from MSW. Its deleterious effects on ground and surface waters become apparent. Just some of the effects include:

1. Soluble organics and some inorganics causing dissolved oxygen depletion in surface waters.
2. Soluble constituents that result in objectionable tastes and odors in water supplies.
3. The obvious health hazards connected with toxic materials and heavy metal ions and microbiological contaminants in excess of drinking water standards.
4. The effects of dissolved solids in excess concentrations limiting the use of ground and surface waters for drinking domestic, industrial or recreational use.

These examples all point to the basic need for monitoring many of these parameters. The reader is referred to the EPA Handbook for Monitoring Industrial Wastewater (5) and to the introductory remarks in Standard Methods (6) for further discussion and background information regarding the undesirable nature and potential effects of the various leachate indicator parameters.

The actual selection and use of indicators for a particular monitoring program will generally be decided from the indicators on Table 6-6 and will depend upon a number of considerations.

1. Type of Monitoring Network - I, II or III as was presented in Chapter 4.
2. Susceptability to attenuation.
3. Background water quality.
4. Location of well being sampled - "A" Wells, "B" Wells, or "C" Wells.
5. Purpose of Monitoring.
6. Other considerations including cost,regulatory standards to be met, availability of laboratory equipment and manpower, simplicity and precision of determination.
7. Type of refuse handled and other site-specific factors.

6.4 INDICATOR GROUPS

Indicators can be categorized into various groups or levels of monitoring which vary in degrees of information obtained in relationship to the purpose for the monitoring. Three such levels widely used by researchers, engineers and regulatory agencies are:

- Specific Conductance Measurements
- Key Indicator Analyses
- Extended Indicator Analyses

6.4.1 SPECIFIC CONDUCTANCE MEASUREMENTS

For monitoring ground-water quality and its fluctuations over a period of time, specific conductance is a useful parameter for approximating the total

amount of inorganic dissolved solids. The real value of specific conductance is that it can be performed easily and quickly, requiring little training, with portable field equipment which is relatively inexpensive, accurate and reliable (approximately $200.00).

Specific conductance has been successfully correlated with total dissolved solids for monitoring leachate-enriched ground water. It also has been used successfully to detect fluctuations and trends for ionic impurities in the ground water. If conductivity is being used as an indicator of total dissolved solids, it is absolutely essential that a correlation be obtained for the specific land disposal site being monitored. Otherwise, gross errors can be expected in data interpretation.

Availability of a conductivity meter at a site would allow an operator to "spot check" monitoring wells at very frequent intervals (say weekly or monthly). It can also be used advantageously during the sampling of a site to prioritize wells to be sampled, where time and budget restrictions are a problem.

6.4.2 KEY INDICATOR ANALYSES GROUP

The intent of this monitoring group is to include highly sensitive analytical parameters, which can be performed rapidly and accurately, at relatively low cost, by personnel of minimum training, to yield reliable, useful data. One should select a group of parameters which will provide information regarding ionic, nonionic, inorganic, organic and suspended constituents of the ground-water sample. Most of the parameters should lend themselves to field analysis using portable equipment. Field analyses have the obvious advantage of eliminating the necessity for low temperature and/or chemical preservation of the sample, therby minimizing labor and deterioration effects on the analysis which can result from sample degradation due to aging. This

monitoring group can be performed at frequent intervals, with low cost, manpower and equipment requirements.

The final selection of analytical parameters must consider the background water quality, the pure leachate quality, as well as the hydrogeologic influences. The group must therefore be site specific as well as remain flexible to change at a site as may be dictated by data interpretation.

For discussion purposes, the following list of key indicators has been widely used in the field to determine the presence of leachate:

- Specific Conductance
- pH
- Temperature
- Chloride
- Iron
- Color
- Turbidity
- COD

This group fits well the criteria for key indicators and, with the exception of COD, can be performed rapidly using portable field equipment.

It is not suggested that all of the indicator parameters mentioned in this list must necessarily be used together to determine the presence of leachate. Rather, this is to be left to the judgment of the individual analyst. It is possible, for instance, that results from just one of the analyses (i.e. specific conductance) could indicate the probable presence of leachate. A decision would then be made whether to run some or all of the remaining parameters, or additonal tests to determine the reason for the high conductance value.

Data obtained from indicator analyses have value in and of themselves, that is, individual determinations will give valuable information regarding the possible presence of leachate. In addition, data obtained from several indicator analyses can be cross-correlated and interpreted so that even more insight can be gained about the nature of the contamination, over and above what is obtained from the individual tests.

The following hypothetical examples serve to illustrate this point:

Example 1. A sample of ground water is analyzed and yields the following results:

High levels of color and COD.

Low levels of iron, turbidity, and conductance.

These results could be interpreted as an indication of the presence of an appreciable concentration of colored organic contamination in a system which is low in soluble and suspended inorganic contaminant levels.

Example 2. A sample of ground water is anlayzed and yields the following results:

High levels of conductance, chloride, pH, and turbidity.

Low levels of COD, color, and iron.

These results could be interpreted as an indication of the presence of an appreciable concentration of inorganic materials, both suspended and in solution and a low concentration level of organic materials.

Example 3. A sample of ground water is analyzed and yields the following results:

High levels of conductance, chloride, iron, color, and COD.

Low levels of turbidity and pH.

These results could be interpreted as an indication of the presence of appreciable concentrations of both inorganic and organic contaminants in acid solution and very low levels of suspended materials.

Further interpretation of the indicator data must then consider background water quality, hydrogeology, attenuation and other pollution sources in the vicinity of the landfill in order to determine whether the presence of leachate is indicated.

6.4.3 EXTENDED INDICATOR ANALYSES GROUP

This monitoring group is a much more comprehensive group of analytical parameters. Table 6-6 presents a comprehensive extended indicator analyses group which provides for a good characterization of the water sample and represents indicators commonly used by researchers and required by many regulatory agencies.(4) Performance of this monitoring group will obviously be costly, requiring trained personnel and an adequately equipped sanitary laboraboty. Very few of the parameters can be analyzed with portable field equipment thus requiring the utilization of acceptable storage and preservation techniques.

There can be a number of reasons for performing extended indicator analyses. The main reason is the need to perform additional analyses as a result of problems which become known from data provided by the key indicator analyses group.

Additional testing, whether instituted by a regulatory agency or the landfill operator, should always be approached conservatively from both a technical and cost standpoint. Arbitrarily requiring an extended program without reasonable technical justification results in a very costly undertaking with little or no regard to its cost benefit effecitveness.

An extended analysis program can only be justified when it can be demonstrated that a basic indicator program does not have the necessary and sufficient capability to assure the absence of leachate contamination or to provide enough information to solve a specific contamination problem or yield required background quality information. When background quality information is being developed, a relatively large number of analytical parameters should be investigated in order to choose the few most valuable ones which will constitute the key indicator analysis program.

A sudden radical change in a key indicator may also point to the necessity for extended analytical work. For example, indicator data might suddenly indicate contamination by a non-specific organic material, as indicated by an elevated COD value. In this case, the indication would be to perform a number of analyses, such as wet chemical tests or even infra-red and gas chromatography (if equipment is available), or in order to determine which specific compound or compounds caused the change in the COD. From a regulatory standpoint, the nature of an extended analysis program would also be related to the standards which have been set to assure the absence of leachate contamination. For example, suppose that a state regulatory agency decides to adopt as its enforcement standard the U. S. Public Health Service Drinking Water Standards of 1962. This would result in an extended analysis

program which would at least require testing for twenty-four parameters, four physical and twenty chemical.

As another example, suppose that a state regulatory agency decides to adopt as its enforcement standards a series of seven physical and chemical parameters, (i.e. - pH, specific conductance, chemical oxygen demand (COD), chloride, iron, color and turbidity). This program would per se constitute an adequate key indicator program, thereby eliminating the need for an extended analysis program for regulatory purposes.

Thus, it can be seen that the concept of an extended analysis program is a relative one. It is usually relative to regulatory requirements, or the need for data, in addition to the key indicator program or specific contamination problems. It must, therefore, be the task of the responsible engineers and analysts to determine what will constitute an extended analysis program, if any, for a given landfill site.

6.5 GUIDELINES FOR USING INDICATORS

For a given land disposal site, the selection and use of indicators will vary with the background water quality, the differential attenuation that may occur and the well being monitored.

6.5.1 BACKGROUND WATER QUALITY MONITORING

6.5.1.1 New Land Disposal Site

For a new land disposal site, the background monitoring will define the naturally occurring constituents in the ground water and contaminants from other possible pollution sources that may be in the area. Section 6.2 presents a good summary of the background quality one might expect to find in different geologic settings and with a variety of "other pollution sources." Usually,

the background quality at a new site can be satisfactorily defined by performing an extended indicator analysis group on an "A" well(s) that has been installed at the site for this purpose. Section 4.2 discusses various types of "A" wells in the different monitoring networks. It may be desirable to have additional "A" wells if other pollution sources that may have a significant impact on the ground-water quality at the site are suspected where there is more than one water-bearing zone to be monitored.

For the first sampling, the extended indicator analyses group should include, as a minimum, all of the parameters on Table 6-6. Additional parameters may be deemed desirable where applicable to define "other pollution sources" in the area. In the case of the latter, the characteristics of other pollution sources should be investigated in selecting any additional parameters for monitoring. It is desirable to perform a few samplings (say 3 or 4) to obtain a more statistically reliable data base for the long-term monitoring prior to commencement of operations. As is usually the case, this is not done due to time and economic considerations. Therefore, it is suggested to at least collect additional data on a few selected key indicators which will likely be used in the long-term monitoring. These can be done quickly and cheaply and will provide valuable data in developing a statistically reliable data profile (see Section 6.6).

While the "A" wells will establish the background water quality for the site, it is also important to develop background quality data for each "B" and "C" well. This is more important for larger landfills where the "A", "B", and "C" wells are relatively far apart in respect to changes in geology and other pollution sources which may influence their quality. Therefore, it is

recommended that an extended indicator analysis group be performed on all monitoring wells as soon as possible after their installation.

6.5.1.2 Existing Land Disposal Site

For an existing land disposal site, where solid waste has already been landfilled, the background quality monitoring should include leachate contamination that may have already occurred on the site. Monitoring in this case must then involve the "A", "B", and "C" wells installed at the site, where as before the "A" well will define the natural background quality, while the "B" and "C" wells define existing leachate contamination.

Here, the quality of the "B" well becomes especially significant in selecting the key and extended indicators for the monitoring program. This well will detect the leachate contaminants that are entering the saturated zone. The analysis of this well will also provide valuable information about the unknown past history of a site. For example, an old site may have disposed of hazardous wastes in the past which could result in significant concentrations of exotic contaminants not normally attributed to municipal solid waste (i.e. certain heavy metals or pesticides). Depending upon the extent of the problem and the degree of potential hazard to the public, a decision can then be made as to whether to include additional parameters into the monitoring program as key or extended indicators. In any event, analysis of the "B" well will represent the "worst case" in terms of leachate contamination at a particular site, and can provide a basis for including or excluding indicator parameters.

6.5.2 ON-GOING MONITORING

The on-going monitoring program should consist of the judicious use of repre-

sentative key and extended indicator analyses groups, the former being run at more frequent intervals and the latter less frequently for verification purposes. The key indicator group is designed for the primary purpose of determining presence or absence of leachate contamination and as a "check" on quality fluctuations. The extended indicator group is designed to provide verification of non-specific key indicators (i.e. COD or specific conductance) and for legal purposes in an enforcement action.

The on-going program will, of course, involve the monitoring of the "A", "B" and "C" wells. After establishing some background quality data for the various wells (whether it be one or a series of samplings), the most representative indicators for the site should be selected. In doing this, one must consider the background quality data, the constituents of the leachate, and the potential influence of attenuation. The information presented in Chapter 3 on attenuation, and in Chapter 6 on background water quality provides some valuable guidelines for selecting indicators.

As an example, suppose the natural background quality is high in iron or total dissolved solids. In this case, the value of iron and specific conductance as key indicators of leachate contamination is lessened because of the high concentrations that would be required to distinguish from background. Or, there may be another pollution source in the vicinity that is affecting the background quality of the ground water, such as deicing of adjacent highways, or local septic tanks, or leaky sewers. These, too, might serve to lessen the value of various parameters as leachate indicators. Section 6.2.1 presents some very useful information on background water quality which should be used as a guide in indicator selection.

Susceptibility to attenuation in different soils would also affect the value of the various parameters as indicators of leachate. The information presented in Section 3.2 and Table 3-2 can be used as a guide in indicator selection for a particular disposal site. Table 3-2 points out the significance of chloride as an indicator due to its freedom from attenuation.

The key indicator program should be followed as long as no presence of leachate is detected or where leachate is already present, no significant fluctuations in the data are observed. Another consideration here might also be the regulatory agency's requirements, as many states do require a periodic (say, annual) testing for an extended analysis group, regardless of the monitoring trends being observed. In most cases, all "A", "B", and "C" wells at the site should be included in the key indicator program. However, this is not to be considered an ironclad rule of thumb. Many large acreage land disposal sites may have as many as 20 monitoring wells. In these cases, one may elect not to sample all 20 wells at each sampling, but to rotate sampling to include say 5 wells each sampling date. Therefore, in the case of quarterly sampling, each well would be sampled once per year. In this case, each sampling should include at least one of each well type, that is, an "A", "B", and "C" well.

The convenience of the specific conductance test can be a valuable asset in this case. Ability to be tested quickly, with a field instrument may allow at least a specific conductance reading on all 20 wells at each sampling date. The need to pump out the well prior to sampling would limit the use of the specific conductance given time restrictions. The specific conductance reading provides the added dimension of deciding on the spot which wells should receive priority for sampling. Having the specific conductance data

profile on hand for quick reference, the field technician can compare today's reading with the profile which may show a significant change and worthy of further investigation. So, a combination of a routine rotational sampling schedule, subject to possible modification due to a significant change in specific conductance, would comprise a sound rationale to the use of the key indicator group.

If a significant change is observed in a key indicator parameter or parameters (i.e. - increase in specific conductance), the possibility of leachate contamination should be suspected and a plan for further investigative and corrective action should be instituted immediately. This plan should include additional sampling and analytical work as determined by the key indicator data obtained, to serve as a data base for developing a satisfactory correction action to eliminate the cause of the contamination problem and to monitor the effect of implementing the same.

Assuming that a leachate contamination problem has been discovered by the key indicator program and corrective action implemented, sampling and indicator testing should be instituted on an increased level of frequency, with the possible inclusion of additional parameters. This should be continued until there is reasonable certitude that conditions have returned to normal and will probably remain so. At this point, a re-assessment of the key indicator analytical program should be made. It should then be decided whether to re-institute the program in its original form or to initiate it in a modified form, based upon experience gained in solving the contamination problem.

The following example will serve to illustrate this approach:

Suppose that sulfate was found to be a principal contributing contaminant which had caused a high specific conductance reading.

It might then be decided to test for sulfate, for a limited time, in addition to the other parameters of the key indicator program. Valuable data could be collected in this manner to show sulfate/conductance ratios which could be used as a guide in monitoring for future problems.

Obtaining background quality data and further investigating a particular problem are the principal technical reasons for implementing an extended indicator analyses group. In the legal sense, enforcement data needs may require extended indicator analyses data. Administratively, many regulatory agencies will require (say, annual or bi-annual) sampling for an extended indicator analyses group.

In any case, the conservative use of the extended indicator parameters should be kept in mind due to the relatively excessive cost and manpower requirements associated with their performance. The fact that the extended indicator parameters are basically serving to verify the results of the key indicator parameters, should provide a basis of a rationale for selecting and ranking the former. In other words, a significant fluctuation in a particular key indicator would warrant further investigation by a select group of extended parameters and not automatically the entire extended indicator analyses group. The implementation of the extended indicator analyses group should be a monitoring well specific decision. For example, a quality change in one "C" well should not immediately require the testing for extended indicators in all of the "C" wells.

The following examples of relationships between key and extended indicators serve to illustrate the point :

1. Specific Conductance:

A significant change in specific conductance would be an indicator of possible changes in levels of one or more of the following extended indicator parameters: pH, total dissolved solids, chloride, sulfate, phosphate, alkalinity, acidity, nitrogen series, sodium, potassium, calcium, magnesium, hardness, heavy metals, cyanide, fluoride, and COD.

2. A significant change in chloride concentration would be an indicator of possible changes in levels of one or more of the following extended indicator parameters: specific conductance, total dissolved solids, pH, acidity, and metal ions.

3. A significant change in iron (total) concentration would be an indicator of possible changes in levels of one or more of the following extended indicator parameters: specific conductance, pH, total dissolved solids, chloride, sulfate, phosphate, manganese, and fluoride.

4. A significant change in color would be an indicator of possible changes in levels of one or more of the following extended indicator parameters: COD, TOC, tannins, lignins, organic N, total dissolved solids, pH, iron, BOD and conductance.

5. A significant change in turbidity would be an indicator of possible changes in levels of one or more of the following extended indicator parameters: pH, conductance, COD, TOC, tannins, lignins, total suspended soilds, phosphate, alkalinity, acidity, calcium, magnesium, hardness, heavy metals, fluoride, and BOD.

6. A significant change in COD would be an indicator of possible changes in levels of one or more of the following extended parameters: BOD, pH, conductance, TOC, volatile acids, tannins, lignins, organic-N, total dissolved solids, total suspended solids, volatile solids.

In a practical situation, several of the key indicators will most probably show variations at the same time. Therefore, looking at combinations of key indicators will provide additional information for the analyst to define the chemistry of the system involved and further assist him in specifying additional extended indicators for analysis.

6.6 MONITORING FREQUENCY

The sampling schedule for a land disposal site should maintain flexibility for modification. Monitoring frequency is greatly influenced by many factors as listed below:

1. Characteristics of ground-water flow.
2. Location and purpose of the particular monitoring well.
3. Climatological characteristics.
4. Trends in the monitoring data.
5. Legal and institutional data needs.
6. Other considerations.

Obviously, the hazardous nature of the leachate and what is being threatened (i.e. a single domestic well versus an entire municipal water supply) will to a large degree dicatate the monitoring effort.

6.6.1 CHARACTERISTICS OF GROUND-WATER FLOW

The principal characteristic of concern in selecting a sampling frequency is the rate of ground-water flow at the land disposal site. As was discussed in Chapter 4, the flow rate will be primarily dependent on the aquifer porosity, permeability as well as the hydraulic gradient existing at the site. The aquifers were generally categorized by porosity into intergranular porosity, fracture porosity, and solution porosity with ground-water flow rates ranging in orders of magnitude from a few feet per year in an impervious intergranular porosity aquifer to tens of feet per day in the more unpredictable fracture and solution porosity aquifers.

The higher the rate of ground-water flow, warrants more frequent monitoring. Two extreme examples would be an intergranular porosity aquifer with impervious clay soils and a fracture or solution porosity aquifer with unpredictable and high flow rates likely. For an example, suppose the closest "C" well is 100 feet from the landfill and the closest downgradient property line or domestic well is 300 feet away. At the site with the clay soils, it would be senseless to do frequent sampling if, theoretically, it would take ten to twenty years for any leachate-enriched ground water to even reach the well. Here, after establishing background quality, an annual or bi-annual monitoring of the well with select key indicator parameters would suffice. In the latter case, however, it is possible that contaminants could migrate off of the property in a matter of weeks or months. Here, a quarterly monitoring with key indicators, with, perhaps, a more frequent (daily) spot checking with specific conductance would be warranted. As was discussed earlier in the Chapter, the extended indicator group would be utilized as needed.

Most landfills will fall between these two extremes, but one can see that careful consideration must be given to the flow rate and the distances involved to select a frequency which will not miss an environmental occurrence.

In a similar sense, the monitoring well would be influenced by the vertical flow rate of leachate-enriched ground water. For example, suppose a disposal site is underlain by a sand aquifer with a relatively high ground-water flow rate, but it is separated from the landfill by a thick layer of impervious clay. Here, a concentrated monitoring effort of the "C" wells in the aquifer would not be justified until the "B" well detected that contaminants have travelled through the clay layer and reached the aquifer.

6.6.2 LOCATION AND PURPOSE OF THE MONITORING WELL

The distance that a monitoring well is located from the land disposal site and its depth will influence monitoring frequency. For example, there may be a case where a line of "C" wells are placed along the property line for legal and administrative reasons due to ground-water protection laws. There is little need to concentrate on monitoring these wells until the monitoring results of closer "C" wells presents some reason to believe that leachate contaminants may be approaching close to the property line. Only minimum monitoring of these wells to establish background quality and meet regulatory requirements would be justified. Anything more than an annual frequency would be considered wasteful.

Another example might be a well located in deep water bearing zones separated from the disposal site by other aquifers and aquicludes. Chapter 4 depicts

examples of this (i.e. coastal plain) where there are a series of alternating aquifers and aquicludes. For institutional reasons, or regional water planning purposes, a monitoring well may be placed in a deep aquifer which has almost no chance of being contaminated by the land disposal site. After an initial sampling, or, perhaps, two for background purposes, such wells would only deserve attention every two or five years, or, of course, in the unlikely event that other monitoring results cause reason for concern.

6.6.3 CLIMATOLOGICAL CHARACTERISTICS

In setting up the initial monitoring schedule for a particular site, one should analyze the fluctuations in leachate generation that occur over the year. The water balance method, which was presented in Chapters 3 and 5, is a very useful tool for this purpose. As an example, suppose it is desired to perform quarterly sampling. Instead of arbitrarily assigning a sampling date every third month, most of the monitoring effort should be concentrated either during and/or after those periods of the year of greatest leachate generation. The reflection of the actual sampling dates should also take into account the well location and depth, ground-water flow rate, saturation condition of the landfill, and other factors to project approximate lag times that may occur between first appearance of leachate and its impact on the monitoring well.

6.6.4 TRENDS IN THE MONITORING DATA

The three factors presented above (ground-water flow rate, well purpose and location and climate) will be used in establishing monitoring frequencies at the outset. However, monitoring frequencies should never be considered ironclad, but should maintain flexibility for modification to respond to

trends in the monitoring data.

As an example, suppose a spot check with a specific conductance meter indicates a significant change in the water quality at a particular well. Further investigation with additional key and extended indicators would be desired immediately, regardless of when the next sampling is scheduled. Concentrating on this well might also reduce the frequency at another well whose recent data has not shown significant changes in water quality.

6.6.5 LEGAL AND INSTITUTIONAL DATA NEEDS

Monitoring frequencies at a site may also be altered for legal and institutional reasons. As an example, suppose an enforcement action is initiated against a landfill. In order to strengthen their case, attorneys for both the state and the disposal site may request that all of the monitoring wells be monitored for an extended indicator analyses group.

6.6.6 OTHER CONSIDERATIONS

Other reasons for modifying the monitoring frequencies at a site would include,

- complaints from neighboring residents.
- an unusually severe climatological event, such as a hurricane with large amounts of rain in a short time period.
- a sudden change in or addition of an "other pollution source", such as an oil spill adjacent to the property.
- an unusual operational occurrence, such as the illegal and/or improper dumping of a large volume of liquids at a site.

A properly planned monitoring program will allow for modification in sampling schedules to respond to the above-mentioned occurrences.

6.7 COST CONSIDERATIONS

In selecting indicator parameters and sampling frequencies, it is important to be mindful of relative costs for performing the monitoring. The three basic levels of indicators used for monitoring presented earlier, vary not only in the depth of analytical data provided but also in the costs for sampling and analysis.

Specific conductance is so valuable because it is so inexpensive to perform. Being analyzed with a portable field meter, the analytical cost is merely the few extra minutes required by the technician to do the test at the site. The meter, itself, is relatively inexpensive, cost approximately $200.00 (1976 prices). The sampling costs are also low primarily because it is not necessary to collect and preserve samples for the laboratory thus lessening the amount of bottles to be carried and the time for adding sample preservatives. This advantage would be somewhat lessened where pre-pumping of the wells is done. There is no way of estimating sampling costs and time requirements since they are site specific depending upon accessibility and number of wells, as well as the pre-pumping (if necessary), and sample withdrawal method used. For order of magnitude comparison purposes only, a typical commercial laboratory would charge approximately $3.00 per sample for a specific conductance analysis (New York Area, 1976 prices).

A typical key indicator analysis group (i.e. specific conductance, pH, temperature, chloride, iron, color, turbidity, and COD) would be more expensive for sampling and analysis. Like specific conductance, all the others, except COD, can be run in the field with portable equipment. Sampling time will be increased by the additional equipment and analyses required, the time to collect, store and preserve the COD sample. Where one man could manage nicely

with specific conductance measurements, an assistant may be desirable in monitoring for the key indicators depending upon the number and accessibility of wells to be sampled and the sample withdrawal method used. Adverse weather conditions may also necessitate transporting samples back to the laboratory for analyses, where specific conductance could still be done in the field. For order of magnitude comparison purposes, a typical commercial laboratory would charge approximately $50.00 per sample for the key indicator analyses listed (New York Area, 1976 prices), exclusive of sampling.

A typical extended indicator analysis group, such as listed in Table 6-6, would be the most expensive level of monitoring for both sampling and analysis. With the exception of some key indicators which might be run in the field, all the indicators require proper storage, preservation and transport of samples to the laboratory for analysis. This will require additional sampling time and possibly additional manpower to perform properly and efficiently in the field. Of course, adverse weather conditions may further complicate sampling efforts.

TABLE 6-7

COMPARATIVE COSTS OF

INDICATOR ANALYSES*

Monitoring Group	Approximate Cost of Analysis + Per Sample ($)
Specific Conductance	$ 3.00
Key Indicators	$50.00
Extended Indicators	$600.00-$700.00

Note: It should be noted that there is an economy of numbers relative to both sampling and analysis. Appreciable quantity discounts are usually available for different levels of sampling and analysis. Additional savings can usually be realized through the use of long-term sampling and analysis contract.

* A comparison of sampling costs has not been made due to its extreme site specificity - such a comparison should consider the number and accessibility of wells, weather conditions, whether or not the wells will be pre-pumped prior to sampling, and the pre-pumping and sample withdrawal methods uses.

+ Based on January 1976 rates of a typical commercial laboratory in the New York Area. Refer to Chapter 8 for laboratory manpower requirements for analyses.

Again, for order of magnitude comparison purposes, a typical commercial laboratory would charge approximately $600.00-$700.00 per sample for the extended indicator parameters listed in Table 6-6, exclusive of sampling.

Table 6-7 summarizes the cost of analysis for the various monitoring groups discussed above. As noted, no comparisons of sampling costs has been made due to its extreme site specificity. In general terms, however, the sampling costs do increase as more indicators are added and that the sampling cost increases are magnified if pre-pumping of the wells is performed.

6.8 DATA MANAGEMENT

6.8.1 GENERAL

In a given sanitary landfill, appreciable quantities of data relative to ground-water quality will be generated over a period of time. Several factors govern the amount of data produced among which are the number of monitoring wells, the number of parameters to be tested and the frequency of testing, both scheduled and unscheduled (response to operational problems).

As a hypothetical case, let us assume that there are 20 monitoring wells in a given landfill and that the following tests are performed in a given year:

Testing Category	No. of Parameters	No. of Wells	No. of Tests
Annual-Extended	30	20	600
Quarterly-Indicator	10	20	600 (200 x 3)
Problem-Unscheduled	30	20	600
		Total	1,800

The total number of tests performed in the landfill over a period of one year

will be 1,800. This figure could approach 20,000 over a 10-year period.

This amount of raw data must be processed, interrelated, statistically analyzed amd stored in readily retrievable form so that it will be of maximum value for quality control, engineering and legal purposes.

The use of digital computer treatment would appear to be an excellent tool, both rapid and cost-effective as a management information system in the handling of this type of data. Other approaches to data management could entail manual processing, storage and retrieval of the data in the form of tables, charts and graphs which can show parameter levels and trends relative to standard values. In both cases, the statistical handling and use of analytical data for quality control purposes, in the form of ranges, means, standard deviations, parameter ratios and control charts will be important.

The technological state of the art of land disposal is still relatively young and is highly dependent on monitoring for its development. Even if a particular design operational strategy is successful at one site, it cannot be automatically assumed acceptable for all sites due to the extreme site specificity which is fundamental to land disposal of solid waste. Again, monitoring becomes critically important. Thus, the parameters monitored and the significant results obtained from the monitoring program will be critically evaluated in assessing a site and related design and operational approaches and in deciding upon modification. Because of the significance which may be placed on the results of the monitoring program, it should be the desire of the landfill management to understand and attempt to identify the causes of fluctuations in monitoring data obtained. Incorrect inter-

pretation of monitoring results may result in unnecessary expenditures or in a false sense of security.

The variability of the indicator parameters measured in a monitoring program may result from various phenomena, some of which are listed below:

1. Natural fluctuations in the background water quality.
2. Occurrence of another pollution source which might cause the background water quality to fluctuate.
3. Attenuation taking place in the subsurface environment.
4. Climatological variations.
5. Operational deficiencies, incidents and modifications.
6. Experimental errors in the analyses of measured parameters.
7. Sampling method utilized.

Variations in the background water quality will occur with location and time. Such variations recorded in the "A" wells should be fingerprinted statistically to allow for more accurate interpretation of the data fluctuations recorded at the "B" and "C" wells. In the same vein, fluctuations in background quality may be aritficially induced by another pollution source. As was stressed earlier, such occurrences must be carefully recorded because of their effect on monitoring data interpretation. As was discussed in Chapter 3, attenuation and climatological variations will have a definite influence on time and distance changes in monitoring data.

Operational factors will have a definite influence on the monitoring data and its evaluation and should be carefully documented. For example, operation changes, such as, type of wastes, a sudden disposal of a large quantity of liquid wastes, a deficiency in cover, or construction of the dikes,

diversionditches and the like, could all significantly affect monitoring results and should be carefully described with dates recorded.

The results of the analysis of a sample, by the same or different technicians, using the same laboratory techniques often fluctuate widely. Even very accurate laboratory analysis cannot prevent a relatively wide range in determined values of parameters,such as BOD, which may experience experimental error as high as ±20%. Variations will also exist with alternate analytical methods, especially field versus laboratory methods. This becomes even more significant for concentrated leachate samples. Where interferences further complicate analysis. All of this information must be carefully recorded because of its significance in data interpretation. A more detailed discussion on analytical methods is presented in Chapter 8.

Differences in the sampling method utilized will be important for monitoring data evaluation due to the variations that can be created. Was the well flushed out prior to obtaining a sample? Was the sample collected aerobically or anaerobically? Was the sample properly preserved? How much time elapsed between sample collection & analysis? Who did the sampling? It is important to know all of this information and understand its implications in evaluating the monitoring results.

All of these possible causes in variations should be carefully recorded and identified for proper evaluation of the monitoring results. It will be important for the monitoring program to distinguish between fluctuations which are significant and attributable to the landfill thus requiring some form of remedial action versus those variations which are insignificant or not attributable to deficiencies at the landfill. Of course, a complicating

feature for a land disposal site, unlike in water and air pollution, is the time lag which inherently exists between cause and effect. For example, it may take months or years for a fluctuation observed in an "A" or "B" well to reach a distant "C" well thus often complicating and retarding data interpretation.

6.8.2 APPLICATION OF STATISTICS

In the Handbook for Monitoring Industrial Wastewater, USEPA, 1973,[5] the value of statistics in monitoring is discussed:

"Statistics aid in the development of general laws resulting from numerous individual determinations which, by themselves, may be meaningless. The resulting relationships are part of the fundamental function of statistics which expresses the data obtained from an investigative process in a condensed and meaningful form. Thus, the average or mean is often used as a single value to represent a group of data. The variability of the group of observations is expressed by the value of the standard deviation and trends in concentrations during the monitoring process are expressed in the form of regression coefficients.

In general, the concern is with the treatment of the collected data. The accuracy of usefulness of these data is greatly enchanced if a full understanding was involved in generating the facts. The balance between use of statistical methods and evaluation based upon physical understanding is extremely important. The use and value of statistics decreases as physical understanding increases. Specifically, the difficulty lies in separating chance effects from valid occurrences. With the knowledge of basic probability theory and the use of statistical techniques, such as Least Squares Curve Fitting, Analysis of Variance, Regressive and Correlation Analysis, Chi-Squared Goodness of Fit, and others, it is possible to construct mathematical models and curves of almost any level of precision desired. Such techniques help to evaluate information having wide variations, so that an estimate of the best value of the parameter being measured can be assigned; and also to assess the precision of that estimate. Statistical procedures may also help in identifying errors and mistakes and are helpful in comparing sampling methods and procedures and in evaluating waste loadings from different process schemes."

Evaluation based upon physical understanding is especially significant for monitoring of a land disposal site due to the extreme site specificity of the various phenomena involved.

Probably, the major use of statistics in a monitoring program is to correlate the data for the proper choice of statistical parameters (mean, range and standard deviation) for the specific indicators for evaluation and comparison purposes.

Statistics and data analyses are very broad topics and are beyond the scope of this manual. The above-referenced EPA Handbook[5] cited several good references on statistics and these have been included in the bibliography at the end of this chapter for additional reading where a statistical approach is desired. It should be emphasized that rules and formulas for data analyses are many and they must be chosen wisely and applied correctly to be of value.

6.8.3 INDICATOR DATA PROFILES

Once a monitoring program has been in operation for an appreciable period of time, the data obtained from it can be used to provide specific analytical profiles for ground water and/or surface water for a given landfill site. These profiles will be characterized by data from a number of sampling points within the landfill and will reflect the influence of the various phenomena which were discussed earlier that result in fluctuations in the indicator parameters. Statistical analyses of the profiles will provide such important statistical values as normal ranges, means and standard deviations for each of the indicator parameters.

Quality control data of the landfill site can be obtained from the profile data. This could take the form of control charts for the various parameters which would indicate whether the operation was "in control" or "out of control" relative to upper and lower control limits provided by the control chart.

Statistics can play an important role in the correlation of specific parameters, especially in the case of specific conductance to other parameters such as total dissolved solids. Reference 5 presents an excellent discussion on the statistics for correlation of specific parameters. The data profile will also provide an insight into the inter-relationships of the various key indicator parameters in the form of normal ratios (i.e. conductance, total dissolved solids, iron, color, etc.) which should be developed for a cost effective monitoring program. When enough data are obtained on indicator parameter ratios (i.e. conductance, total dissolved solids, etc.) for a given landfill site, statistical values of range, mean and standard deviation can be developed, as is done for the individual indicator parameters themselves. This information can be used as a valuable statistical tool for quality control of the landfill and as an aid in the diagnosis of leachate contamination problems and their probable causes.

An indicator program, based on sufficient background quality data and on-going statistical information, should provide a basic, cost-effective, reliable monitoring tool for the quality control of a landfill.

In a monitoring program, data profiles can be used in a variety of ways, some of which are discussed below:

1. Concentration of the various indicator parameters versus time for each monitoring well.

   This is perhaps the most common use of a data profile in monitoring programs. It provides an immediate visual picture of the trends in quality and is nicely defined with the basic statistical values (mean, range and standard deviation). It provides a

valuable tool in comparing monitoring results of the "A", "B" and "C" wells for operational and enforcement purposes. It provides a readily available and convenient tool for comparing water quality trends to trends and occurrences in the various phenomena that influence the ground-water quality which were discussed earlier in this section.

2. Concentration of the various indicator parameters versus distance from the landfill. This type of profile would be constructed by plotting the data for selected indicator parameters which are obtained on a particular date for the various "C" wells located at different distances from the landfill. The quality of the "B" well would represent the concentration at zero distance from the landfill. Chapter 3 (Section 3.2.4) discussed the use of this type of profile in the measurement of attenuation at a land disposal site. Figure 3-1 shows an example of this profile.

3. Other profiles providing "physical understanding" information. In the evaluation of the monitoring data obtained at a land disposal site, it would be of value to be readily accessible to information on the phenomena which may be the potential cause of fluctuations and trends in the monitoring data. This might include a water balance profile and a "chronological events" profile of important occurrences. A water balance profile, such as shown in Figure 6-2 could be developed for each site as part of the permit applications or for several representative "typical sites" throughout the state. The actual quantities are not as

important as the trends they would depict. A profile of "chronological events" might look like Figure 6-3 and could be kept on file and up-to-date easily by the inspection and monitoring personnel. Reference to such a profile would be of obvious value providing physical understanding in evaluating monitoring results. It should be recognized, however, that one must be mindful of the time lag between cause and effect that is inherent at land disposal sites, when using such profiles in the evaluation of monitoring results.

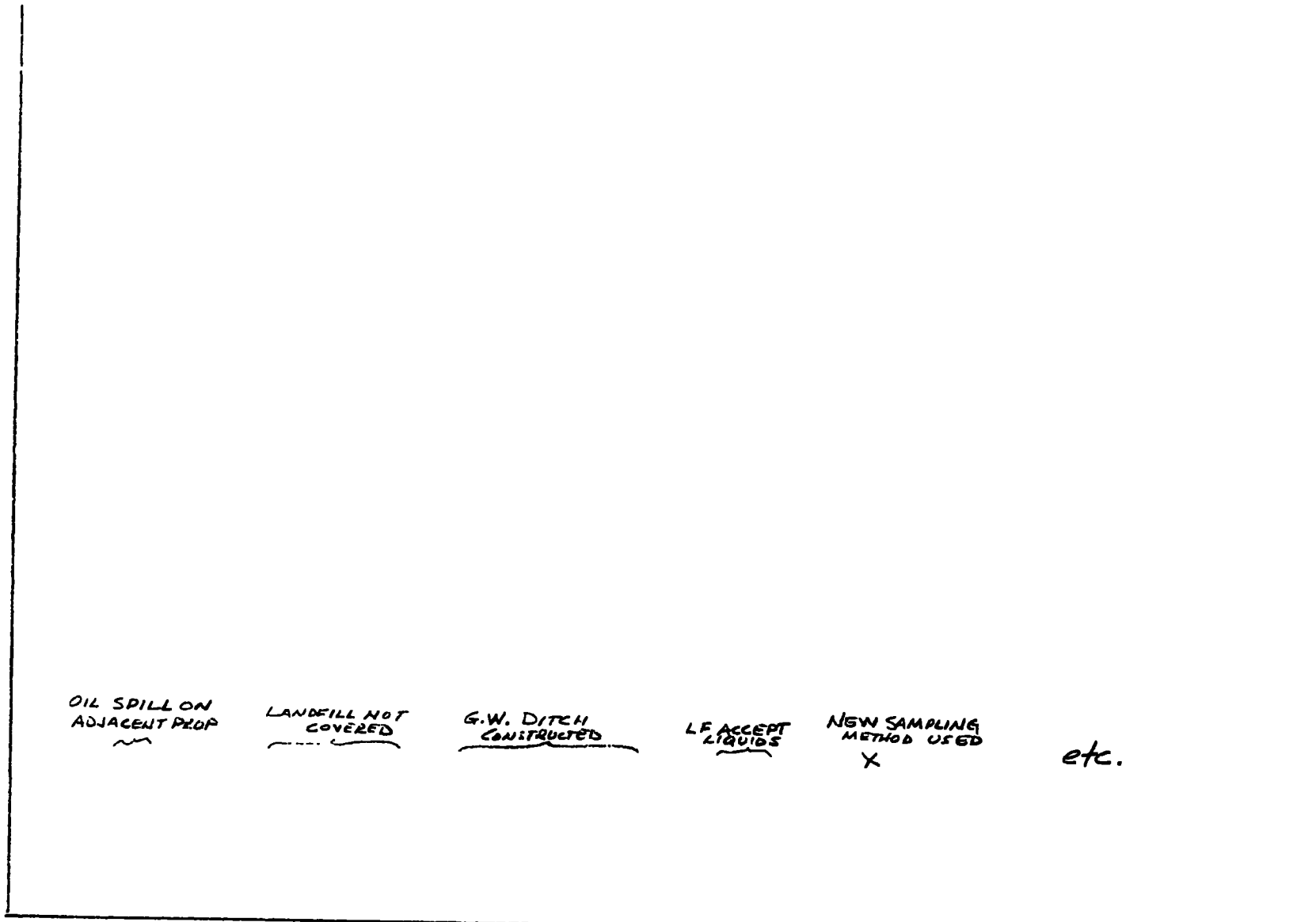

FIGURE 6-3 CHRONOLOGICAL EVENTS PROFILE

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PRELIMINARY

CHAPTER 7

SAMPLING, STORAGE AND PRESERVATION

7.1 INTRODUCTION

The sampling of ground and surface waters associated with sanitary landfill monitoring is a critically important operation. The analytical results obtained from the samples and the subsequent decisions which are based on the analytical data, are vitally dependent upon the validity of the samples obtained.

Every effort must be made to assure that the sample is representative of the particular body of water being sampled. A detailed sampling plan, acceptable to all interested parties, should be developed prior to any sampling operations.

The physical, chemical and bacteriological integrity of the sample must be maintained from the time of sampling to the time of testing in order to keep any changes at a minimum. The time between sampling and testing should be kept at the absolute minimum which is practicable.

7.2 SAMPLE COLLECTION

The following, from Standard Methods, p. 36, is a useful guide:

> "A record should be made of every sample collected and every bottle should be identified, preferably by attaching an appropriately inscribed tag or label. The record should contain sufficient information to provide positive identification of the sample at a later date, as well as the name of the sample collector, the date, hour and exact

> location, the water temperature, and any data which may be needed in the future for correlation, such as weather conditions, water level, stream flow, or the like. Sampling points should be fixed by detailed description, by maps, or with the aid of stakes, buoys or landmarks in such a manner as to permit their identification by other persons without reliance upon memory or personal guidance. ..... Samples from wells should be collected only after the well has been pumped for a sufficient time to insure that the sample will represent the ground water which feeds the well. Sometimes it will be necessary to pump at a specified rate to achieve a characteristic drawdown, if this determines the zones from which the well is supplied. It may be desirable to record the pumping rate and the drawdown as part of the sample record."

The quality of water pumped should equal approximately 3 to 5 well volumes. If the well is pumped dry, sufficient time should be allowed for full recovery prior to sampling.

7.2.1 SAMPLE COLLECTION TECHNIQUES

Various water withdrawal techniques were discussed in Chapter 5 including vacuum pressure and bailing methods. The important underlying principle, of course, being to obtain a representative sample of the ground water and to minimize degradation. If the well depth is within pumpable limits, a vacuum sampling technique can be used to obtain the sample under anaerobic conditions. Figure 7-1 shows a typical vacuum sampling technique using a vacuum pump and portable generator, used successfully in Orange County, Florida. [1] Vacuum can also be supplied from an automobile or truck engine or a hand pump manifold which could replace the vacuum pump/portable generator combination. Note in Figure 7-1, a 1/2-inch tube is permanently installed in the monitoring well which would eliminate the possibility of cross-contamination between wells.

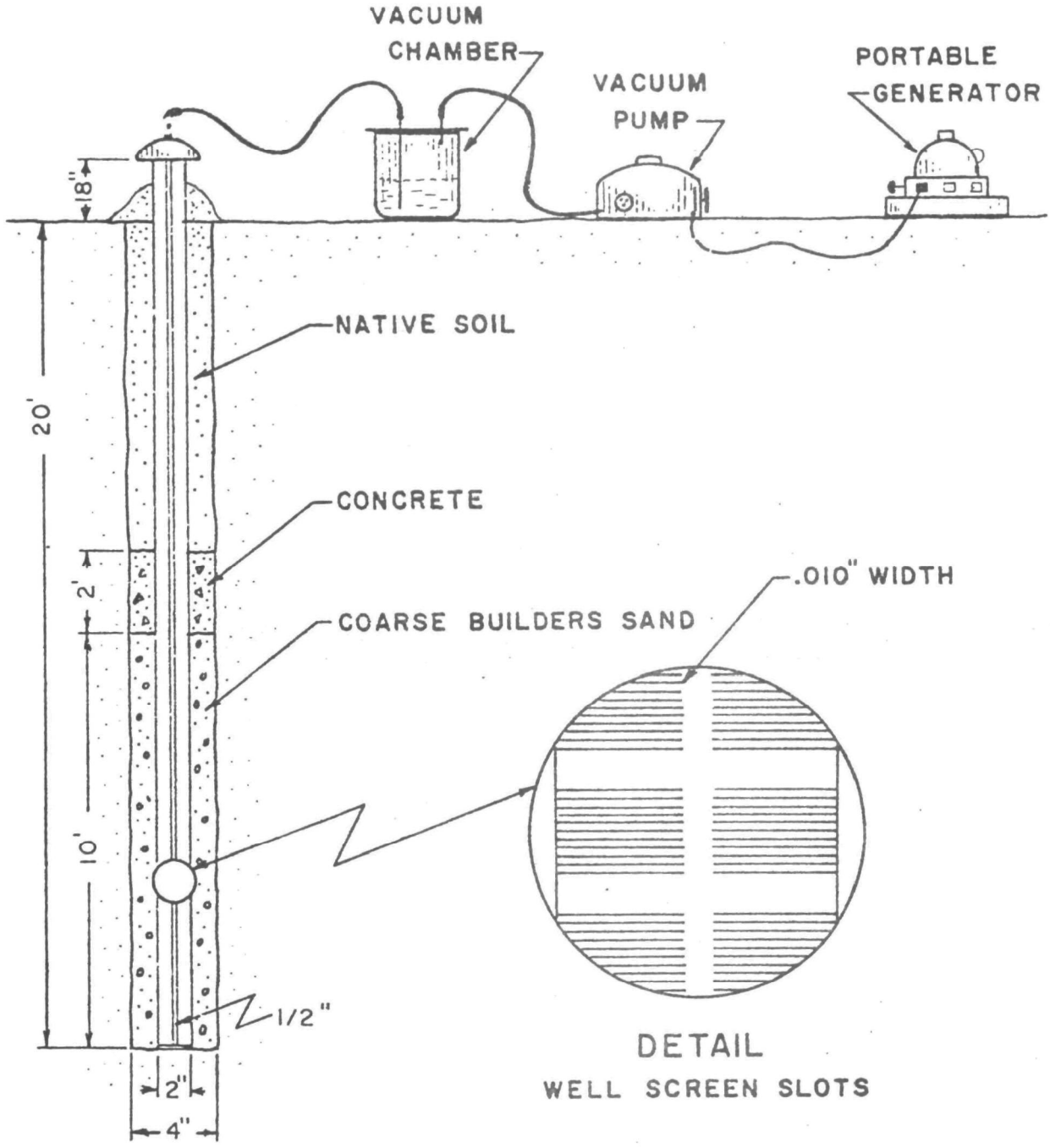


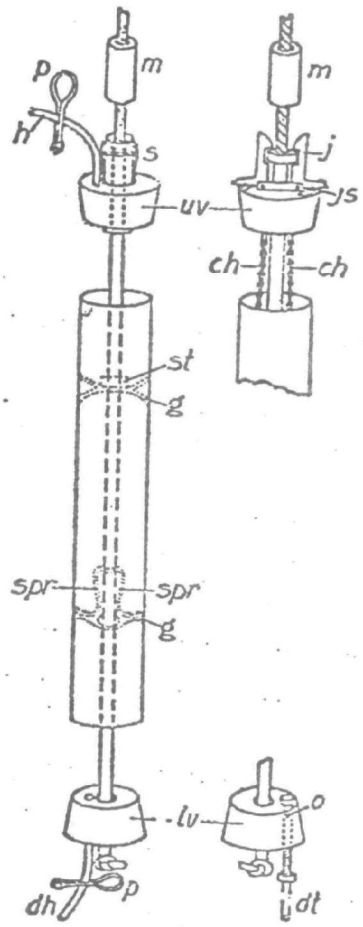
FIGURE 7-1 Profile Of Shallow Sampling Well. (REF. 1)

With a pressure-type sampling method, such as the type shown on Figure 24 (Chapter 5), the sample is obtained by connecting the sample bottle directly to the 1/2-inch water discharge outlet. To insure anaerobic conditions, the sample bottle should be flushed out with an inert gas prior to collecting a sample. The built-in feature of this method, being used to both pre-pump and sample the well can effect considerable savings on labor and also eliminate the possibility of cross-contamination between wells, as can occur with portable pumping and sampling devices. They are also of notable value in obtaining bacteriological samples where external sources of contamination must be avoided.

Bailers are also used to collect a sample. A Kemmerer water bottle sampler, as shown on Figure 7-2 is a bailer commonly used. In transferring the sample from the sampler to the sample bottle, contact with air and agitation of the sample should be minimized slow and careful transfer, placing the tip of the sampler's exit tube to the side of the sample bottle is recommended. To minimize cross-contamination, the bailers should be thoroughly flushed out with tap water with the first sample from the next well to be sampled prior to collecting the sample for analysis.

Samples for bacteriological examination must be collected in sterile containers. Detailed sampling procedures for bacteriological samples are given in Standard Methods, 13th Ed., pp. 657-660 and Biological Analysis of Water and Wastewater, AM 302, Millipore Corp., 1974, pp. 4-6.

Samples can be taken directly from wells with a sterile bottle in a weighted frame which can be lowered below the water surface and opened below surface. Samples can also be obtained by means of various pumping devices, as described



ch—chain which anchors upper valve to upper interior guide.
dh—rubber drain tube.
dt—brass drain tube.
g—interior guide fastened to inner surface of sampler.
h—rubber tube.
j—jaw of release.
js—jaw spring.
lv—lower valve.
m—messenger.
o—opening interior of drain tube.
p—pinch cock.
s—upper release spring operating on horizontal pin, one end of which fits into groove on central rod.
spr—spring fastened to lower internal guide and operating in groove on central rod to provide lower release.
st—stop on central rod.
uv—upper valve.
Left—View of complete sampler with valves open.
Top right—Another type of construction of upper valve and tripping device.
Bottom right—Another type of construction of lower valve and drain tube.

FIG. 4 Diagram Showing Structural Features of Modified Kemmerer Sampler.[21]

[21] P. S. Welch, *Limnological Methods*, p. 200, Fig. 59.

FIGURE 7-2

previously. The same pumping schedule should be observed as for non-sterile samples. Sample volumes of approximately 250 ml. are usually satisfactory for bacteriological testing.

Sampling and preservation of samples are addressed in the 1973 Annual Book of ASTM Standards, Part 23, Water; Atmospheric Analysis, pp. 72-75, Standard Methods of Sampling Homogeneous Industrial Waste Water; and pp. 76-91, Standard Methods of Sampling Water.

7.2.2 RECORDS

Adequate records should be maintained on each sample that is taken. Record information should include:

Sample description: Type (ground water, surface water) volume.

Sample source: Well number, location.

Sampler's Identity: Chain of evidence should be maintained; each time transfer of a sample occurs a record including signatures of parties involved in transfer should be made. This procedure can have legal significance .

Time and date of sampling.

Significant weather conditions.

Sample laboratory number.

Pertinent well data: Depth, depth to water surface, pumping schedule and method.

Sampling method: Vacuum, Kemmerer, pressure.

Preservatives,if any: Type and number - i.e. NaOH for cyanide, H_3PO and $CuSO_4$ for phenols, etc.

Sample containers: Type, size and number - i.e. - three (3) liter glass stoppered bottles, one (1) one gallon screw-cap bottle, etc.

Reason for sampling: Initial sampling of new landfill; annual sampling, quarterly sampling, special problem sampling, in conjunction with contaminant discovered in nearby domestic well.

Appearance of sample: Color, turbidity, sediment, oil on surface.

Any other information which appears to be significant: i.e. sampled in conjunction with state, county, local regulatory authorities.

Sampled for specific conductance value only.

Sampled for key indicator analysis.

Sampled for extended analysis.

Re-sampled following engineering corrective action.

Name and location of laboratory performing analyses.

Sample temperature upon sampling.

Thermal preservation: i.e. - transportation in ice chest.

Analytical determinations, if any, preformed in the field at the time of sampling and results obtained, analysts identity and affiliation.

7.3 SAMPLE CONTAINERS

For most samples and analytical parameters, either glass or plastic containers are satisfactory. Some exceptions are encountered, such as the use of plastic for silica determinations and glass for phenols, or oil and grease determination. Containers should be kept full until samples are analyzed, in order to maintain anaerobic conditions.

As a general guide in choosing a container for a sample, the ideal material of construction should be non-reactive with the sample and especially, the

particular analytical parameter to be tested. Table 2 lists the recommended containers for various analyses.

Cleanliness of containers is of utmost importance. An effective procedure for cleaning containers is to wash sequentially with a detergent, tap water rinse, nitric acid rinse, tap water rinse, hydrochloric acid rinse, tap water rinse and finally, deionized or distilled water. In addition, the containers should be rinsed several times with the sample at the time of sampling.

7.4 PRESERVATION OF SAMPLES AND SAMPLE VOLUME REQUIREMENTS.

The following excerpt from Methods for Chemical Analysis of Water and Waste, EPA-625/6-74-003, pp. vi-xii, is a useful guide for sample preservation, sample volume requirements and sample containers.

Additional useful information relative to preservation of polluted waters, wastewaters, etc., is available in Standard Methods, 13th Ed., 1971, pp. 368-369.

Additionally, Standard Methods provides a very useful "Sampling and Storage" section for many of the analytical methods offered.

7.5 PRESERVATION OF SAMPLES IN THE FIELD

Samples should be preserved at low temperatures during transport to the laboratory for analysis. A convenient method is to use an insulated cooler containing ice so that a temperature of 0 to 10°C is maintained.

If possible, appropriate chemical preservation should be performed in the

field for various analytical parameters at the time of sampling. In this case, separate bottles and chemical preservatives are required for particular parameters. As an example, for the extended analyses group in Chapter 6, proper preservation techniques would require splitting the sample into as many as approximately ten (10) bottles. Thus, one can see that sampling a large number of wells for several analyses can become a cumbersome procedure in the field for this reason.

Regardless of the method of preservation, analyses should be performed as soon as is practicably possible after sampling.

SAMPLE PRESERVATION

Complete and unequivocal preservation of samples, either domestic sewage, industrial wastes, or natural waters, is a practical impossibility. Regardless of the nature of the sample, complete stability for every constituent can never be achieved. At best, preservation techniques can only retard the chemical and biological changes that inevitably continue after the sample is removed from the parent source. The changes that take place in a sample are either chemical or biological. In the former case, certain changes occur in the chemical structure of the constituents that are a function of physical conditions. Metal cations may precipitate as hydroxides or form complexes with other constituents; cations or anions may change valence states under certain reducing or oxidizing conditions; other constituents may dissolve or volatilize with the passage of time. Metal cations may also adsorb onto surfaces (glass, plastic, quartz, etc.), such as, iron and lead. Biological changes taking place in a sample may change the valence of an element or a radical to a different valence. Soluble constituents may be converted to organically bound materials in cell structures, or cell lysis may result in release of cellular material into solution. The well known nitrogen and phosphorus cycles are examples of biological influence on sample composition.

Methods of preservation are relatively limited and are intended generally to (1) retard biological action, (2) retard hydrolysis of chemical compounds and complexes and (3) reduce volatility of constituents.

Preservation methods are generally limited to pH control, chemical addition, refrigeration, and freezing. Table 1 shows the various preservatives that may be used to retard changes in samples.

(12)

TABLE 1

Preservative	Action	Applicable to:
$HgCl_2$	Bacterial Inhibitor	Nitrogen forms, Phosphorus forms
Acid (HNO_3)	Metals solvent, prevents precipitation	Metals
Acid (H_2SO_4)	Bacterial Inhibitor	Organic samples (COD, oil & grease organic carbon)
	Salt formation with organic bases	Ammonia, amines
Alkali (NaOH)	Salt formation with volatile compounds	Cyanides, organic acids
Refrigeration	Bacterial Inhibitor	Acidity-alkalinity, organic materials, BOD, color, odor, organic P, organic N, carbon, etc., biological organism (coliform, etc.)

In summary, refrigeration at temperatures near freezing or below is the best preservation technique available, but it is not applicable to all types of samples.

The recommended choice of preservatives for various constituents is given in Table 2. These choices are based on the accompanying references and on information supplied by various Regional Analytical Quality Control Coordinators.

TABLE 2

RECOMMENDATION FOR SAMPLING AND PRESERVATION OF SAMPLES ACCORDING TO MEASUREMENT (1)

Measurement	Vol. Req. (ml)	Container	Preservative	Holding Time(6)
Acidity	100	P, G[2]	Cool, 4°C	24 Hrs.
Alkalinity	100	P, G	Cool, 4°C	24 Hrs.
Arsenic	100	P, G	HNO_3 to pH <2	6 Mos.
BOD	1000	P, G	Cool, 4°C	6 Hrs.[3]
Bromide	100	P, G	Cool, 4°C	24 Hrs.
COD	50	P, G	H_2SO_4 to pH <2	7 Days
Chloride	50	P, G	None Req.	7 Days
Chlorine Req.	50	P, G	Cool, 4°C	24 Hrs.
Color	50	P, G	Cool, 4°C	24 Hrs.
Cyanides	500	P, G	Cool, 4°C NaOH to pH 12	24 Hrs.
Dissolved Oxygen Probe	300	G only	Det. on site	No Holding
Winkler	300	G only	Fix on site	No Holding

TABLE 2 (Continued)

Measurement	Vol. Req. (ml)	Container	Preservative	Holding Time(6)
Fluoride	300	P, G	Cool, 4°C	7 Days
Hardness	100	P, G	Cool, 4°C	7 Days
Iodide	100	P, G	Cool, 4°C	24 Hrs.
MBAS	250	P, G	Cool, 4°C	24 Hrs.
Metals				
Dissolved	200	P, G	Filter on site HNO_3 to pH <2	6 Mos.
Suspended			Filter on site	6 Mos.
Total	100		HNO_3 to pH <2	6 Mos.
Mercury				
Dissolved	100	P, G	Filter HNO_3 to pH <2	38 Days (Glass) 13 Days (Hard Plastic)
Total	100	P, G	HNO_3 to pH <2	38 Days (Glass) 13 Days (Hard Plastic)

TABLE 2 (Continued)

Measurement	Vol. Req. (ml)	Container	Preservative	Holding Time(6)
Nitrogen				
Ammonia	400	P, G	Cool, 4°C H_2SO_4 to pH <2	24 Hrs.(4)
Kjeldahl	500	P, G	Cool, 4°C H_2SO_4 to pH <2	24 Hrs.(4)
Nitrate	100	P, G	Cool, 4°C H_2SO_4 to pH <2	24 Hrs.(4)
Nitrite	50	P, G	Cool, 4°C	24 Hrs. (4)
NTA	50	P, G	Cool, 4°C	24 Hrs.
Oil & Grease	1000	G only	Cool, 4°C H_2SO_4 to pH <2	24 Hrs.
Organic Carbon	25	P, G	Cool, 4°C H_2SO_4 to pH <2	24 Hrs.
pH	25	P, G	Cool, 4°C Det. on site	6 Hrs.(3)
Phenolics	500	G only	Cool, 4°C H_3PO_4 to pH <4 1.0 g $CuSO_4$/l	24 Hrs.
Phosphorus				
Ortho-phosphate, Dissolved	50	P, G	Filter on site Cool, 4°C	24 Hrs.(4)

(16) 129

TABLE 2 (Continued)

Measurement	Vol. Req. (ml)	Container	Preservative	Holding Time(6)
Hydrolyzable	50	P, G	Cool, 4°C H_2SO_4 to pH <2	24 Hrs.(4)
Total	50	P, G	Cool, 4°C	24 Hrs. (4)
Total, Dissolved	50	P, G	Filter on site Cool, 4°C	24 Hrs.(4)
Residue				
Filterable	100	P, G	Cool, 4°C	7 Days
Non-Filterable	100	P, G	Cool, 4°C	7 Days
Total	100	P, G	Cool, 4°C	7Days
Volatile	100	P, G	Cool, 4°C	7 Days
Settleable Matter	1000	P, G	None Req.	24 Hrs.
Selenium	50	P, G	HNO_3 to pH <2	6 Mos.
Silica	50	P only	Cool, 4°C	7 Days
Specific Conductance	100	P, G	Cool, 4°C	24 Hrs. (5)
Sulfate	50	P, G	Cool, 4°C	7 Days

TABLE 2 (Continued)

Measurement	Vol. Req. (ml)	Container	Preservative	Holding Time(6)
Sulfide	50	P, G	2 ml zinc acetate	24 Hrs.
Sulfite	50	P, G	Cool, 4°C	24 Hrs.
Temperature	1000	P, G	Det. on site	No Holding
Threshold Odor	200	G only	Cool, 4°C	24 Hrs.
Turbidity	100	P, G	Cool, 4°C	7 Days

1. More specific instructions for preservation and sampling are found with each procedure as detailed in this manual. A general discussion on sampling water and industrial wastewater may be found in ASTM, Part 23, p. 72-91 (1973).

2. Plastic or Glass

3. If samples cannot be returned to the laboratory in less than 6 hours and holding time exceeds this limit, the final reported data should indicate the actual holding time.

4. Mercuric chloride may be used as an alternate preservative at a concentration of 40 mg/1, especially if a longer holding time is required. However, the use of mercuric chloride is discouraged whenever possible.

5. If the sample is stabilized by cooling, it should be warmed to 25°C for reading, or temperature correction made and results reported at 25°C.

6. It has been shown that samples properly preserved may be held for extended periods beyond the recommended holding time.

CHAPTER 8

ANALYTICAL METHODS

8.1 INTRODUCTION

Reliable, cost effective analytical methods must be selected and applied in order to successfully carry on Basic Indicator, and Extended Analysis programs.

The parameters of interest in the analytical characterizations of leachate are usually physical, chemical and biological. Normally, the desired information is quantitative rather than qualitative, although qualitative data may be required at times for special problems. For purposes of this Manual, consideration will be given only to the quantitative aspect of the analytical data.

As stated previously, "Leachate represents an extremely complex system containing soluble, insoluble, organic, inorganic, ionic, nonionic and bacteriological constituents in an aqueous medium. Actual types, numbers and levels of constituents are widely variable...."

When dealing with a complex material of variable composition, such as leachate, it is recognized that there is a serious potential for numerous interferences in the determination of a given parameter.

The physical measurements, such as specific conductance and pH, are not normally subject to appreciable interference,but many of the chemical and biological determinations are readily affected by matrix interferences.

When an analyst wishes to perform a quantitative determination on a particular parameter, he must decide which analytical method will be used. There is usually a choice among several standard methods which can be applied to a given determination. Among the many factors which must be considered in the choice of an analytical method are the following: sensitivity, precision and accuracy required, nature of the matrix and its effect upon the determination (interferences), available equipment, manpower and instrumentation, level of expertise of the analyst, number of samples to be analyzed, turn-around time, history and available information regarding the sample, reason for performing the analysis, how the analytical data will be offered other parameters, if any, to be determined on the sample, importance of cost factors. When all pertinent considerations of this nature have been carefully weighed the decision is then made to apply a particular standard method to the problem. There are several literature sources of standard analytical methods which can be applied, either directly or with modification, to the analysis of leachate samples. There are three references which are in wide use for this purpose, namely:

1. Standard Methods for the Examination of Water and Wastewater, 13th Ed. APHA, 1971
2. Manual of Methods for Chemical Analysis of Water and Wastes, U. S. Environmental Protection Agency, 1974
3. 1973 Annual Book of ASTM Standards, Part 23, Water; Atmospheric Analysis

Some comments are made relative to the analysis of polluted waters and other similar samples in Standard Methods for the Examination of Water and Wastewater, P. 367. These comments, following, are appropriate to review at this point. "These procedures described in Part 200 of this manual are intended for the

physical and chemical examination of wastewaters of both domestic and industrial origin, treatment plant effluents, polluted waters, sludges and bottom sediments. An effort has been made to present methods which apply as generally as possible and to indicate modifications which are required for samples of unusual composition, such as certain industrial wastes. However, because of the wide variety of industrial wastes, the procedures given here cannot cover all possibilities and may not be suitable for all wastes and combination of wastes. Hence, some modification of a procedure may be necessary in specific instances. Whenever a procedure is modified, the nature of the modification must be plainly stated in the report of results. The procedures which are indicated as being intended for the examination of sludges and bottom sediments may not apply without modification to chemical sludges or slurries."

In this same vein, the following comments are made in Handbook for Analytical Quality Control in Water and Wastewater Laboratories, U.S.E.P.A., 1973, P 1-3. "Regardless of the analytical method used in the laboratory, the specific methodology should be carefully documented. In some water pollution reports it is customary to state that Standard Methods have been used throughout. Close examination indicates, however, that this is not strictly true. In many laboratories, the standard method has been modified because of recent research or personal preferences of the laboratory staff. In other cases, the standard method has been replaced with a better one. Statements concerning the methods used in arriving at laboratory data should be clearly and honestly stated. The methods used should be adequately referenced and the procedures applied exactly as directed.

Knowing the specific method which has been used, the reviewer can apply the associated precision and accuracy of the method when interpreting the laboratory results. If the analytical methodology is in doubt, the data user may honestly inquire as to the reliability of the result he is to interpret.

The advantages of strict adherence to accepted methods should not stifle investigations leading to improvements in analytical procedures. In spite of the value of accepted and documented methods, occasions do arise when a procedure must be modified to eliminate unusual interference, or to yield increased sensitivity. When modification is necessary, the revision should be carefully worked out to accomplish the desired result. It is advisable to assemble data using both the regular and the modified procedure to show the superiority of the latter. This useful information can be brought to the attention of the individuals and groups responsible for methods standardization. For maximum benefit, the modified procedure should be rewritten in the standard format so that the substituted procedure may be used throughout the laboratory for routine examination of samples. Responsibility for the use of a non-standard procedure rests with the analyst and his supervisor, since such use represents a departure from accepted practice."

8.2 Alternate Analytical Methods

8.2.1 Method Comparability

Relative to the use of alternate analytical methods for the National Pollution Discharge Elimination System, the EPA has published guidelines in the Federal Register, October 16, 1973, as follows:

"Typical Comparability Testing Procedure.

This procedure is designed to provide data on the comparability(equivalency) of two dissimilar analytical methods for measurement of the same property or constituent.

In regarding the comparison, one method is assumed to be satisfactory (standard) and the second or alternate method is compared for equivalency.

To provide sufficient data to apply statistical measurements of significance, the following determinations are required:

1. Using an effluent sample representative of normal operating processes, well mixed between aliquot withdrawal, run seven replicate determinations by each method.

Report values in the following manner:

TABLE 1

Effluent Sample Representative of Normal Operating Conditions.

Aliquot	Standard Method*	Alternate Method*
List 1 through 7		

*Cite method reference

2. If variations occur in the concentration of the measured constituent in the plant effluent, report the above testing on two more samples, one collected at the highest level of constituent normally encountered in the waste samples examined by the laboratory and one having a concentration at or near the lowest level usually examined. Report values in the following manner:

TABLE 2

Effluent Samples of Varying Composition

Aliquot	Low Level	High Level
List 1 through 7		

3. Using the sample from 1, add a small volume of standard solution sufficient to double the concentration. Run 7 replicate determinations by each method. Report values as Table 3:

Effluent Samples Plus Standard Solution, in the same way as Table 1. Cite source and amount of standard solution; it should be proportioned to the

to the original concentration. The above procedure must be followed on each outfall for which a permit is issued, unless it can be shown that the outfalls in question are comparable."

A comparability of that procedure for analytical methods used on landfill leachate samples can be modelled after the above-cited EPA procedure. Samples, instead of representing plant effluents, will represent potentially leachate-enriched ground and/or surface waters. The results of the standard and alternate methods should be compared for statistically significant differences. If the alternate method proves to be equal to or better than the standard method, it should be considered an acceptable analytical method for the determination of the particular parameter in the leachate sample.

8.2.2 Other Analytical Methods

A considerable amount of valuable pertinent information on analytical methodology and data is available in Standard Methods for the Examination of Water and Wastewater, 13th Edition, 1971. Several sections of this work are reprinted here to be used as a guide in the analysis of leachate samples. The particular subjects of interest which are treated are:

1. Other (instrumental) methods of analysis, including Atomic Absorption Spectroscopy, FlamePhotometry, Emission Spectroscopy, Polarography, Potentiometric Tiliation, Specific Ion Electrodes and Probes, Gas Chromotography and Automated Analytical Instrumentation. (pp. 12-15)
2. Interferences and methods used for their elimination. (pp. 15-18)
3. Expression of Results. (pp. 18-20)
4. Siginficant Figures. (pp. 20-21)

5. Precision and Accuracy. Statistical Approach, Standard Deviation, Range, Rejection of Experimental Data, Presentation of Precision and Accuracy Data, Quality Control. (pp. 22-25)

6. Graphical Representation of Data, Method of Least Squares. (pp. 25-27)

7. Self-Evaluation (Desirable Philosophy for the Analyst). (p. 27)

8. Methods Evaluation by the Committee on Standard Methods of the Water Pollution Control Federation. (pp.369-370)

an alkali, as is done in the direct nesslerization method for ammonia nitrogen. For samples of relatively coarse turbidity, centrifuging may suffice. In some instances, glass fiber filters, filter paper or sintered-glass filters of fine porosity will serve the purpose. For very small particle sizes, the more recently developed cellulose acetate membrane filters may provide the required retentiveness. Used with discretion, each of these methods will yield satisfactory results in a suitable situation. However, it must be emphasized that no single universally ideal method of turbidity removal is available. Moreover, the analyst should be perpetually alert to adsorption losses possible with any flocculating or filtering procedure and an attendant alteration in the sample filtrate.

8. Other Methods of Analysis

The use of an instrumental method of analysis not specifically described in procedures in this manual is permissible, provided that the results so obtained are checked periodically, either against a standard method described in this manual or against a standard sample of undisputed composition. Identification of any such instrumental method used must be included in the laboratory report along with the analytical results.

a. Atomic absorption spectroscopy: Atomic absorption spectrophotometry has been applied to the determination of a growing number of metals in drinking water without the need for prior concentration or extensive sample pretreatment. The use of organic solvents coupled with oxyacetylene, oxyhydrogen or nitrous oxide-acetylene flames enables the determination of metals which form refractory oxides. This manual presents atomic absorption methods for many metals. Although not described in the text, calcium, lithium, potassium, sodium and strontium can also be determined readily by the atomic absorption approach.

b. Flame photometry: Flame photometry is used for the determination of sodium, potassium, lithium and strontium. To some extent it is also useful for the determination of calcium and other ions.

c. Emission spectroscopy: Arc-spark emission spectroscopy is becoming an important analytical tool for water analysis and is proving valuable both for trace analysis and for certain determinations not easily made by any other method. Considerable specialized training and experience with this technic are required to obtain satisfactory results, and frequently it is practical to obtain only semiquantitative results from such methods in water analysis. It should be noted that an arc-spark emission spectrograph is relatively expensive when used exclusively for routine water testing, but its purchase is justified if it can be used as a general laboratory analytical instrument.

Among the advantages of arc-spark spectrographic analyses are: (1) the minute size of sample required; (2) elimination of the necessity for bringing solids, such as precipitates and corrosion products, into solution; (3) detection of all determinable elements present in a sample, whether specifically looked for or not; and (4) their unexcelled sensitivity for some elements Among the disadvantages of spectrographic analyses are: (1) the high cost of first-class equipment; (2) the need for special training and experience; (3) the possible occurrence of severe

interferences which must be taken into account if reasonable accuracy is to be achieved; and (4) the inability to distinguish between different valence states of an element, as, for instance, between chromic and chromate or ferric and ferrous.

Silver is the only element for which a spectrographic method is described in this manual. The following can also be determined spectrographically: aluminum, barium, boron, chromium, copper, iron, lead, lithium, magnesium, manganese, nickel, silicon, strontium and zinc. Among the elements for which there is no standard method in this manual but which are determinable by arc-spark spectrography are cobalt, molybdenum, tin, titanium, vanadium and a number of others.[1, 2]

d. Polarography: Polarography is suggested for scanning industrial wastes for various metal ions, especially where the possible interferences in the precise colorimetric procedures are unknown. The older polarographic method for dissolved oxygen also remains from the past.

Recent developments in polarography include the introduction of pulse polarographs with dual synchronized electrodes capable of differential derivative output. Operation in the pulse mode permits determination of seven or more metals on a single portion of the sample after ashing with nitric acid. If a 100-ml portion of the sample is ashed, determinations may be made in the low microgram-per-liter range.

A method closely allied to polarography is amperometric titration, which is suitable for the determination of residual chlorine and other iodometric methods by titrimetry.

e. Potentiometric titration: Growing in acceptance for titrimetric work are electrical instruments called titrimeters, or electrotitrators. If used discreetly with an understanding of their limitations, these instruments can be applied to many of the titrimetric determinations described, including those for acidity and the alkalinities. In addition, titrimetric precipitation reactions such as those for chloride, as well as titrimetric procedures based on complexometric and oxidation-reduction reactions, can be performed with these instruments. To be suitable for these extensive applications, an instrument must be equipped with all the necessary special electrodes. Some recent electrotitrator models embody automatic features by which a titration is self-executing after the preliminary settings are made. In order to avoid spurious readings, the analyst is urged to check instrument operation against representative known samples in the same concentration range as the water under examination.

f. Specific ion electrodes and probes: The past decade has witnessed the advent of specific ion electrodes and probes for the rapid estimation of certain constituents in water. These electrodes function best in conjunction with the concurrently developed expanded-scale pH meters. For the most part, the new electrodes operate on the ion-exchange principle. The specific ion electrodes available at this time are designed for the measurement of calcium, divalent copper, divalent hardness, potassium, sodium, total monovalent and total divalent cations, and bromide, chloride, cyanide, fluoride, iodide, nitrate, perchlorate and sulfide anions among others. Additional specific ion electrodes can doubtless be anticipated in the future.

These devices are subject to varying

degrees of interference from other ions in the sample and must still receive the thorough study that would warrant their adoption as tentative and standard methods. Nonetheless, their value for monitoring activities is readily apparent. To remove all doubt of variations in reliability, each electrode should be checked in the presence of interferences as well as the ion for which it is intended. This manual details the electrode method for fluoride after a collaborative study established its credibility in the presence of common interferences (Section 121).

The commercial dissolved oxygen probes vary considerably in their dependability and maintenance requirements. Despite these shortcomings, they have been applied to the monitoring of dissolved oxygen levels in a variety of waters and wastewaters. Most probes embody an electrode covered by a thin layer of electrolyte held in place by an oxygen-permeable membrane. The oxygen in solution diffuses through the membrane and electrolyte layer to react at the electrode, inducing a current which is proportional to the activity (and concentration) of the dissolved oxygen. Satisfactory dissolved-oxygen electrodes are also available without a membrane. In either case, the face of the dissolved oxygen sensor should be kept well agitated, and temperature compensation should be provided, in order to insure acceptable results in the laboratory or monitoring application.

g. Gas chromatography: Considerable work is under way in the development of gas chromatographic methods suitable for water and wastewater analysis. Two such methods appear in this manual: one for the determination of chlorinated hydrocarbon pesticides in drinking water; the second for the determination of the components in sludge digester gas. Investigations reveal that gas chromatography may also be useful for the determination of phenols. The skill of the operator and the expense entailed in its purchase will probably limit use of this specialized instrumentation to the larger organization which can afford the sizable financial outlay involved.

h. Automated analytical instrumentation: Automated analytical instruments are now available and in use to run individual samples at rates of 10 to 60 samples per hour. The same instruments can be modified to perform analyses for two to twelve constituents simultaneously from one sample. The instruments are composed of a group of interchangeable modules joined together in series by a tubing system. Each module performs the individual operations of filtering, heating, digesting, time delay, color sensing, etc., that the procedure requires.

The read-out system employs sensing elements with indicators, alarms and/or recorders. For monitoring applications, automatic standardization-compensation, electrical and chemical, is done by a self-adjusting recorder when known chemical standards are sent periodically through the same analytical train. Such instrument systems are presently available.

Appropriate methodology is supplied by the manufacturer for many of the common constituents of water and wastewater. Some methods are based on procedures described in this manual, while others originate from the manufacturer's adaptation of published research. Since a number of methods of

varying reliability may be available for a single constituent of water and wastewater, a critical appraisal of the method adopted is obviously mandatory.

Automated methodology is susceptible to the same interferences as the original method from which it derives. For this reason, new methods developed for automated analysis must be subjected to the exacting tests for accuracy and freedom from adverse response already met by the accepted standard methods.

Off color and turbidity produced during the course of an analysis will be visible to an analyst manually performing a given determination, and the result will be properly discarded. Such abnormal effects caused by unsuspected interferences might escape notice in an automated analysis. Calibration of the instrument system at least once each day with standards containing interferences of known concentration could help to expose such difficulties. Routine practice is to check instrument action and guard against questionable results by the insertion of standards and blanks at regular intervals—perhaps after every 10 samples in the train. Another important precaution is proper sample identification by arrangement into convenient groups.

In brief, a fair degree of operator skill and knowledge, together with adequately detailed instructions, is required for successful automated analysis.

i. Other newer methods of analysis: Instrumentation and new methods of analysis are always under development. The analyst will find it to his advantage to keep abreast of current progress. Reviews of each branch of analytical chemistry are published regularly in the periodical, *Analytical Chemistry.*

9. *Interferences*

Many analytical procedures are subject to interference from substances which may be present in the sample. The more common and obvious interferences are known, and information about them has been given in the details of individual procedures. It is inevitable that the analyst will encounter interferences about which he is not forewarned. Such occurrences are unavoidable because of the diverse nature of waters and particularly of wastewaters. Therefore, the analyst must be alert to the fact that hitherto untested ions, new treatment compounds—especially complexing agents—and new industrial wastes constitute an ever-present threat to the accuracy of chemical analyses. He must be on his guard at all times to detect the occurrence of such interferences.

Any sudden change in the apparent composition of a supply which has been rather constant, any off color observed in a colorimetric test or during a titration, any unexpected turbidity, odor or other laboratory finding is cause for suspicion. Such a change may be due to a normal variation in the relative concentrations of the usual constituents, but it may be caused by the introduction of an unforeseen interfering substance.

A few substances—such as chlorine, chlorine dioxide, alum, iron salts, silicates, copper sulfate, ammonium sulfate and polyphosphates—are so widely used in water treatment that they deserve special mention as possible causes of interference. Of these, chlorine is

probably the worst offender, in that it bleaches or alters the colors of many of the sensitive organic reagents which serve as titration indicators and as color developers for photometric methods. Among the methods which have proved effective in removing chlorine residuals are: the addition of minimal amounts of sulfite, thiosulfate or arsenite; exposure to sunlight or an artificial ultraviolet source; and prolonged storage.

Whenever interference is encountered or suspected, and no specific recommendations are found in this manual for overcoming it, the analyst must endeavor to determine what technic, if any, suffices to eliminate the interference without adversely affecting the analysis itself. If two or more choices of procedure are offered, often one procedure will be less affected than another by the presence of the interfering substance. If different procedures yield considerably different results, it is likely that interference is present. Some interferences become less severe upon dilution, or upon use of smaller aliquots; any tendency of the results to increase or decrease in a consistent manner with dilution indicates the likelihood of interference effects.

a. Interference may cause the analytical results to be either too high or too low, as a consequence of one of the following processes:

1) An interfering substance may react like the substance sought, and thus produce a high result—for example, bromide will respond to titration as though it were chloride.

2) An interfering substance may react with the substance sought and thus produce a low result—for example, chloride will react with a portion of the nitrate in the presence of the sulfuric acid, using the phenoldisulfonic acid method.

3) An interfering substance may combine with the analytical reagent and thus prevent it from reacting with the substance sought—for example, chlorine will destroy many indicators and color-developing reagents.

Nearly every interference will fit one of these classes. For example, in a photometric method, turbidity may be considered as a "substance" which acts like the one being determined—that is, it reduces the transmission of light. Occasionally, two or more interfering substances, if present simultaneously, may interact in a nonadditive fashion, either canceling or enhancing one another's effects.

b. The best way to minimize interference is to remove the interfering substance or to render it innocuous by one of these methods:

1) Either the substance sought or the interfering substance may be removed physically: For example, fluoride and ammonia may be distilled off, leaving interferences behind; chloride may be converted to silver chloride and filtered off, leaving nitrate behind. The interferences may also be adsorbed on an ion-exchange resin, a process described more fully in Section 100B.

2) The pH may be adjusted so that only the substance sought will react.

3) The sample may be oxidized or reduced to convert the interfering substance to a harmless form—for example, chlorine may be reduced to chloride by adding thiosulfate.

4) The addition of a suitable agent may complex the interfering substance so that it is innocuous although still present: For example, iron may be complexed with pyrophosphate to pre-

vent it from interfering with the copper determination; copper may be complexed with cyanide or sulfide to prevent interference with the titrimetric hardness determination.

5) A combination of the first four technics may be used: For example, phenols are distilled from an acid solution to prevent amines from distilling; thiosulfate is used in the dithizone method for zinc to prevent most of the interfering metals from passing into the carbon tetrachloride layer.

6) Color and turbidity may sometimes be destroyed by wet or dry ashing, or may be removed by use of a flocculating agent. Some types of turbidity may be removed by filtration. These procedures, however, introduce the danger that the desired constituent will also be removed.

c. If none of these technics is practical, several methods of compensation can be used:

1) If the color or turbidity initially present in the sample interferes in a photometric determination, it may be possible to use photometric compensation. The technic is described in Section 000A.7 preceding.

2) The concentration of interfering substances may be determined and then identical amounts may be added to the calibration standards. This involves much labor.

3) If the interference does not continue to increase as the concentration of interfering substance increases, but tends to level off, then a large excess of interfering substance may be added routinely to all samples and to all standards. This is called "swamping." For example, an excess of calcium is added in the photometric magnesium determination.

4) The presence in the chemical reagents of the substance sought may be accounted for by carrying out a blank determination.

10. Recovery

A qualitative estimate of the presence or absence of interfering substances in a particular determination may be made by means of a recovery procedure. Although this method does not enable the analyst to apply any correction factor to the results of an analysis, it does give him some basis for judging the applicability of a particular method of analysis to a particular sample. Furthermore, it enables the analyst to obtain information in this regard without an extensive investigation to determine exactly which substances can interfere in the method used. It also does away with the necessity of making separate determinations on the sample for the interfering substances themselves.

A recovery may be performed at the same time as the determination itself. Of course, recoveries would not be run on a routine basis with samples whose general composition is known or when using a method whose applicability to the sample is well established. Recovery methods are to be regarded as tools to remove doubt about the applicability of a method to a sample. In brief, the recovery procedure involves applying the analytical method to a reagent blank; to a series of known standards covering the expected range of concentration of the sample; to the sample itself, in at least a duplicate run; and to the recovery samples, prepared by adding known quantities of the substance sought to separate portions of the sample itself, each portion equal to

the size of sample taken for the run. The substance sought should be added in sufficient quantity to overcome the limits of error of the analytical method, but without causing the total in the sample to exceed the range of the known standards used.

The results are first corrected by subtracting the reagent blank from each of the other determined values. The resulting known standards are then graphically represented. From this graph, the amount of sought substance in the sample alone is determined. This value is then subtracted from each of the determinations consisting of sample plus known added substance. The resulting amount of substance divided by the known amount originally added and multiplied by 100 gives the percentage recovery.

The procedure outlined above may be applied to colorimetric or instrumental methods of analysis. It may also be applied in a more simple form to titrimetric, gravimetric and other types of analyses.

Rigid rules concerning the percentage recoveries required for acceptance of results of analyses for a given sample and method cannot be stipulated. Recoveries of substances in the range of the sensitivity of the method may, of course, be very high or very low and approach a value nearer to 100% recovery as the error of the method becomes small with respect to the magnitude of the amount of substance added. In general, intricate and exacting procedures for trace substances which have inherent errors due to their complexity may give recoveries that would be considered very poor and yet, from the practical viewpoint of usefulness of the result, may be quite acceptable. Poor results may reflect either interferences present in the sample or real inadequacy of the method of analysis in the range in which it is being used.

It must be stressed, however, that the judicious use of recovery methods for the evaluation of analytical procedures and their applicability to particular samples is an invaluable aid to the analyst in both routine and research investigations.

000 B. Expression of Results

1. Units

Analytical results should be expressed in milligrams per liter (mg/l). Assuming that 1 liter of water, sewage or industrial waste weighs 1 kilogram, milligrams per liter is equivalent to parts per million.* Only the significant figures (see Section 000B.2 below) should be recorded.

If the concentrations are generally less than 1 mg/l, it may be more convenient to express the results in micrograms per liter (μg/l). This is equivalent to parts per billion (ppb), where billion is understood to be 10^9. If the concentration is greater than 10,000

* It should be noted that, in water analysis, "parts per million" is *always* understood to imply a weight/weight ratio, even though in practice a volume may be measured instead of a weight. By contrast, "percent" may be either a volume/volume or a weight/weight ratio.

mg/l, the results should be expressed in percent, 1% being equivalent to 10,000 mg/l.

In reporting analyses of stream pollution or evaluating plant operation and efficiencies, it is desirable to express the results on a weighted basis, including both the concentration and the volume of flow in cubic feet per second (cfs) or million gallons daily (mgd). These weighted results may be expressed as *quantity units* (*QU*) according to the practice of the U.S. Public Health Service; as *pounds per 24 hr;* or as *population equivalents* based on biochemical oxygen demand (BOD). Totals of the weighted units may be converted to the weighted average mg/l. The various units are calculated as follows:

$$QU_1 = (\text{mg/l}) \times (1{,}000 \text{ cfs})$$

$$QU_2 = (\text{mg/l}) \times (\text{mgd})$$

$$\text{lb/24 hr} = (\text{mg/l}) \times (\text{mgd}) \times 8.34$$

$$\text{lb/24 hr} = (\text{mg/l}) \times (\text{cfs}) \times 5.39$$

$$\text{Population equivalent} = (\text{mg/l 5-day BOD}) \times (\text{mgd}) \times \frac{8.34}{0.17}$$

Table 000(1) presents the factors which are useful for converting the concentrations of the common ions found in water—from milligrams per liter to milliequivalents per liter, and vice versa. The term milliequivalent used in this table represents 0.001 of an equivalent weight. The equivalent weight, in turn, is defined as the weight of the ion (sum of the atomic weights

TABLE 000(1): CONVERSION FACTORS *

(Milligrams per Liter—Milliequivalents per Liter)

Ion (Cation)	me/l = mg/l ×	mg/l = me/l ×
Al^{3+}	0.1112	8.994
B^{3+}	0.2775	3.604
Ba^{2+}	0.01456	68.67
Ca^{2+}	0.04990	20.04
Cr^{2+}	0.05770	17.33
Cu^{2+}	0.03148	31.77
Fe^{2+}	0.03581	27.92
Fe^{3+}	0.05372	18.62
H^+	0.9921	1.008
K^+	0.02557	39.10
Li^+	0.1441	6.939
Mg^{2+}	0 08226	12.16
Mn^{2+}	0.03640	27.47
Mn^{4+}	0.07281	13.73
Na^+	0.04350	22.99
NH_4^+	0.05544	18.04
Pb^{2+}	0.009653	103.6
Sr^{2+}	0.02283	43.81
Zn^{2+}	0.03060	32.69

Ion (Anion)	me/l = mg/l ×	mg/l = me/l ×
BO_2^-	0.02336	42.81
Br^-	0.01251	79.91
Cl^-	0.02821	35.45
CO_3^{2-}	0.03333	30.00
CrO_4^{2-}	0.01724	58.00
F^-	0.05264	19.00
HCO_3^-	0.01639	61.02
HPO_4^{2-}	0.02084	47.99
$H_2PO_4^-$	0.01031	96.99
HS^-	0.03024	33.07
HSO_3^-	0.01233	81.07
HSO_4^-	0.01030	97.07
I^-	0.007880	126.9
NO_2^-	0.02174	46.01
NO_3^-	0.01613	62.00
OH^-	0.05880	17.01
PO_4^{3-}	0.03159	31.66
S^{2-}	0.06238	16.03
SiO_3^{2-}	0.02629	38.04
SO_3^{2-}	0.02498	40 03
SO_4^{2-}	0.02083	48 03

* Factors are based on ion charge and not on redox reactions which may be possible for certain of these ions. Cations and anions are listed separately in alphabetical order.

of the atoms making up the ion) divided by the number of charges normally associated with the particular ion. The factors for converting results from mg/l to me/l were computed by dividing the ion charge by the weight of the ion. Conversely, the factors for converting results from me/l to mg/l were calculated by dividing the weight of the ion by the ion charge. This table is offered for the convenience of laboratories which report results in me/l as well as mg/l.

2. *Significant Figures*

To avoid ambiguity in reporting results or in presenting directions for a procedure, it is the custom to use "significant figures." All the digits in a reported result are expected to be known definitely, except for the last digit, which may be in doubt. Such a number is said to contain only significant figures. If more than a single doubtful digit is carried, the extra digit or digits are not significant. If an analytical result is reported as "75.6 mg/l," the analyst should be quite certain of the "75," but may be uncertain as to whether the ".6" should be .5 or .7, or even .4 or .8, because of unavoidable uncertainty in the analytical procedure. If the standard deviation were known from previous work to be ±2 mg/l, the analyst would have, or at least should have, rounded off the result to "76 mg/l" before reporting it. On the other hand, if the method were so good that a result of "75.61 mg/l" could have been conscientiously reported, then the analyst should not have rounded it off to 75.6.

A report should present only such figures as are justified by the accuracy of the work. The all too common practice of requiring that quantities listed in a column have the same number of figures to the right of the decimal point is justified in bookkeeping, but not in chemistry.

a. Rounding off: Rounding off is accomplished by dropping the digits which are not significant. If the digit 6, 7, 8 or 9 is dropped, then the preceding digit must be increased by one unit; if the digit 0, 1, 2, 3 or 4 is dropped, the preceding digit is not altered. If the digit 5 is dropped, the preceding digit is rounded off to the nearest even number: thus 2.25 becomes 2.2, and 2.35 becomes 2.4.

b. Ambiguous zeros: The digit 0 may record a measured value of zero, or it may serve merely as a spacer to locate the decimal point. If the result of a sulfate determination is reported as 420 mg/l, the recipient of the report may be in doubt whether the zero is significant or not, because the zero cannot be deleted. If an analyst calculates a total residue (total solids) content of 1,146 mg/l, but realizes that the 4 is somewhat doubtful and that therefore the 6 has no significance, he will round off the answer to 1,150 mg/l and so report, but here, too, the recipient of the report will not know whether the zero is significant. Although the number could be expressed as a power of 10 (e.g., 11.5×10^2 or 1.15×10^3), this form is not generally used, as it would not be consistent with the normal expression of results and might also be confusing. In most other cases, there will be no doubt as to the sense in which the digit 0 is used. It is obvious that the zeros are significant in such numbers as 104, 40.08, and 0.0003. In a number written as 5.000, it is understood that all the zeros are significant, or else the number could have been

rounded off to 5.00, 5.0, or 5, whichever was appropriate. Whenever the zero is ambiguous, it is advisable to accompany the result with an estimate of its uncertainty.

Sometimes, significant zeros are dropped without good cause. If a buret is read as "23.60 ml," it should be so recorded, and not as "23.6 ml." The first number indicates that the analyst took the trouble to estimate the second decimal place; "23.6 ml" would indicate that he read the buret rather carelessly.

c. The plus-or-minus (±) notation: If a calculation yields as a result "1,476 mg/l" with a standard deviation estimated as ±40 mg/l, it should be reported as 1,480 ± 40 mg/l. But if the standard deviation is estimated as ±100 mg/l, the answer should be rounded off still further and reported 1,500 ± 100 mg/l. By this device, ambiguity is avoided and the recipient of the report can tell that the zeros are only spacers. Even if the problem of ambiguous zeros is not present, showing the standard deviation is helpful in that it provides an estimate of reliability.

d. Calculations: As a practical operating rule, the result of a calculation in which several numbers are multiplied or divided together should be rounded off to as few significant figures as are present in the factor with the fewest significant figures. Suppose that the following calculation must be made in order to obtain the result of an analysis:

$$\frac{56 \times 0.003462 \times 43.22}{1.684}$$

A ten-place desk calculator yields an answer of "4.975740996," but this number must be rounded off to a mere "5.0" because one of the measurements, 56, which entered into the calculation has only two significant figures. It was a waste of time to measure the other three factors to four significant figures because the "56" is "the weakest link in the chain" and limits the accuracy of the answer. If the other factors were measured to only three, instead of four, significant figures, the answer would not suffer and the labor would be less.

When adding or subtracting numbers, that number which has the fewest decimal places, not necessarily the fewest significant figures, puts the limit on the number of places that may justifiably be carried in the sum or difference. Thus the sum

```
     0.0072
    12.02
     4.0078
    25.9
 4,886
-----------
 4,927.9350
```

must be rounded off to a mere "4,928," no decimals, because one of the addends, 4,886, has no decimal places. Notice that another addend, 25.9, has only three significant figures and yet it does not set a limit to the number of significant figures in the answer.

The preceding discussion is necessarily oversimplified, and the reader is referred to the bibliography for a more detailed discussion.

000 C. Precision and Accuracy

A clear distinction should be made between the terms "precision" and "accuracy" when applied to methods of analysis. *Precision* refers to the reproducibility of a method when repeated on a homogeneous sample under controlled conditions, regardless of whether or not the observed values are widely displaced from the true value as a result of systematic or constant errors present throughout the measurements. Precision can be expressed by the standard deviation. *Accuracy* refers to the agreement between the amount of a component measured by the test method and the amount actually present. *Relative error* expresses the difference between the measured and the actual amounts, as a percentage of the actual amount. A method may have very high precision but recover only a part of the element being determined; or an analysis, although precise, may be in error because of poorly standardized solutions, inaccurate dilution technics, inaccurate balance weights, or improperly calibrated equipment. On the other hand, a method may be accurate but lack precision because of low instrument sensitivity, variable rate of biologic activity, or other factors beyond the control of the analyst.

It is possible to determine both the precision and the accuracy of a test method by analyzing samples to which known quantities of standard substances have been added. It is possible to determine the precision, but not the accuracy, of such methods as those for suspended solids, BOD, and numerous physical characteristics because of the unavailability of standard substances that can be added in known quantities on which percentage recovery can be based.

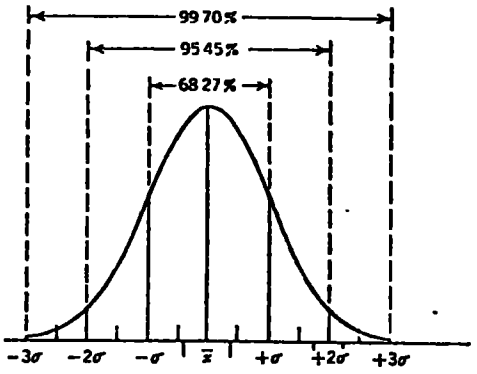


Figure 1. Gaussian or normal curve of frequencies.

1. Statistical Approach

a. Standard deviation (σ): Experience has shown that if a determination is repeated a large number of times under essentially the same conditions, the observed values, x, will be distributed at random about an average as a result of uncontrollable or experimental errors. If there are an infinite number of observations from a common universe of causes, a plot of the relative frequency against magnitude will produce a symmetrical bell-shaped curve known as the Gaussian or normal curve (Figure 1). The shape of this curve is completely defined by two statistical parameters: (1) the mean or average, $\bar{x}$, of n observations; and (2) the standard deviation, σ, which fixes the width or spread of the curve on each side of the mean. The formula is:

$$\sigma = \sqrt{\frac{\Sigma (x - \bar{x})^2}{n - 1}}$$

The proportion of the total observations lying within any given range

about the mean is related to the standard deviation. For example, 68.27% of the observations lie between $\bar{x} \pm 1\ \sigma$; 95.45%, between $\bar{x} \pm 2\ \sigma$; and 99.70%, between $\bar{x} \pm 3\ \sigma$. These limits do not apply exactly for any finite sample from a normal population; the agreement with them may be expected to be better as the number of observations, n, increases.

b. Application of standard deviation: If the standard deviation, σ, for a particular analytical procedure has been determined from a large number of samples, and a set of n replicates on a sample gives a mean result $\bar{x}$, there is a 95% chance that the true value of the mean for this sample lies within the values $\bar{x} \pm 1.96\ \sigma/\sqrt{n}$. This range is known as the 95% confidence interval. It provides an estimate of the reliability of the mean, and may be used to forecast the number of replicates needed to secure suitable precision.

If the standard deviation is not known and is estimated from a single small * sample, or a few small samples, the 95% confidence interval of the mean of n observations is given by the equation $\bar{x} \pm t\sigma/\sqrt{n}$, where t has the following values:

n	t
2	12.71
3	4.30
4	3.18
5	2.78
10	2.26
∞	1.96

The use of t compensates for the tendency of small samples to underestimate the variability.

* A "small sample" in statistical discussions means a small number of replicate determinations, n, and does not refer to the quantity used for a determination.

c. Range (R): The difference between the smallest and largest of n observations is also closely related to the standard deviation. When the distribution of errors is normal in form, the range, R, of n observations exceeds the standard deviation times a factor d_n only in 5% of the cases. Values for the factor d_n are:

n	d_n
2	2.77
3	3.32
4	3.63
5	3.86
6	4.03

As it is rather general practice to run replicate analyses, use of these limits is very convenient for detecting faulty technic, large sampling errors, or other assignable causes of variation.

d. Rejection of experimental data: Quite often in a series of observations, one or more of the results deviate greatly from the mean, whereas the other values are in close agreement with the mean value. The problem arises at this point as to rejection of the disagreeing values. Theoretically, no results should be rejected, since the presence of disagreeing results shows faulty technics and therefore casts doubt on all the results. Of course the result of any test in which a known error has occurred is rejected immediately. For methods for the rejection of other experimental data, standard texts on analytical chemistry or statistical measurement should be consulted.

e. Presentation of precision and accuracy data: The precision and accuracy data are presented in one of three ways in this volume, depending on when and how the information was originally assembled.

In point of time, the oldest data are given in the wastewater section and present for the most part the precision with which certain determinations can be performed. These data first appeared in the 10th Edition and survive unchanged in the current volume. The complex character of wastewater samples initially dictated this approach.

Beginning with the 11th Edition, a concerted effort was made to offer an idea of the precision and accuracy with which selected methods can be applied on a broad geographic basis in examination of the relatively simpler water samples. The manner of best expressing the resulting data has remained to this day a matter of relentless study. The 11th and 12th Editions presented both precision and accuracy in terms of mg/l. This practice is retained for the time being where such data continue to be cited in this manual. However, experience of the past decade suggests that data can be presented with greater brevity and easier understanding in the form of a percentage. By this system, the standard deviation is expressed as a percentage of the mean and is now termed the relative standard deviation. It measures the precision or reproducibility of a method, independent of the known concentration of the sample constituent. Similarly, the relative error gives the difference between the mean of a series of test results and the true value, expressed as a percentage of the true value. Thus, the relative error represents the measure of the accuracy of a method. The relative standard deviation and relative error are preferred in quoting the precision and accuracy of a method because they are independent of the concentration.

f. Quality control: Quality control may be defined for the purpose of this manual as a statistical system for monitoring the precision (variation), or reproducibility, of analytical procedures in a given laboratory.

The control chart provides an important tool for identifying the causes of variation in the quality of a procedure. Certain variations in chemical procedures occur by chance, about which little or nothing can be done. However, variations can also result from "assignable causes" such as differences in methods, reagents, equipment, and the skill of persons performing the tests. Chance variations behave in a random manner and show no cycles, runs, or similarly recognizable pattern. If, on the other hand, the variations in the data exhibit cycles, runs, or a definite pattern, at least one assignable cause may be at work, and the conditions producing the variations are said to be "out of control."

Two basic types of control charts have proved valuable. The $\bar{x}$-chart is used to monitor the average of a procedure, while the R-chart is used to monitor the variability of a procedure. An $\bar{x}$-chart discloses the variation in the averages of a number of replications of a given procedure. It consists of a central line, $\bar{\bar{x}}$, and upper and lower control limits, which may range from $+1\sigma$ to $+3\sigma$ and -1σ to -3σ standard deviations from the center line. (The values of $\bar{\bar{x}}$ and the standard deviation are derived from past data.) Figure 5 in Section 100C.1 illustrates one application of control charts. As long as the sample averages remain inside the control limits and show only random variation within the limits, the procedure is said to be "in control" with respect to its central tendency. If an average falls outside the control

limits, or if there is nonrandom variation within the limits, the process is said to be "out of control" with respect to its central tendency. Such a condition should prompt an investigation into the assignable cause or causes of the extreme variation.

The same basic principles which apply to the $\bar{x}$-chart also hold for the R-chart, except that the R-chart is a plotting of the ranges of samples. It reveals variations in the ranges of samples rather than variation in the averages of samples.

One of the most important factors in a quality control program is an adequate supply of a stable known control. This control can be a large sample from a natural source known to contain the constituent of concern or a synthetic sample prepared in the laboratory from chemicals of the highest purity grade. Once the test to be controlled has been selected, 20 or more determinations for the same constituent in the control sample are made under routine daily conditions. The values are then totaled and the average value is obtained. The standard deviation is calculated to ascertain the range of allowable variation that can be expected in routine work for this particular constituent. If this same sample is then treated as a routine daily control sample, it is possible to determine by the use of a control chart constructed from the original 20 determinations whether the daily assays for this constituent are in or out of control. When the control sample is prepared from chemicals of the highest purity, the probable accuracy of the determination can also be estimated.

Duncan's volume on *Quality Control and Industrial Statistics* describes the $\bar{x}$- and R-charts in detail and their relevance to quality control. Youden's book on *Statistical Techniques for Collaborative Tests* offers valuable information on collaborative tests.

2. *Graphical Representation of Data*

Graphical representation of data is one of the simplest methods for showing the influence of one variable on another. Graphs are frequently desirable and advantageous in colorimetric analysis because they show any variation of one variable with respect to the other within specified limits.

a. General: Ordinary rectangular-coordinate paper is satisfactory for most purposes. Twenty lines per inch is recommended. Semilogarithmic paper is convenient when one of the coordinates is to be the logarithm of an observed variable.

The five rules listed by Worthing and Geffner for choosing the coordinate scales are useful. Although these rules are not inflexible, they are satisfactory. When doubt arises, common sense should prevail. The rules are:

1) The independent and dependent variables should be plotted on abscissa and ordinate in a manner which can be easily comprehended.

2) The scales should be chosen so that the value of either coordinate can be found quickly and easily.

3) The curve should cover as much of the graph paper as possible.

4) The scales should be chosen so that the slope of the curve approaches unity as nearly as possible.

5) Other things being equal, the variables should be chosen to give a plot which will be as nearly a straight line as possible.

The title of a graph should adequately describe what the plot is intended to show. Legends should be presented on

the graph to clarify possible ambiguities. Complete information on the conditions under which the data were obtained should be included in the legend.

b. Method of least squares: If sufficient points are available and the functional relationship between the two variables is well defined, a smooth curve can be drawn through the points. If the function is not well defined, as is frequently the case when using experimental data, the method of least squares is used to fit a straight line to the pattern.

Any straight line can be represented by the equation $x = my + b$. The slope of the line is represented by the constant m and the slope intercept (on the x axis) is represented by the constant b. The method of least squares has the advantage of giving a set of values for these constants not dependent upon the judgment of the investigator. Two equations besides the one for a straight line are involved in these calculations:

$$m = \frac{n\Sigma xy - \Sigma x \Sigma y}{n\Sigma y^2 - (\Sigma y)^2}$$

$$b = \frac{\Sigma y^2 \Sigma x - \Sigma y \Sigma xy}{n\Sigma y^2 - (\Sigma y)^2}$$

n being the number of observations (sets of x and y values) to be summed. In order to compute the constants by this method, it is first necessary to calculate Σx, Σy, Σy^2, and Σxy. These operations are carried out to more places than the number of significant figures in the experimental data because the experimental values are assumed to be exact for the purposes of the calculations.

Example: Given the following data to be graphed, find the best line to fit the points:

Absorbance	Solute Concentration *mg/l*
0.10	29.8
0.20	32.6
0.30	38.1
0.40	39.2
0.50	41.3
0.60	44.1
0.70	48.7

Let y equal the absorbance values which are subject to error, and x the accurately known concentration of solute. The first step is to find the summations (Σ) of x, y, y^2, and xy:

x	y	y^2	xy
29.8	0.10	0.01	2.98
32.6	0.20	0.04	6.52
38.1	0.30	0.09	11.43
39.2	0.40	0.16	15.68
41.3	0.50	0.25	20.65
44.1	0.60	0.36	26.46
48.7	0.70	0.49	34.09
$\Sigma = 273.8$	2.80	1.40	117.81

The next step is to substitute the summations in the equations for m and b; $n = 7$ as there are seven sets of x and y values:

$$m = \frac{7\,(117.81) - 2.80\,(273.8)}{7\,(1.40) - (2.80)^2} = 29.6$$

$$b = \frac{1.4\,(273.8) - 2.80\,(117.81)}{7\,(1.40) - (2.80)^2} = 27.27$$

To plot the line, three convenient values of y are selected—say, 0, 0.20, 0.60—and corresponding values of x are calculated:

$x_0 = 29.6(0) + 27.27 = 27.27$
$x_1 = 29.6(0.20) + 27.27 = 33.19$
$x_2 = 29.6(0.60) + 27\,27 = 45.04$

When the points representing these values are plotted on the graph, they will lie in a straight line (unless an error in calculation has been made), which is the line of best fit for the

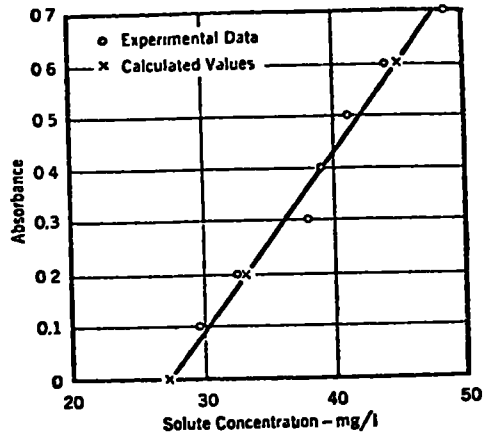


Figure 2. Example of least-squares method.

given data. The points representing the latter are also plotted on the graph, as in Figure 2.

3. *Self-Evaluation (Desirable Philosophy for the Analyst)*

A good analyst continually tempers his confidence with doubt. Such doubt stimulates a search for new and different methods of confirmation for his reassurance. Frequent self-appraisals should embrace every step—from collecting samples to reporting results.

The analyst's first critical scrutiny should be directed at the entire sample collection process in order to guarantee a representative sample for the purpose of the analysis and to avoid any possible losses or contamination during the act of collection. Attention should also be given to the type of container and to the manner of transport and storage, as discussed elsewhere in this volume.

A periodic reassessment should be made of the available analytical methods, with an eye to applicability for the purpose and the situation. In addition, each method selected must be evaluated by the analyst himself for sensitivity, precision and accuracy, because only in this way can he determine whether his technic is satisfactory and whether he has interpreted the directions properly. Self-evaluation on these points can give the analyst confidence in the value and significance of his reported results.

The benefits of less rigid intralaboratory as well as interlaboratory evaluations deserve serious consideration. The analyst can regularly check standard or unknown concentrations with and without interfering elements, and compare results on the same sample with other workers in the laboratory. Such programs can uncover weaknesses in the analytical chain and enable improvements to be instituted without delay. The results can disclose whether the trouble stems from faulty sample treatment, improper elimination of interference, poor calibration practices, sloppy experimental technic, impure or incorrectly standardized reagents, defective instrumentation, or even inadvertent mistakes in arithmetic.

Other checks of a water analysis are described in Section 100C and involve anion-cation balance, specific conductance, ion exchange, and the recovery of added substance in the sample (see also Section 000A.10 preceding).

All these approaches are designed to appraise and upgrade the level of laboratory performance and thus inspire greater faith in the final reported results.

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200 D. Methods Evaluation by the Committee

The Committee on Standard Methods of the Water Pollution Control Federation has attempted to establish the precision and accuracy of the methods in Part 200. For many methods, results were obtained from ten replicate determinations on 10 different days or, when necessary, from five replicate samples on 20 days.

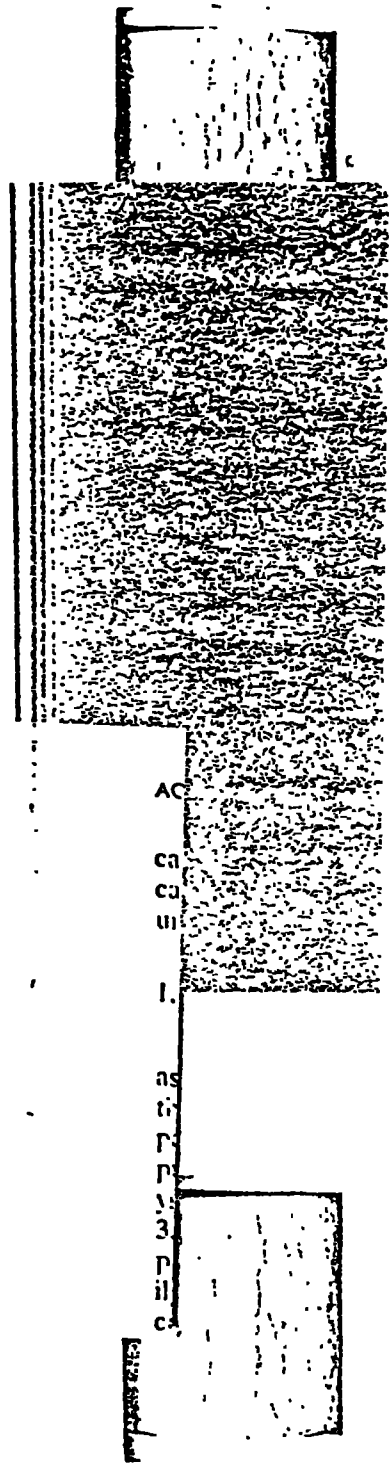
Most methods studied were found to be statistically reliable, and the standard deviations given may be used with some confidence in statistical prediction. If a method has been found statistically unreliable, this is indicated in the statements on precision under the method. The standard deviations of unreliable methods cannot safely be used for statistical prediction, but may be of some value for indicating roughly the variation that may be expected.

In expressing the evaluation data on

each test, the number of analysts and determinations is given in shorthand form; for example, "$n = 5$; 56×10," which means that 5 different analysts ran 56 separate sets of 10 determinations each, making a total of 560 determinations. Usually the precision is expressed as the standard deviation in original units of measurement—i.e., milligrams or milliliters. In a few instances, the precision is expressed as the coefficient of variation C_v (the ratio of the standard deviation to the average), expressed as a percentage:

$$C_v = \frac{100\,\sigma}{\bar{x}}$$

The standard deviation given with each method is based on careful laboratory examination. No attempt has been made to obtain the standard deviation under research conditions, or with the use of specially calibrated apparatus or glassware. The values given are to be regarded as provisional in nature and subject to change on further study. In general, the standard deviations given may be regarded as being too high rather than too low.



8.3 Specific Analytical Methods for the Analysis of Relatively Concentrated Leachate Samples

8.3.1 Introduction

Specific analytical methods for the analysis of relatively concentrated leachate samples were investigated in the report "Compilation of Methodology for Measuring Pollution Parameters of Landfill Leachate": by E.S.K. Chian and F.B. DeWalle, University of Illinois, EPA Program Element No. 1DB064. It is stated in the abstract, P. IV of the subject report:

"Since different analytical methods can be used to determine a specific parameter, a preliminary laboratory evaluation was made of those methods least subject to interferences. All analyses were conducted with a relatively concentrated leachate sample obtained from a lysimeter filled with milled solid waste. The results indicate that strong interferences are sometimes encountered when using colorimetric tests due principally to the color and suspended solids present in leachate. In such instances, alternative methods were evaluated or recommendations were made to reduce the interfering effects. Automated chemical analysis using colorimetric methods can sometimes experience significant interferences.

Further research is necessary to evaluate additional methods using leachate samples of different strengths and collected from landfills of different ages. The precision and sensitivity of each method will also have to be determined. The interfering parameter should be quantified to allow predictions of its magnitude with leachate samples of different strengths."

Also, in the above-cited report, Introduction, P.3, it is stated:

"It is the purpose of the present study to review the analytical methods to determine contaminants as reported in the literature. The methods compiled and

evaluated in this study were generally reported in the literature; additional information was obtained by contacting the principal investigators. Interferences in the chemical analysis due to the complex nature of the leachate as enumerated in the reported studies are listed in this report.

The compilation showed that different methods subject to different interferences are used to determine a certain parameter. For each parameter, only that method was evaluated in this laboratory which was found to have the smallest interference. The laboratory evaluation tested the method for its susceptibility to certain interferences commonly found in leachate. In addition, the accuracy of the method was tested. All laboratory analyses were performed using a high strength leachate sample obtained from a recently installed lysimeter filled with milled refuse. Recommendations made in this report, therefore, only apply to leachate of similar strength. No evaluation was made of precision and sensitivity of each method since this was beyond the scope of the work. Realizing the above restrictions, recommendations were made in the present study for the selection of those methods least subject to interference. Further recommendations were made concerning modifications of the selected methods."

8.3.2 Measurement of Interference Effects

Two general procedures were used by Chian and DeWalle to deal with interference effects in their evaluation of specific analytical methods. These procedures were the Standard Addition Method and the Dilution Method. These methods are discussed in this report on pp. 12-15, which are reproduced herewith.

In general, it would be expected that interferences encountered in concentrated leachates would be relatively severe and constitute "worst case" effects when compared with more dilute leachates. Leachates obtained in the field (landfills) for analysis may vary greatly in total concentration, i.e. from total concentration

SECTION 5

METHODOLOGY OF METHOD EVALUATION

Since most of the leachate studies have been conducted by researchers in the sanitary or environmental engineering fields, the methods that are used closely reflect those of Standard Methods (APHA, 1971). Studies between 1960 and 1965 used the 11th edition, between 1965 and 1971 the 12th edition and after 1971 the 13th edition. Laboratories not employing complicated instruments, sometimes use methods listed by Hach Chemical Company, Handbook of Water Analysis (Hach Chemical Company, 1973). Methods used by geologists are generally those reported in Techniques of Water Resources Investigation of the U. S. Geological Survey (U. S. Geological Survey, 1970). Recent studies use the EPA procedures in Methods for Chemical Analysis of Water and Wastes (EPA, 1974) which also contain optional procedures for automated analysis. Most studies employing automated chemical analysis, however, use methods recommended by Technicon Industrial Systems, Industrial Methods (Technicon, 1973).

The different parameters that have been determined in the studies reported in the literature are listed below. Each section contains a survey of the different methods used to analyze a certain parameter, and the obtained experiences. The method least interferred with by the matrix of the leachate sample was selected and then evaluated in greater detail in the present study. The method was evaluated with the standard addition method and by using progressively increasing dilutions.

5.1 STANDARD ADDITION METHOD

The standard addition method is widely used in chemical analysis when interferences present in the sample cannot be avoided. An advantage of this method is that it avoids the necessity of preparing synthetic standards of a composition similar to that of the sample (Geological Survey, 1970). In this method equal volumes of sample are added to a water blank and standards containing increasing but known amounts of the test element. The volume of the blank and the standards must have the same volume to result in a similar dilution of the sample. The diluted samples containing increasing amounts of the test element are then analyzed according to the standard procedures. The obtained values are then plotted on the vertical axis of a graph while the concentration of the known standards are plotted on the horizontal axis (Figure 3). When the resulting line is extrapolated to zero measured concentration, the point of interception of the abscissa is the concentration of the unknown element. The abscissa on the left of the ordinate is scaled the same as on the right side, but in the opposite direction of the ordinate. Since the scale of the ordinate and abscissa are identical, a line drawn under 45° from the extrapolated point on the abscissa to the ordinate represents a 100 percent recovery of the added element. Thus 100 percent of the known amount added to the diluted sample is recovered. If the actual line connecting the points has a slope lower than 45° the recovery of the added element is less than 100 percent while a slope higher than 45° represents a higher than 100 percent recovery. The

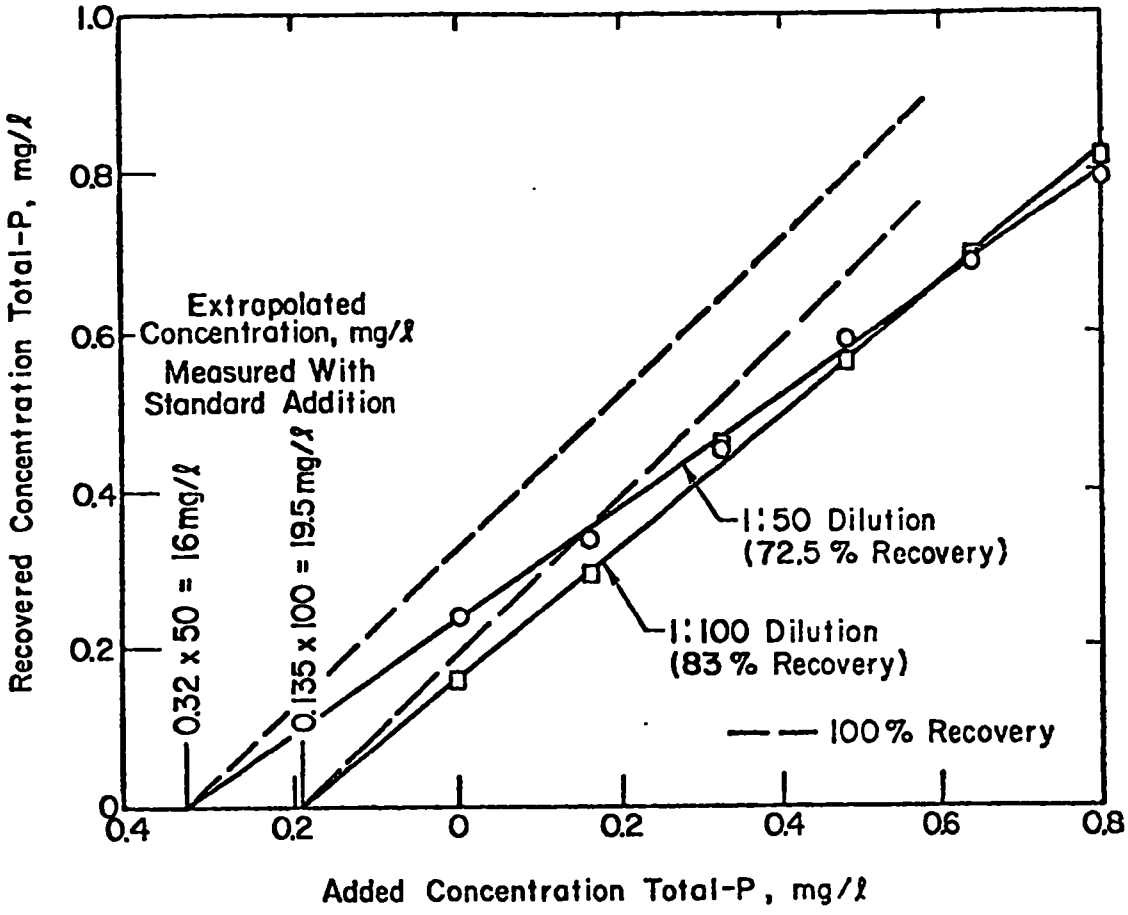


Figure 3. The Total-P Determination with the Ascorbic Acid Method in the 1:50 and 1:100 Diluted Leachate Sample Using the Standard Addition Method

actual percentage recovery can be calculated from the tangent of the line connecting the data points. This is identical to taking the intersect of the ordinate and dividing it by the extrapolated value on the left side of the abscissa.

5.2 DILUTION METHOD

In the dilution method the sample is diluted with increasing amounts of distilled deionized water and analyzed according to the standard procedures. The obtained concentration is then adjusted for its dilution effect to give the apparent initial concentration before dilution (Figure 4). The apparent concentration is then plotted vertically and the dilution factor horizontally. The apparent concentration generally increases at increasing dilution till it reaches a plateau value. This indicates that matrix interferences of the sample, which generally result in lower aparent concentrations, are reduced with increasing dilution. The dilution associated with this plateau value are then used to determine the other leachate samples in the batch. At very high dilution the spectrophotometer is not able to record absorbances accurately, resulting in high apparent concentrations in the initial sample. Comparison of the dilution method and the standard addition method as evaluated for the total phosphate determination, showed that the latter is generally more accurate. The former, however, is easier to administer, and requires less time. Both the standard addition method and the dilution method require the analysis of a "sample blank", i.e. the sample in its final dilution containing all the reagents except the color forming reagent. In the heavy metal analysis the sample blank is analysed by measuring the broad non specific absorption at a wavelength bordering the analyte line.

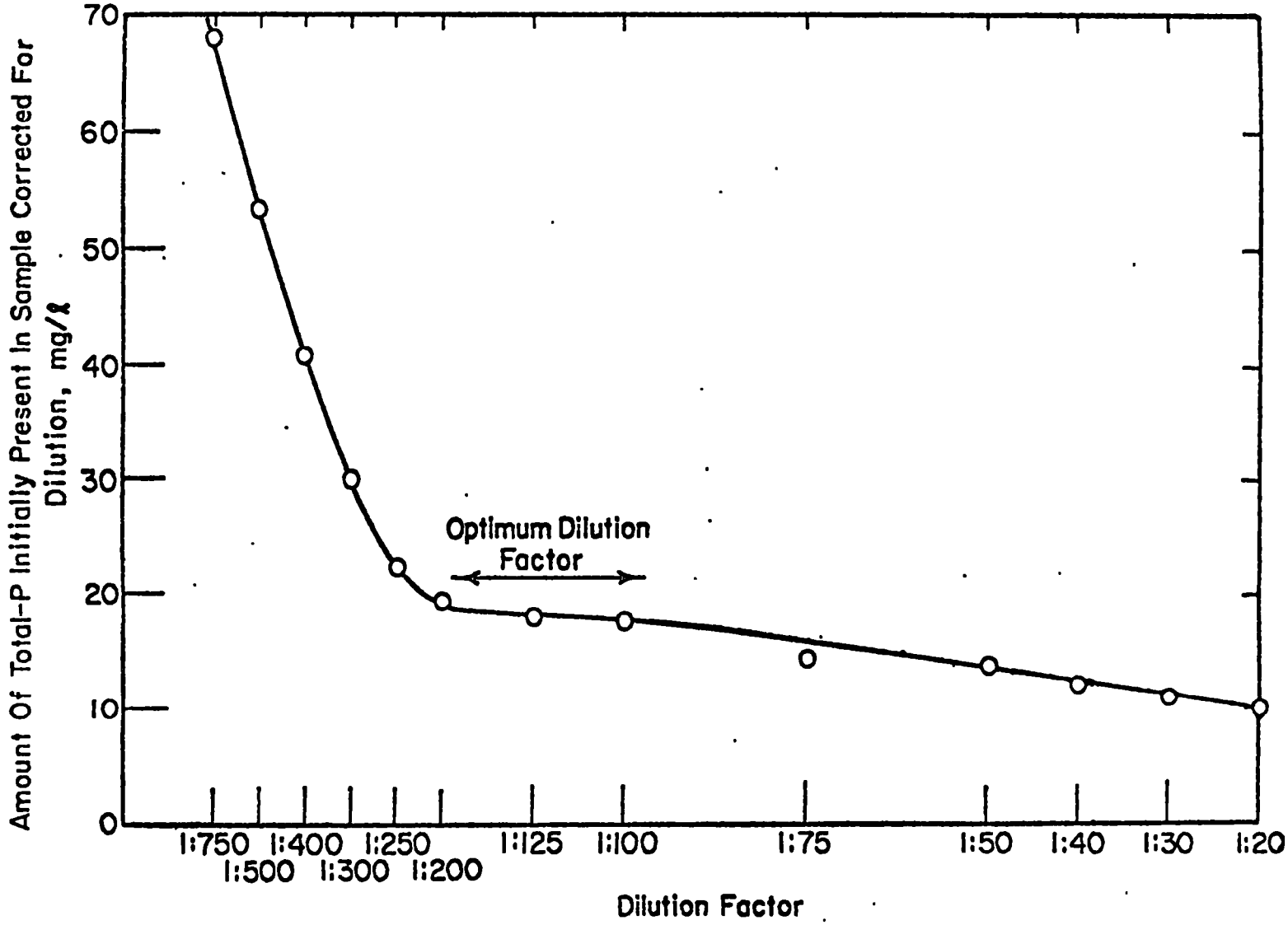


Figure 4. The Total-P Determination with the Ascorbic Acid Method in a Leachate Sample Using the Progressive Dilution Method

of minimum detectability to a highly concentrated product. The ratios of the individual contaminants present in the leachate are also variable and must be considered when evaluating interference effects in a given analytical method.

The analyst, therefore, must always evaluate a specific analytical method relative to a specific leachate sample. Several guidelines for handling interferences are of great value. The judgment of the analyst is of prime importance in applying the guidelines to the specific problems at hand. Experience with a given leachate is obviously of practical value. The degree of accuracy, sensitivity and precision required in a specific analytical problem will constitute foremost considerations in the final selection of the method and possible modifications.

8.4 Analytical Methods

In discussing individual analytical methods in their report, Chian and DeWalle address the following aspects in each case:
Principle, Interferences, Previous Studies, Evaluation of The Method, Recommendations and Procedures.

The methods discussed in the report are as follows:

1. Physical Parameters: pH, Oxidation Reduction Potential (ORP) and Specific Conductance, Residue.
2. Organic Chemical Parameters: C.O.D., T.O.C., Volatile Acids, Tannin and Lignin, Organic Nitrogen.
3. Inorganic Chemical Parameters: Chloride, Sulfate, Phosphate, Alkalinity and Acidity, Nitrate, Nitrite, Ammonia, Sodium and Potassium, Calcium and Magnesium.

4. Biological Parameters: B.O.D., Coliform Bacteria (Total and Fecal).

5. Miscellaneous Determinations

The report also contains a useful appendix of parameters and methods used by various investigators. (Appendix A, P. 125 - Survey of physical, chemical and biological methods used by various investigators).

8.5 Brief Description of Specific Analytical Methods for Leachate Analysis

Following is a brief description of the analytical methods as recommended by Chian and DeWalle for the analysis of concentrated leachate:

1. Physical Parameters:

A. pH Determination:

Electrometric determination using a glass indicating electrode and a calomel reference electrode or a combination electrode. The procedure is according to Standard Methods, 13th Edition, 1971, p.279.

B. Oxidation Reduction Potential (ORP):

The measurement is made with a pH meter, using a platinum indicating electrode and a calomel reference electrode. The pH determination is made concurrently.

C. Specific Conductance Determination:

The determination is performed with a commercially available meter and an electrode with a cell constant of 1.0. Both temperature and pH are determined concurrently, as they affect the results. Reference is made to Standard Methods, 13th Edition, 1971, pp.326-327.

D. Residue Determination:

Total solids is determined after drying to constant weight at 103-105°C. and the volatile solids is determined from the weight

loss at 550°C. for one hour. The suspended solids(filterable residue) is determined using a glass fiber filter and drying to constant weight at 103-105°C. The following reference is given: Standard Methods, 13th Edition, 1971, pp. 289,292,293

2. Organic Chemical Parameters:

A. Chemical Oxygen Demand (C.O.D.):

The C.O.D. determination is performed according to Standard Methods, 13th Edition, 1971, pp. 496-499. If the C.O.D. is less than 100 mg/liter, more accurate results may be obtained by using the low level C.O.D. procedure given on p. 498 of the same reference.

B. Total Organic Carbon (T.O.C.):

The T.O.C. analysis is run according to Standard Methods, 13th Edition, 1971, pp. 257-259.

C. Volatile Acids (Total Organic Acids):

Volatile acids are determined by the column-partition chromotographic method as listed in Standard Methods, 13th Edition, 1971, pp. 577-580. Standard amounts of acid are added to determine the recovery of the method.

D. Tannin and Lignin

The tannin and lignin procedure is according to Standard Methods, 13th Edition, pp. 346-347.

E. Organic Nitrogen:

Organic nitrogen is determined according to Standard Methods, 13th Edition, pp. 244-248. A 300 ml. sample, 50 ml. digestion reagent and 30 min. digestion period are used.

3. Inorganic Chemical Parameters:

A. Chloride:

In biologically stabilized leachate samples in which color does not cause any interference, the chloride determination is conducted with the mercuric nitrate method (Standard Methods,13th Edition, pp. 97-99). In strongly polluted leachate, chloride is determined by the potentiometric titration method (Standard Methods, 13th Edition, pp. 377-380).

B. Sulfate:

Sulfate is determined by the gravimetric method with drying of residue, according to Standard Methods, 13th Edition, 1971, pp. 332-333.

C. Phosphate:

The aminonaphthol sulfonic acid or ascorbic acid method is used to measure total phosphorus concentration in leachate using the persulfate digestion. The amount of recommended persulfate digestion reagent is 400 mg./100 ml. sample, while the digestion time recommended by Standard Methods is sufficient to hydrolyze the phosphorus. The ortho-phosphate test as determined by the ascorbic acid method does not experience significant interference and should be run on the anaerobically stored leachate after as little dilution as possible. In order to obtain reliable results, a standard addition or progressive dilution curve should be established for the total phosphorus determination. Such steps are not necessary for the orthophosphate determination References for the total phosphate and ascorbic acid methods are Standard Methods, 13th Edition, 1971, pp. 524-526 and 532-534.

The aminonaphthol sulfonic acid method reference is Physical, Chemical and Microbiological Methods of Solid Waste Testing; Four Additional Procedures: N. S. Ulmer, U.S.EPA, NERC, Cincinnati, 1974.

D. Alkalinity and Acidity:

The alkalinity and acidity determinations are made potentiometrically on undiluted samples. The endpoints used are those determined from the titration curve. Standard 0.02N NaOH is used for the acidity determination and standard 0.02N H_2SO_4 or HCl is used for the alkalinity determination. Reference is Standard Methods, 13th Edition, 1971, pp. 52 and 55.

E. Nitrate:

Nitrate is determined with the specific ion electrode instead of the brucine-sulfanilic acid colorimetric method. It is preferable to measure the nitrate with the electrode in the undiluted sample. Standard amounts of nitrate should be added to the sample to determine the recovery of the method. When the brucine-sulfanilic acid method is used, the suspended solids and color may be removed with a massive lime dosage of 5,000 to 10,000 mg./l. $Ca(OH)_2$. Aluminum hydroxide is not as effective as a coagulant.

F. Nitrite:

Nitrite is determined by the naphthylamine colorimetric procedure as outlined in Standard Methods, 13th Edition, 1971,pp.240-243. The naphthylamine reagent is replaced by n- (1-naphthyl) ethylenediamine dihydrochloride.. Standard amounts of nitrite nitrogen are added to the filtered sample.

G. Ammonia

Two methods are recommended for determination of ammonia. One uses the selective ion electrode with sufficient sample dilution

to reduce matrix interference of the leachate. (Reference: U.S. EPA Methods for Chemical Analysis of Water and Wastes, 1974, pp. 165-167). The other method uses distillation followed by titration of the ammonia in the distillate with standard 0.02N H_2SO_4, with mixed methyl red-methylene blue as the indicator. For this method, a maximum concentration of 75 mg/l. ammonia in the diluted sample is recommended, unless additional buffer is used. A pH of 7.4 is sufficiently high to distill off the ammonia. A pH 9.4 is too high, causing partial destruction of the organic nitrogen. (Reference: Standard Methods, 13th Edition, 1971, pp. 224-226 and 246-247).

H. Sodium and Potassium:

Sodium and potassium are determined by flame photometry (Reference: Standard Methods, 13th Edition, 1971, pp. 316-320, Sodium and pp. 283-284, Potassium). Sodium and potassium may also be determined by atomic absorption spectroscopy, in which case it is recommended that cesium be added at a concentration of 1,000 mg./l. to suppress ionization of the analyte ion in the flame. (Reference: Methods for Chemical Analysis of Water and Wastes, U.S.EPA, 1974: Sodium, pp. 147-148 and Potassium, pp. 143-144).

I. Calcium and Magnesium:

Calcium and magnesium are determined by atomic absorption spectroscopy, using 10,000 mg./l. lanthanum to reduce interference. (References: Standard Methods, 13th Edition 1971, pp. 212-213 and Methods or Chemical Analysis of Water and Wastes, U.S. EPA, 1974, pp. 103-104, Calcium and pp. 114-115, Magnesium).

J. Hardness:

Hardness is calculated from the concentrations of the individual polyvalent metals as determined by atomic absorption spectroscopy and should include Ca, Mg, Fe, Al, Zn, Cu and other polyvalent cations, expressed as $CaCO_3$ equivalents.

K. Heavy Metals:

The heavy metals are determined with atomic absorption techniques. Standard additions are used for leachates of high strength to determine the magnitude of the interference. Standard additions should be used for the elements lead, copper, nickel and chromium but may be omitted for zinc and cadmium. For total metal analysis, the sample should be collected in a polyethylene bottle and acidified to pH2 with 1:1 redistilled nitric acid. When the dissolved metals, those filterable through a 0.45μ filter, are determined, the suspended metals should be determined concurrently.

The determination of arsenic and selenium by atomic absorption using the gaseous hydride method may not be satisfactory, since reduction to the trivalent form with $SnCl_2$ may not be complete. The conversion to gaseous arsine after addition of zinc metal may also not be complete. Colorimetric methods are therefore recommended. The analysis of mercury by atomic absorption with the cold vapor technique also depends on the reduction of the sample with $SnSO_4$ or $SnCl_2$, which may not be complete when other oxidants in high concentrations are present. (References: Standard Methods, 13th Edition, 1971, p. 213; Methods for Chemical Analysis of Water and Wastes, U.S.EPA, 1974, pp. 213 and 295-299, Selinium; Methods for

Chemical Analysis of Water and Wastes, U.S.EPA, 1974, Calcium pp. 103-104, Magnesium pp. 114-115, Iron pp. 110-111, Aluminum pp. 92-93, Zinc pp. 155-156, Copper pp. 108-109, Arsenic pp.9-10 and 95-96, Selenium p. 145, Mercury pp. 118-122).

4. Biological Parameters:

A. Biochemical Oxygen Demand (B.O.D.):

The B.O.D. determination is run according to Standard Methods, using dilution water which is seeded with settled domestic sewage. B.O.D. values obtained should be judged carefully and be determined parallel with comparable chemical tests such as free volatile fatty acids, C.O.D. or T.O.C. (Reference: Standard Methods, 13th Edition, 1971, pp.489-495).

B. Coliform Bacteria (Total and Fecal):

The most probable number (MPN) technique should be selected for leachate monitoring purposes, as opposed to the membrane filter (MF) technique, since it is able to detect bacteria at lower concentrations and is less subject to suspended solids interference. Inactivation studies, however, in which a certain amount of bacteria is added to a sample to study its subsequent reduction with time, should be conducted if the MF technique is used. Presumptive and confirmed tests are run for total coliforms and the completed coliform test is run in those instances where leachate causes pollution of drinking water supplies. (Reference: Standard Methods, 13th Edition, 1971, pp. 664-668).

The fecal coliform MPN procedure is used as a confirmatory test procedure in conjunction with prior enrichment in a presumptive test medium for optimum recovery of fecal coliforms. (Reference: Standard Methods, 13th Edition, 1971, pp.669-672)

5. Miscellaneous Determinations:

Some of the miscellaneous leachate parameters which have been given attention in various studies are: Methylene blue active substances, cyanide, fluoride, sulfide, silica, hexane solubles, ether solubles, color, visual appearance and odor.

8.5.1 Additional valuable information on specific analytical methods is available in "Procedures for the Analysis of Landfill Leachate", Proceedings of an Internation Seminar, Environmental Conservation Directorate, Ottawa, Ontario, Report EPS-4EC-75-2, October 1975.

8.6 Field Testing Versus Testing in the Laboratory

The majority of tests performed on leachate samples are carried out in the analytical laboratory on samples which have been preserved by refrigeration or chemical means. A limited number of tests, however, can be performed at the sampling site on a freshly drawn sample. There are a number of advantages in field testing, among which are that sample degradation is practically eliminated, along with the need for sample preservation, transportation and handling. An added advantage is the ability to re-sample and re-analyze immediately, on site, if it is suspected that a particular sample is not representative or valid. There are also disadvantages encountered in field testing and these usually relate to the reliability of the particular method and equipment used for the test.

Some tests can be run in the field with the same methods and equipment which would be used in the laboratory and yield the same reliability. Among such tests are those involving the measurement of pH, oxidation reduction potential, specific conductance, turbidity, dissolved oxygen and specific ions by means of specific ion electrodes. The equipment used in these tests is available in portable models which are of equal applicability in the field and laboratory.

Other tests are sometimes performed in the field using methods and equipment

specifically designed for field use. A number of commercially available kits are available for such purposes. These methods are not usually used in the analytical laboratory and are generally recognized as being applicable only to field testing.

While offering distinct advantages, there are also disadvantages inherent in the use of field kits. The following evaluation of field kit usage is given in Handbook for Monitoring Industrial Wastewater, U.S. EPA, Technology Transfer, August 1973, p. 5-14:

"Estimating the Amounts of Pollutants Present by Use of "Kits""

Companies, such as the Hach Chemical Company, Delta Scientific, Inc., and Koslow Scientific Company have manufactured "Kits" for the analysis of various constituents of wastewater. The kits consist of a small portable container in which all the necessary equipment and instructions are conveniently packaged and arranged to perform a variety of tests. No previous laboratory training is required and, within minutes, an indication of the chemical constituents in wastewater can be determined.

Koslow Scientific and the Hach Company provide kits for determining the presence of heavy metals, such as Cd, Hg and Pb, and includes reagents for masking interferences.

The major disadvantage in using kits is the inability of the pre-packaged devices and reagents to effectively cope with interferences. Reference 2 (Standard Methods for the Examination of Water and Wastewater, 13th Edition, American Public Health Association, 1971) outlines procedures for the removal of interferences by pretreatment techniques and the reagents necessary for masking these interferences that are usually not available in the kits. The accuracy of the tests performed with kits is usually less than that obtainable with precise laboratory techniques. Kits give good results in relatively

clean water but pose problems when used to anlayze wastewaters. They are nevertheless useful in preliminary surveys performed to determine overall characteristics of a wastewater."

This evaluation of the use of field kits for the analysis of industrial wastewater is equally applicable in the case of leachate analysis.

Mobile Laboratories

Although not in widespread usage, mobile analytical laboratories have the potential of providing a combination of laboratory capability and field-testing convenience. The instrumentation and general capability of a mobile laboratory can vary over a wide range, depending upon its application, manpower, and the capital investment involved. By using normal laboratory equipment and methods, the mobile laboratory can obtain results equivalent to those of a conventional analytical laboratory, while incorporating all of the advantages of field kits. Limitations imposed by sample degradability and work load will be encountered by the mobile laboratory in much the same way as experienced by the conventional laboratory under certain conditions. If a sample or samples presented to a mobile laboratory must be analyzed for a large number of parameters (i.e. 20 or 30), then sample degradation versus work load will have to be addressed. The sample will have to be preserved and the analyses prioritized relative to order of degradability. In this respect, the mobile laboratory shares the disadvantages of the conventional laboratory, along with its advantages.

8.7 Automated Methods

Automated wet chemistry methods offer many advantages, among which are economy, increased precision and accuracy when applied to repetitive analytical work loads of significant volumes. Federal, state and local regulatory agencies, industry, educational institutions and independent testing laboratories, among

others, use automated methods to handle large, demanding repetitive analytical work loads.

Automated wet chemistry is addressed in the "Handbook for Monitoring Industrial Wastewater", U.S. EPA, August 1973, pps. 5-14, as follows:

> "Automated wet chemistry is frequently used in analysis of wastewaters and for automated monitoring of waste effluents. When used, the system consists of a sampler to select air, reagents, diluents and filtered samples. From the sampler, the fluids pass through a proportioning pump and manifold where the fluids are aspirated, proportioned and mixed. The samples are then ready for separation by passing through any one of the following units: a dialyzer (continuously separates interfering materials in the reaction mixture) a digester (used for digestion, distillation or solvent evaporation), a continuous filter (for on-stream separation of particulate matter by a moving belt of filter paper) or a distillation head (separates high vapor pressure components).
>
> After separation, the samples can be conditioned in a constant temperature heating bath. After conditioning, the samples pass through a detection system which may be a colorimeter, a flame photometer, a fluorometer, a UV spectrophotometer, an IR spectrophotometer, an atomic absorption spectrophotometer, or a dual differential colorimeter. The signals from the detection system are sent to a recorder or a computer system."

In the May 1975 issue of "Environmental Newsletter", a publication of Technicon Industrial Systems, a list of water quality major automated methods is presented. The list is reproduced below. It should be noted that ten (10) of the methods are Federal Register approved and eleven (11) of the methods are presented in the U.S. EPA manual "Methods for Chemical Analysis of Water and Wastes", 1974.

Figure 1 CLASSIFICATION OF MAJOR METHODS FOR AUTOANALYZER I AND II SYSTEMS

Parameter	Technicon or Other Method	Federal Register Approved	Individual Variance Approvals	1974 EPA Methods Book Reference (1)	Practical Method (2)	Usual Method Range (3)
Acidity (Thymol Blue)	164-71W				X	0-500mg/l
Alkalinity (Methyl Orange)	111-71W	X		P. 5		0-500mg/l
Aluminum	(4)				X	
Ammonia (Dialysis)	270-73W				X	0-1mg/l
Boron	202-72W				X	0-1mg/l
Chloride	99-70W	X		P. 31		0-10mg/l
Chromium (Hexavalent)	162-71W				X	0-0.2mg/l
COD	137-71W				X	0-100mg/l
Color	181-72W				X	0-250 Units
Copper	211-72W				X	0-2mg/l
Cyanide	315-74W				X	0-500 µg/l
Fluoride	129-71W			P. 61	X	0-2mg/l
Hardness (Total)	165-71W	X		P. 70		0-250mg/l
Iron	109-71W		X		X	0-1mg/l (ppm)
Mercury	— — —			P. 127		0-20 µg/l
rate & Nitrite (Dialysis)	271-73W				X	0-1mg/l
gen (Ammonia)	98-70W	X		P. 168		0-10mg/l
Nitrogen (Kjeldahl, Total)	146-70W	X		P. 182 (HgO)		0-10mg/l
				P. 190 (SeO_2)		
Nitrogen (Nitrite)	102-70W	X				0-1mg/l (N)
Nitrogen (Nitrate & Nitrite)	100-70W	X				0-2mg/l (N)
Nitrogen (Organic plus Ammonia)	325-74W(5)		X			0-0.40mg/l or 0-10mg/l
NTA	— — —			P. 220		0-10mg/l
Phenol	127-71W		X	P. 243		0-500 µg/l
Phosphorus (Total)	94-70W(6)	X				0-10mg/l
Phosphorus (Total)	297-73W			P. 256	X	0-10mg/l
	188-72W					0-1mg/l
Phosphorus (O-phosphate)	94-70W	X				0-10mg/l
Silicates	105-71W				X	0-10mg/l
Sucrose	274-73W				X	0-1000mg/l
	289-73W				X	0-100mg/l
Sulfate	118-71W	X		P. 279		0-300mg/l
Sulfite	173-72W				X	0-3mg/l

1) Some EPA Methods may differ in detail from the Technicon Listed Method. In such cases the method to be selected for use is determined by review of sample matrices and ranges required

2) Method in practical use but general regulatory approval not yet obtained

3) Method ranges can be adjusted to suit particular needs. Method resolution is typically 1% of full scale.

4) Norwegian Institute for Water Research, Oslo, Norway, A. Henderson

5) Manual digestion followed by autoanalysis for ammoniacal nitrogen.

6) Manual digestion followed by autoanalysis for phosphorus.

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Automated methods are discussed in "Standard Methods for the Examination of Water and Wastewater", 13th Ed., 1971, pps. 14-15, as follows:

"Automated analytical instrumentation: Automated analytical instruments are now available and in use to run individual samples at rates of 10 to 60 samples per hour. The same instruments can be modified to perform analyses for two to twelve constituents simultaneously from one sample. The instruments are composed of a group of interchangeable modules joined together in series by a tubing system. Each module performs the individual operations of filtering, heating, digesting, time delay, color sensing, etc. that the procedure requires.

The read-out system employs sensing elements with indicators, alarms and/or recorders. For monitoring applications, automatic standardization-compensation, electrical and chemical, is done by a self-adjusting recorder when known chemical standards are sent periodically through the same analysis train. Such instrument systems are presently available.

Appropriate methodology is supplied by the manufacturer for many of the common constituents of water and wastewater. Some methods are based on procedures described in this manual, while others originate from the manufacturer's adaptation of published research. Since a number of methods of varying reliability may be available for a single constituent of water and wastewater, a critical appraisal of the method adopted is obviously mandatory.

Automated methodology is susceptible to the same interferences as the original method from which it derives. For this reason, new methods developed for automated analysis must be subjected to the exacting tests for accuracy and freedom from adverse response already met by the accepted standard methods.

Off color and turbidity produced during the course of an analysis will be visible to an analyst manually performing a given determination and the result will be properly discarded. Such abnormal effects caused by unsuspected inferferences might escape notice in an automated analysis. Calibration of the instrument system at least once each day with standards containing interferences of known concentration could help to expose such difficulties. Routine practice is to check instrument action and guard against questionable results by the insertion of standards and blanks at regular intervals - perhaps after every 10 samples in the train. Another important precaution is proper sample identification by arrangement into convenient groups.

In brief, a fair degree of operator skill and knowledge, together with adequately detailed instructions, is required for successful automated analysis."

In the report "Compilation of Methodology For Measuring Pollution Parameters of Landfill Leachate" by E.S.K. Chian and F.B. DeWalle, the following comments are made concerning automated methods:

"All automated methods as recommended by EPA (1974) for water and wastewater and Technicon (1973) for industrial waste should be evaluated for possible interferences since most tests are based on colorimetric analyses which are generally subject to strong interference by the color and suspended solids present in leachate. Such evaluation is necessary since increasing amounts of leachate samples will be analyzed by automated methods at a future date."

Laboratory Quality Control

The subject of laboratory quality control is treated in detail in "Handbook For Analytical Quality Control in Water and Wastewater Laboratories",

U.S. EPA Technology Transfer, June 1972. The various topics covered include: Importance of Quality Control, Laboratory Services, Instrumental Quality Control, Glassware, Reagents, Solvents and Gases, Control of Analytical Performance, Data Handling and Reporting, Special Requirements for Trace Organic Analysis and Skills and Training. A number of valuable references are provided in each section. Chapter 1, Importance of Quality Control is reprinted below:

(pp. 8-47, 8-48, 8-49)

The technical and legal aspects of an adequate quality control program are of prime importance in the analysis of sanitary landfill leachate samples. The investment of time and effort needed for a quality control program are well compensated in the resultant reliability of and confidence in the data obtained.

Chapter 1

IMPORTANCE OF QUALITY CONTROL

1.1 General

The role of the analytical laboratory is to provide qualitative and quantitative data to be used in decision making. To be valuable, the data must accurately describe the characteristics or the concentration of constituents in the sample submitted to the laboratory. In many cases, an approximate answer or incorrect result is worse than no answer at all, because it will lead to faulty interpretations.

Decisions made using water and wastewater data are far-reaching. Water quality standards are set to establish satisfactory conditions for a given water use. The laboratory data define whether that condition is being met, and whether the water can be used for its intended purpose. If the laboratory results indicate a violation of the standard, action is required on the part of pollution control authorities. With the present emphasis on legal action and social pressures to abate pollution, the analyst should be aware of his responsibility to provide laboratory results that are a reliable description of the sample. Furthermore, the analyst must be aware that his professional competence, the procedures he has used, and the reported values may be used and challenged in court. To satisfactorily meet this challenge, the laboratory data must be backed up by an adequate program to document the proper control and application of all of the factors which affect the final result.

In wastewater analyses, the laboratory data define the treatment plant influent, the status of the steps in the treatment process, and the final load imposed upon the water resources. Decisions on process changes, plant modification, or even the construction of a new facility may be based upon the results of laboratory analyses. The financial implications alone are significant reasons for extreme care in analysis.

Research investigations in water pollution control rest upon a firm base of laboratory data. The final result sought can usually be described in numerical terms. The progress of the research and the alternative pathways available are generally evaluated on the basis of laboratory data. The value of the research effort will depend upon the validity of the laboratory results.

1.2 Quality Control Program

Because of the importance of laboratory analyses and the resulting actions which they produce, a program to insure the reliability of the data is essential. It is recognized that all analysts practice quality control to varying degrees, depending somewhat upon their training, professional pride, and awareness of the importance of the work they are doing. However, under the pressure of daily workload, analytical quality control may be easily neglected. Therefore, an established, routine control program applied to every analytical test is important in assuring the reliability of the final results.

The quality control program in the laboratory has two primary functions. First, the program should monitor the reliability (truth) of the results reported. It should continually provide an answer to "How good (true) are the results submitted?" This phase may be termed "measurement of quality." The second function is the control of quality in order to meet

the program requirements for reliability. For example, the processing of spiked samples is the measurement of quality, while the use of analytical grade reagents is a control measure. Just as each analytical method has a rigid protocol, so the quality control associated with that test must also involve definite required steps to monitor and assure that the result is correct. The steps in quality control will vary with the type of analysis. For example, in a titration, standardization of the titrant on a frequent basis is an element of quality control. In an instrumental method, the check-out of instrument response and the calibration of the instrument in concentration units is also a quality control function. Ideally, all of the variables which can affect the final answer should be considered, evaluated, and controlled.

This handbook considers the factors which go into creating an analytical result, and provides recommendations for the control of these factors in order to insure that the best possible answer is obtained. A program based upon these recommendations will give the analyst and his supervisor confidence in the reliability and the representative nature of the sample charactieristics being reported.

Without exception, the final responsibility for the reliability of the analytical results submitted rests with the Laboratory Director.

1.3 Analytical Methods

In general, the widespread use of an analytical method indicates that it is a reliable means of analysis, and this fact tends to support the validity of the test result reported. Conversely, the use of a little-known technique forces the data user to place faith in the judgement of the analyst. When the analyst uses a "private" method, or one not commonly accepted in the field, he must stand alone in defining both his choice of the method and the result obtained.

The need for standardization of methods within a single laboratory is readily apparent. Uniform methods between cooperating laboratories are also important in order to remove the methodology as a variable in comparison or joint use of data between laboratories. Uniformity of methods is particularly important when laboratories are providing data to a common data bank, such as STORET*, or when several laboratories are cooperating in joint field surveys. A lack of standardization of methods raises doubts as to the validity of the results reported. If the same constituent is measured by different analytical procedures within a single laboratory, or in several laboratories, the question is raised as to which procedure is superior, and why the superior method is not used throughout.

The physical and chemical methods used should be selected by the following criteria:

a. The method should measure the desired constituent with precision and accuracy sufficient to meet the data needs in the presence of the interferences normally encountered in polluted waters.

b. The procedure should utilize the equipment and skills normally available in the average water pollution control laboratory.

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*STORET is the acronym used to identify the computer-oriented U.S. Environmental Protection Agency Water Quality Control Information System for STOrage and RETrieval of data and information.

c. The selected methods should be in use in many laboratories or have been sufficiently tested to establish their validity.

d. The method should be sufficiently rapid to permit routine use for the examination of large numbers of samples.

The use of EPA methods in all EPA laboratories provides a common base for combined data between Agency programs. Uniformity throughout EPA lends considerable support to the validity of the results reported by the Agency.

Regardless of the analytical method used in the laboratory, the specific methodology should be carefully documented. In some water pollution reports it is customary to state that Standard Methods (1) have been used throughout. Close examination indicates, however, that this is not strictly true. In many laboratories, the standard method has been modified because of recent research or personal preferences of the laboratory staff. In other cases the standard method has been replaced with a better one. Statements concerning the methods used in arriving at laboratory data should be clearly and honestly stated. The methods used should be adequately referenced and the procedures applied exactly as directed.

Knowing the specific method which has been used, the reviewer can apply the associated precision and accuracy of the method when interpreting the laboratory results. If the analytical methodology is in doubt, the data user may honestly inquire as to the reliability of the result he is to interpret.

The advantages of strict adherence to accepted methods should not stifle investigations leading to improvements in analytical procedures. In spite of the value of accepted and documented methods, occasions do arise when a procedure must be modified to eliminate unusual interference, or to yield increased sensitivity. When modification is necessary, the revision should be carefully worked out to accomplish the desired result. It is advisable to assemble data using both the regular and the modified procedure to show the superiority of the latter. This useful information can be brought to the attention of the individuals and groups responsible for methods standardization. For maximum benefit, the modified procedure should be rewritten in the standard format so that the substituted procedure may be used throughout the laboratory for routine examination of samples. Responsibility for the use of a non-standard procedure rests with the analyst and his supervisor, since such use represents a departure from accepted practice.

In field operations, the problem of transport of samples to the laboratory, or the need to examine a large number of samples to arrive at gross values will sometimes require the use of rapid field methods yielding approximate answers. Such methods should be used with caution, and with a clear understanding that the results obtained do not compare in reliability with those obtained using standard laboratory methods. The fact that "quick and dirty" methods have been used should be noted, and the results should not be reported along with more reliable laboratory-derived analytical information. The data user is entitled to know that approximate values have been obtained for screening purposes only, and that the results do not represent the customary precision and accuracy obtained in the laboratory.

1.4 References

1. Standard Methods for the Examination of Water and Wastewater, 13th Edition, American Public Health Association, New York (1971).

The economics of quality control is greatly favored in the use of automated analysis systems as compared to manual systems. In a recent issue of the U.S. EPA Analytical Quality Control Newsletter (October 1975, p. 5), it is stated that for a particular automated system, the additional personnel work load required to provide an analytical quality assurance overhead of 40%, is estimated to be about 1%. The 40 to 1 advantage is most impressive. The newsletter article is reprinted below:

(p. 8-51)

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ENVIRONMENTAL MONITORING AND SUPPORT LABORATORY-CINCINNATI, OHIO

AUTOMATED QUALITY ASSURANCE

In a number of progress reports on the Laboratory Automation System in this Newsletter, it was pointed out that one of the goals of the project was a significant improvement in analytical quality assurance techniques. Results obtained since installation last May have led us to the conclusion that a quality assurance overhead as high as 40% may be accommodated in many analyses with a nominal increase in personnel workload.

Quality assurance overhead is defined as the percentage of analytical measurements made that do not generate environmental data, but either provide control information to the operator, or assure better overall quality of data. Check standards, spikes, replicates, blanks, reagent blanks, and calibration standards all contribute to this overhead:

$$\text{AQC OVERHEAD} = \frac{\text{Sum of Blanks + Replicates + Spikes + Standards}}{\text{Sum of All Measurements Made}} \times 100$$

Check standards, replicates, and spikes should be measured at regular intervals during the analysis of a series of environmental samples. The results of these measurements should be compared with historical data for that operator and that method, and the information used to determine whether the analytical procedure is in or out of control.

Calibration standards, baseline blanks, and reagent blanks should be measured to assure better overall quality in the data. Clearly the more calibration standards, the better the definition of the working calibration curve over a wider dynamic range. No assumptions of linearity need be made. Similarly, frequent checks of baseline blanks are checks on baseline drift and no assumptions of quiet baseline need be made. Frequent checks of reagent blanks assure that no reagent contamination is a source of error. All AQA overhead measurements contribute to cost in a manual system since each involves the attention of personnel and time for calculations. Often quality assurance overhead is simply deleted to improve environmental sample throughput and reduce costs. With the on-line, real time laboratory automation system, all of the above quality assurance overhead has been fully integrated into the programs for operation of several instruments. In the specific case of the Technicon Auto Analyzer, the additional personnel workload required to provide an AQA overhead of 40% is estimated to be about 1%. This consists largely of the time required to prepare spikes, blanks, and additional standards, and include them in the analytical sequence. At the end of a series of measurements the instrument operator devotes very little time to data evaluation. Output reports contain clear presentations of all quality assurance information and the frequent time wasted because of uncertainties about the quality of the data is eliminated. (Bill Budde, 513-684-2918)

Manpower and Skill Requirements

Manpower and skill requirements for analytical work are dependent upon a number of factors, including nature of the sample, work load, analytical parameter to be tested, method used, sensitivity, precision and accuracy desired and equipment and facilities available. A consideration of major importance, of course, is whether the analyses will be performed by manual or automated methods.

In "Handbook For Analytical Quality Control In Water and Wastewater Laboratories", U.S. EPA, June 1972, pp. 9-2, 9-3, 9-4, skills and skill-time ratings for standard manual analytical operations are discussed in detail. A reprint of this section is given below: pp. 8-53, 8-54, 8-55.

Manpower and skill requirements are reduced dramatically when automated methods are used. The usual skill requirements are those of a technician for preparation of samples, solutions, calibration and glassware handling. Automated data processing affords additional manpower savings.

9.2 Skills

The cost of data production in the analytical laboratory is based largely upon two factors—the pay scale of the analyst, and the number of data units produced per unit of time. However, estimates of the number of measurements that can be made per unit of time are difficult, because of the variety of factors involved. If the analyst is pushed to produce data at a rate beyond his capabilities, unreliable results may be produced. On the other hand, the analyst should be under some compulsion to produce a minimum number of measurements per unit of time, lest the cost of data production become prohibitive. In the following table, estimates are given for the number of determinations that an analyst should be expected to perform on a routine basis. The degree of skill required for reliable performance is also indicated. The arbitrary rating numbers for the degree of skill required are footnoted in the tables, but are explained more fully below:

a. Rating 1—indicates an operation that can be performed by a semi-skilled sub-professional with limited background; comparable to GS-3 through GS-5.

b. Rating 2—operation requires an experienced aide (sub-professional) with background in general laboratory technique and some knowledge of chemistry, or a professional with modest training and experience; comparable to GS-4 through GS-7.

c. Rating 3—indicates a complex procedure requiring a good background in analytical techniques; comparable to GS-7 through GS-11.

d. Rating 4—a highly involved procedure requiring experience on complex instruments; determination requires specialization by analyst who interprets results; comparable to GS-9 through GS-13.

The time limits presented in the table are based on use of EPA methods.

A tacit assumption has been made that multiple analytical units are available for measurements requiring special equipment, as for cyanides, phenols, ammonia, nitrogen and COD. For some of the simple instrumental or simple volumetric measurements, it is assumed that other operations such as filtration, dilution or duplicate readings are required; in such cases the number of measurements performed per day may appear to be fewer than one would normally anticipate.

Table 9-1

SKILL-TIME RATING OF STANDARD ANALYTICAL OPERATIONS

Measurement	Skill Required (Rating No.)	No./Day
	(Simple Instrumental)	
pH	1	100-125
Conductivity	1	100-125
Turbidity (HACH 2100)	1	75-100
Color	1	60-75
DO (Probe)	1, 2	100-125
Fluoride (Probe)	1, 2	100-125
	(Simple Volumetric)	
Alkalinity (Potentiometric)	1	50-75
Acidity (Potentiometric)	1	50-75
Chloride	1	100-125
Hardness	1	100-125
DO (Winkler)	1, 2	75-100
	(Simple Gravimetric)	
Solids, Suspended	1, 2	20-25
Solids, Dissolved	1, 2	20-25
Solids, Total	1, 2	25-30
Solids, Volatile	1, 2	25-30
	(Simple Colorimetric)	
Nitrite N (Manual)	2	75-100
Nitrate N (Manual)	2	40-50
Sulfate (Turbidimetric)	2	100-125
Silica	2	100-125
Arsenic	2, 3	20-30

SKILL REQUIRED

1 - aide with minimum training, comparable to GS-3 through GS-5

2 - aide with special training or professional with minimum training, comparable to GS-5 through GS-7.

3 - experienced analyst, professional, comparable to GS-9 through GS-12.

Table 9-1 (continued)

SKILL-TIME RATING OF STANDARD ANALYTICAL OPERATIONS

Measurement	Skill Required (Rating No.)	No./Day
	(Complex, Volumetric or Colorimetric)	
BOD	2, 3	30-40*
COD	2, 3	25-30
TKN	2, 3	25-30
Phosphorus, Total	2, 3	50-60
Phenol (Dist'n only)	2, 3	20-30
Oil & Grease (Soxhlet)	2, 3	15-20
Fluoride (Dist'n)	2, 3	25-30
Cyanide	2, 3	10-15
	(Special Instrumental)	
Metals by AA (No preliminary treatment)	2, 3	150
Metals by AA (With preliminary treatment)	2, 3	60-80
Pesticides by GC (Without cleanup)	3, 4	3-5
Pesticides by GC (With cleanup)	3, 4	2-4

SKILL REQUIRED

2 - aide with special training or professional with minimum training, comparable to GS-5 through GS-7.

3 - experienced analyst, professional, comparable to GS-9 through GS-12.

4 - experienced analyst, professional, comparable to GS-11 through GS-13.

* - depends on type of sample.

Table 3. With automation, 3 people can handle three times the workload formerly handled by 12 people at Pa.'s Bureau of Water Quality Management Lab

	Manual Method[a,b]			Automated Method		
Parameter	Number per hour	Bench space required	Personnel	Number per hour	Bench space required	Personnel[c]
Distilled fluoride	3–4	10 ft	1	40	↑	↑
Distilled cyanide	1–2	15 ft	1	25	15 ft	1
Distilled phenol	3–4	15 ft	1	20	↓	↓
Total phosphorus	6–8	10 ft	1	45		
Sulfate	13–16	5 ft	1	30		
					↑	↑
Iron	10–12	20 ft[d]	1	30	15 ft	1
					↓	↓
Alkalinity	10–12	5 ft	1	30		
Cl	13–16	5 ft	1	30	↑	↑
NH_3-N	13–16 est.	8 ft	1	30	15 ft	1
NO_2-N	10–13	8 ft	1	30	↓	↓
NO_3-N	5–7	8 ft	1	30		
ardness	13–16	5 ft	1	30		
TOTAL	100–126	114 ft	12	370	45 ft	3

[a] Source—Handbook for Analytical Quality Control in Water and Wastewater Laboratories, E.P.A. [b] Includes sample prep and cleanup. [c] One technician needed to prepare samples and glassware. [d] AutoAnalyzer Unit can be used for other analyses. Source: American Water Works Association, Floyd D. Kefford.

A comparison of manual versus automated analytical methods for throughput, space and personnel requirements is given in "Automated Methods For Assessing Water Quality Come Of Age", by M.J.F. Du Cros and J. Salpeter, Environmental Science and Technology, Vol. 9, Number 10, Oct. 1975, p. 932. See Table No. 3 below (p.8-57).

Data are presented in tabular form for the analysis of 12 water quality parameters. For the manual methods, 12 personnel and 114-feet of bench space are required to perform 100 to 126 determinations per hour. For the automated methods, 3 personnel and 45-feet of bench space are required to perform 370 determinations per hour. In addition to an appreciable savings of space, the automated throughput is approximately 12 times that achieved with manual methods.

Records, Data Handling and Reporting

A significant amount of analytical data are generated in a leachate testing program. The data must be handled, interpreted, checked of validity, recorded and reported. This is an important aspect of the testing program and should be given appropriate attention. If the data are not properly handled, the considerable effort and expense involved in sampling and analysis can be lost or applied wrongly. It should be noted that legal, as well as technical considerations can be associated with records, data handling and reporting.

Reprinted below is Chapter 7, pp. 7-1 to 7-11, entitled "Data Handling and Reporting", from Handbook for Analytical Quality Control In Water and Wastewater Laboratories, U.S.E.P.A., June 1972. Among the topics treated in this chapter are: Significant Figures, Accuracy Data, Precision Data, Report Forms, Digital Read-Out, Key Punch Cards and Paper Tape, Storet Computerized Storage and Retrieval of Water Quality and Data and SHAVES - a Consolidated Data Reporting and Evaluation System.

CHAPTER 7

DATA HANDLING AND REPORTING

7.1 Introduction

To obtain meaningful data on water quality, the laboratory must first collect a representative sample and deliver it unchanged for analysis. The analyst must then complete the proper analysis in the prescribed fashion. Having accomplished these steps, one other important step must be completed before the data are of use. This step includes the permanent recording of the analytical data in meaningful, exact terms, and reporting it in proper form to some storage facility for future interpretation and use.

The brief sections that follow discuss the data value itself, recording and reporting the value in the proper way, means of quality control of data, and storage and retrieval.

7.2 The Analytical Value

7.2.1 Significant Figures

The term significant figure is used rather loosely to describe some judgment of the number of reportable digits in a result. Often the judgment is not soundly based and meaningful digits are lost or meaningless digits are accepted.

Proper use of significant figures gives an indication of the reliability of the analytical method used. The following definitions and rules are suggested for retention of significant figures:

A number is an expression of quantity. A figure or digit is any of the characters 0, 1, 2, 3, 4, 5, 6, 7, 8, 9, which, alone or in combination, serves to express a number. A significant figure is a digit that denotes the amount of the quantity in the place in which it stands.

Reported values should contain only significant figures. A value is made up of significant figures when it contains all digits known to be true and one last digit in doubt. For example, if a value is reported as 18.8 mg/l, the "18" must be firm values while the "0.8" is somewhat uncertain and may be "7" or "9".

The number zero may or may not be a significant figure:

a. Final zeros after a decimal point are always significant figures. For example, 9.8 grams to the nearest mg is reported as 9.800 grams.

b. Zeros before a decimal point with other preceding digits are significant. With no other preceding digit, a zero before the decimal point is not significant.

c. If there are no digits preceding a decimal point, the zeros after the decimal point but preceding other digits are not significant. These zeros only indicate the position of the decimal point.

d. Final zeros in a whole number may or may not be significant. In a conductivity measurement of 1000 μmhos/cm, there is no implication that the conductivity is 1000 ± 1 μmho. Rather, the zeros only indicate the magnitude of the number.

A good measure of the significance of one or more zeros before or after another digit is to determine whether the zeros can be dropped by expressing the number in exponential form. If they can, the zeros are not significant. For example, no zeros can be dropped when expressing a weight of 100.08 grams in exponential form; therefore the zeros are significant. However, a weight of 0.0008 grams can be expressed in exponential form as 8×10^{-4} grams, and the zeros are not significant. Significant figures reflect the limits of the particular method of analysis. It must be decided beforehand whether this number of significant digits is sufficient for interpretation purposes. If not, there is little that can be done within the limits of normal laboratory operations to improve these values. If more significant figures are needed, a further improvement in method or selection of another method will be required to produce an increase in significant figures.

Once the number of significant figures is established for a type of analysis, data resulting from such analyses are reduced according to set rules for rounding off.

7.2.2 Rounding Off Numbers

Rounding off of numbers is a necessary operation in all analytical areas. It is automatically applied by the limits of measurement of every instrument and all glassware. However, it is often applied in chemical calculations incorrectly by blind rule or prematurely, and in these instances, can seriously affect the final results. Rounding off should normally be applied only as follows:

7.2.2.1 Rounding-Off Rules

a. If the figure following those to be retained is less than 5, the figure is dropped, and the retained figures are kept unchanged. As an example: 11.443 is rounded off to 11.44.

b. If the figure following those to be retained is greater than 5, the figure is dropped, and the last retained figure is raised by 1. As an example: 11.446 is rounded off to 11.45.

c. When the figure following those to be retained is 5, and there are no figures other than zeros beyond the 5, the figure is dropped, and the last place figure retained is increased by 1 if it is an odd number, or it is kept unchanged if an even number. As an example: 11.435 is rounded off to 11.44, while 11.425 is rounded off to 11.42.

7.2.2.2 Rounding Off Single Arithmetic Operations

a. Addition: When adding a series of numbers, the sum should be rounded off to the same numbers of decimal places as the addend with the smallest number of places. However, the operation is completed with all decimal places intact and rounding off is done afterward. As an example:

11.1
11.12
11.13

33.35 The sum is rounded off to 33.4.

b. Subtraction: When subtracting one number from another, rounding off should be completed before the subtraction operation, to avoid invalidation of the whole operation.

c. Multiplication: When two numbers of unequal digits are to be multiplied, all digits are carried through the operation, then the product is rounded off to the number of significant digits of the less accurate number.

d. Division: When two numbers of unequal digits are to be divided, the division is carried out on the two numbers using all digits. Then the quotient is rounded off to the number of digits of the less accurate of the divisor or dividend.

e. Powers and Roots: When a number contains n significant digits, its root can be relied on for n digits, but its power can rarely be relied on for n digits.

7.2.2.3 Rounding Off the Results of a Series of Arithmetic Operations

The rules for rounding off are reasonable for simple calculations, however, when dealing with two nearly equal numbers, there is a danger of loss of all significance when applied to a series of computations which rely on a relatively small difference in two values. Examples are calculation of variance and standard deviation. The recommended procedure is to carry several extra figures through the calculation and then to round off the final answer to the proper number of significant figures.

7.2.3 Glossary of Terms

To clarify the meanings of reports and evaluations of data, the following terms are defined. They are derived in part from American Chemical Society and American Society for Quality Control usage (1, 2).

7.2.3.1 Accuracy Data

Measurements which relate to the difference between the average test results and the true result when the latter is known or assumed. The following measures apply:

> Bias is defined as error in a method which systematically distorts results. The term is used interchangeably with accuracy in that bias is a measure of inaccuracy.
>
> Relative error is the mean error of a series of test results as a percentage of the true result.

7.2.3.2 Average

In ordinary usage, the arithmetic mean. The arithmetic mean of a set on $\underline{n}$ values is the sum of the values divided by $\underline{n}$.

7.2.3.3 Characteristic

A property that can serve to differentiate between items. The differentiation may be either quantitative (by variables), or qualitative (by attributes).

7.2.3.4 Error

The difference between an observed value and its true value.

7.2.3.5 Mean

The sum of a series of test results divided by the number in the series. Arithmetic mean is understood ($\bar{X}$).

7.2.3.6 Population

Same as Universe. (See subparagraph 7.2.3.13).

7.2.3.7 Precision

Degree of mutual agreement among individual measurements. Relative to a method of test, precision is the degree of mutual agreement among individual measurements made under prescribed, like conditions.

7.2.3.8 Precision Data

Measurements which relate to the variation among the test results themselves, i.e., the scatter or dispersion of a series of test results, without assumption of any prior information. The following measures apply:

a. Standard Deviation (σ). The square root of the variance.

$$\sigma = \sqrt{\frac{\sum_{i=1}^{n} X_i^2}{n} - \bar{X}^2}$$

b. Standard Deviation, estimate of universe (s).

$$s = \sqrt{\frac{\sum_{i=1}^{n} X_i^2 - \frac{\left(\sum_{i=1}^{n} X_i\right)^2}{n}}{n-1}}$$

c. Coefficient of Variance (V). The ratio of the standard deviation (s) of a set of numbers, n, to their average, $\bar{X}$, expressed as a percentage:

$$V = \frac{S}{X}$$

d. Range. The difference between the largest and smallest values in a set.

e. 95% Confidence Limits. The interval within which one estimates a given population parameter to lie, 95% of the time.

7.2.3.9 Sample

A group of units, or portion of material, taken from a larger collection of units, or quantity of material, which serves to provide information that can be used as a basis for judging the quality of the larger quantity as a basis for action on the larger quantity or on the production process. Also used in the sense of a "sample of observations."

7.2.3.10 Series

A number of test results which possess common properties that identify them uniquely.

7.2.3.11 Skewness (k)

A measure of the lopsidedness or asymmetry of a frequency distribution defined by the expression:

$$\frac{(X_i - \overline{X})^3}{n\sigma^3}$$

This measure is a pure signed number. If the data are perfectly symmetrical, the skewness is zero. If k is negative, the long tail of the distribution is to the left. If k is positive, the long tail extends to the right.

7.2.3.12 Unit

An object on which a measurement or observation may be made.

7.2.3.13 Universe

The totality of the set of items, units, measurements, etc., real or conceptual, that is under consideration.

7.2.3.14 Variable

A term used to designate a method of testing, whereby units are measured to determine, and to record for each unit, the numerical magnitude of the characteristic under consideration. This involves reading a scale of some kind.

7.3 Report Forms

The analytical information reported should include the parameter, the details of the analysis such as burette readings, absorbance, wavelength, normalities of reagents, correction factors, blanks, and finally, the reported value.

To reduce errors in manipulation of numbers, a good general rule is to keep data transposition to an absolute minimum. If this were pursued, the ideal report form would include all preliminary information of the analysis, yet it would be possible to use the same form through to the final reporting of data into a computer or other storage device. However, the ideal report form is not usually in use. Rather, a variety of methods are used to record data. They are:

7.3.1 Loose Sheets

Reporting of data onto loose or ring-binder forms is an older, but much used means of recording data. It does allow easy addition of new sheets, removal of older data, or collection of specific data segments. However, the easy facility for addition or removal also permits easy loss or misplacement of sheets, mix-ups as to date sequence, and questionable status in formal display, or for presentation as evidence.

7.3.2 Bound Books

An improvement in data recording is use of bound books which force the sequence of data insertion. Modification beyond a simple lined book improves its effectiveness with little additional effort. Numbering of pages encourages use in sequence and aids also in referencing data, through a table of contents, according to time, type of analysis, kind of sample, analyst, etc.

Validation can be easily accomplished by requiring the analyst to date and sign each analysis on the day completed. This validation can be strengthened further by providing space for the laboratory supervisor to sign off as to the date and acceptability of the analysis.

A further development of the bound notebook is the commercially available version designed for research-type work. These note books are preprinted with book and page numbers and spaces for title of project, project number, analyst signature, witness signature and dates. Each report sheet has its detachable duplicate sheet which allows for up-to-date review by management without disruption of the book in the laboratory. The cost is about four times that of ordinary notebooks.

Use of bound notebooks is essentially limited to research and development work where an analysis is part of a relatively long project, and where the recording in the notebook is the prime disposition of the data until a status or final report is written.

7.3.3 Pre-Printed Report Forms

Most field laboratories or other installations doing repetitive analyses for many parameters day in and day out, develop their own system of recording and tabulating laboratory data. This may include bound notebooks; but a vehicle for forwarding data is also required. In many instances, laboratory units tailor a form to fit a specific group of analyses, or to report a single type of analysis for series of samples, with as much information as possible preprinted to simplify use of the form. With loose-sheet multicopy forms (use of carbon or NCR paper) information can be forwarded daily, weekly, or on whatever schedule is necessary, while allowing retention of all data in the laboratory. Still, the most common record is an internal bench sheet, or bound book, for recording of all data in rough form. The bench sheet or book never leaves the laboratory but serves as the source of information for all subsequent report forms (See Figure 7-1).

In most instances the supervisor and analyst wish to look at the data from a sample point in relation to other sample points on the river or lake. This review of data by the supervisor, prior to release, is a very important part of the laboratory's quality control program; however, it is not easily accomplished with bench sheets. For this purpose, a summary sheet can be prepared which compares a related group of analyses from a number of stations. An example is shown in Figure 7-2. Since the form contains all of the information necessary for

Figure 7-1. EXAM OF BENCH SHEET

NL-C-88
(7-68)

Spectrographic Analyses Bench Data

Sample #____ Date_____ Source_____ TDS_____

_____ ml. conc. to_____ ml. Factor _____

Test Count	Sec
1.	
2.	
3.	

	Count			Rerun Count			Av. Count	PPM In Conc. Sample	Less Than	(μg/l)PPB Orig. Sample
1. Zn										
2. Cd										
3. As										
4. B										
5. P										
6. Fe										
7. Mo										
8. Mn										
9. Al										
10. Be										
11. Cu										
12. Ag										
13. Ni										
14. Co										
15. Pb										
16. Cr										
17. V										
18. Ba										
19. Sr										

Figure 7-2. EXAMPLE OF SUMMARY REVIEW SHEET

Table 2. MINERALS ANALYSES OF ZONE B, OHIO RIVER SAMPLES, CONC., mg/l.

STATION	Storet Number	Date, 1969	Alkalinity	Hardness	Chloride	Sulfate	Fluoride	SOLIDS		
								Total	Diss.	Susp.
Ohio at Ironton	200152									
Ohio at Greenup Dam	200001									
Ohio at Portsmouth	200139									
Scioto at Lucasville	381710									
Ohio at Maysville	200153									
Ohio at Meldahl Dam	383070									
Little Miami at Cincinnati	380090									
Ohio at Cincinnati	380037									
Licking at 12th Street	200523									
Ohio at Miami Fort	383072									
Ohio at Markland Dam	200521									
Kentucky at Dam I	200522									
Ohio at Madison	174304									
Great Miami at Eldean										
Great Miami at Sellars Road	383047									
Great Miami at Liberty-Fairfield Road	383015									
Great Miami at American Materials Bridge	383007									
Whitewater at Suspension										
Great Miami at Lawrenceburg (Lost Bridge)	383071									

reporting data it is used also to complete the data forms forwarded to the storage and retrieval system.

The forms used to report data to data storage systems require a clear identification of the sample point, the parameter code, the type of analysis used, and the reporting terminology. Failure to provide the correct information can result in rejection of the data, or insertion in an incorrect parameter. As a group of analyses is completed on one or more samples, the values are reported in floating decimal form, along with the code numbers, for identifying the parameter and the sampling point (station). Figure 7-3 shows an example of a preprinted report form for forwarding data to keypunch.

7.3.4 Digital Read-out

Instrumental analyses, including automated, wet-chemistry instruments, such as Technicon AutoAnalyzer, atomic absorption spectrophotometer. pH meter, selective electrode meter, etc., now can provide direct digital readout of concentration, which can be recorded directly onto report sheets without further calculation. Electronics manufacturers now produce computer-calculators that will construct best-fit curves. integrate curves, and/or perform a pre-set series of calculations required to obtain the final reported value for recording by the analyst.

7.3.5 Key Punch Cards and Paper Tape

Since much of the analytical data generated in laboratories is recorded on bench sheets, transferred to data report forms, key-punched, then manipulated on small terminal computers, or manipulated and stored in a larger data storage system, there is a built-in danger of transfer error. This increases with each transposition of data. It is suggested that the analyst can reduce this error by recording data onto punch cards directly from bench sheets. The cards can be retained, or forwarded immediately to the data storage system as desired. IBM now offers a small hand-operated key-punch for this purpose.

It is anticipated that in future water quality systems, the intermediate report sheets will be eliminated and the data will be punched automatically by the analytical instrument system onto key-punch cards and/or paper tapes for direct use as computer input.

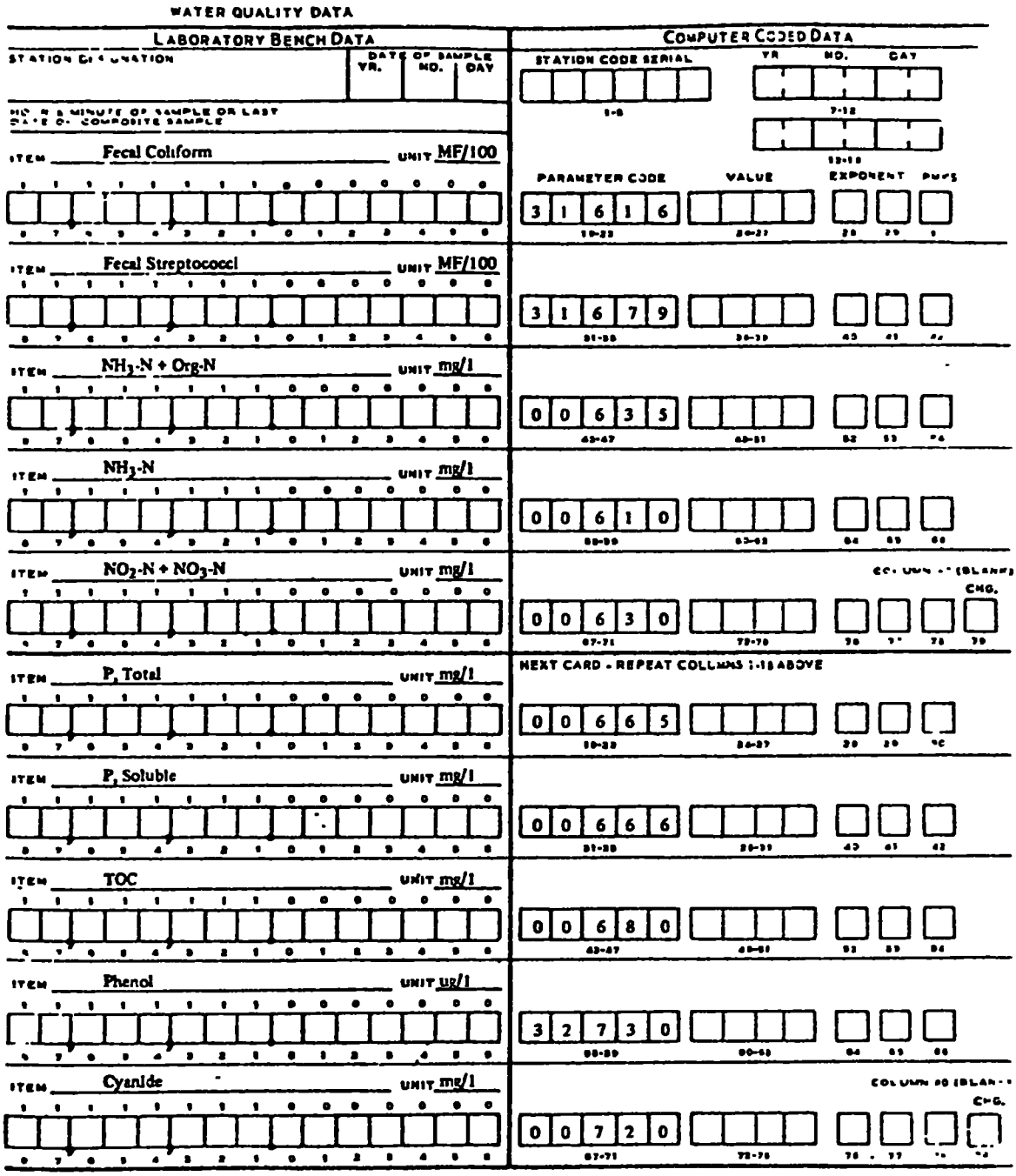
7.4 STORET—Computerized Storage and Retrieval of Water Quality Data

The use of computers with their almost unlimited ability to record, store, retrieve, and manipulate huge amounts of data is a natural outgrowth of demands for meaningful interpretation of the great masses of data generated in almost any technical activity.

In August 1961, an informal conference was held in the Basic Data Branch, Division of Water Supply and Pollution Control, U.S. Public Health Service. A number of ideas were brought together in the basic design of a system for storage and retrieval of data for water pollution control, called STORET. In 1966, the STORET system was transferred. with the Division, into the Federal Water Pollution Control Administration. U.S. Department of the Interior. A refinement of this system is now operated by the Technical Data and Information Branch, Division of Applied Technology. EPA.

If properly stored, the data can be retrieved according to the point of sampling. the date, the specific parameters stored, etc., or all data at a sample point or series of points can be

Figure 7-3. EXAMPLE OF STORET REPORT FORM



extracted as a unit.

There is a State/Federal cooperative activity which provides State water pollution control agencies with direct, rapid access into a central computer system for the storage, retrieval, and analysis of water quality control information.

Full details on use of the STORET system are given in the STORET handbook recently revised (3).

7.4 SHAVES—A Consolidated Data Reporting and Evaluation System

Information systems have been developed to bridge the gap between the analyst and his raw data, and a complex data storage and control system. These systems include preprinted report forms, computerized verification, and evaluation of data and data storage. An example is the SHAVES system.

The term, SHAVES, is an acronym for "Sample Handling and Verification System," which originated at the Great Lakes-Illinois River Basin Comprehensive Project Laboratory at Grosse Isle, Michigan. Although the system's original purpose was verification of the calculations following laboratory analyses, it now includes data storage, checks for completeness and consistency of data, procedures for submitting analytical requests, a set of forms for recording sampling and analytical information, and a clerical procedure to account for analyses completed and pending. The primary purposes of SHAVES are the standardization, automation and control of reporting analyses. All samples received at the Pacific Northwest Water Laboratory for routine analysis are processed through the system.

Although SHAVES uses a computer to perform its operations, it is not primarily a computer program. It is intended for use as an intra-laboratory quality control tool, and as such compliments the STORET system. It is described in detail elsewhere (4).

7.6 References

1. "Guide for Measure of Precision and Accuracy," Anal. Chem., Vol. 33, p. 480, (1961). p. 480.

2. "Glossary of General Terms Used in Quality Control," Quality Progress, Standard Group of the Standards Committee, ASQC, II, (7), pp. 21-2, (1969).

3. Water Quality Control Information System (STORET), EPA, Washington, D.C. 20460, Nov. 15, 1971.

4. Byram, K. V. and Krawczyk, D. F., "An Evaluation of SHAVES: A Water Quality Sample Handling System," Environmental Protection Agency, Pacific Northwest Water Laboratory, 1969.

9. STEP OUTLINE OF MONITORING PROCEDURES

9.1 INTRODUCTION

For every landfill both existing and proposed, the necessity of establishing a ground-water monitoring program should be investigated. The methodology to make this determination and, if necessary, to define the specifics of a monitoring program, can be described in a logical sequence of individual steps. Following the generalized steps presented in this chapter will allow these determinations to be based on the proper factual information. As with any complex situation, however, original thought will be required during each step to insure arriving at the best possible answers.

The steps presented in this chapter are intended to indicate the logical progression of required efforts and therefore not accompanied by detailed descriptions. Such descriptions can be found in other chapters of this manual or in the references cited at the end of the appropriate chapter.

9.2 STEP 1 - INITIAL SITE INSPECTION

All the information would be gathered from an inspection of the landfill, examination of landfill records and other existing information such as topographic maps, and discussion with land-

fill operating personnel. The purpose is to define, with a minimum expenditure of time and money, the probable magnitude of the ground-water contamination problem and thus the urgency of conducting a detailed study and establishing a monitoring program.

9.2.1 NATURE OF THE WASTE

The types of waste accepted or rejected varies widely from landfill to landfill depending largely on the types of waste generated in the area, the regulatory agency for the area, the landfill operator, and economics. A determination of the types of wastes accepted at a particular landfill (both currently and historically) is critical to the monitoring evaluation, i.e., the contaminants likely to be present in the ground water. Wastes can be generally categorized as follows:

- municipal refuse (paper, household garbage, leaves and grass, wood, synthetics, cloth, glass and metal)
- bulky refuse (tree stumps, car bodies, and demolition debris)
- municipal sewage sludge
- industrial solid wastes (defective raw materials and products, packaging and scrap)
- industrial chemical wastes (liquid or solid)

- industrial sludge or residue (fly and bottom ash, waste water treatment sludge and pollution control systems residue)

- chemical waste in sealed drums (of particular concern because of delayed release factor)

- low level radioactive wastes (contaminated laboratory equipment, clothing and building debris)

The categories which are accepted at the landfill should be determined and, in the case of industrial wastes, more detailed information should be sought.

9.2.2 AREAL EXTENT AND THICKNESS OF THE LANDFILL

The size and thickness of a landfill are important factors in establishing the volume of leachate generated as well as the concentration of contaminants in the leachate. The areal extent of a landfill may be measured directly or indirectly from an accurate large scale map or aerial photograph. In addition, the extent of flat and sloping portions of the landfill should be determined. Landfill thickness may, in some cases, be determined from a recent topographic map or by measuring the difference in elevation between the toe and top surface. If the landfill fills a depression, pre-landfilling elevations of the base of the depression may be available,

otherwise one or more borings will have to be drilled through the landfill to directly determine its thickness.

9.2.3 PRETREATMENT AND IN-PLACE TREATMENT OF REFUSE

In most cases, refuse is compacted after it has been placed in the landfill. This is accomplished either by special equipment or by the bulldozers used to spread and cover the refuse. The method of in-place compaction should be determined to allow estimates to be made of density and field capacity of the landfill. In some cases refuse receives treatment prior to landfilling. Shredding and/or pre-disposal compaction and baling of refuse will significantly increase its density and thus its field capacity. Incineration and resource recovery operations will alter its composition and consequently change the nature of the leachate generated after landfilling. In addition, the percentage of the total refuse received which is treated, the types of refuse receiving treatment and the placement of the treated and untreated refuse in the landfill should be determined.

9.2.4 LANDFILLING PROCEDURES

The procedures used in placing and covering refuse at the landfill site will influence the volumes and characteristics of leachate generated. Such practices as separation of different types of refuse at the landfill site, thickness of refuse

layers between cover layers, thickness of cover layers, and type of material used for cover will be of importance. For example, if chemical wastes are accepted at a landfill but segregated from municipal refuse in one area, leachate generated within this chemical disposal area will follow a flow path which may be predictable and thus would influence the selection of monitoring points. Thickness of refuse and cover layers may affect volumes and characteristics of leachate and would therefore also influence the design of a monitoring system.

9.2.5 RATE OF LANDFILLING AND REFUSE AGE

The rate at which the thickness of a landfill increases will affect the volume of leachate generated since a thick section of refuse can absorb more water (field capacity). If the landfill thickness increases at a sufficient rate relative to precipitation, and is covered upon completion to exclude precipitation, very little leachate will be generated. Thus, the rate of filling will influence the design of a monitoring program.

The rates of increase in thickness of various portions of the landfill may be extraplated from the records of weights of refuse accepted over the landfilling period, if such records are available. If not, recollections of landfill operators regarding volumes of refuse accepted over past years may provide

some useful information.

Refuse layers of different ages produce leachates of different chemical characteristics. This factor may be useful in designing a monitoring program; however, other factors, such as refuse composition have a more noticeable influence on leachate than does refuse age.

9.2.6 LINERS AND COVERS

A landfill equipped with an underliner and leachate collection system would be assumed not to be contaminating ground water and a monitoring system would be designed only to test the validity of this assumption. Similarly, if the landfill were completed and covered to prevent the infiltration of precipitation, monitoring, at least initially, would be necessary only to establish if this were indeed the case. If the liner or cover system was shown to be ineffective by initial monitoring data, an expanded monitoring program would be designed to define the extent of the problem and necessary corrective measures.

When analyzing the effectiveness of a bottom liner and collection system, the volume of leachate actually collected should correspond to the predicted volume of leachate being generated. With covers and surface drainage systems, runoff and evapotranspiration accounts for a large percentage of precipitation. Factors such as cover permeability, slopes and vegetation type

would be considered in this determination.

9.2.7 VISUAL SURVEY OF TOPOGRAPHY AND GEOLOGY

The primary purpose of this effort would be to establish estimates of surface runoff and infiltration patterns, and general direction of ground-water flow. The topography of the areas surrounding the landfill will establish the direction of surface-water flow, either towards or away from the landfill surface. Recharge and discharge areas at the site can be determined and, based on these, the general direction of ground-water flow approximated.

9.2.8 GROUND-WATER USE (PRELIMINARY)

A preliminary check into ground-water use in the vicinity of the landfill should be made at this time. Simply ascertaining the existence of supply wells in the vicinity of the landfill is sufficient for this step, with additional data regarding such wells obtained as part of the next step.

9.3 STEP 2 - PRELIMINARY INVESTIGATIONS

Once the need for a detailed ground-water investigation and monitoring program has been established (Step 1), the program should be carefully planned. To accomplish this task efficiently, all existing pertinent data is gathered and examined at the outset. The data would include all information from Step 1 and

any useful data available from outside sources. In addition, certain data not gathered during Step 1, but which can be readily obtained at the site, e.g., analyses of water samples from existing wells, are now gathered.

9.3.1 EXISTING DATA

Information which should be sought other than that gathered in Step 1 includes: historical precipitation records for the site or a nearby area, geologic and topographic maps which include the landfill site, geologic logs of any existing wells or test borings at or near the site, and a recent aerial photograph of the site from which to prepare an accurate base map. In addition, if potential sources of contamination other than the landfill are located in the vicinity, all available information regarding these sources (type and volume of waste, methods of disposal, etc.) should be collected and reviewed.

9.3.2 PRELIMINARY SITE INVESTIGATION

Additional data which will improve the efficiency of a hydrogeologic investigation of a landfill (Step 3) include: analyses of water samples from surface-water bodies and existing wells located on or near the site; analyses of samples of leachate from surface seeps; examination of site vegetation, by a botanist, for signs of stress; observations of surface drainage

patterns during a rainfall; and a check of building basements and other subsurface structures at the site for landfill gas accumulations.

The most critical areas for monitoring in the vicinity of a landfill will be where industrial, domestic, or public-supply wells are threatened by leachate contamination. In the previous step, a preliminary check of the number and location of all such wells was made. In addition to well locations, such information as screened interval, pumping rates and periods of pumping, as well as water use should be compiled. If possible, the size and shape of the cone of influence should be estimated for each well located. Historical water-quality data for the wells, where available, should be examined and, where not available or insufficient, water samples should be taken for analyses.

Many states and regional authorities have regulations regarding minimum distances from landfills within which supply wells must be monitored for contamination. These regulations would have to be followed, but the actual distance monitored should be based on the results of a hydrogeologic investigation. Constituents determined in water-quality analyses should meet applicable regulations and expanded, if necessary, based on anticipated leachate characteristics and ultimate use of water from each supply well.

Besides supply wells, surface-water use in the area must be considered. Nearby surface water may be used for potable supply, fishing or shellfishing, swimming or other recreation, or wildlife habitat. Surface-water bodies are often discharge areas, and as such are subject to contamination from leachate in the ground-water system. Such information as location, use and rate of flow for all surface-water bodies in the vicinity of a landfill should be established. Natural water quality, existing contamination, and sources of contamination should be investigated. Surface-water bodies may form an important part of a monitoring program because, at discharge points, they are the places where ground-water contamination is most conspicuous.

9.4 STEP 3 - DEFINITION OF THE HYDROGEOLOGIC SETTING

Probably the most important factor in establishing the need for and design of a landfill monitoring system is the hydrogeologic setting of the landfill. Such information as surficial and bedrock geology, depth to the water table and direction and rate of ground-water flow should be determined prior to selecting a landfill site. In the past this has not been the case and landfills have been located primarily on land of low economic value, such as swamps or abandoned gravel pits. In such areas the ground-water pollution potential is high and the need for monitoring and abatement procedures is acute.

9.4.1 SURFICIAL GEOLOGY

A survey of surficial geology should establish the areal extent and thickness of the layers of various types of deposits under and adjacent to the landfill, and the permeabilities and interconnections of these layers. The survey can be divided into three sequential parts: 1) a review of geologic data gathered during Steps 1 and 2; 2) geophysical surveys designed to fill in missing subsurface information; and 3) test drilling to provide direct control for the geophysics, obtain more precise data in critical areas, and allow detailed analyses of geologic samples.

9.4.2 BEDROCK GEOLOGY

In some cases bedrock will act as a barrier to leachate movement and in others leachate may move into bedrock aquifers. The type of rock beneath the site and the amount of fracturing will determine the role of bedrock in the movement of leachate. Determination of bedrock geology will essentially follow the steps outlined for surficial geology.

9.4.3 GROUND WATER

The ground-water investigation should be designed to answer such questions as depth to the water table, extent of ground-water mounding caused by the landfill, natural flow direction

and rate, influence of the landfill on flow direction and rate, locations of recharge and discharge areas, types and interconnection of aquifers, and infiltration at the site relative to total ground-water flow. Much of this information will be obtained during and immediately following the previously outlined geologic investigation. During test drilling, such data as water levels and head differences with increasing depth would be recorded. Test borings can be equipped with screens and test pumped at various intervals, using other borings as observation wells, to establish aquifer characteristics and interconnection between aquifers.

Historical precipitation records and estimation of surface runoff and evapotranspiration will provide information regarding recharge at the site. ~~This~~ These data ~~is~~ are necessary to predict the probable size and shape of the leachate plume. Ground-water recharge and discharge areas are determined by observation of surface features and relative water levels in observation wells.

Information gained from the hydrogeologic investigation is directly applicable to the subsequent monitoring program. The potential for the landfill to contaminate an aquifer or surface-water body, and thus the need for a monitoring program, will be established. Information regarding the depth and extent of contaminated and threatened aquifers will help determine

the best locations and depths of monitoring wells. The size and complexity of a monitoring program will be partially based on the calculated volume of recharge through the landfill, and the volume and rate of ground-water flow. Subsequent steps in this chapter will provide data necessary for refinement of the initially outlined monitoring program and elimination of its less important features.

9.4.4 DETERMINE EXISTING WATER QUALITY

An accurate and complete record of existing water quality, both ground and surface, is very useful in a monitoring program. If contamination has already occurred, water samples from uncontaminated areas should be collected and analyzed to establish natural water quality. If sources of contamination other than the landfill are present, the effects of these sources should be determined. Since the object of a monitoring program is to determine change, the importance of historical data is obvious.

If sources of contamination other than the landfill are present, (determined in Step 2) and if existing information is insufficient to define the problem, additional investigation will be necessary. Such an investigation would include direction of ground-water flow in the vicinity of the source, rate of contaminant generation, nature of the contaminants, and result-

ing degree of ground-water degradation. The monitoring system must then be designed to account for these "outside" contaminants, so that they, or their effects, are not inadvertently attributed to the landfill.

In addition to directly introduced contaminants, landfill leachate may cause secondary reactions to occur when it reaches and blends with ground water. For example, a mixing of chemically reduced leachate with ground water may lower the oxidation potential of the leachate-enriched ground water. This, in turn, may reduce and dissolve iron or manganese occurring in the aquifer materials as coatings. Cation exchange reactions which release calcium and magnesium, changes in pH, or precipitation of some leachate constituents are other reactions which could occur and change water quality.

9.4.5 DETERMINATION OF THE RATE OF LEACHATE GENERATION

The leachate generation rate, which will influence the extent of the necessary monitoring program, is determined by a water balance study of the landfill. Data necessary for water balance determination include: precipitation data, landfill surface characteristics, vegetation type and density, landfill site topography, ground-water underflow rate, rate of landfilling and pretreatment and compaction of refuse. A discussion of water balance calculations is given in Chapter 5 of this manual.

9.5 STEP 4 - DETERMINE THE POLLUTING POTENTIAL OF THE LANDFILL

The extent and design of a monitoring system will be largely determined by the pollution potential of the landfill. Estimation of the pollution potential is essentially by consolidation of all data gathered in Steps 1, 2 and 3. Determinations made would include: the location, size and rate of movement of the contaminated plume; the aquifers affected and those which may be affected in the future; the types of contaminants present, and the degree of attenuation of those contaminants by the subsurface sediments. ~~This~~ These data can then be used to predict the total pollution damage that may be caused by the landfill if no action is taken, or to estimate the influence of various possible abatement procedures. Monitoring program data ~~is~~ are then used to establish the accuracy of these predictions or provide a warning of abatement system ineffectiveness or failure.

9.6 STEP 5 - ESTABLISH THE MONITORING PROGRAM

The information gathered in the previous steps would now be written into a detailed report describing the investigations and defining the ground-water contamination problem at the landfill site. Based on this report, the monitoring system would be designed. The methods and purposes for such a monitoring program are outlined below, with detailed discussions of the various topics included in other chapters of this manual.

9.6.1 SELECT THE MONITORING SITES

Data from the previous steps is used to rank all potential monitoring sites in order of importance. High priority sites would include currently developed aquifers, aquifers with good development potential, and discharge areas, such as marshland, which could be damaged by the anticipated leachate discharges. Monitoring sites should be selected to provide sufficiently early warning to allow corrective action to be taken. Ideally, monitoring should be sufficient to indicate the size and type of abatement program necessary to correct a problem once it has been detected. At the very least, the monitoring program should insure that a health hazard does not arise.

9.6.2 DETERMINE MONITORING OBJECTIVES

Following selection of the sites to be monitored the specific objectives of the monitoring program should be determined. Such objectives might include: defining the rate of leachate plume movement, monitoring the concentration of a specific contaminant(s), early warning of an unexpected change in direction or enlarging of the leachate plume, or unexpected inter-aquifer movement of the plume to a previously unpolluted aquifer.

Once the monitoring objectives have been determined, the data requirements to satisfy these objectives must be defined.

Data requirements would include: specific chemical constituents to be included in analyses of water samples, physical measurements to be made on site, and the frequency of sampling or measurement. For example, if an objective of a monitoring program is to insure that leachate does not migrate into a particular aquifer, monthly measurements of the specific conductance of water samples from that aquifer might be made, with routine detailed chemical analyses run only on a semi-annual or annual basis. Such a program might be selected to provide information to protect the aquifer at minimal cost.

9.6.3 ESTABLISH THE MONITORING METHODS AND PROCEDURES NECESSARY TO ACCOMPLISH OBJECTIVES

Certain monitoring devices will be required to accomplish the specific monitoring program objectives. For example, at a specific point, a single well screened over a small section of an aquifer may suffice, or a cluster of several wells, screened over different portions of the aquifer or in separate aquifers may be required. Wells of a particular material may be necessary to avoid interference with leachate sample chemistry, or devices other than wells might be required. A discussion of monitoring and sampling techniques is given in Chapter 5.

A detailed sampling or measuring procedure should be established to insure uniform results. If possible, one person should be

responsible for sampling or overseeing the sampling to insure uniform procedure. This would be especially important for complex procedures but less so for simpler procedures such as conductance measurements. The handling and storage of water samples is also extremely important. For example, if nitrogen analyses are to be made, chilling or acidification of the sample is required, and if metals are to be tested, acidification with nitric acid is necessary. A discussion of preservation of samples is given in Chapter 7. The cost of monitoring will vary widely depending on the sampling procedures and analyses used, thus the program should be designed to be properly operable within the available budget.

Sufficient budget must also be reserved for proper data reduction, record keeping and periodic data review. Records of data should be in three forms: the original data as gathered along with explanatory notes, continuous tabular form, and continuous graph form. Plotting data on approximately uniform grids permits relative values and trends to be easily distinguishable. Periodic review of all data by a qualified scientist followed by a written summary, and distribution and review of the summary by all involved parties ~~is~~ are the proper procedures for handling monitoring data.

9.6.4 ESTABLISH MANAGEMENT PROGRAM

Conditions under which abatement procedures, other corrective

measures, or additional monitoring steps will be taken, should be outlined at the outset of monitoring. Such conditions might include constituent limits, physical parameter limits, or trend shifts. Possible steps which might be taken in the event the established conditions are exceeded should be determined. An understanding should be reached as to where the responsibility lies for all phases of the monitoring and potential abatement programs.

9.7 EXAMPLES OF LANDFILL CONTAMINATION PROBLEMS

Following are two scenarios of fictitious landfill investigations leading to ground-water monitoring programs. The conditions of the two investigations are somewhat different, the first starting with a landfill and defining the pollution problem, and the second starting with a problem and looking for its cause. The first scenario closely follows the preceding step outline; however, the second is an example of a problem requiring a somewhat different approach. The scenarios are intended as illustrative examples and as such are necessarily simplified, i.e., some points included in the step outline have been omitted. Approaches and conclusions other than those presented may be equally valid, as no attempt has been made to include all possibilities. Rather, it is left to the reader to expand upon the two cases using the factual information presented in the other chapters of this manual.

9.7.1 SCENARIO 1 - A LANDFILL CONTAMINATION STUDY

A large county maintained landfill is found to be in violation of the 1899 Harbors and Rivers Act allowing leachate to flow into and contaminate an adjacent river. As a result of a Federal lawsuit, a court order is issued ordering county officials to take the necessary steps to abate this condition. The county officials retain a ground-water consulting firm to investigate leachate conditions at the landfill site, determine if leachate is actually discharging to the river, and if so, what steps to take to abate this problem.

The hydrogeologist assigned this project makes a visit to the landfill for a preliminary inspection tour with the landfill operator. During this tour, he learns that the landfill receives approximately 1,000 tons of refuse per day, about 90% of which is municipal; the remaining 10% is of industrial origin. The refuse receives no pretreatment but after landfilling, it is spread into thin layers by a bulldozer, and compacted by a specially designed landfill compaction machine. The layers of compacted refuse are covered daily by sandy fill material. Small amounts of industrial chemical waste are accepted at the landfill but it is not separated from the other refuse. The rate and method of landfilling, the type of cover material used, and local precipitation rates, indicate that the refuse has all reached field capacity.

The landfill is 40 acres in size and is approximately 60 feet thick with a generally flat top surface. Directly north of the landfill is a hill with an elevation of approximately 80 feet. South of the landfill is a tidal marsh which separates the landfill from the river. The distance between the landfill and the river is approximately 1,000 feet. The landfill is not lined or covered with impermeable materials nor does it utilize any other leachate prevention techniques.

The topography of the site indicates to the hydrogeologist that ground-water flow is from north to south with the hill and landfill acting as recharge areas, and the marsh and river as discharge areas. The nearest supply well is located approximately one-half mile north of the landfill and serves as a supply well for an individual residence. No other wells or borings ~~can~~ could be ~~located~~ found in the landfill vicinity.

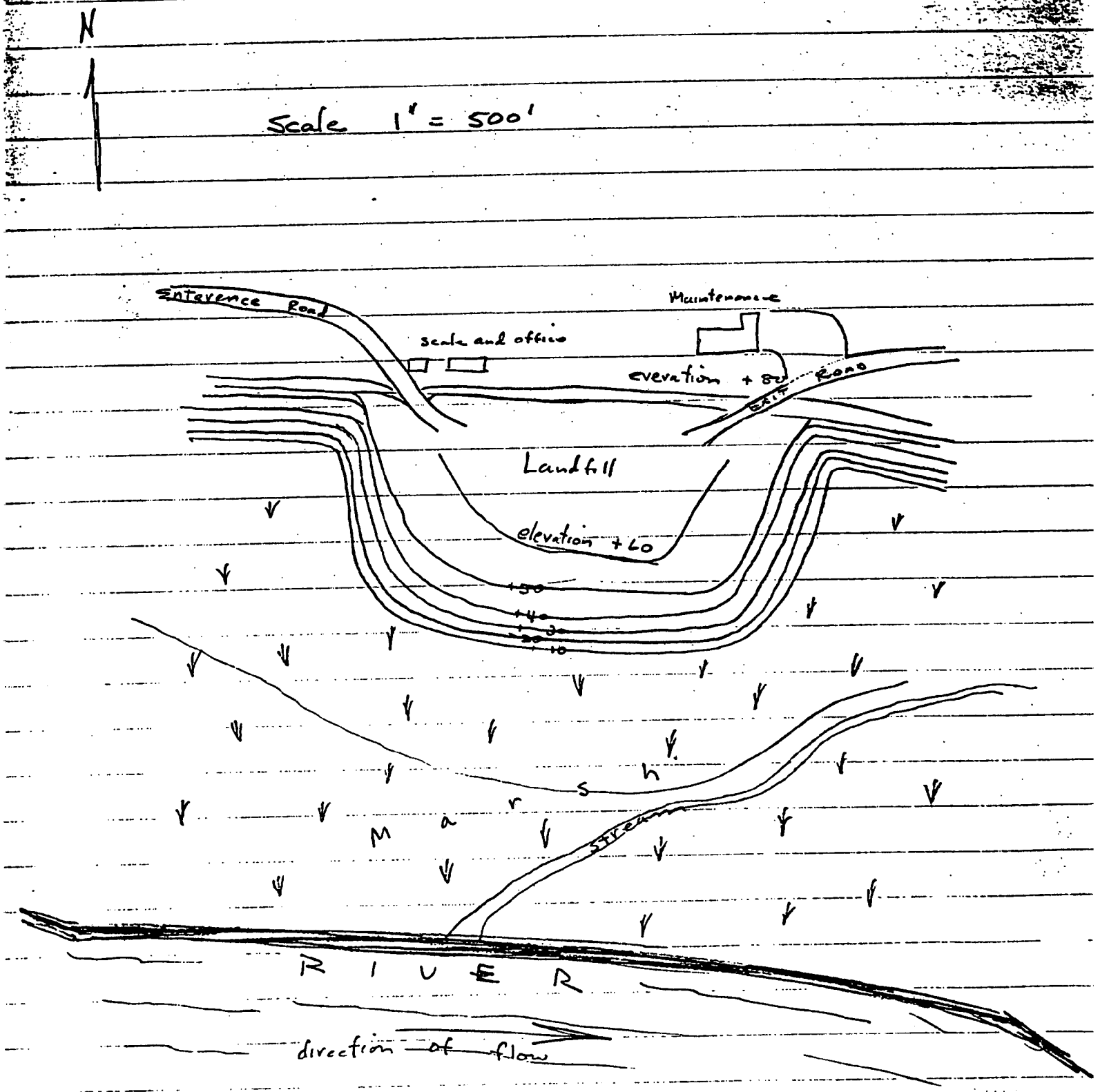
The landfill has been in existence for approximately 12 years. There was no special site preparation prior to landfilling; refuse was simply dumped into the edge of the marsh. There is presently little or no vegetation apparent on most of the landfill surface and erosion channels on the steeper slopes are apparent. Snall leachate seeps are evident along most of the top of the landfill. These flow directly into the marsh forming leachate pools which are periodically flushed out into the river during periods of heavy rainfall. A sketch

map prepared by the hydrogeologist during his field inspection and showing important features of the landfill site is shown on Figure / .

During his discussion with the landfill operator, the hydrogeologist learns that the county is considering the construction of a berm, or dike, around the southern toe of the landfill to prevent leachate from migrating into the marsh area. County officials feel that leachate can be trapped behind such a berm and pumped to an evaporation pit or back to the top of the landfill for recirculation. The hydrogeologist is asked to evaluate the effectiveness of this scheme.

Additional observations by the hydrogeologist include the fact that the flat, highly permeable top surface of the landfill would allow a large percentage of precipitation to percolate into the refuse. In addition, surface runoff from the hilly area to the north is free to flow onto the top surface of the landfill and infiltrate into the refuse. The volume of leachate likely to be generated from these two recharge sources would be considerably greater than the volume discharged by the surface seeps. Thus, a considerable volume of leachate must be moving with the ground-water system beneath the landfill and discharging into the marsh or river. If this is the case, a surface berm would do little to abate the problem. Final observations of the tour include the obvious stress on

Figure 1 – general map of County landfill Site



vegetation in portions of the marsh directly south of the landfill. However, there is no visible effect of discharging leachate on the river.

Based on his preliminary investigation, the hydrogeologist recommends a detailed ground-water investigation to determine if contaminated ground water is actually discharging directly into the river and if so, the nature of the contaminants and the rate of their discharge. In addition, he states that in this case, the construction of a berm may do little to abate leachate discharge to the river if it is in fact occurring. The results of the ground-water study, however, will suggest what other abatement steps might be more effective.

After being advised to proceed with the ground-water investigation, the hydrogeologist obtains the following:

1. Precipitation records for the past three years from a weather station located twelve miles from the landfill site.

2. A U.S. Geological Survey geologic map showing bedrock and overburden materials in the vicinity of the site.

3. Information regarding the depth and construction of the domestic supply well north of the landfill.

4. A water sample from the domestic supply well.

5. Water samples from the river both upstream from and adjacent to, the landfill.

6. A sample from one of the leachate seeps.

7. A recent aerial photograph of the site.

Analysis of the data gathered indicates that precipitation on the landfill surface averages approximately 40 inches per year. The hydrogeologist then estimates that a minimum of 50% of this precipitation infiltrates the surface of the landfill. Since the landfill area is 40 acres, at least 20 million gallons per year of leachate ~~is~~ are generated from this source. The low permeability crystalline bedrock which underlies the site probably acts as a barrier to leachate flow. Details regarding the nature of the surficial materials at the landfill site are not available. The domestic supply well north of the landfill was drilled to a depth of 100 feet and screened in a coarse sand aquifer with a high yield. Water from this well shows no indication of leachate contamination, nor do any of the water samples taken from the river. The seep sample, however, is highly mineralized and contains contaminants typically found in municipal refuse leachate. A base map of the landfill site is traced from the aerial photograph.

To further define the location of contaminated ground water at the landfill site, an electrical resistivity survey is

conducted. The results of this survey, shown on Figure 2, indicate that highly mineralized ground water is confined to an area of the marsh directly south of the landfill. Some attenuation of contaminants in the ground water appears to be occurring in the direction of the river.

While the results of the resistivity survey indicate that contaminated ground water is indeed flowing from the landfill to the river, additional geologic and water-quality data are needed to further define the problem and suggest effective abatement procedures. To obtain this information, a well drilling contractor is hired to install a series of test borings and wells. Subsequently, five test borings are drilled on and to the north of the landfill. Two casings with well-points are installed in each boring. The locations of these borings, designated A through E for the deep wells and A^1 and E^1 for the shallow wells, are shown on Figure 3. As the drilling rig cannot be operated in the marsh area, ten additional test wells are installed in this area by hand. The locations of these wells, designated 1 through 10, are also shown on Figure 3.

Construction details along with ground-water elevations, temperature and specific conductance of ground water for all the wells installed are shown on Table /. Based on ~~this~~ these data, a water-table contour map and geologic cross section are drawn

Figure 2 - Location of resistivity measurement Points

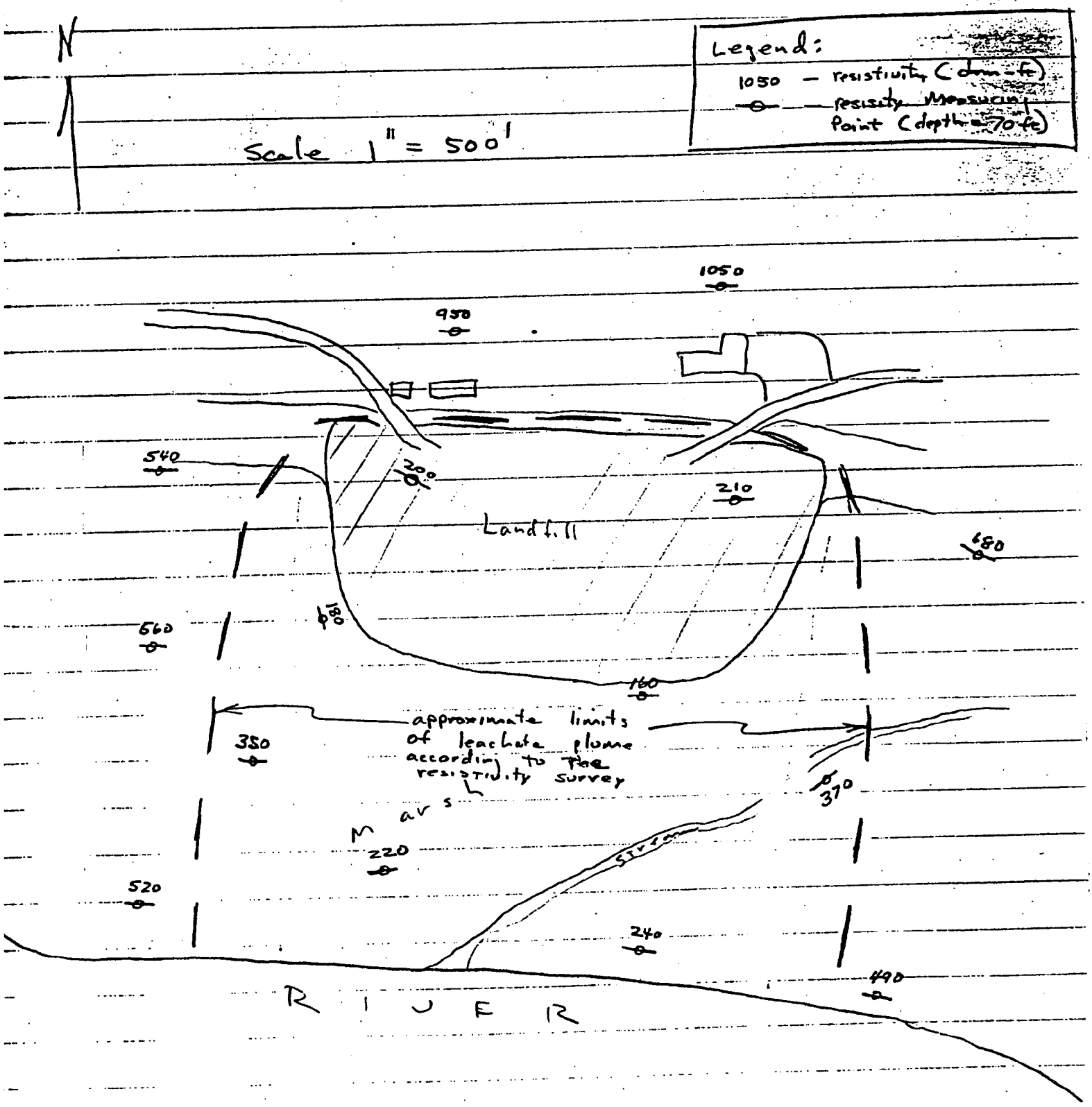


Figure 3 = Location of Test wells - ~~County Landfill~~

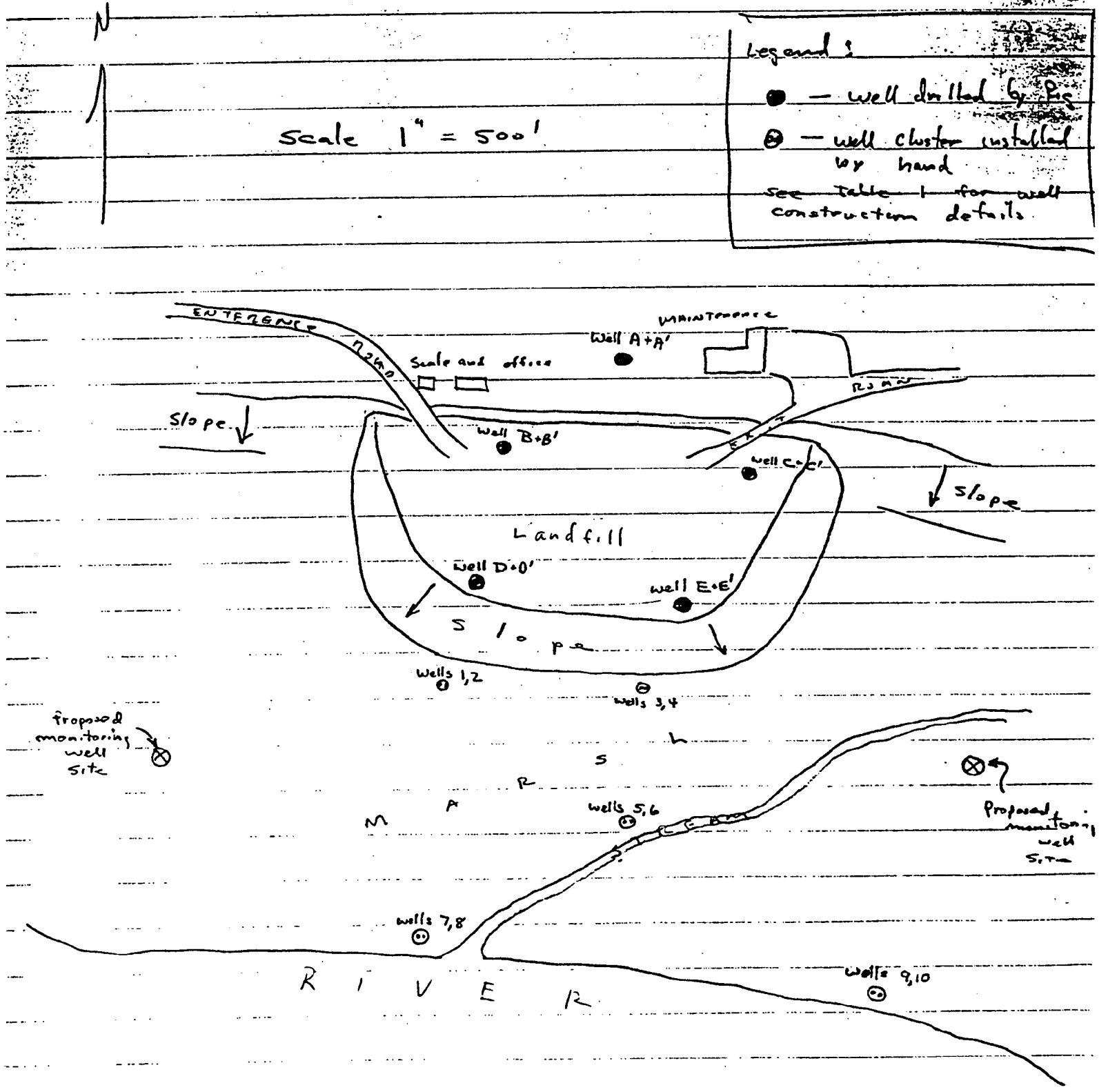


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Table 1 - Well construction details, water levels and water quality (Physical) County Landfill
(all wells equiped with a 2-foot screen or well point)

Well	well diameter (inches)	well depth (feet)	elevation (top of casing) msl	depth to water (feet)	elevation of water (feet)	specific conductance (micromhos/cm)	Temperature (°F)	
A	2.5	~~102~~	82.20	~~74.20~~	+8.00	210	62	
A'	1.5	~~83~~	82.20	74.10	+~~8~~.10	220	62	
B	2.5	~~90~~	67.80	60.15	+~~7~~.65	210	62	
B'	1.5	70	67.80	59.80	+8.00	270	65	
C	2.5	94	68.44	60.59	+7.85	940	72	
C'	1.5	73	68.48	59.98	+~~8~~.50	1060	90	
D	2.5	90	58.50	51.35	+~~7~~.15	890	71	
D'	1.5	58	58.50	51.15	+~~7~~.35	1120	94	
E	2.5	92	57.21	49.91	+7.30	990	75	
E'	1.5	60	57.32	49.61	+~~7~~.71	1220	101	
1	1.5	30	7.10	2.00	+5.10	920	82	
2	1.5	10	7.10	2.40	+~~4~~.70	820	68	
3	1.5	30	8.00	2.80	+5.20	1000	83	
4	1.5	10	8.00	3.18	+~~4~~.82	900	67	
5	1.5	35	6.60	4.20	+2.40	720	69	
6	1.5	10	6.60	3.70	+2.90	690	65	
7	1.5	30	6.90	3.90	+3.00	480	64	
8	1.5	10	6.90	4.60	+2.30	480	63	
9	1.5	30	7.10	4.00	+3.10	360	63	
10	1.5	10	7.14	5.14	+2.00	420	63	

(Figures 4 and 5 respectively). Also shown on Figure is the ground-water head at each well point. As ground water ~~flows~~ moves along flowlines from areas of higher head to areas of lower head, examination of the figures shows that highly contaminated ground water from the base of the landfill is flowing downward into the deeper sediments beneath the landfill and then upward and discharging directly into the river. This analysis is supported by the specific conductance and temperature data.

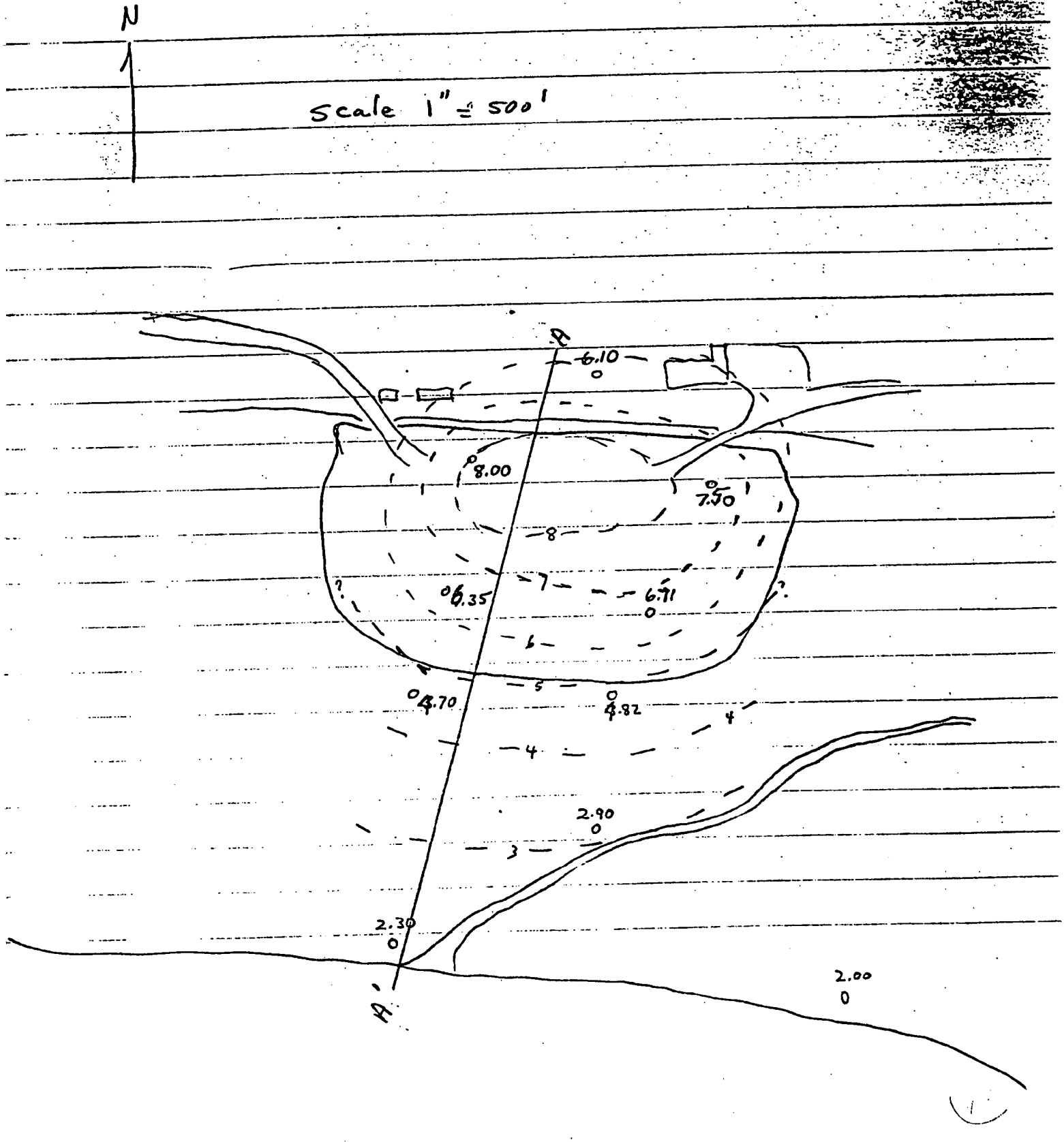
While some attenuation of contaminants is occurring along the flow path, the attenuation is by no means complete. Detailed chemical analyses of water samples from all the test wells (not given here) confirms this. In addition, contaminated ground water from portions of the landfill is discharging directly into the marsh (Figure 5) and probably responsible for for the observed vegetation stress. Figure 5 also shows contaminated water discharging directly to the river. The dilution is so great, however, that this source of contamination is not detectable in river water samples.

Possible actions that might be considered with regard to this problem are as follows:

1. Do nothing.

2. Remove the landfill to a more hydrologically acceptable site.

Figure 4 Water Table Contour map, [redacted]



3. Construct a shallow surface berm around the toe of the landfill.

4. Install pumping wells directly beneath the landfill to reverse the hydraulic gradient.

5. Install a line of interceptor wells along the toe of the landfill to restrict movement of leachate away from the landfill toward the river.

6. Reduce leachate generation by restricting recharge to the landfill.

The first possibility is unacepptable because of the severe stress placed on the marsh by the discharging leachate. The second possibility is prohibitively expensive and the third possibility would do little or nothing to abate the problem due to the deep migration of the leachate. The fourth and fifth possibilities may be technically feasible, but would be difficult to accomplish due to the low permeability of the sediments beneath the landfill. In addition, these two solutions create the new problem of what to do with the large volume of contaminated water pumped from the wells. The final possibility appears to be the best solution and is so recommended to the county.

The procedures recommended to reduce infiltration and thus

leachate generation are as follows:

1. Close the landfill to dumping as soon as an alternate site can be located. Prepare the new site using the latest technology to reduce environmental impact.

2. Immediately eliminate runoff onto the landfill surface by means of a cutoff trench and drain the collected uncontaminated runoff directly into the marsh for its beneficial flushing action.

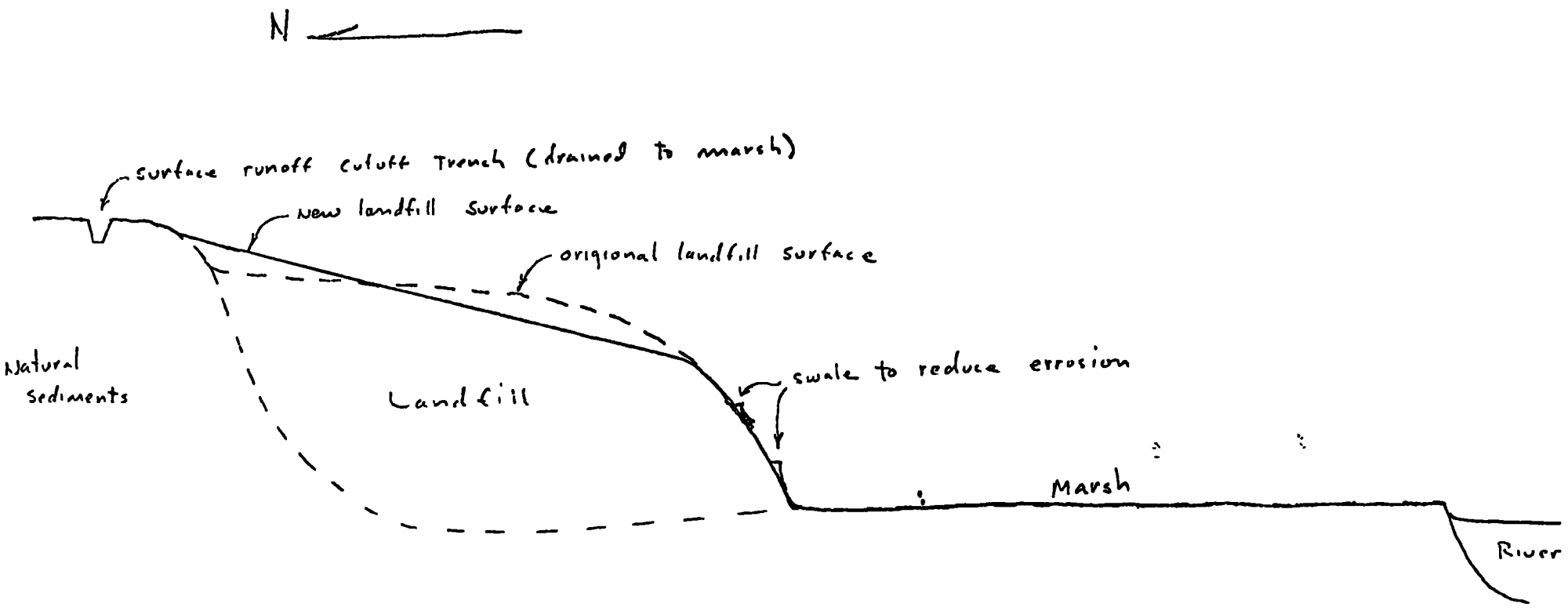
3. When the landfill is closed to further dumping, regrade the landfill surface to eliminate its presently flat top surface and create a continuous grade from the top of the hill to the toe of the landfill at the marsh (see Figure .

4. Cover the entire landfill surface with a compacted soil of low permeability. Cover this compacted material with a layer of top soil and plant a high water use grass species such as alfalfa.

5. Construct a series of swales and channels to further increase surface runoff, reduce erosion and direct the surface runoff into the marsh beyond the toe of the landfill (see Figure 6).

In addition, the consultants recommend that the county institute a monitoring program to determine the effectiveness of the

Figure 6 – Suggested landfill grade to promote surface runoff



abatement plan. It is recommended that the monitoring data collection begin as soon as possible to obtain antecedent information prior to making the abatement improvements on the complete landfill.

The recommended program is as follows:

1. Install monitoring wells of the same design as Well 2 at the two locations marked X on Figure 2. Use these, plus the 15 existing test wells as monitoring wells.

2. Measure the water level in each well monthly.

3. Measure the specific conductance of the water in each well monthly.

4. Take a water sample from each well yearly and conduct a detailed chemical analysis of each sample.

5. If any well shows a marked change in specific conductance, analyze a water sample from that well immediately.

6. Install a rain gauge on the landfill surface and record monthly precipitation.

7. Reduce all data to both tabular and graph form.

8. Review all data annually and, if necessary, adjust the monitoring program as suggested by the data analyses.

9.7.2 SCENARIO 2 - A GROUND-WATER CONTAMINATION PROBLEM

For two months, a growing number of complaints have been registered by residents of a housing development at the northern edge of a small city. So far, eight citizens from the development have visited the Board of Health to complain, each with the problem that something has suddenly gone wrong with their drinking water.

The city sanitarian sends an inspector to investigate the eight complaints. He returns with the following report. In two of the houses, slightly reddish water comes from the faucets, even after running for prolonged periods. In a third house, the water is slightly gray. The remaining houses are not experiencing discoloration, but the water has a peculiar taste and there is a slight odor apparent in the water of some of the houses he visited.

The inspector has collected a water sample from each house, directly from the kitchen sink as none of the houses are using water softeners. The water samples are sent to a laboratory for analysis. Meanwhile, the sanitarian marks the location of each of the affected houses on a map. He notes that each house is connected to the city sewer system, but each has its own water-supply well. All of the houses where the problem has occurred are in the northern half of the development, and

within an area a quarter-mile wide. Dozens of other houses are interspersed with the ones inspected and each has the same type of water supply and waste disposal system.

Only two possible causes of the water-quality problems are apparent. The more likely of the two is that the 10-year old sewer system has suddenly developed several large leaks and the raw sewage is seeping into the ground and contaminating the wells. The second, and seemingly more remote, potential cause is the 45-acre county landfill located more than a mile north of the nearest affected house. The landfill seems even more unlikely when the topography of the area is considered. ~~As shown in Figure 7~~, North of the development the terrain rises gently for several hundred yards and is then broken by a steep, elongated hill, which blocks a view of the landfill from the city. Beyond the hill, the ground slopes gently downward for more than half a mile to the edge of the landfill, located in an old gravel quarry.

Thus, for contaminated water from the landfill to reach the northern development, the sanitarian concludes it would have to travel in an uphill direction for over a mile. If this were the case, why weren't the houses affected sooner, as both they and the landfill have been there for ten years. In addition, why were only a few houses in the development affected and not the others? And what about the four houses located north of

the hill, between the development and the landfill, shouldn't they be affected if the landfill were the cause? In an effort to define the problem, the sanitarian sends his inspector to obtain water samples from several additional houses in the development where no problem had yet been reported and also from two of the four houses between the landfill and the development.

When the results of the water analyses came back from the lab they did little to indicate the source of the problem. Each of the original eight samples, taken from houses where the owners had complained, contained constituents well above the recommended limits. The constituents in highest concentrations were not the same for each house, however. Three of the houses have water supplies with abnormally high iron content and low pH. In all of the samples, chloride is well above normal for the area, but the concentrations differ from sample to sample. Significantly, concentrations of calcium and sodium are abnormally high in two samples and manganese in one. Ammonia is found to be above normal in five of the samples. The analysis of the three samples from the houses in the development whose owners had not complained, and the two samples from outside the development indicated the wells at these locations are producing high quality water.

The levels of chloride and metals in several of the samples were too high to have originated from the sanitary sewer. In

addition, the high-quality water in other houses in the development would be unlikely if large leaks had developed in the sewer line. On the other hand, the landfill is more than a mile away, downhill from the development, and high-quality water is being pumped from wells between the landfill and the development so the landfill still seems an unlikely cause of the problem. The sanitarian now believes that some completely unknown source is responsible and decides to hire a ground-water expert to determine what it is.

A ground-water consulting firm is retained by the city, presented with the analyses of water samples from the thirteen houses along with a map showing the location of those houses, and charged with locating the source of the contaminants apparent in eight of the samples.

The hydrogeologist assigned the task first obtains topographic maps and geologic maps of the area from the U.S. Geological Survey. In addition, he contacts a company providing aerial photography services in a city nearby and is able to obtain black and white aerial photographs of the city and the region to the north. A visit to the local Health Department provides well records for the houses in the affected development. These records indicate the depth of each well, the geologic materials penetrated during drilling, the static water level, and the yield of the well as estimated by the driller. Calls

to three local drilling firms produce similar records for the wells serving the four houses between the development and the landfill. A visit to the landfill site and discussions with the operator disclose the age of the landfill, the methods of landfilling used and the surface conditions and drainage characteristics of the landfill.

With these data available, the hydrogeologist is able to establish the following:

1. The houses in the development and the four houses north of the development are resting on a layer of glacial till between 10 and 30 feet thick.

2. Beneath this till layer is an extensive sand and gravel aquifer which is probably about 50 to 100 feet thick. Underlying this aquifer is crystalline bedrock.

3. The general direction of ground-water flow in the area is from the mountainous area ten miles north of the city.

4. The gradient of the water table in the vicinity of the development is low, but the permeability of the aquifer is quite high. The rate of ground-water flow in the area is about 2 feet per day.

5. The wells belonging to houses in the development, with the exception of the eight contaminated wells, are screened

near the top of the sand and gravel aquifer in an interval of about 50 to 60 feet below land surface.

6. With corrections for differences in elevation, the four wells belonging to the houses north of the development are screened at approximately the same depth in the aquifer as the majority of the development houses.

7. The eight contaminated wells in the development are screened substantially deeper than the other development wells. In four of these, a 10-foot thick clay lens was penetrated at the normal screening depth of 40 to 60 feet, and the wells were drilled an additional 20 feet into the sand and gravel beneath the clay. The remaining four wells were drilled at a later date by a different drilling firm and were inexplicably deeper.

8. The landfill, located 6,000 feet upgradient of the development, is situated in an abandoned gravel pit, which is probably connected directly to the aquifer serving the development.

9. The landfill is roughly circular, covering an area of about 45 acres, and is about 1,600 feet in diameter.

10. The contaminants reported in high concentrations in the eight wells in the development are characteristic of typical municipal landfill leachate.

11. No significantly large source of contamination other than the landfill and the sewer system is located in the immediate vicinity of the development or upgradient of the development as far as the mountains 10 miles north.

12. The landfill is 10 years old, has a broad, flat upper surface, and the deposited refuse is covered daily with sand taken from an unfilled portion of the old gravel bank.

13. Rainfall in the area averages 40 inches per year.

Based on these findings, the hydrogeologist concludes that it is indeed possible, in fact probable, that the landfill is the source of the contamination found in the eight wells. He rules out the sewer as the source of contamination since it is the deeper wells, rather than the shallow ones, which had become contaminated.

Using the available geologic and hydrologic data, a cross section of the area, including the landfill and the development, is drawn illustrating how only the deeper wells would become contaminated (see Figure 7). Since the landfill is probably resting directly on top of the aquifer, leachate generated in the landfill would flow into and move with the natural ground water. From other landfill investigations, however, it is known that leachate can flow as a distinct plume with relatively

Fig 7

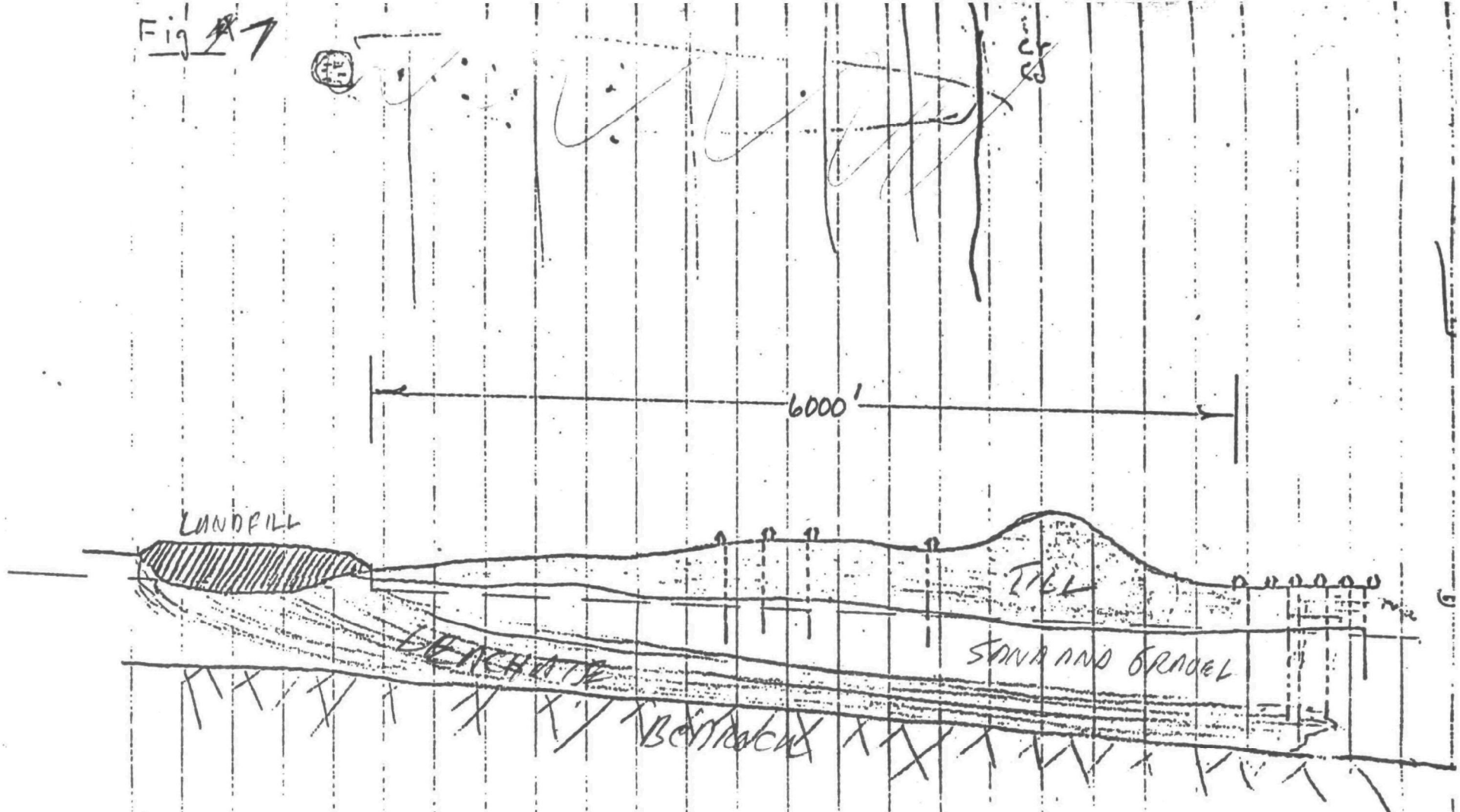


Fig 7 Geologic Cross Section

⇧ - Naval

⏚ - Domestic well.

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little dispersement in the ground-water system. Furthermore, this plume may tend to sink toward the bottom of the aquifer as it moves. Thus, the plume might be just thick enough to be picked up by the deeper wells but still could flow underneath the shallower ones, as illustrated in Figure 7. The second part of the problem, the 10-year delay for the contamination to appear, is answered by the estimated flow rate of the ground water. Assuming the leachate began to move into the aquifer during the first year of landfilling, it took approximately 3,000 days to reach the vicinity of the first well. Because the distance from the landfill to the well is 6,000 feet, ground-water velocity would have to be 2 feet per day, which is what it is estimated to be. An explanation for how contamination traveled the 400 feet from the first well affected to the last well affected in only 60 days (rather than 200) is provided by the change in velocity of the ground water as it enters the cone of influence created by the large number of pumping wells in the development area.

The width of the affected area, one-quarter mile, is explained by the width of the landfill itself (see Figure 8). Since it is possible for a leachate plume to migrate without substantial dispersion and remain at approximately its original width for substantial distances, and thus, should be at least 1,600 feet wide (probably somewhat wider) as it reaches the development.

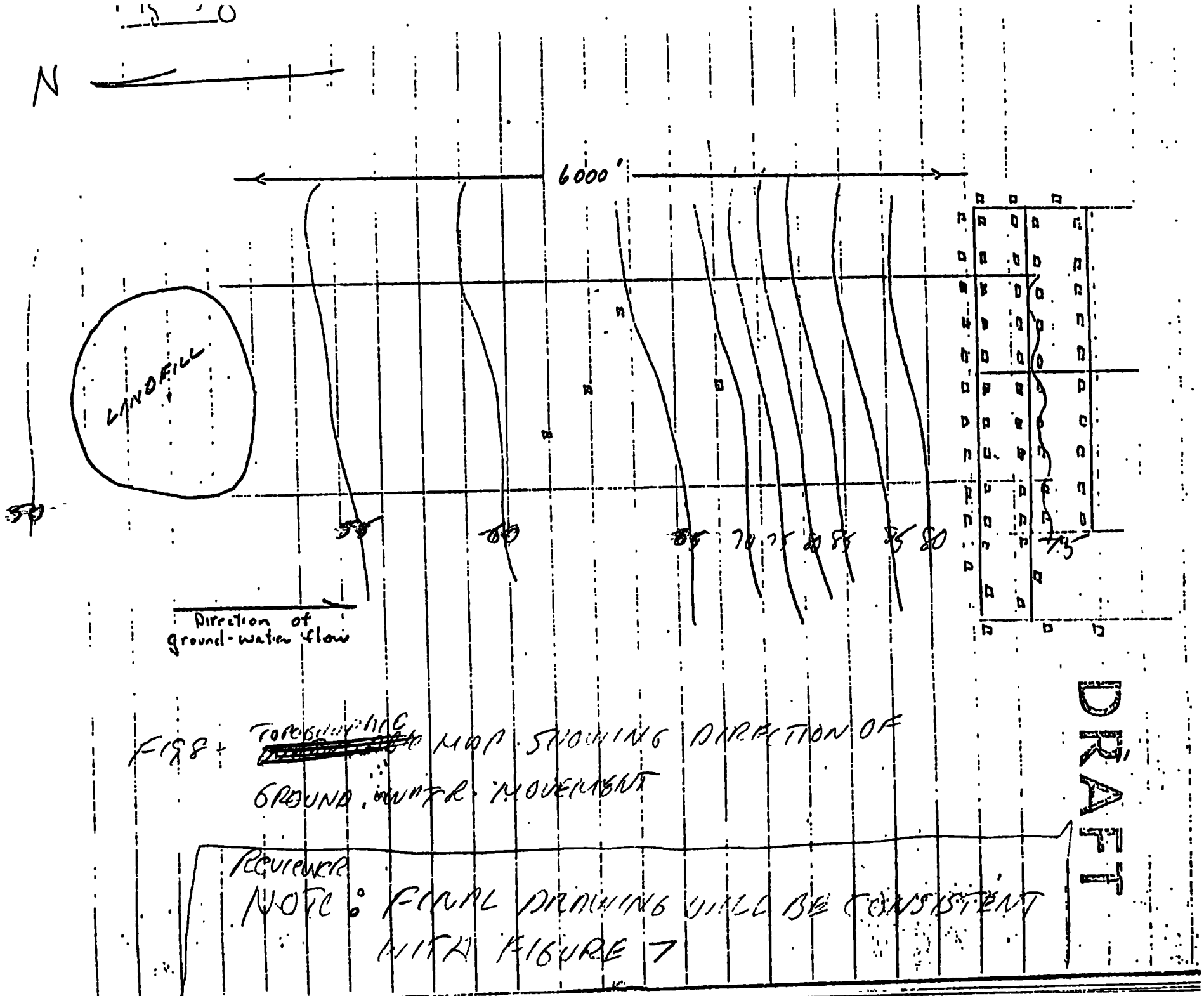

FIG 8 FORESHORTENING MAP SHOWING DIRECTION OF GROUNDWATER MOVEMENT

REQUIREMENT NOTE: FINAL DRAWING WILL BE CONSISTENT WITH FIGURE 7

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Along with the data and findings, the ground-water consultants include the following recommendations in their report to the city.

1. Immediately advise the owners of the contaminated wells to obtain their drinking water from other sources.

2. Collect water samples from all the unsampled wells in the subdivision and analyze for chloride, calcium, and iron. If any abnormal concentrations are found, advise the owners of those wells not to drink the water.

3. Immediately institute an investigation to positively establish the landfill as the source of the problem, and define the actual extent and rate of movement of the contaminants. In addition, detailed water analyses should be performed to determine what potential health hazards exist.

4. When the problem has been defined, establish what abatement procedures might be effective. Evaluate the various possible procedures and define which will be the most effective.

The City Department of Health decided to carry out the first two recommendations themselves. The consulting firm is contracted to undertake the work necessary to satisfy the third and fourth recommendations.

9.7.2.1 LANDFILL INVESTIGATIONS

The first phase of the consultant's investigation, to establish the landfill as the actual cause of the problem and to define the nature and extent of the leachate plume is undertaken as a series of tasks.

Task 1 - Assemble and analyze all available background data (already done during preliminary investigations).

Task 2 - Conduct a field inspection of landfill site (already done during preliminary investigation).

Task 3 - Conduct a resistivity survey to attempt to define the depth and lateral extent of the leachate plume.

Task 4 - Drill a total of 6 wells down to bedrock to verify the results of the resistivity survey and obtain geologic and water samples. Conduct pumping tests to determine actual hydrologic characteristics of the aquifer.

Task 5 - Construct a water-balance model of the landfill to accurately determine the contributions of precipitation and underflow to the volume of leachate generation. This task would be accomplished entirely with existing data.

The results of the Phase 1 investigation indicate that the preliminary analysis of the situation was essentially correct. Furthermore, the leachate plume is found to contain hazardous constituents originating from industrial wastes which are traditionally accepted at the landfill. The volume of leachate being generated by the landfill is calculated at approximately 80,000 gallons per day from precipitation with no significant contribution from underflow.

Based on these results, two alternative abatement programs are presented to the city. The first half of both programs is the same, eliminate the source of the pollution. The second half of the program deals with what to do about the leachate that is already in the ground. Monitoring recommendations are included with both programs.

9.7.2.2 ABATEMENT PROGRAM 1

It is recommended that placement of refuse of the existing county landfill be stopped as soon as an alternate disposal site can be located and prepared. The selection of a new site should be based on geologic and hydrologic considerations so that a new ground-water contamination problem is not created. Site preparation and landfilling methods should be based on the latest technology to minimize the possibility of leachate contamination of ground or surface water.

Preparation should begin at once to regrade the existing landfill to eliminate the flat top surface and provide adequately steep side slopes to promote runoff of precipitation. The upper surface of the landfill should be covered with a minimum of two feet of compacted soil with a low permeability to minimize infiltration. This upper layer should then be covered with a one-foot layer of top soil and seeded. A dense vegetation cover should be maintained on the landfill surface to maximize evapotranspiration.

It has been determined by aquifer tests that the leachate presently in the ground-water system can be removed by a series of high-capacity pumping wells. Three 10-inch diameter wells would be installed 400 feet apart across the plume (shown in Figure) and 500 feet north of the development. The wells would be drilled to rock and be screened from 80 feet below land surface to rock, to include the entire thickness of the plume in the screened zone. The wells would be pumped continuously at a rate of 2,000 gpm (gallons per minute). This will establish a hydraulic barrier which will block leachate flowing south from the landfill site toward the development. In addition, leachate south of the barrier wells will be drawn back toward the wells by the induced reversal in gradient. When the polluted water has been removed from the aquifer beneath the development, the pumping rate of the barrier wells can be reduced to 700 gpm

and the eight deep development wells can be returned to use, but with continual monitoring of their quality for a period of time.

While this program will effectively eliminate the present problem, a new problem of what to do with the contaminated water pumped from the barrier wells will arise. Since the capacity of the treatment plant is insufficient to handle this additional volume, the water would have to be piped away and discharged untreated either back at the landfill site or into the river at the south end of the city. There are many serious problems with these possibilities, however, and subject to further investigation, both may prove unacceptable. The only remaining alternative then would be the construction of additional treatment facilities.

9.7.2.3 ABATEMENT PROGRAM 2

Discontinue landfilling and complete the existing landfill as described in Program 1. Abandon use of the eight contaminated wells for water supply but keep them intact for use as observation wells. Drill eight new wells to a depth of 50 feet below land surface. These replacement wells would then be screened above the contaminated zone, at approximately the same depth as the other wells in the development.

Allow the contaminated plume to flow along its natural course toward the river. Since there are no city supply wells or other private wells in its path, no additional effects will be apparent until the plume reaches the river. The additional three miles the plume must travel might be sufficient to attenuate most of the contaminants. The progress of the plume should be monitored by a series of observation wells placed along its route. These monitoring wells will determine if attenuation is actually occurring at a significant rate and if the plume is altering its course. If at some time in the future it is determined that the contaminants within the plume are not being sufficiently attenuated and will be deleterious to the river, a series of barrier wells should be installed to intercept the plume prior to its reaching the river.

The city should consider the possibility of connecting the northern development to the city water supply. While this does not appear to be immediately necessary, continued close monitoring of the location of the plume in the vicinity of the development may detect an enlargement of the plume and all the wells would have to be abandoned.

A-1 FUNDAMENTALS OF LEACHATE

In their publication, Summary Report: Gas and Leachate from Land Disposal of Municipal Solid Waste, U.S.E.P.A., Cincinnati, Ohio, 1974, the U.S.E.P.A. presents an excellent comprehensive summary on leachate, its production and characteristics. Much of this material has been reproduced and included in this appendix for the convenience of the user of the manual. Hereinafter, the above-referenced report will be called the leachate summary report.

Two other reports on leachate which are pertinent to assessing potential leachate contamination at land disposal sites are:

- Use of the water balance method for predicting leachate generation from solid waste disposal sites, Office of Solid Waste Management Program, U.S.E.P.A., October 1975, (EPA/530/SW-168).
- An environmental assessment of potential gas and leachate problems at land disposal sites. Office of Solid Waste Management Programs, U.S.E.P.A., 1973, (SW-110).

These reports will also be referenced in this section and will be referred to as the water balance report and environmental assessment report respectively. It is intended that this appendix provide the user of this manual with sufficient information to assist in performing an assessment of potential leachate contamination at land disposal sites. This, in turn, is used to determine the need, type and intensity of monitoring that should be assigned to a

particular land disposal site.

LEACHATE PRODUCTION

In their environmental assessment report, EPA puts leachate production into perspective. It states:

> "It becomes quite evident that the main parameter affecting leachate quality and quantity is purely and simply the quantity of water flow through the solid wastes. Generally, the more water that flows through the solid waste, the more pollutants will be leached out. Therefore, the proper sanitary landfill design and operational approach is to eliminate or minimize percolation through the solid waste. With the smaller amounts of percolation, the pollutants tend to be more concentrated, but the rate at which they are transmitted to the surrounding environment is not so apt to exceed the capability of the natural surroundings to accept and attenuate most of them to some degree."

Therefore, one can see that the volume of leachate generation is influential in both the extent of a leachate contamination problem and the relative strength of the leachate and its concentration in the ground water being monitored.

Estimating leachate generation can be useful in designing a monitoring program, and interpreting the data collected in the following ways:

- Predicting the time of first appearance of leachate.
- Predicting the potential quantity of pollutants generated at a land disposal site.
- Help explain fluctuations in monitoring well data that may occur, and
- Relate operational characteristics and site conditions to potential leachate generation.

In the leachate summary report, and the water balance report, EPA applies the water balance method as a useful tool in estimating leachate generation at a land disposal site.

In addition, the leachate summary report provides an excellent summary of leachate characteristics as has been observed by many researchers in the field. Data are presented on the quality of pure leachate as well as samples of leachate-enriched ground water. Examples are also given depicting the relationship of leachate concentrations to quantity produced and season of the year. A comprehensive list of references on leachate is also presented.

For the convenience of the manual user, sections have been reproduced from the leachate summary report and the water balance report and included as part of this appendix.

The following section has been reproduced from:

"SUMMARY REPORT: GAS AND LEACHATE FROM LAND DISPOSAL OF MUNICIPAL SOLID WASTE", U.S. EPA, Cincinnati, Ohio, 1974.

SECTION VI

LEACHATE PRODUCTION

The various physical, chemical, and biological processes that occur when solid wastes are disposed on land produce compounds that are susceptible to solution or suspension in water percolating through the disposed solid waste. This percolating water containing solids derived from the solid waste is called leachate. The volume of leachate produced at any particular site is dependent on many factors, but generally, is determined by the quantities of surface water infiltration and/or interception of groundwater. Composition of leachate is highly dependent on the composition of solid waste, its age, and the environment in which it is located. Environmental conditions, such as temperature, moisture regimen, and the availability of oxygen are significant factors in determining the exact chemical constituents contained within leachate.

VOLUME

The sanitary landfill site is a part of the classical hydrologic cycle. The governing criteria for determining leachate volume are those describing the phenomena occurring at the cover material surface. A water balance can be written:

$$W_R + W_{SR} + W_{GW} + W_{IR} = I + R + E \ldots\ldots [1]$$

where W_R = input water from precipitation

W_{SR} = input water from surrounding surface runoff

W_{GW} = input water from groundwater

W_{IR} = input water from irrigation

I = Infiltration

R = Surface Runoff

E = Evapotranspiration

Infiltration can be defined:

$$I = \Delta S_s + \Delta S_R + L + W_D \text{ ..... } [2]$$

where ΔS_s = change in moisture storage in soil

ΔS_R = change in moisture storage in solid waste

L = leachate

W_D = water contributed by solid waste decomposition

Proper design and operation can eliminate input water from surrounding surface runoff, groundwater and irrigation. Some control can be exerted over infiltration, evaporation, surface runoff, and moisture storage capacity of soils and solid waste. The volume of water produced during solid waste decomposition is generally considered negligible.

Use of the water balance has been proposed by Remson, et al.[1] Fenn and Hanley,[2] Salvato, et al.[3] and California.[4] The volume of leachate, time of initial occurrence, and subsequent flow rate and allowable volume of irrigation water can all be determined by appropriate use of the water balance. The Remson work, supported in part by U.S. EPA Research Grant R801947, provided a useful computerized moisture routing technique. Salvato, et al., and the California summary discuss the various factors that influence runoff and infiltration and provide guidance for determining approximate values. Fenn and Hanley applied the water balance to hypothetical landfills in Cincinnati, Orlando, and Los Angeles.

Determination of runoff from landfill surfaces by the rational runoff formula was proposed by Salvato, et al. They provided tables for a rainfall of 25.4 mm/hour (1 inch/hour) intensity and 6-hour duration. The empirical runoff coefficients (C) used were from Frevert, et al.[5] and are provided in Table 5 along with calculated quantities of runoff for a 25.4 mm (1-inch) rainfall. The influence of slope, surface condition, and soil type on the quantity of runoff and the potential for leachate production is clearly demonstrated in Table 5. As great as 55 percent change in runoff and infiltration is attributed to slope. As great as 173 percent change in runoff and infiltration is attributed to soil type. As great as 71.5 percent change in runoff and infiltration is attributed to surface condition (vegetation, bulk density). A silt or clay loam, in a pasture land area at a 5 to 10 percent slope is generally recommended for encouraging runoff, limiting erosion, and avoiding soil shrinkage problems. As such, a coefficient of 0.36 would apply and one might expect from a storm of 25.4 mm/hr (1 inch/hr) intensity and 1 hour duration, approxi-

mately 9.06 m^3/ha (9,690 gal/acre) runoff and 16.3 m^3/ha (17,400 gal/acre) potential infiltration. Of course a large amount of this infiltration is lost by evaporation and transpiration. The remainder goes first to meeting moisture retention (storage) capacity of the soil and solid waste and then to leachate in accordance with Equation 2.

Determination of the runoff from storms by the rational runoff formula is largely dependent on the accuracy of the coefficient, C, chosen for the specific site. Not specifically considered in the rational runoff formula are: the previous moisture conditions of the site and the limitation imposed by the hydraulic conductivity of the soil. Infiltration rates generally cannot exceed the hydraulic conductivity (permeability) of the soil. The hydraulic conductivity of soils has been conveniently tabulated.[6] Table 6 provides estimates of maximum hourly infiltration for these soils assuming Q = CiA is applicable, the soils are saturated and uniform, and sufficient water is available. Comparison of Tables 5 and 6 indicates the necessity for careful determination of the hydraulic conductivity of proposed cover soils (differences as great as 10^3 in the same soil group) and interpretation of the results of the rational runoff formula.

Infiltration is dependent on the frequency, duration, and intensity of rainfall. These precipitation characteristics are significant in determining the previous moisture conditions of the soil and hence the amount of water required to reach saturation when the hydraulic conductivity of the soil will control infiltration rates. The Bureau of Reclamation[7] has related rainfall intensity to infiltration, and has accounted for differences due to vegetation, soil type, precipitation, and evaporation. Appropriate relationships are depicted in Figure 2 and Table 7.

The importance of vegetation in promoting infiltration is clearly shown in Table 5. The density and type of vegetation are also important in determining evaporation and transpiration. Consumptive use of water, Table 8, is determined largely by the vegetative-soil system, but ranges have been compiled.

Moisture retention by soil is dependent on soil type and previous wetting. Soil will retain a characteristic amount of water against the pressures exerted by gravity and plant roots. These are referred to as field capacity and wilting point respectively. They are commonly expressed as a percent of volume or as a depth per unit depth of soil. Examples are provided in Table 9. The difference in water retained between the wilting point and the field capacity is that amount available for evapotranspiration and storage.

Table 5. RUNOFF AND INFILTRATION FOR A 2.5 cm RAINFALL[a]

Surface condition	Slope	Rational runoff coefficient[b,c]			Runoff in m^3/ha[d,e]			Infiltration in m^3/ha[e]		
Pasture or meadow (cover crop)	%	Sandy loam	Clay or silt loam	Clay	Sandy loam	Clay or silt loam	Clay	Sandy loam	Clay or silt loam	Clay
Flat	<2 0-5	0.05-0.10 0.10	0.13-0.17 0.30	0.40	25.7 (2,730)	77.1 (8,200)	102 (10,900)	230 (24,500)	180 (19,100)	153 (16,300)
Rolling	2-7 5-10	0.10-0.15 0.16	0.18-0.22 0.36	0.55	41.0 (4,360)	91.1 (9,690)	141 (15,000)	215 (22,900)	164 (17,400)	115 (12,200)
Hilly	>7 10-30	0.15-0.20 0.22	0.25-0.35 0.42	0.60	56.3 (5,900)	107 (11,400)	155 (16,500)	199 (21,200)	149 (15,800)	102 (10,900)
Cultivated (no vegetatio) not compacted										
Flat	0-5	0.30	0.50	0.60	76.7 (8,160)	128 (13,600)	59.2 (6,300)	180 (19,100)	128 (13,600)	102 (10,900)
Rolling	5-10	0.40	0.60	0.70	102 (10,900)	153 (16,300)	180 (19,100)	153 (16,300)	102 (10,900)	76.7 (8,160)
Hilly	10-30	0.52	0.72	0.82	133 (14,100)	184 (19,600)	210 (22,300)	123 (13,100)	77.7 (7,630)	46.1 (4,900)

[a]After Salvato, J. A., Wilkie, N. G. and Mead, B. E. "Sanitary Landfill Leachate Prevention and Control." Journal WPCF, 43 (10). 2084 (1971).

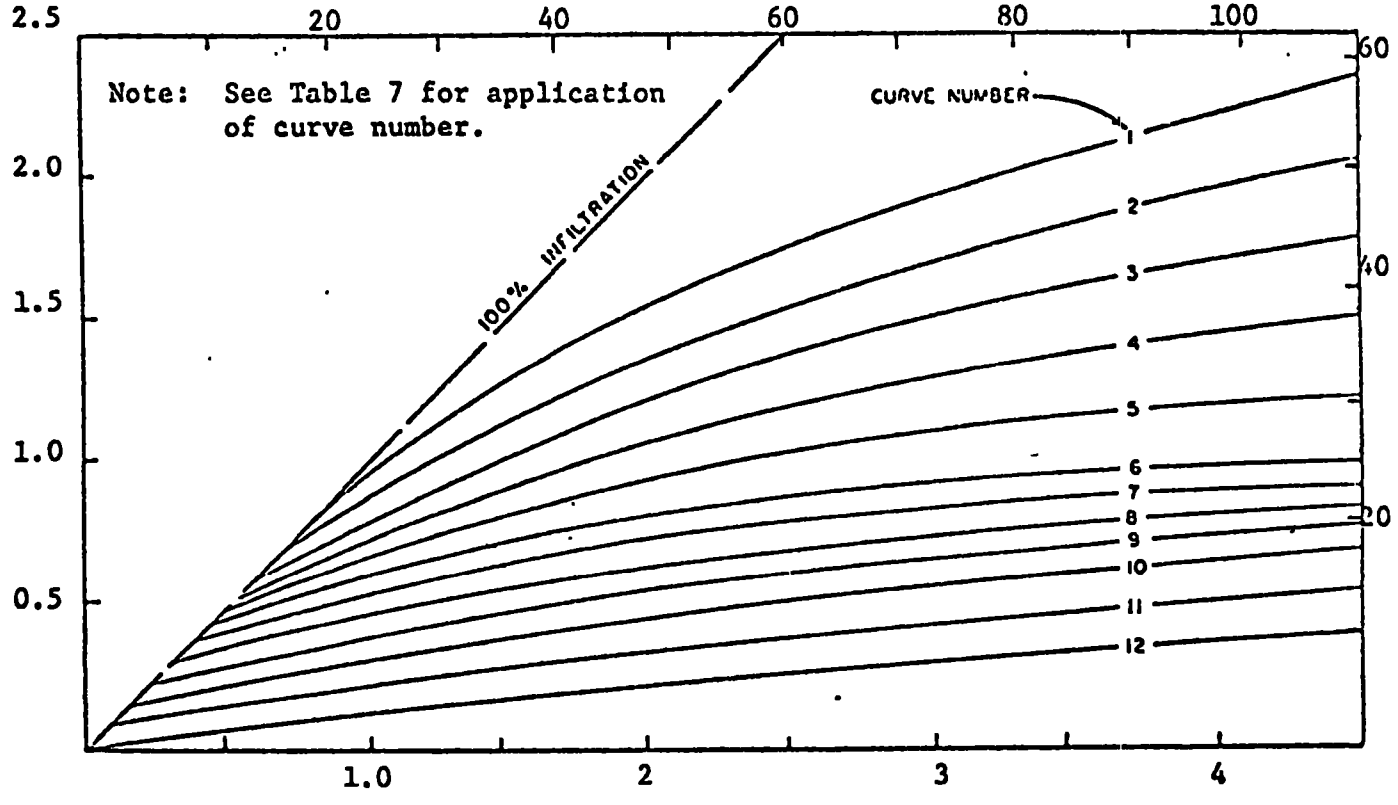
[b]Frevert, Schwab, Edminster and Barnes. Soil and Water Conservation Engineering, Wiley, pp. 439 (1963).

[c]Ven TeChow, "Handbook of Applied Hydrology." (1964).

[d]Q = CiA.

[e]Numbers in parenthesis refer to gallons/acre.

RAINFALL, mm/hr.



INFILTRATION, in/hr.

INFILTRATION, mm/hr.

RAINFALL, in/hr.

Figure 2. INFILTRATION CALCULATION CURVES ADAPTED FROM "DESIGN OF SMALL DAMS"[7]

Table 6. MAXIMUM HOURLY WATER TRANSMISSION UNDER SATURATION

Soil description[a]	Hydraulic conductivity cm/sec	Hourly transmitted volume[b] m^3/ha
Well-graded gravels or gravel-sand mixtures, little of no fines	$>10^{-2}$	$>3.6x10^{3}$ ($>3.85x10^{5}$)
Poorly graded gravels or gravel-sand mixtures, little or no fines	$>10^{-2}$	$>3.6x10^{3}$ ($>3.85x10^{5}$)
Silty gravels, gravel-sand-silt mixtures	10^{-3} to 10^{-6}	$3.6x10^{2}$ to $3.6x10^{-1}$ ($3.85x10^{4}$ to $3.85x10^{1}$)
Clayey gravels, gravel-sand-clay mixtures	10^{-6} to 10^{-8}	$3.6x10^{-1}$ to $3.6x10^{1}$ ($3.85x10^{1}$ to $3.85x10^{-1}$)
Well-graded sands or gravelly sands little of no fines	10^{-3}	$>3.6x10^{2}$ ($>3.85x10^{4}$)
Poorly graded sands or gravelly sands, little or no fines	10^{-3}	$>3.6x10^{2}$ ($>3.85x10^{4}$)
Silty sands, sand-silt mixtures	10^{-3} to 10^{-6}	$3.6x10^{2}$ to $3.6x10^{-1}$ ($3.85x10^{4}$ to $3.85x10^{1}$)
Clayey sands, sand-clay mixtures	10^{-6} to 10^{-8}	$3.6x10^{-1}$ to $3.6x10^{1}$ ($3.85x10^{1}$ to $3.85x10^{-1}$)
Inorganic silts and very fine sands rock flour, silty or clayey fine sands or clayey silts with slight plasticity	10^{-3} to 10^{-6}	$3.6x10^{2}$ to $3.6x10^{-1}$ ($3.85x10^{4}$ to $3.85x10^{1}$)

Table 6. MAXIMUM HOURLY WATER TRANSMISSION UNDER SATURATION (Continued)

Soil description[a]	Hydraulic conductivity cm/sec	Hourly transmitted volume[b] m^3/ha
Organic silts and organic silt-clays of low plasticity	10^{-4} to 10^{-6}	$3.6x10^{1}$ to $3.6x10^{-1}$ ($3.85x10^{3}$ to $3.85x10^{1}$)
Inorganic silts, micaceous or diatomaceous fine sandy or silty soils, elastic silts	10^{-4} to 10^{-6}	$3.6x10^{1}$ to $3.6x10^{-1}$ ($3.85x10^{3}$ to $3.85x10^{1}$)
Inorganic clays of high plasticity, fat clays	10^{-6} to 10^{-8}	$3.6x10^{-1}$ to $3.6x10^{-3}$ ($3.85x10^{1}$ to $3.85x10^{-1}$)
Organic clays of medium to high plasticity, organic silts	10^{-6} to 10^{-8}	$3.6x10^{-1}$ to $3.6x10^{-3}$ ($3.85x10^{1}$ to $3.85x10^{-1}$)
Inorganic clays of low to medium plasticity, gravelly clays, sandy clays, silty clays, lean clays	10^{-6} to 10^{-8}	$3.6x10^{-1}$ to $3.6x10^{-3}$ ($3.85x10^{1}$ to $3.85x10^{-1}$)

[a]Soil description according to USCS.

[b]Numbers in parenthesis are gal/acre.

Table 7. REPRESENTATIVE VARIATION OF RAINFALL - INFILTRATION CURVES WITH SOIL TYPE, COVER, AND PRECEDING MOISTURE CONDITIONS[a]

Soil type	Cover	M values[b] 0.2	0.4	0.6	0.8	1.0
		Curve number[c]				
Sandy loam	Turf	1	2	4	6	8
Sandy loam	Bare	3	4	6	8	10
Clayey loam	Turf	2	3	5	7	9
Clayey loam	Bare	5	6	8	10	12

[a]Sanitary Landfill Studies: Appendix A -- Summary of Selected Previous Investigations. California Department of Water Resources. Sacramento. 1969. 115 p.

[b]M increases with degree of soil saturation.

[c]Curve number refers to Figure 2.

$^1M = \frac{e\ (^d60 + 1)}{10}$ for nonirrigated areas.

$M = \frac{e\ (^d60 + 1)}{10} + A$ for irrigated area such as parks, where A is allowance for irrigation = 0.11.

where: e = evaporation = $(0.9 - \frac{^e60}{e\ annual})$

where: $(^e60)$ = pan evaporation for preceding 60 days.

(e annual) = man annual pan evaporation.

$^d60$ = weighted preceding 60-day precipitation as:

$$^d60 = p(1\text{-}4) + p(\tfrac{5\text{-}9}{2}) + p(\tfrac{10\text{-}14}{4}) + p(\tfrac{15\text{-}30}{6.67}) + p(\tfrac{31\text{-}60}{10})$$

Table 8. APPROXIMATE SEASONAL CONSUMPTION OF WATER[a]

Vegetation	mm/unit area
Coniferous trees	102-229
Deciduous trees	177-254
Potatoes	177-280
Rye	457-up
Wheat	509-560
Grapes	152-up
Corn	509-191
Oats	711-1020
Meadow grass	560-1525
Lucern grass	660-1400

[a]Adapted from Urquhart, L. C., Civil Engineering Handbook. New York, McGraw-Hill. p. 907. January, 1950.

Table 9. MOISTURE CRITERIA OF SOILS

Soil	Field capacity			Wilting point			Ref.
	%	in/ft	mm/m	%	in/ft	mm/m	
Sand	7.5	0.9	75	3.33	0.4	33.3	(1)
Loam	25.8	3.1	258	13.3	1.6	133.0	(1)
Clay	43.3	5.2	433	24.2	2.9	242.0	(1)
Fine sand	12.0	1.4	120	2.0	0.24	20.0	(2)
Sandy loam	20.0	2.4	200	5.0	0.6	50.0	(2)
Silty loam	30.0	3.6	300	10.0	1.2	100.0	(2)
Clay loam	37.5	4.5	375	12.5	1.5	125.0	(2)
Clay loam	45.0	5.4	450	15.0	1.8	150.0	(2)

1. Sanitary Landfill Studies, Appendix A - Summary of Selected Previous Investigations. California Department of Water Resources, Sacramento. 1969.

2. Thornthwaite, C. W. and Mather, J. R., "Instructions and Tables for Computing Potential Evapotranspiration and the Water Balance". Publications in Climatology, X(3), Drexel Institute of Technology, Laboratory of Climatology. 1957.

Table 10. MOISTURE RETENTION OF SOLID WASTE[a]

Initial Moisture % Wet Weight	Added moisture mm/m	Added moisture in/ft.	Solid Waste Density[b] kg/m^3	Solid Waste Density[b] lb/yd^3	Reference	Notes
29.9	425	5.1	392	661	1	c,d,e
25.1	258	3.1	430	727	1	c,d,e
32.0	441	5.3	417	705	1	c,d,e
27.6	200	2.4	592	1000	1	c,d,e
62	100	1.2	337	570	2	c,d,e
50	142	1.7	314	530	2	c,d,e
	108-125	1.3-1.5			3	
	208	2.5			4	c,e
	125	1.5			5	

Notes: a adapted from A. A. Fungaroli and R. L. Steiner "Investigation of Sanitary Landfill Behavior" Research Grant R800777 October 1973.

b unit wet density

c not corrected for evaporation and transpiration losses.

d includes H_2O retained in soil cover.

e includes H_2O retained in sub-drain.

References:

1. "Project Plan. Test Cell 2, Boone County Field Site." Solid and Hazardous Waste Research Laboratory, U.S. EPA, Cincinnati Jan. 1973 (manuscript).

2. Rovers, F. A. and Farquhar, G. J. "Infiltration and Landfill Behavior" in Proceedings of the American Society of Civil Engineers. 99 (EE5), pp 671 - 690. October 1973.

3. "Pollution of Water by Tipped Refuse" Ministry of Housing and Local Government, Her Majesty's Stationery Office, London, 1961.

4. Merz, R. C., Final Report on the Investigation of Leaching of a Sanitary Landfill. State Water Pollution Control Board. Publication 10. Sacramento, California 1954. 91 p.

5. Qasim, S. R. and J. C. Burchinal "Leaching from Simulated Landfills," Journal of the Water Pollution Control Federation 42, pp 371-379, March 1970.

Moisture retention in solid waste is similar in concept to that of soil except no data is available on the wilting point. It is generally assumed that water is lost from the solid waste in significant amounts only through percolation. Table 10 and Figure 3 provide a summary of data on the field capacity of solid waste. Decomposition of the solid waste, particle size, density, and initial moisture content account for the wide range in reported values.

The moisture retention capacity of solid waste has been proposed for exploitation as a receptor of liquid wastes and sludges. Examples are municipal water and wastewater treatment plant sludges, commercial wastes such as vegetable market and restaurant wastes, and industrial liquid and sludge wastes not permitted to be discharged to streams. Such a contribution of water to the solid waste in the sanitary landfill does not necessarily create a leachate problem or significantly affect the total volume of leachate produced. The following hypothetical example will indicate the quantities involved.

A typical sanitary landfill, 592 kg/m^3 (1,000#/cu yd) of municipal solid waste, placed 3 meters deep (9 feet) will hold approximately 760 mm (30 inches) of H_2O/unit surface area (18.7 gallons of H_2O/sq ft) before steady state leaching occurs. If no sludge or high moisture bearing solid waste is added to this waste, then 3 years will likely elapse before steady state leaching is established (based on assumption of 254 mm (10 inches/sq ft) annual net infiltration). If 508 mm (20 inches) of excess moisture (12.4 gal/ft^2) is added to the solid waste during deposition, then steady state leaching will likely occur 1 year later. Total leachate volume produced in the first 10 years with no intentional moisture addition during deposition is approximately 2,540 mm (100 inches), (62.2 gallons/sq ft); if 503 mm (20 inches) of moisture is added during deposition, then leachate volume after 10 years is approximately 3,050 mm (120 inches), (74.6 gallons/sq ft).

It is obvious from the example that the occurrence of leachate will be accelerated if water is added to the landfill. It is not as obvious, nor is it as easy to evaluate the impact on leachate and gas characteristics. Additions of water and the compounds dissolved in it may accelerate decomposition as well as inhibit it; it may create, magnify, or reduce the impact of leachate on the environment; operational problems may be solved as well as created.

The seasonal dependence of evaporation, transpiration, and infiltration and the dependence of all these factors on the distribution of rainfall and available moisture throughout the year create a complex problem that has not been rigorously solved. Remson presented a moisture routing procedure easily adaptable to electronic computa-

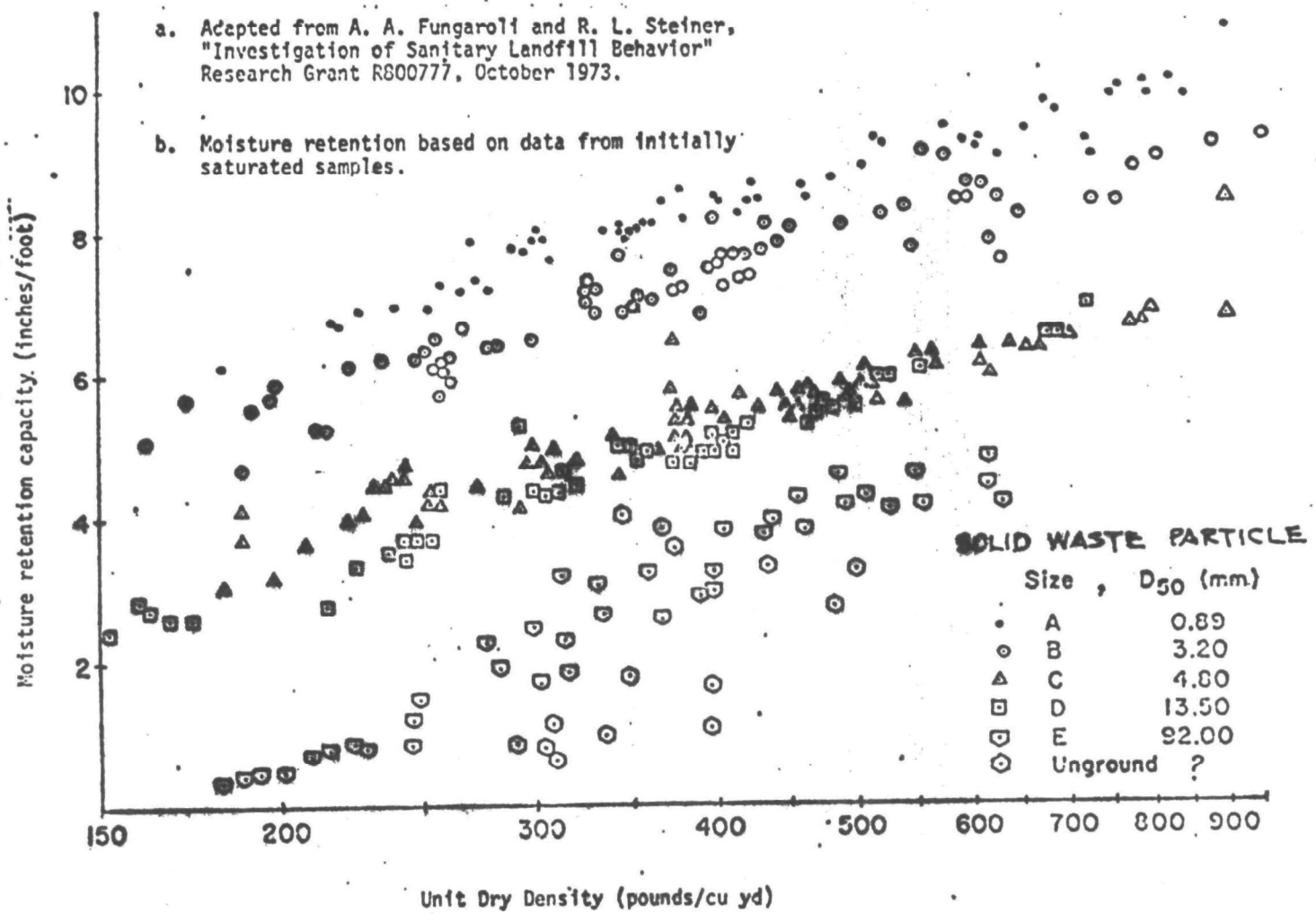


Figure 3. Moisture retention capacity of solid waste[a, b]

tion. Fungaroli[8] developed a national evaluation of potential infiltration, Figure 4, based on annual averages for evapotranspiration and rainfall and no surface runoff. Inclusion of surface runoff requires identification of site-specific characteristics; evaluation of leachate volume is best left to analysis of specific sites, rather than regional generalizations. Fenn and Hanley applied the water balance, including provision for surface runoff and evapotranspiration for Cincinnati, Orlando, and Los Angeles.

Control measures, such as diversion of upland drainage, sloping of cover material, use of relatively impermeable soils for cover material, rapid attainment of final elevations, planting of high transpiring vegetation, use of impermeable membranes overlying the final lift of solid waste, maintenance of final grades, and use of subsurface drains and ditches to control groundwater, are available to the design engineer and operator. Use of impermeable membranes requires vents to manage landfill gases and drains to manage the intercepted infiltrating water. There is a general paucity of quantitative information on the use of these controls.

Hughes, et al.[9] calculated from piezometer calculations that 40 to 50 percent of the annual precipitation of Illinois of 838 mm (33 inches) will infiltrate the surface of landfills to produce leachates. In the dry California climate more than two-thirds of the simulated rainfall applied to a solid waste cell was evaporated.[10] The only landfills where the amount of runoff is actually measured are the test fills in Sonoma County, California.[11]

The results of the test cells in Figure 5 show that with a low amount of rainfall approximately 40 percent leaves the landfill as overland runoff. At higher intensities, a constant amount of approximately 14 mm (0.55 inch) is retained at the surface of the fill while precipitation in excess of this amount appears as runoff water. Schoenberger and Fungaroli[12] found that during the winter period two landfills in Pennsylvania produced leachate at a rate of 0.29 cm/day and 0.23 cm/day which amount is equal to the net precipitation (rainfall-evaporation) in that period. Lower percentage infiltration are experienced in Europe. Only one European study measured a leachate volume equal to 44 percent of the yearly precipitation.[13] More typical values lie around 10 to 26 percent (Reuss, 1971) or 10 percent[14] (Pierau, 1968) corresponding with an amount of leachate of 0.03 to 0.10 l/sec/ha or 0.3 to 0.9 mm/day. Klotter and Hantge[15] (1969) measured a flow rate of 0.06 l/sec/ha for a 9 ha. landfill, while Knock and Stegman[16] (1971) calculated a volume of 0.08 l/sec/ha. The considerable spread in the relative amount of leachate generated from landfills may indicate that by properly manipulating the nature and the slope of the surface cover, the amount of leachate can be reduced or enlarged as desired.

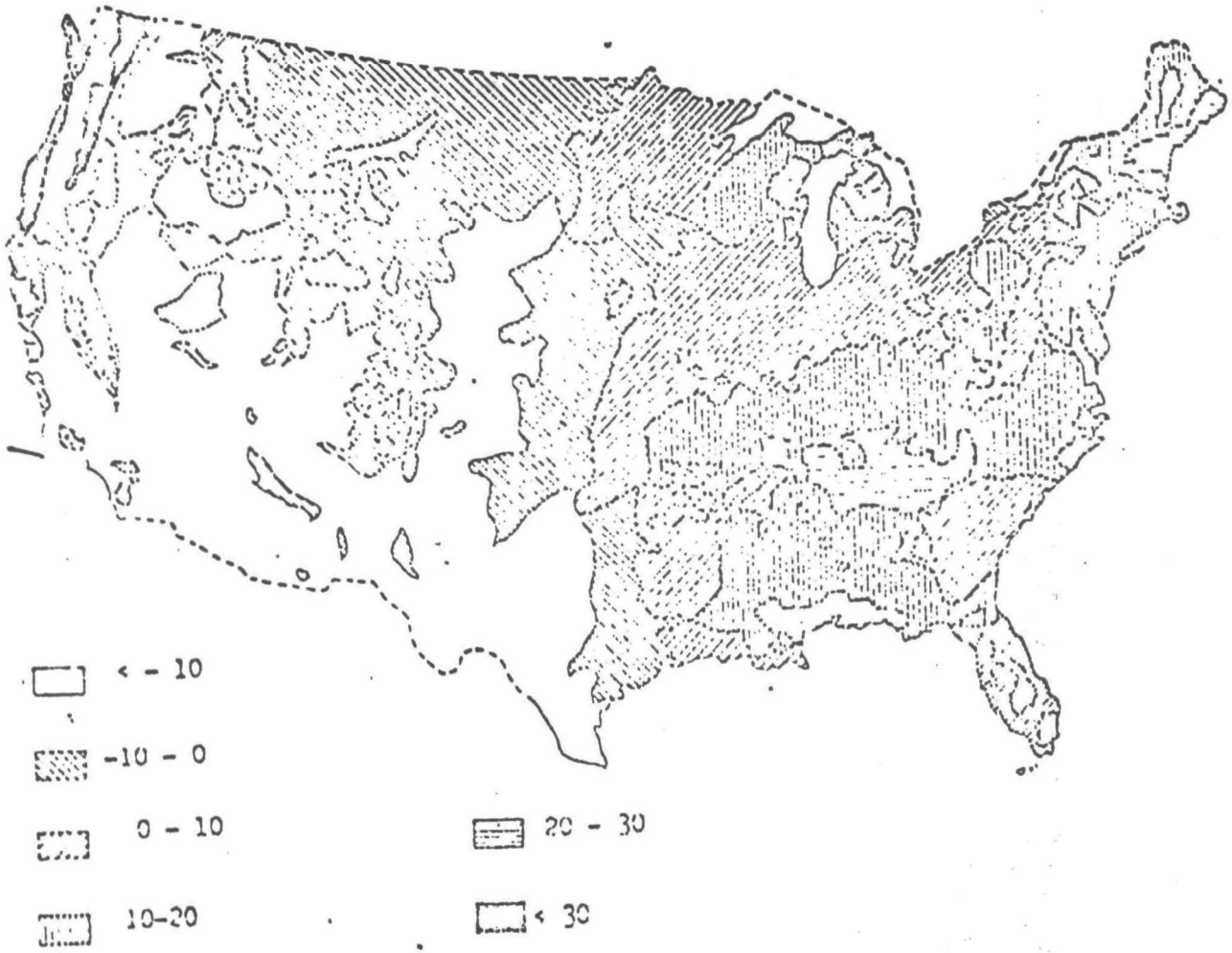


Figure 4. AVERAGE POTENTIAL INFILTRATION - INCHES

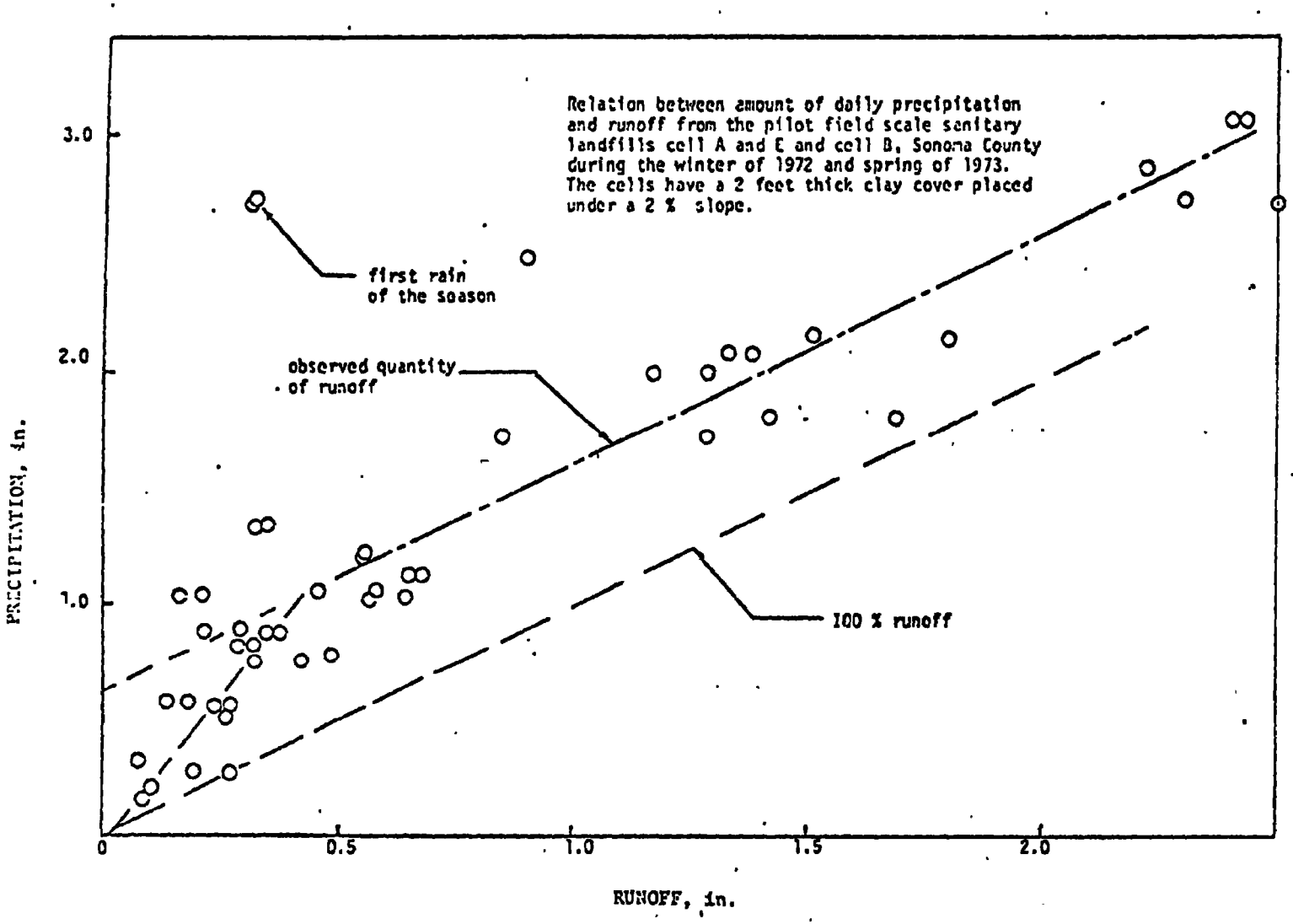


Figure 5 . OBSERVED RUNOFF FOR PRECIPITATION EVENTS.

The critical area of limited information appears to be determination of surface runoff/infiltration under surface conditions prevalent at sanitary landfill sites. Such factors as slope, erosion, vegetation, soil density, and soil type are factors that need to be studied further. Work presently being conducted under U.S. EPA Research Grant R802412 is evaluating the net infiltration through simulated sanitary landfill cover materials with three soil types, three soil densities, and three types of vegetation. This work needs to be expanded and the influence of slope determined. Hydraulic properties of cover material and solid waste need to be determined. An extensive review of existing water accounting and routing methods, culminating in a method for leachate volume prediction and its verification is needed.

CHARACTERISTICS

The compositions of leachates reported in the literature are quite diverse. Ranges of specific chemical characteristics of those studies listed in Table 11 are typical. The breadth of reported data are also typical for individual studies[17] over a long period of time. The many factors that contribute to the spread of data are time since deposition of the solid waste; the moisture regimen, such as total volume, distribution, intensity, and duration; solid waste characteristics; temperature; and sampling and analytical methods. Other factors such as landfill geometry and interaction of leachate with its environment prior to sample collection also contribute to the spread of data. Most of these factors are rarely defined in the literature, making interpretation and comparison with other studies difficult, if not rather arbitrary.

Some comments regarding the studies tabulated are warranted. Cursory examination of Table 11 indicates leachate is generally high in organic content (BOD_5 =10,000, COD =15,000, TOC =5,700) and total solids (>1 percent), is slightly acid (pH 5.0±1.0) and contains low heavy metals (<1.0 ppm) except for iron which is commonly present in levels of 1,000 ppm.

The data presented from the Solid and Hazardous Waste Research Laboratory were obtained at the Boone County Field Site Test Cell 1 which contains 395 tons of municipal solid waste compared to 592 kg/m^3 (1,000 lb/yd^3. Test Cell 1 was constructed in 1971 in accordance with best available sanitary landfill technology at that time.

The data reported from the University of Illinois were obtained under U.S. EPA Research Contract 68-02-0162. The leachate was generated from a laboratory lysimeter, 1.22 m diameter (4 feet), containing 1,520 kg (3,358 pounds) of shredded solid waste, 38 mm (1.5 inch) grate opening, compacted to 330 kg/m^3 (556.9 lb/yd^3). Water was

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TABLE 11

LEACHATE COMPOSITION

CONSTITUENT	SHWRL[1]	U OF ILL.[2]	DREXEL[3]	GA. TECH[4]	GA. TECH RECIRCULATE	OLIN AVE LANDFILL[5]	DUPAGE[6] DM61	DUPAGE DM63	DUPAGE LW5B	DUPAGE LW6B	WINNETKA[6] LW17	WINNETKA LW13	WINNETKA LW5B	RIVERSIDE[7] BIN#1
COD	16,000-22,000	45,000-71,000	1,000-51,000	4,320-12,000	4,280-9,280	2,700-10,650	360	2,940	8,000	40	4260	153	415	
BOD_5	7,500-10,000			2,500-11,000	2,750-6,900	1,550-8,450	125*	4,560*	14,080*	225*	6400*	105*	250*	81-33,100
TOC	13,840	14,000-28,000		1,230-5,000	256-2,798				6.3	7.0				5.6-7.6
pH	5.2-6.4	4.5-6.0	3.7-8.5	5.2-5.6	4.8-5.4	5.9-8.1								
TS	10,000-14,000	34,000	0-42,000	2,442-12,500	1,627-6,918	4,029-7,790								
TLS	10,000-14,000	15,600-23,000					1,104	5,910	6,794	1198	2306	584	594	
TSS	100-700	139	10-26,300	34-610	12-385									
SPECIFIC CONDUCTANCE	6,000-9,000	9,400-16,800												
ALKALINITY ($CaCO_3$)	800-4,000		0-9,700	558-2,280	302-1,370	1800	1630	4,720	5,810	2250	3370	1450	3080	730-9500
HARDNESS ($CaCO_3$)	3,500-5,000		0-5,500	450-1,940	370-1,040	1400	690+	2250+	2200+	540+	1180+	710+	970+	650-8120
TOTAL-P	25-35	20-100	0-130	2.8-26	0.65-22	12	0.5	0.17	1.2	8.9	2.6	1.3	2.75	0.2-29
ORTHO-P	23-33	6.5												0.2-890
NH-N	247.7	400-1,000	0-482	56-187	68-114		0.14Δ	0.5Δ	0.7	1.6Δ	0.5Δ	0.2Δ	0.43Δ	
NO_3+NO_2-N	0.2-0.8	0.5-10.0Δ												
Ca	900-1,700	2,300-4,000		125-750	60-435	80	156	447	308	102	100	109	109	115-2570
Cl	600-800	1,500-2,500	4.7-2,340	98-385	91-248	400	205	946	1330	135	429	70	701	96-2350
Na	450-500	750-1,600	0-7,700	64-143	62-109		63	615	810	74	298	34	348	85-1205
K	295-310	860-2,300					85	220	610	100	168	39	220	28-1860
SULFATE	400-650	900-1,550	25-450	81-156	12-138		1	1	2	2	13.6	5	1	
Mn	75-125			3-10	4-65	20	0.24	0.09	0.00	0.04	1.14	0.2	0.1	
Mg	160-250	530-1,100		26-75	17-63		110	725	450	90	198	75	200	64-410
Fe	220-325	700-2,200	0-1716	9-95	4-110	330	106	12	6.3	0.6	96	11	13.6	6.5-305
Zn	10-30	104	0-167	74	<0.05	5.5	0.10	0.05	0.4	4.5	1.5	0.1	0.6	
Cu	0.5	0.5	0-9.9	0.48	<0.03		<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	
Cd	0.4			17	<0.03		BDL	BDL	BDL	BDL	BDL	BDL	BDL	
Pb	1.6	1.25		1.16	<0.10		1.0	1.0	0.5	1.0	1.0	1.0	0.5	

NOTE: All figures in Mg/l except SPECIFIC CONDUCTANCE which is measured as μmhos/cm.

* Based on 20 Day BOD

\+ Blackwell value calculated from Mg and Ca concentration

Δ Using only NO_3 analyses, NO_2 not determined

BDL = below detectable limits, for Cd = <0.05 mg/l

TABLE 11 (Continued)

CONSTITUENT	MISSION CANYON LANDFILL 3-15-68	MISSION CANYON LANDFILL 3-24-71	SONOMA, CALIF. (9) CONTROL	SONOMA, CALIF. (9) RECIRCULATE	W.VA. U.(10) CYLINDER C	MADISON, WISC. MILLED UNCOVERED	MADISON, WISC.(11) UNMILLED COVERED	RANGE OF ALL VALVES
COD	76,800	3042	26,750-33,500	89,520		71,680	16,580	40- 89,520
BOD_5	10,900	908	15,900-24,600	20,400	33,360			81- 33,360
TOC						27,700	5,906	256- 28,000
pH	5.75	7.4	4.3-5.4	4.6	5.88	5.97	5.72	3.7- 8.5
TS					59,200	55,348	7930	0- 59,200
TDS	44,900	13,409	15,190-16,890	21,010				584-44,900
TSS	172	220	128-323	238		202	192	10- 700
SPECIFIC CONDUCTANCE						16,800	2810	2810- 16,800
$CaCO_3$ ALKALINITY	9860	8677	0-5480	3050	20,850			0- 20,850
$CaCO_3$ HARDNESS	22,800	8930			10,950			0-22,800
TOTAL-P	0.24	0.65	-0.38-9.8	79.2	128	98	85	0- 130
ORTHO-P						29	85	6.5- 85
NH_3-N	0	270	.33-304	194	1,106	1028	347.4	0-1,106
NO_3+NO_2-N			3.2-4.3	4.7[4]		10.29	4.29	0.2-10.29
Ca	7200	216	1041-1700	1560	2,790	3900	572	50-7200
Cl	660	2355	1200-1300	1210	2,310	2467	474	4.7-2467
Na	767	1160	820	980	1,439	1580	330	0-7700
K	68	440	445	910	3,770	2300	900	28-3770
SO_4	1190	19			768	1558	77	1-1558
Mn								0.09- 125
Mg	15,600	8714	725-1070	560	420	1140	220	17-15,600
Fe	2820	4.75			860	1046	91	0-2820
ZN			28	95		370	13	0- 370
Cu			2.15	0.4		0.75	0.65	0- 9.9
Cd						0.375	0.05	<0.03-17
Pb			<2.0	2.0		1.65	0.80	<0.10-2.0

applied daily to bring the solid waste to field capacity within 30 days, and thereafter an equivalent of 0.89 mm/week (0.035 in) was added to generate sufficient leachate for evaluation of leachate treatment methods.

Leachate data reported by Drexel were obtained under U.S. EPA Research Grant R800777. The solid waste used was not processed; the laboratory column was 1.83 m (6 feet) square, and initially was packed 2.44 m (8 feet) deep.[29]

Data presented from Georgia Institute of Technology are from ongoing U.S. EPA Research Grant, R801397, to investigate the feasibility of recirculating leachate back through the landfill as a treatment method. Again, the simulated landfill conditions were utilized for study purposes. The refuse was compacted into a 3.05 m x .915 m (10 ft x 3 ft) column in two 1.52 m (5 ft) lifts to a dry density of about 318 kg/m^3 (535 lb/yd^3). To expedite the production of leachate, 948 l (250 gal) of tap water were added after placement of the soil cover. A more detailed description of the project and the results to date are provided by Pohland and Mao.[18]

The Olin Avenue data are from the work done at the University of Wisconsin to determine the treatability of leachate using a variety of classical treatment methods. From this work the concept of anaerobic digestion followed by aerobic polishing was formulated and led directly to the pilot plant studies that are still ongoing at a landfill in the Milwaukee area. The original laboratory scale studies are available in a progress report for the period June 1, 1970 to August 31, 1971, entitled "The Treatability of Leachate from Sanitary Landfills,"[19] authored by R. K. Ham under U.S. EPA Research Grant R801814.

The DuPage and Winnetka data were taken from Hydrogeology of Solid Waste Disposal Sites in Northeastern Illinois.[20] Specifically, groundwater quality in the near vicinity of sanitary landfills is represented.

Research was conducted by Merz[21] at Riverside, California, in a test bin. This represents the earliest extensive leachate study. The basic constituents identified appear to agree with the most recent analysis of leachate.

The data presented for the Mission Canyon landfill[22] gives a good indication of the age factor when analyzing leachate. As can be seen the early leachate analyses reported high COD and BOD_5 values very typical of fresh leachates. Some 3 years later the effects of age and materials are noted in the low COD--all readily oxidizable organics

already removed--and the BOD_5 is also low indicating material remaining is not conducive to biological degradation or that not much material is even there. The significant increase in chlorides, sodium, and potassium over a 3-year period compared to decreases in all other parameters is interesting but unexplained.

The Sonoma data[23] are from an ongoing Demonstration Grant 1-GOG-EC-00351. Conclusions have not as yet been made; the data are presented to indicate the effect of high moisture throughput on leachate characteristics.

Another study using shredded solid waste, but in combination with unshredded solid waste, was conducted by Qasim and Burchinal.[24]

Chian, et al.[25] reported on the characteristics of two leachate samples provided by R. K. Ham at the University of Wisconsin. The increased solid waste surface area and lack of cover material has a drastic effect on increasing the quantities of materials leached.

A detailed characterization study by Chian and DeWalle[25] showed that the majority, 78 percent, of the organic matter in fresh leachate, Figure 6, consisted of low molecular weight compounds, 78 percent of which was free volatile fatty acids; also, significant amounts of the heavy metals were chelated by the humic carbohydrate-like large molecules and the fulvic acid small molecules. Leachate from an old landfill, Figure 7, was found to consist almost entirely of small molecules of fulvic acid and humic-like materials capable of chelating heavy metals; no free volatile fatty acids were detected.

Heavy metal concentrations listed in Table 11 are generally less than 1 mg/l. Table 12 indicates the solubility of several heavy metal salts in water. The extent to which a substance will dissolve in another varies greatly with different substances and depends on the nature of the solute (in this case the heavy metals) and solvent, the temperature, and the pressure. In general, the effect of pressure on solubility is small unless gases are involved. However, the effect of temperature is usually very pronounced, as can be seen from the table of solubilities for various temperatures.

In general, compounds of similar chemical character are more readily soluble in each other than are those whose chemical character is entirely different. As presented in the table, inorganic materials are dissolved in water, some to a large extent because of chemical similarities, some hardly at all due to vast chemical differences.

The substances presented in the table were selected as representative of those that have the highest interest at this time. The tabulated data are for water at or near pH 7.0. When applying these figures

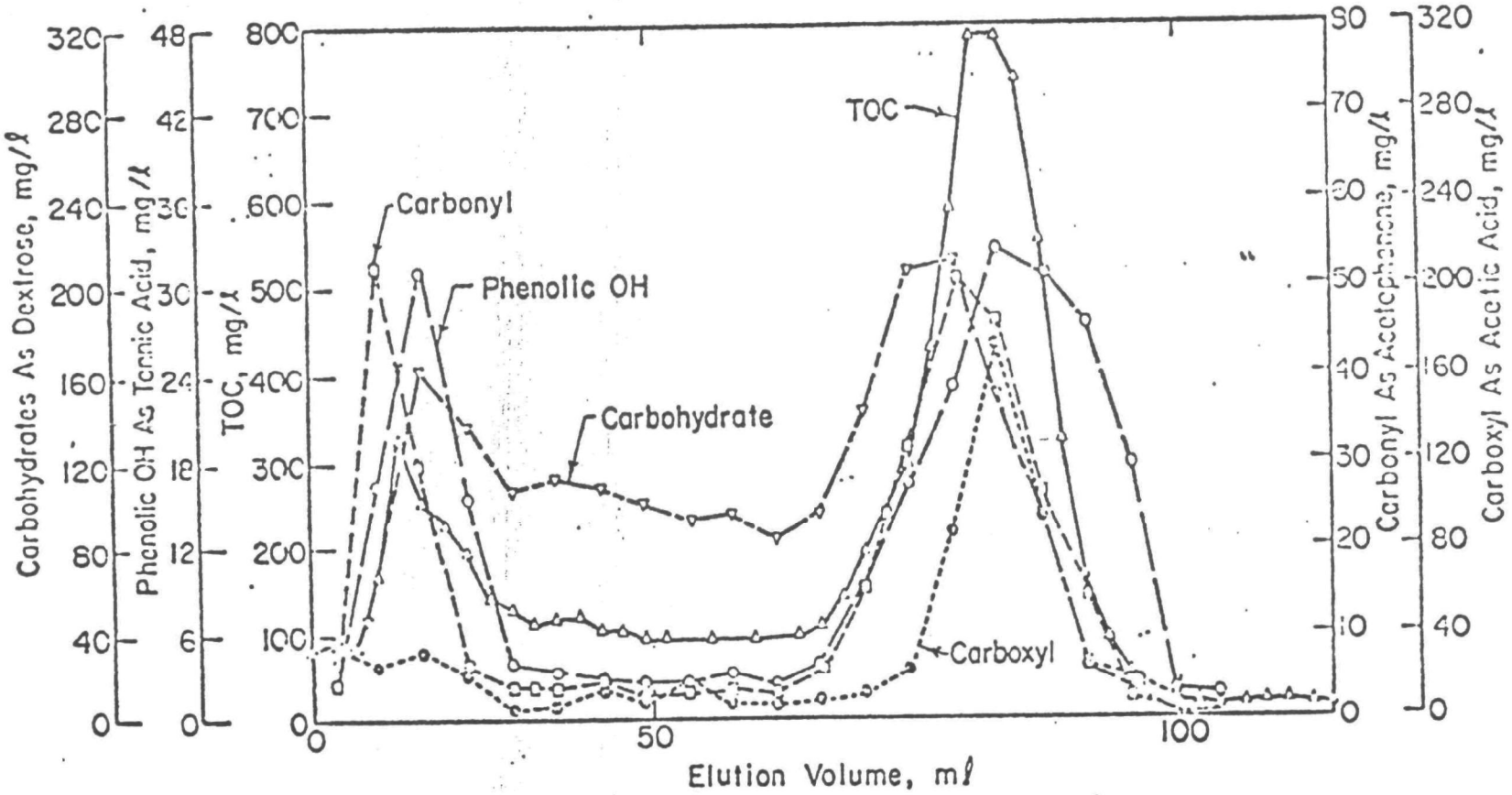


Figure 6. ELUATE OF THE 500 mw ULTRAFILTRATION RETENTATE OF A YOUNG LEACHATE ON A G-75 SEPHADEX COLUMN.

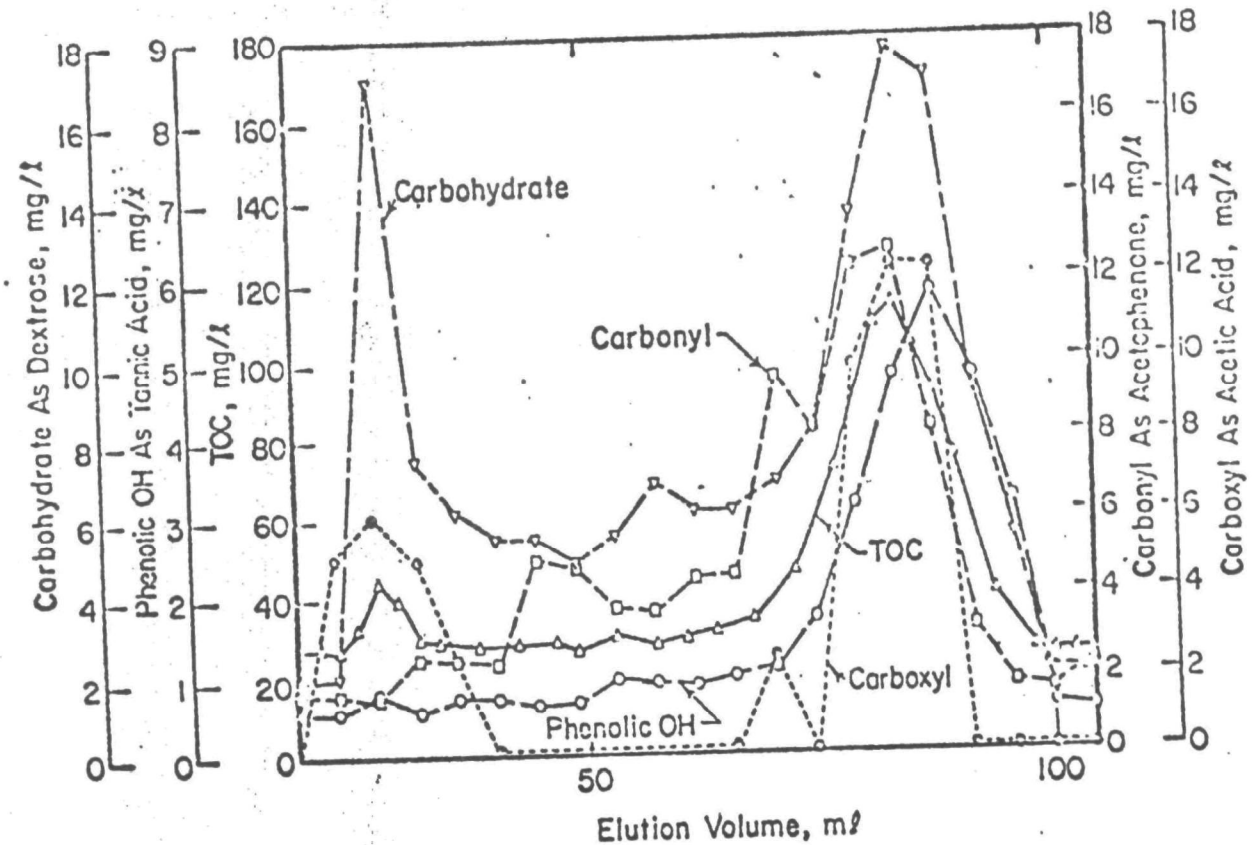


Figure 7. EL ATE OF THE 500 mw ULTRAFILTRATION RETENTATE OF AN OLD LEACHATE ON A G-75 SEPHADEX COLUMN.

Table 12. SOLUBILITIES OF SELECTED HEAVY METAL COMPOUNDS[a,b]

Substance	TEMPERATURE 0 C	20 C	40 C	100 C
$CdCl_2$	90.0	134.5	135.3	147.0
$CdSO_4$	76.48	76.60	78.54	60.77
$CuCl_2.SH_2O$	70.7	77.0	83.8	107.9
$CuSO_4.5H_2O$	14.3	20.7	28.5	75.4
$FeCl_3$	74.4	91.8	--	535.7
$FeSO_4.7H_2O$	15.65	26.5	40.2	--
$PbCl_2$	0.6728	0.99	1.45	3.34
$PbSO_4$	0.0028	0.0041	0.0056	--
$MgCl.6H_2O$	52.8	54.5	57.5	73.0
$MgSO_4.7H_2O$	--	35.5	45.6	--
$HgCl_2$	3.6[c]	--	--	61.3[c]
$ZnCl_2$	--	432.0[c,d]		615.0[c]
$ZnSO_4 17H_2O$	41.9	54.4	--	--

[a]This table shows the amount of substance (anhydrous) which is soluble in 100g of water at the temperature in degrees centigrade.

[b]Adapted from Perry's Chemical Engineers' Handbook, Fourth Edition, McGraw-Hill, New York. 1950.

[c]Parts by weight of substance soluble in 100 parts by weight in water.

[d]Measured at 25 C.

to leachate some variances should and will be expected. In leachate the pH is usually toward the acidic side of the scale, pH 5.0 to 5.5, and thereby significantly affecting microbial and chemical reactions acting on the heavy metals. Ionic and nonionic materials also influence the overall solubility scheme of a particular metal. Additionally, the dissolved solids along with the potential buffering effect of the carbonate-carbonic acid and the volatile acid systems further increase the likelihood for deviation from the pure water system. Thus, it can be stated that the table gives an indication of the solubility of certain metal compounds in water at different temperatures and some inferences can be made from them regarding heavy metal concentrations in leachate.

Extensive bacteriologic content of leachate is not available. Fecal coliform and fecal streptococci data from the Boone County Field Site are presented in Figures 8 and 9. Unpublished data[26] obtained from the Illinois laboratory study[25] indicate a similar trend of high numbers of initial fecal coliform and fecal streptococci, followed by a gradual, but definitive decline to very low numbers of bacteria. Peterson[27] has reported isolation of poliovirus from a laboratory landfill. Thus the public health importance of leachate discharge to a stream is further supported, since an operational landfill will continually receive new pathogen-containing solid waste.

The relative environmental significance of leachate is difficult to determine on a national basis due to the specificity of site conditions that control the moisture regimen and hence, leachate and gas production. The following hypothetical example is offered only for illustration; it represents a typical case east of the Mississippi River (Figure 4). It does not represent a worst case condition. Pierau[14] measured leachate production at 0.9 mm/day (12 in/year). A rule of thumb for annual utilization of landfill space is 1.85 m/ha/10,000 population (15 acre feet per 10,000 population). Landfill depth of 4.56 m (15 ft) is not unusual. Leachate volume then is 123 l/cap/yr (32.6 gal/cap/yr). If one assumes a solid waste generation rate of 970 kg/cap/yr (2,000 lb/cap/yr), then from Rovers and Farquhar,[28] the annual per capita extraction of materials from solid waste initially at 30 percent moisture by wet weight would be 2.58 - 7.8 kg (5.7 to 17.2 lb) COD, 16.1 - 33 kg (35.6 to 72.8 lb) BOD_5, 0.137 - 2.6 kg (0.3 to 5.7 lb) chloride, 0.0318 - 1.0 kg (.07 to 2.2 lb) ammonia nitrogen, 0.0318 - 0.59 (.07 to 1.3 lb) organic nitrogen, and 0.136 - 1.31 kg (0.3 to 2.9 lb) sulfate. The duration of this rate of extraction is not known but would eventually decrease.

The above ranges of leached material quantities were determined experimentally over finite time periods. Fungaroli and Steiner[29] have indicated a relatively constant leaching phenomenon; the quantity

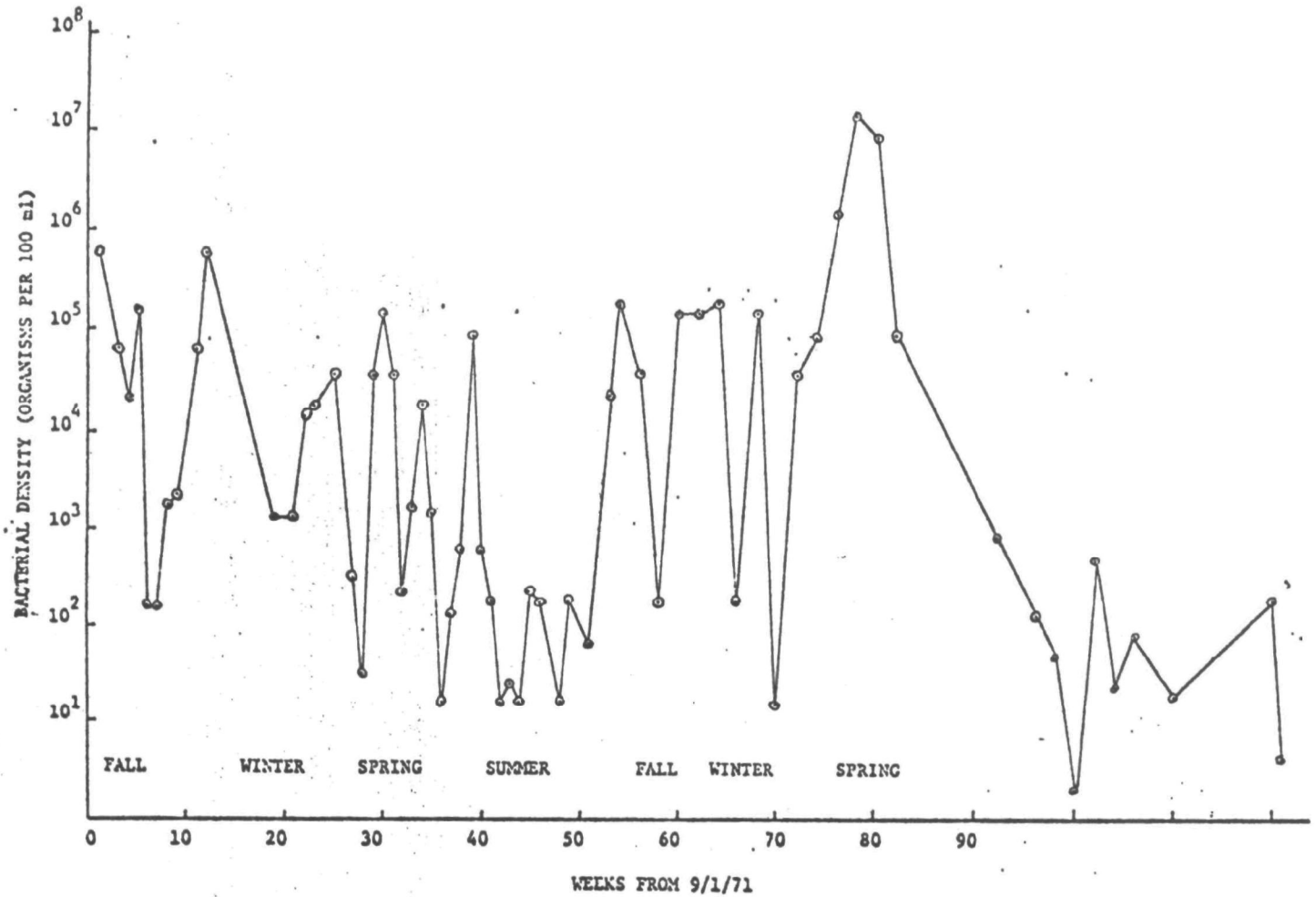


Figure 9. FECAL STREPTOCOCCI IN LEACHATE FROM THE UPPER PIPE OF THE BOONE COUNTY FIELD SITE, TEST CELL 1.

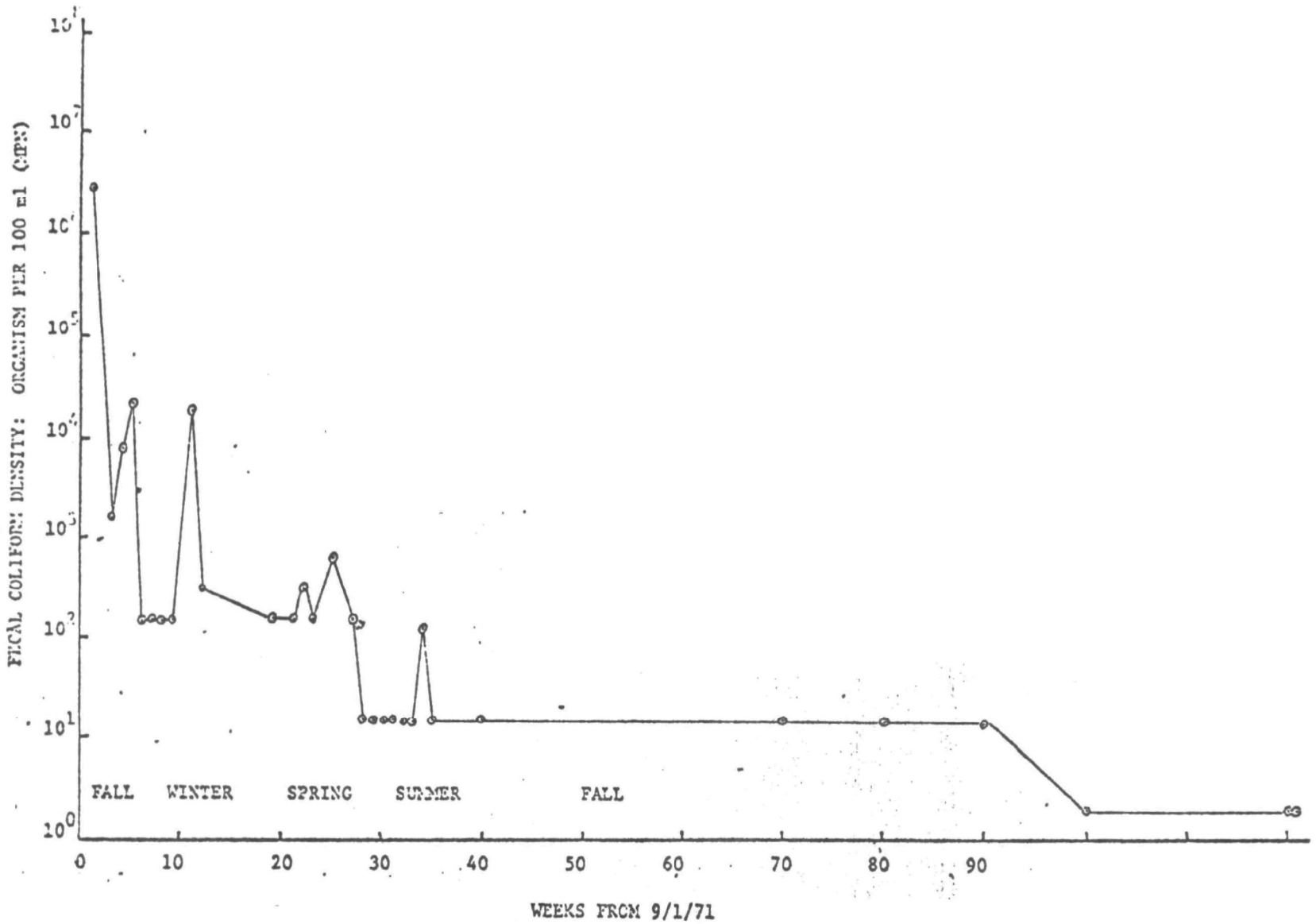
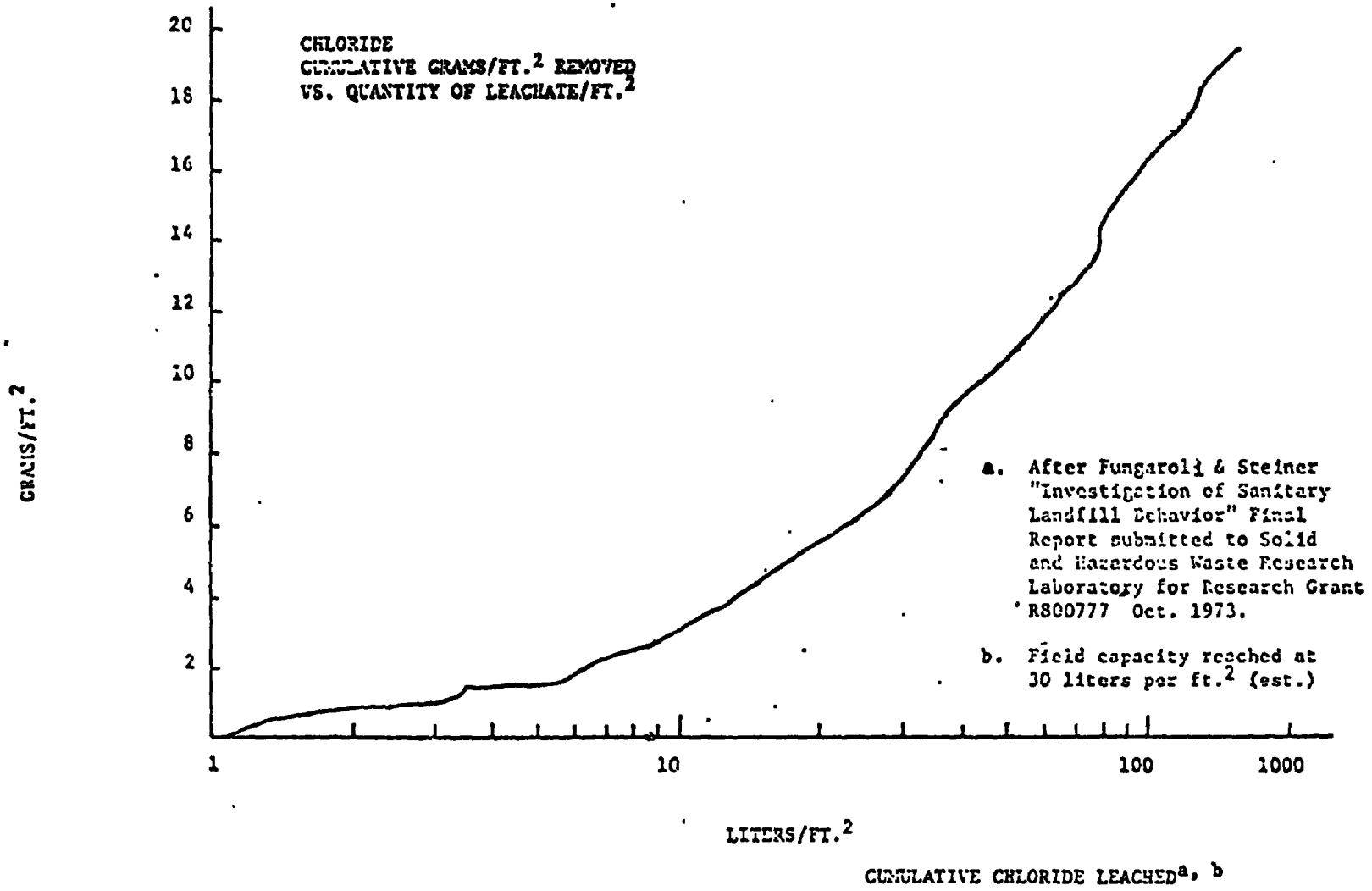


FIGURE 8. FECAL COLIFORM ISOLATED FROM THE UPPER PIPE OF THE BOONE COUNTY FIELD SITE, TEST CELL 1



CUMULATIVE CHLORIDE LEACHED[a, b]

Figure 10.

of materials leached per unit surface area is related to the leaching volume per unit surface area. Typical results are shown in Figures 10, 11, and 12. The duration of this study indicates sanitary landfills have a long-term effect on the environment.

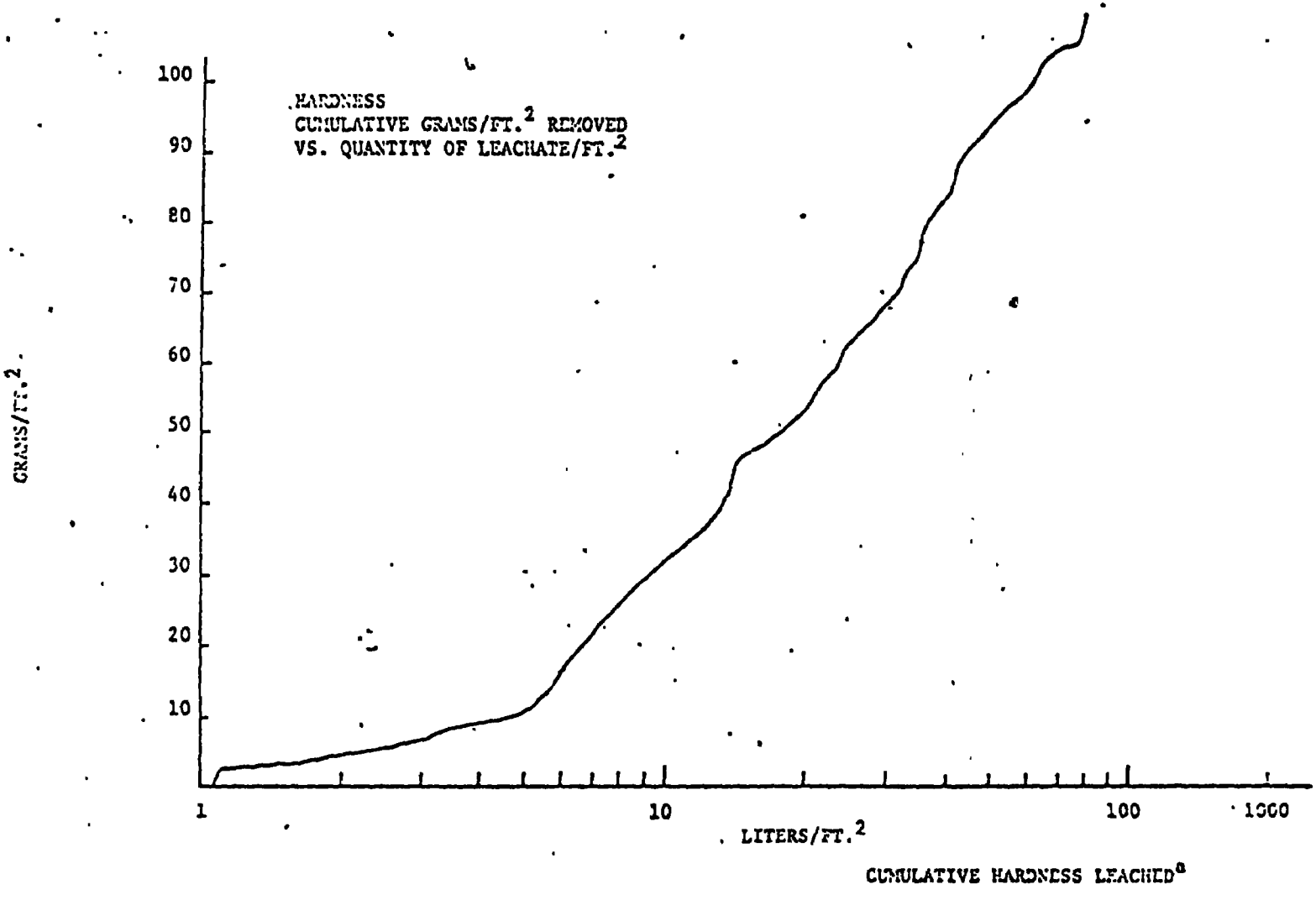

HARDNESS
CUMULATIVE GRAMS/FT.^{2} REMOVED
VS. QUANTITY OF LEACHATE/FT.^{2}
GRAMS/FT.^{2}
100
90
80
70
60
50
40
30
20
10
1
10
100
1000
LITERS/FT.^{2}
CUMULATIVE HARDNESS LEACHED[a]

Figure 11.

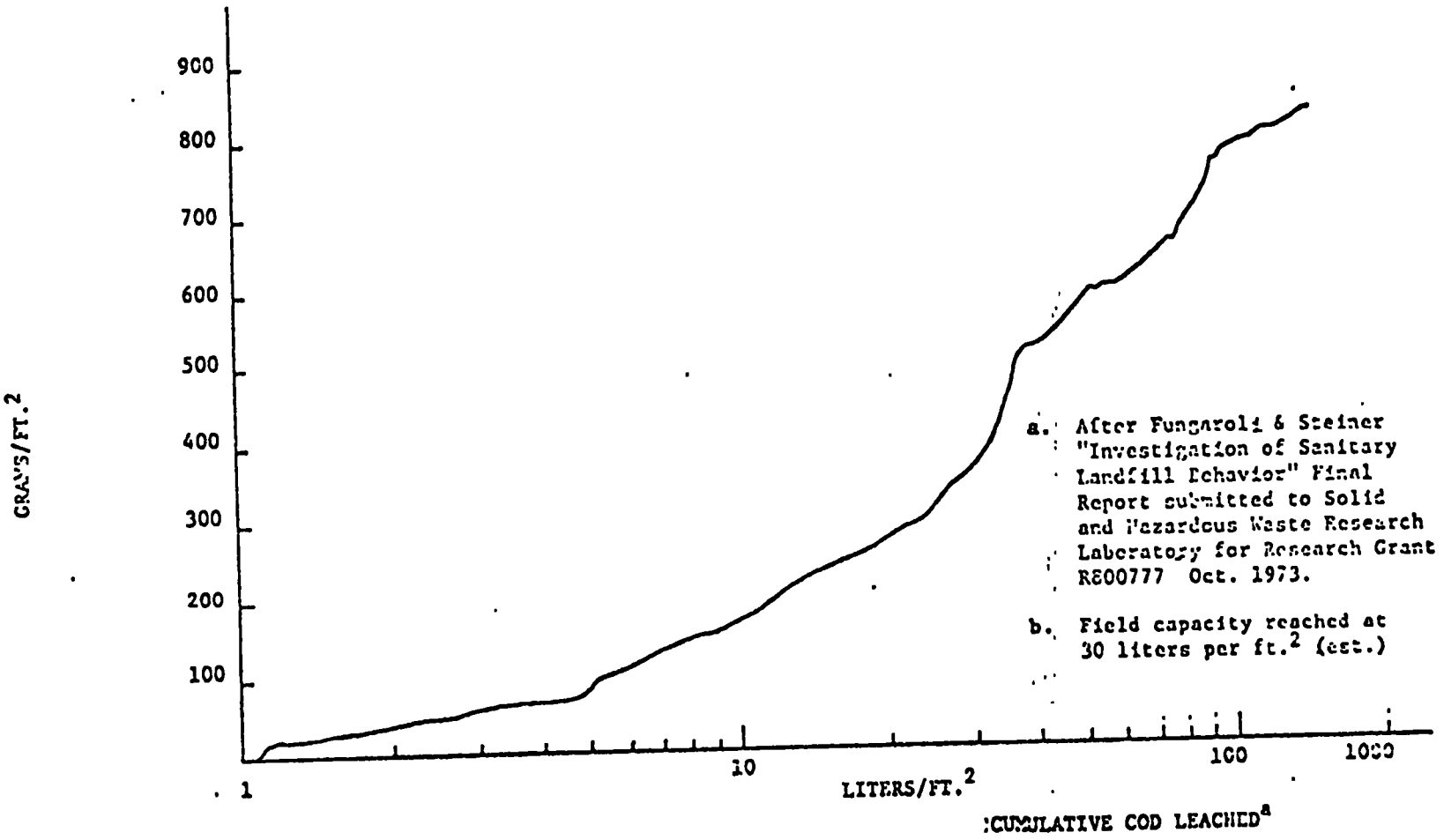

900
800
700
600
500
400
300
200
100
GRAMS/FT.2
1
10
100
1000
LITERS/FT.2
CUMULATIVE COD LEACHED[a]
a. After Fungaroli & Steiner "Investigation of Sanitary Landfill Behavior" Final Report submitted to Solid and Hazardous Waste Research Laboratory for Research Grant R800777 Oct. 1973.
b. Field capacity reached at 30 liters per ft.2 (est.)

Figure 12.

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The following section has been reproduced from:

"USE OF THE WATER BALANCE METHOD FOR PREDICTING LEACHATE GENERATION FROM SOLID WASTE DISPOSAL SITES", OSWMP, U.S. EPA, October 1975, (EPA/530/SW-168)

TABLE 3

RUNOFF COEFFICIENTS*

Surface conditions	Runoff coefficient
Grass cover:	
Sandy soil, flat, 2%	0.05 - 0.10
Sandy soil, average, 2-7%	0.10 - 0.15
Sandy soil, steep, 7%	0.15 - 0.20
Heavy soil, flat, 2%	0.13 - 0.17
Heavy soil, average, 2-7%	0.18 - 0.22
Heavy soil, steep, 7%	0.25 - 0.35

* Chow, V. T., ed. Handbook of applied hydrology; a compendium of water resources technology. New York, McGraw-Hill, [1964]. 1v. (various pagings).

Water Balance Calculations for a Sanitary Landfill

As shown in Figure 2, the water routing through a sanitary landfill basically consists of two phases--routing through the soil cover and routing through the compacted solid waste beneath. The soil cover is that phase which interfaces directly with the atmosphere and will determine the amount of infiltration into the soil and percolation into the solid waste. The solid waste phase and its attendant moisture storage capacity will determine the quality and time of first appearance of the leachate. Therefore, a water balance can be performed on the soil cover phase to determine the amount of percolation. The solid waste phase can then be analyzed in relation to the percolation amounts to determine the extent of potential leachate problems.

Treating the moisture regime of the soil cover as a one dimensional system, the water balance method can be used to calculate the percolation of water into the solid waste. In applying the method, the surface conditions of the sanitary landfill site must be well defined. The type and thickness of the cover soil, the presence or absence and type of vegetative cover, and the topographical features are the primary surface conditions that will affect percolation.

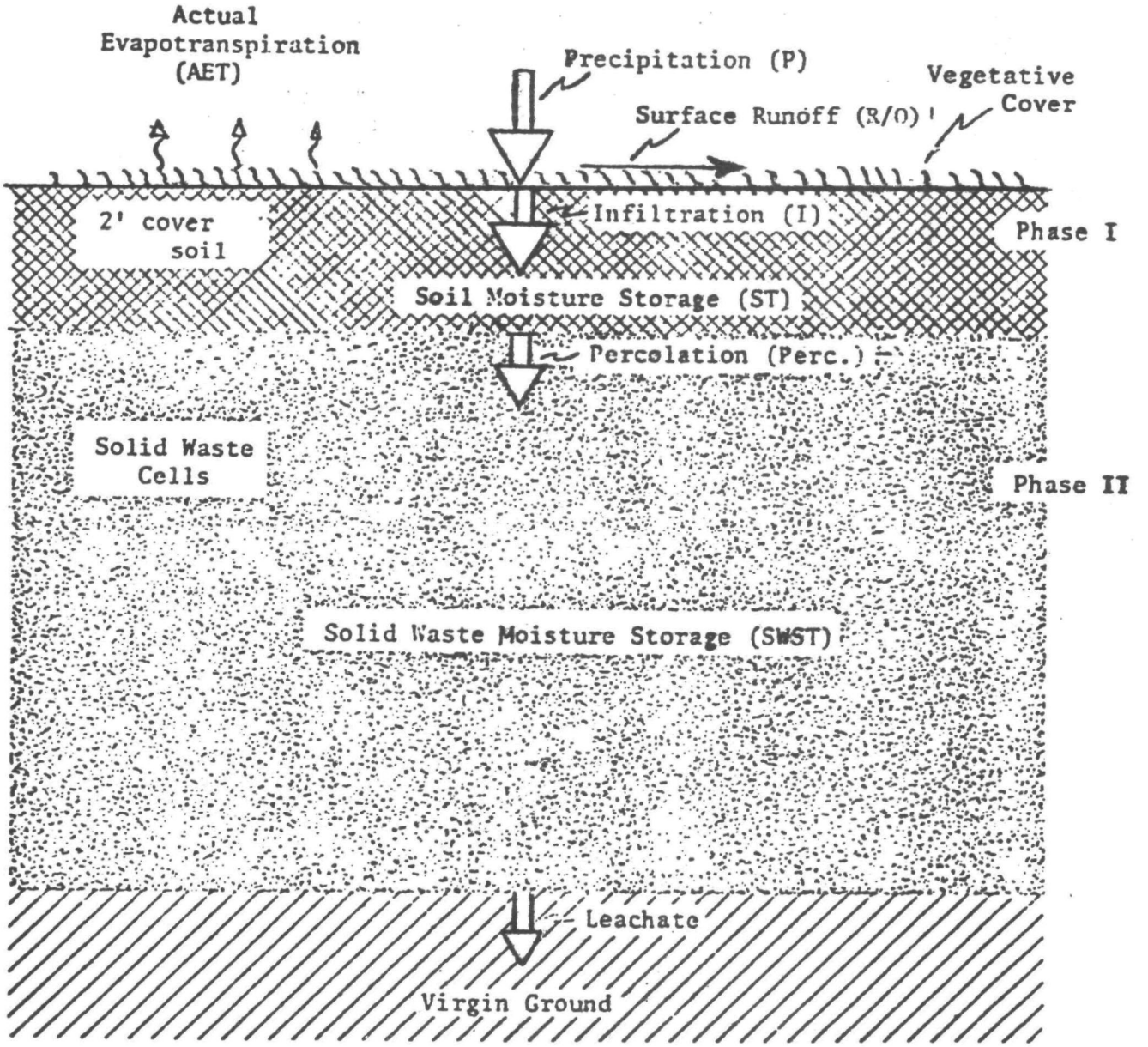


Figure 2. Sanitary Landfill Water Balance

To best illustrate the water balance of a sanitary landfill, three case studies have been selected to reflect various climatic and soil conditions. Cincinnati, Ohio, was selected to represent a humid climate with a sandy type soil; Orlando, Florida, to represent a humid climate with a sandy type soil; and Los Angeles, California, to represent a dry climate with a fine grained soil.

Conditions will vary among sites and among the stages of a given site's life. These conditions must be considered in applying the water balance method. For illustrative purposes, the water balance analysis was simplified by the following basic assumptions:

1. The landfill has been completed with 0.6 meters (2 feet) of final cover and graded with a 2 to 4 percent slope over most of the surface area.

2. The solid waste, cover soil, and vegetative cover were emplaced instantaneously at the beginning of the first month of the computation initiation. Practically speaking, this ignores any percolation that may occur prior to the placement of the final cover soil.

3. The final use of the site is an open green area to be used for recreation or pasture.

4. The surface is fully vegetated with a moderately deep-rooted grass, the roots of which draw water directly from all parts of the soil cover but not from the underlying solid waste.

5. The sole source of infiltration is precipitation falling directly on the landfill's surface. All surface runoff from adjacent drainage areas is diverted around the landfill surface. All ground water infiltration is prevented through proper site selection and design.

6. The hydraulic characteristics of the soil cover and compacted solid waste are uniform in all directions.

7. The depth of the landfill is much less than its horizontal extent. Thus, all water movement is vertically downward.

The water balances for the three case studies are presented and depicted in Tables 4, 5, and 6 and Figures 3, 4, and 5 for Cincinnati, Orlando, and Los Angeles respectively. In order to fully understand the calculations and manipulations involved in the water balance procedure, refer to the Appendix which presents the basic calculations, a discussion of each of the parameters and their manipulations, and copies of the three soil moisture retention tables used in the calculations.

TABLE 4

WATER BALANCE DATA FOR CINCINNATI, OHIO

Parameter *	J	F	M	A	M	J	J	A	S	O	N	D	Annual
PET	0	2	17	50	102	134	155	138	97	51	17	3	766
P	80	76	89	82	100	106	97	90	73	65	83	84	1025
$C_{R/O}$	0.17	0.17	0.17	0.17	0.17	0.13	0.13	0.13	0.13	0.13	0.13	0.17	
R/O	14	13	15	14	17	14	13	12	9	8	11	14	154
I	66	63	75	68	83	92	84	78	64	57	72	70	872
I-PET	+66	+61	+58	+18	-19	-42	-71	-60	-33	+6	+55	+67	+106
Σ NEG (I-PET)				(0)	-19	-61	-132	-192	-225				
ST (Table C)	150	150	150	150	131	99	61	41	33	39	94	150	
Δ ST	0	0	0	0	-19	-32	-38	-20	-8	+6	+55	+56	
AET	0	2	17	50	102	124	122	98	72	51	17	3	658
PERC	+66	+61	+57	+18	0	0	0	0	0	0	0	+11	213

*The parameters are as follows: PET, potential evapotranspiration; P, precipitation; $C_{R/O}$ surface runoff coefficient; R/O, surface runoff; I, infiltration; ST, soil moisture storage; Δ ST, change in storage; AET, actual evapotranspiration; PERC, percolation. All values are in millimeters (1 inch = 25.4 mm). See Appendix for discussion of parameters.

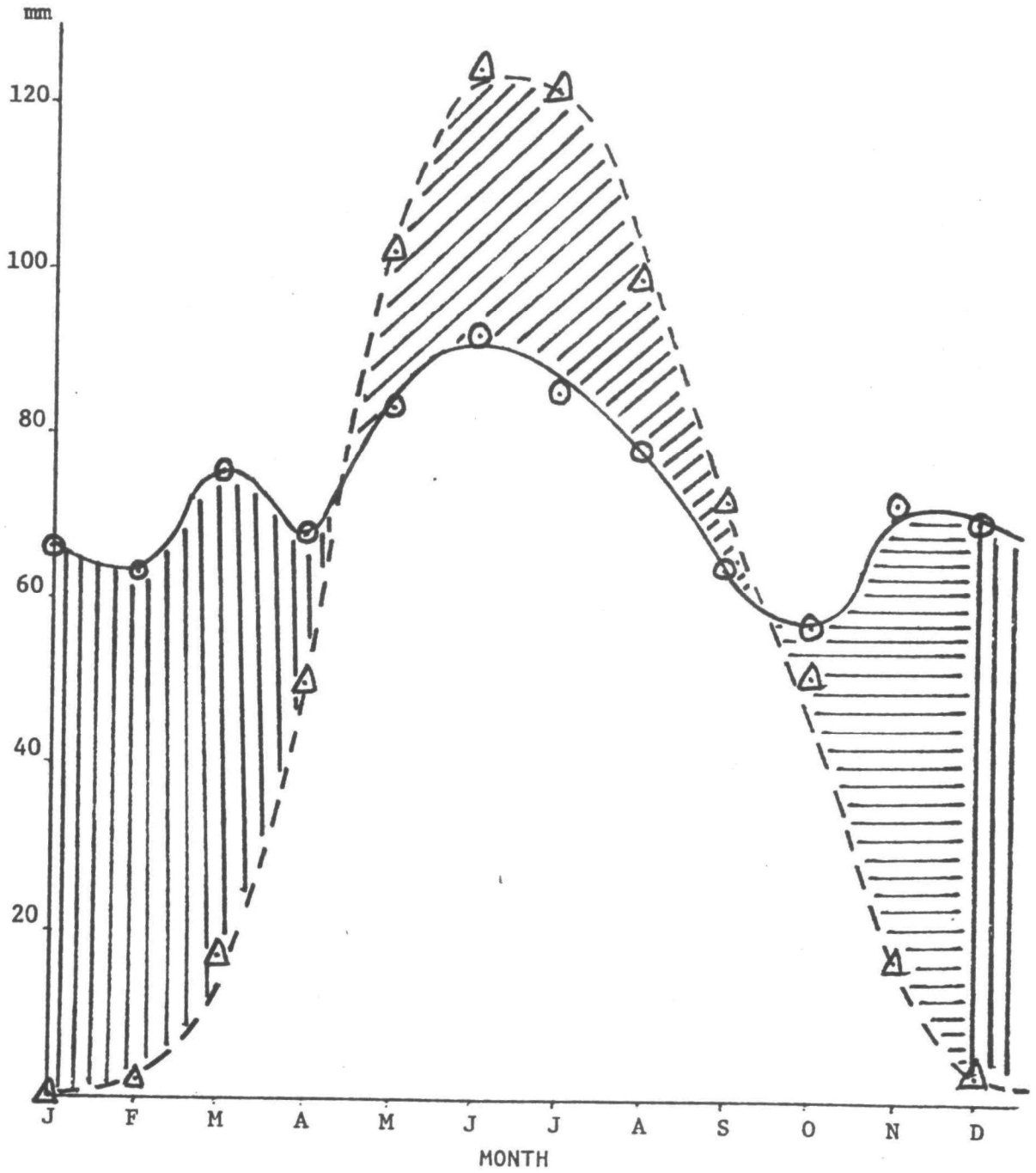


Figure 3. Water Balance for Cincinnati, Ohio

||||||| Percolation

Soil Moisture Recharge

///// Soil Moisture Utilization

θ——θ Infiltration

Δ----Δ Actual Evapotranspiration

TABLE 5

WATER BALANCE DATA FOR ORLANDO, FLORIDA

Parameter*	J	F	M	A	M	J	J	A	S	O	N	D	Annual
PET	33	39	59	90	140	167	175	173	142	100	53	35	1206
P	50	56	91	88	81	161	230	180	200	121	39	45	1342
$C_{R/O}$	.075	.075	.075	.075	.075	.075	.075	.075	.075	.075	.075	.075	
R/O	4	4	7	6	6	13	17	13	15	9	3	3	100
I	46	52	84	82	75	148	213	167	185	112	36	42	1243
I-PET	+13	+13	+25	-8	-65	-19	+38	-6	+43	+12	-17	+7	36
Σ NEG (I-PET)			(0)	-8	-73	-92	-25+	-31			-17		
ST (Table A)	100	100	100	92	47	39	77	73	100	100	84	91	
Δ ST	+9	0	0	-8	-45	-8	+38	-4	+27	0	-16	+7	
AET	33	39	59	90	120	156	175	171	142	100	52	35	1172
PERC	+4	13	25	0	0	0	0	0	16	12	0	0	70

*See footnote, Table 4.

+The situation where a positive I-PET value occurs between two negative values is a special case. Here, ST is found by direct addition of I-PET to the preceding ST. The Σ NEG (I-PET) value is then found from the soil moisture retention table for the ST value.

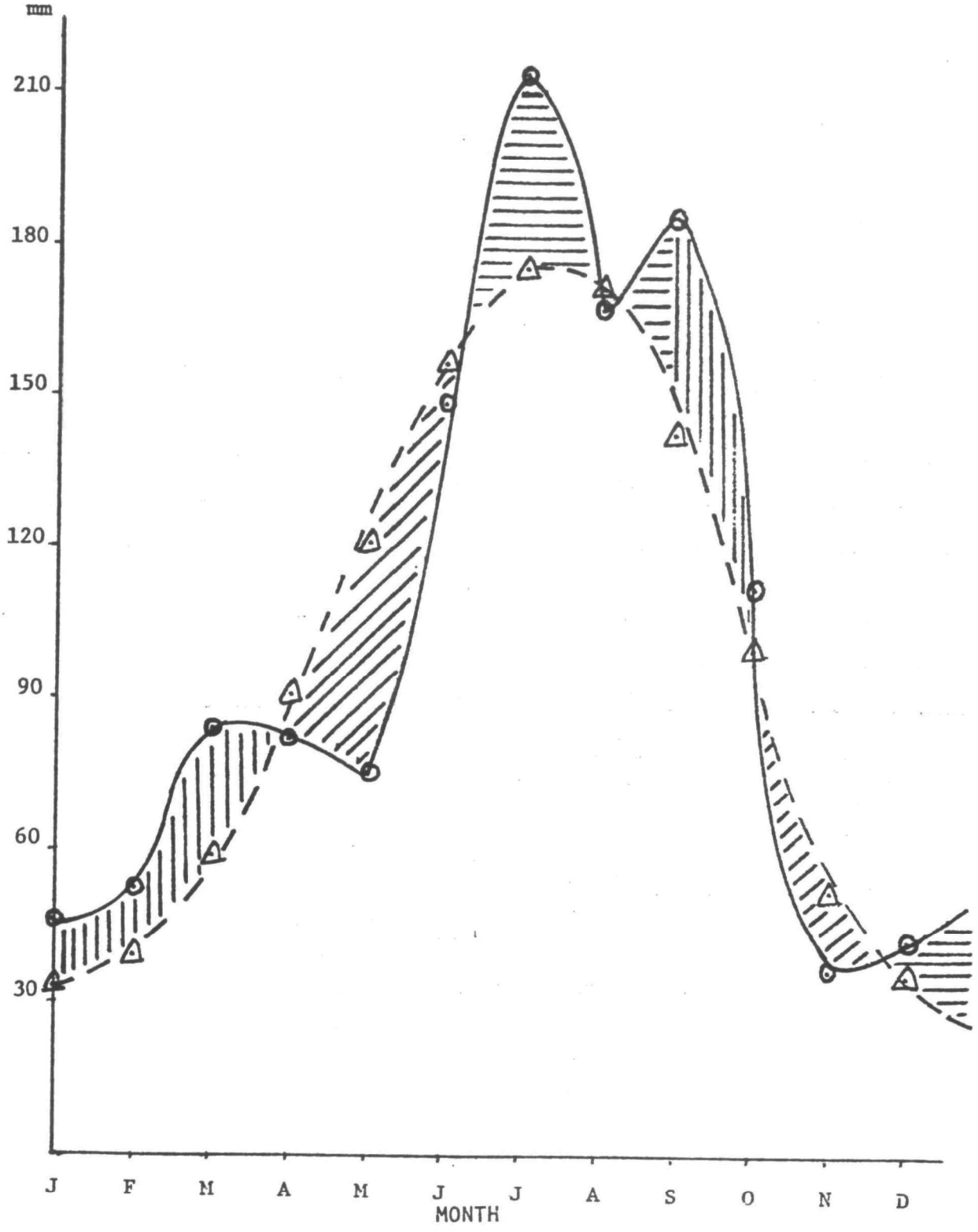


Figure 4. Water Balance for Orlando, Florida

Percolation

Soil Moisture Recharge

Soil Moisture Utilization

θ——θ Infiltration

Δ-----Δ Actual Evapotranspiration

TABLE 6

WATER BALANCE DATA FOR LOS ANGELES, CALIFORNIA

Parameter*	J	F	M	A	M	J	J	A	S	O	N	D	Annual
PET	34	36	49	59	76	94	117	115	96	73	52	39	840
P	78	79	66	27	9	2	0	1	5	14	29	68	378
$C_{R/O}$	0.15	0.15	0.15	0	0	0	0	0	0	0	0	0.15	
R/O	12	12	10	0	0	0	0	0	0	0	0	10	44
I	66	67	56	27	9	2	0	1	5	14	29	58	334
I-PET	+32	+31	+7	-32	-67	-92	-117	-114	-91	-59	-23	+19	-506
Σ NEG (I-PET)			-39	-71	-138	-230	-347	-461	-552	-611	-634		
ST (Table B)	52	83	90	70	40	19	7	3	1	1	1	20	
Δ ST	+32	+31	+7	-20	-30	-21	-12	-4	-2	0	0	+19	
AET	34	36	49	47	39	23	12	5	7	14	29	39	334
PERC	0	0	0	0	0	0	0	0	0	0	0	0	0

* See footnote, Table 4.

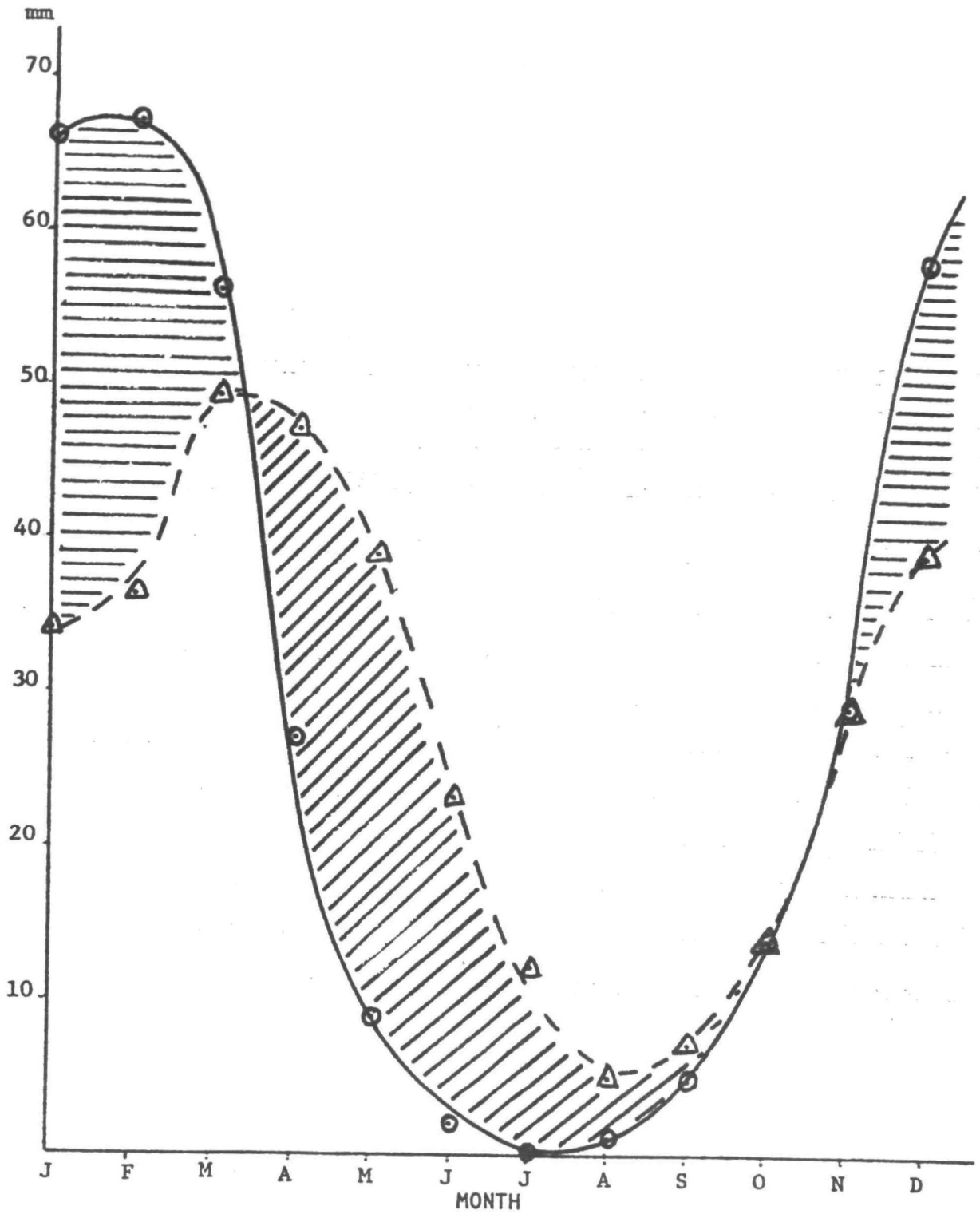


Figure 5. Water Balance for Los Angeles, California

Soil Moisture Recharge

Soil Moisture Utilization

Infiltration

Actual Evapotranspiration

Table 7 presents a summary of the water balances for the three case studies. As expected, the locations in the humid areas experienced percolation while the dry location experienced no significant percolation. It is interesting to note that all three cases are characterized by at least one wet season and one dry season during the one-year cycle. However, only in the humid areas is the precipitation sufficiently greater than the evapotranspiration to exceed the soil moisture storage capacity and produce percolation.

The fluctuating nature of percolation during the one-year cycle is an interesting phenomena to analyze. For example, examine the percolation in Cincinnati. During the dormant season (December to April), little or no evapotranspiration occurs, resulting in a high soil moisture content and significant amounts of percolation. During the growing season (May to September), the large evapotranspiration demand utilizes all of the infiltration moisture. The effect of the soil moisture storage is clearly seen in the fall months of October and November when the infiltration exceeds the potential evapotranspiration. This excess infiltration recharges soil moisture storage, resulting in no significant percolation until December. The fluctuating nature of percolation will cause variations in leachate generation.

Leachate Generation

Knowing the amount of water that percolates through the cover material (phase I), an analysis of the water routing through the solid waste (phase II) can now be performed to determine the magnitude and timing of leachate generation (refer to Figure 2).

Like its cover material, the underlying solid waste cells (including the relatively thin layers of daily cover material) will exhibit a certain capacity to hold water. The field capacity of solid waste has been determined by many investigators to vary from 20 percent to as high as 35 percent by volume.[3,12] In other words, the field capacity would vary from about 200 mm water/meter refuse (2.4 inches/foot) to about 350 mm water/meter refuse (4.2 inches/foot). For present purposes, a value of 300 mm/meter (3.6 inches/foot) will be used.

TABLE 7

SUMMARY OF WATER BALANCE CALCULATIONS

Location	Parameters - mean annual (mm)				
	Precipitation	Runoff	Infiltration	AET	Percolation
Cincinnati, Ohio	1025	154	872	658	213
Orlando, Florida	1342	100	1243	1172	70
Los Angeles, California	378	44	334	334	0

The amount of water which can be added to the solid waste before reaching field capacity depends also on its moisture content when delivered to the landfill site. This value will vary over a wide range depending on the composition of the waste and the climate. Several analyses performed on municipal solid waste show its moisture content to range anywhere from 10 to 20 percent by volume.[3,12,13] A moisture content of 15 percent by volume or about 150 mm/m (1.8 inches/foot) will be used here. Therefore, with a field capacity of 300 mm/m and an initial moisture content of 250 mm/m the compacted waste would have an adsorbtion capacity of about 150 mm of water per meter of solid waste (1.8 inches/foot).

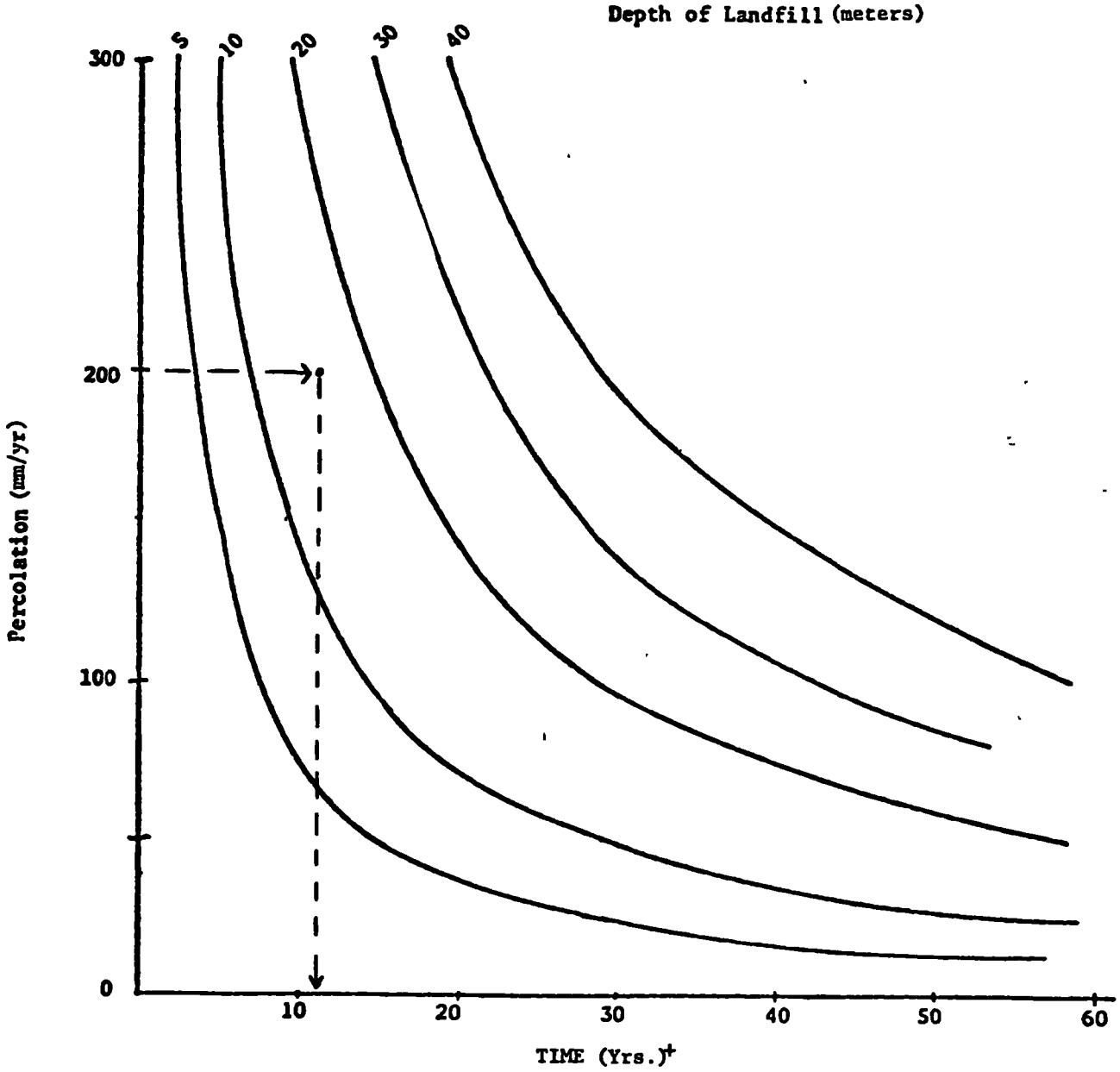
Theoretically, the water movement through a compacted solid waste cell will act like water movement through a soil layer. In other words, the field capacity of a given solid waste level must be exceeded before any significant leachate to a lower level will occur. For the examples, this means that 150 mm of percolation would have to be applied to a municipal solid waste layer one meter deep before any significant leachate would be generated from the bottom of that layer. Practically speaking, due to the heterogeneous nature of the solid waste, some channeling of water will occur causing some leaching to occur prior to attainment of field capacity. However, this amount should be small and certainly not a continuous flow and will be assumed negligible.

Employing the above concepts, one can assess the extent of the leachate problem for a given sanitary landfill site. The time of first appearance of leachate would be influenced by the landfill's depth and the leachate quantities by the landfill surface area (size). Figure 6 shows the relationship between annual percolation amounts and time of first appearance of leachate for various landfill depths. Figure 7 shows the relationship between annual percolation amounts and leachate quantities for various size landfills.

This methodology will be illustrated by application to the three case studies. Equal amounts of solid waste will be assumed for all three cases in determining the relative depths and acreage requirements at the different locations.

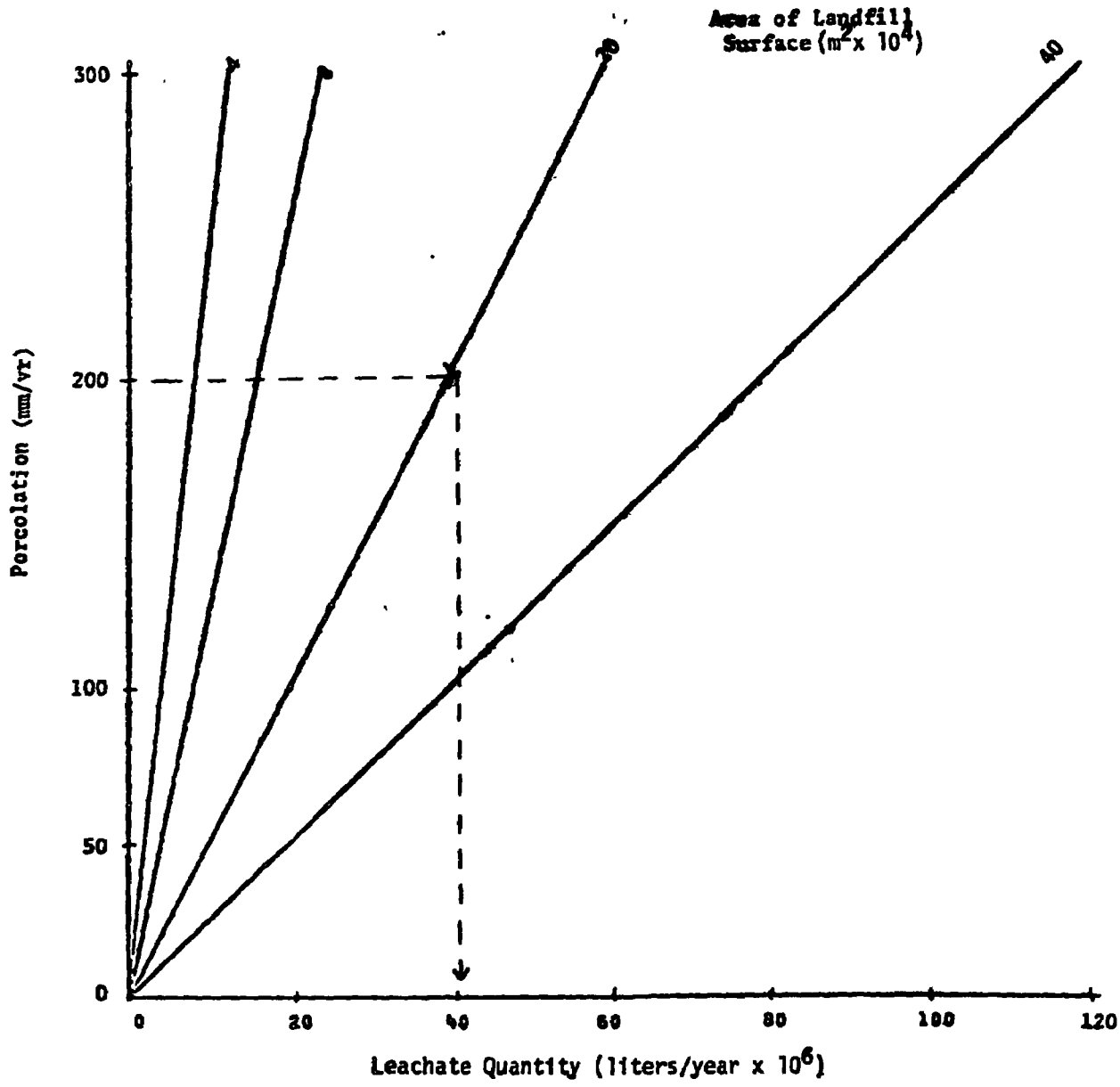
Case 1--Cincinnati, Ohio. The landfills in this location, as in most of the northern part of the country, are generally trench operations or area fills in small ravines. The depth of these operations would be expected to range between 10 and 20 meters, with the surface area usually above 50 acres (ca. $2X10^5 m^2$). A site will be assumed here with an average depth of 15 meters and a surface area of 202,000 m^2 (50 acres). Therefore, with an average annual percolation of slightly more than 200 mm

Figure 6. Time of First Appearance of Leachate *



*Based on a solid waste moisture absorption capacity of 150 mm/m.
+Time zero is defined as that time when the field capacity of the soil cover is first exceeded, producing the first amounts of percolation.

Figure 7. Annual Leachate Quantities
After Time of First Appearance



(Table 4), it would take close to 11 years (Figure 6) for significant amounts of leachate to appear at the bottom of the fill, at which time the average annual leachate quantity would be about 40 million liters (Figure 7).

Case 2--Orlando, Florida. The depth of landfills in this location and most of the coastal United States are limited due to proximity of the water table to the ground surface. The regulations of most state agencies prohibit dumping of solid waste directly into the ground water and, in fact, require a few feet of undisturbed soil between the high ground water level and the bottom of the landfill. With these restrictions, most landfills will fill below ground only one or two meters and above ground as high as availability of cover material will allow. Assuming an average depth of 7.5 meters, only half the depth as Case 1, the surface area required would be doubled to 100 acres (ca. $4x10^5 m^2$). Therefore, if the average annual percolation is 70 mm (Table 5), it would take close to 15 years for significant amounts of leachate to appear (Figure 6), at which time the average leachate quantity would be about 30 million liters/year (Figure 7).

Case 3--Los Angeles, California. The landfills in this area are generally area fills in deep canyons with depths ranging between 30 and 60 meters. Assuming an average depth of 40 meters, the surface area required would only be about one-fourth that of case 1, or 12 acres (ca. $5x10^4 m^2$). As noted in Table 6, percolation is negligible and one can easily assess the leachate problem as being insignificant for such a location.

A summary of the results for the three case studies is presented in Table 8.

Analysis of the sanitary landfill water balance calculations presented above points out some very interesting aspects of leachate generation of importance to the design engineer. These aspects should be considered in the overall assessment of the problem and may enter into the selection and design of leachate control measures.

First, in most cases leachate generation presents a potential problem principally in humid (low AET and high precipitation) areas of the country. Therefore, except for those sites where irrigation is utilized (discussed later), leachate problems will be virtually nonexistent at sanitary landfills in arid parts of the country.

TABLE 8

THEORETICAL LEACHATE QUANTITIES
AND TIME OF FIRST APPEARANCE

Location	Leachate	
	Time of first appearance (years)	Average annual quantity (liters/year) x 10
Cincinnati, Ohio	11	40
Orlando, Florida	15	30
Los Angeles, California	--	0

Second, there may not be a continuous flow of leachate throughout the year. Percolation and generation of leachate will most likely follow a pattern similar to that of the precipitation. This will result in the major portion of the leachate being produced during those months of significant percolation, with much lower flows occurring during the rest of the year.

Third, there will be a variation in the leachate generation pattern and amounts from year to year. The water balance calculations presented in this paper use mean monthly climatic values determined over a 25-year period. However, a brief analysis of precipitation data for any given location will indicate significant variations from year to year. So, while the average year might indicate a relatively minor leachate problem requiring little or no leachate control measures, an above average year may result in an entirely different assessment of the problem. Therefore, the engineer may wish to base his design on monthly precipitation values higher than the average values in order to provide a factor of safety in the estimation of leachate flow.

Other Considerations

The above methodology is presented with the intention of being a basic tool for engineers in assessing and designing sanitary landfills. The presentation was purposely kept straightforward since the concern was more to develop a clear understanding of the basic concepts and methods involved rather than a full scale design manual that would assess leachate problems for all conditions in all areas of the country.

Precipitation percolates into materials deposited in a solid-waste landfill and lixiviation (dissolving of soluble components) produces a solution called leachate. The landfill leachate under conditions where infiltration is greater than runoff and evapotranspiration combined, moves downward through refuse, and through underlying soil and sediment until it reaches an impermeable layer or ground water. In its journey, leachate traverses three zones of geochemical activity with certain characteristics which are shared and others which are unique to each. The ensuing discussion will attempt to describe some of the characteristics in each of the zones and ways in which they interact with the constituents of leachate.

3.5.1 REFUSE ZONE

Solid waste deposited in municipal landfills is a heterogeneous mixture of organic and inorganic materials and living organisms. Upon deposition, and frequently before, microbial activity begins the degradative process on organic matter. The microbial decomposition of organic

matter is encouraged by moisture and warm temperatures. Moisture is provided through precipitation, and temperature increases from the release of energy from oxidation of the organic substrate. Temperatures as high as 120°F have been observed in sealed test cells filled with mixed refuse.[1] Under aerobic conditions temperatures as high as 190°F have been observed.[2]

The microbial activity soon uses up the supply of oxygen and causes the refuse beyond the zone of rapid air diffusion to go anaerobic. Anaerobic conditions cause the end products of decomposition to be somewhat different from carbon dioxide and water which are the products of complete oxidation. Notable among the products of anaerobic decomposition is methane gas. Other organic anaerobic decomposition products such as alcohols, aldehydes, and thiols tend to be more odoriferous than their aerobic counterparts. Of particular importance with regard to leachate are the anaerobic forms of sulfur, nitrogen, iron, and manganese. Sulfur is present as sulfide, nitrogen as ammonia, iron in the ferrous (+2) form, and manganese in the manganous (+2) form. The latter two metals are more soluble in their reduced forms than in their oxidized forms.

The decomposition process provides carbon, hydrogen, nitrogen, oxygen, sulphur, phosphorus, and some metals which are fixed by microorganisms in their tissues. Metabolic products and some inorganic residues from organic decomposition are released to percolate.

The percolate flows downward through the refuse which is in progressively advanced stages of decomposition, and it passes through layers of buried cover material. Percolate shows a net gain in dissolved constituents as it progresses downward, but may lose some individual ions from cation exchange or other reactions encountered en route. Attenuation of percolate constituents within the landfill is not well documented, therefore predictions concerning it must be based upon known geochemical principles and final leachate composition. The attenuation within the refuse zone is not of immediate practical importance because those species which are attenuated are not contributing to ground-water contamination. However, some discussion of processes occuring in the refuse zone is important as it assists in the interpretation of leachate composition.

Elements which are fixed in microbial tissue will not be mobilized until the microorganisms die and the cells break

down. Even then, some of the compounds released are only slightly susceptible to further biodegradation, and represent a stabilized organic material much like soil humus. humus-like material is also a product of refuse decomposition. This material is a polymeric organic colloid with various chemically active functional groups, such as acid, alcohol, and phenol, which can react with metal cations to form complexes.

In this way, metals may be sorbed on the organic colloid and removed from solution. Cation exchange reactions occur in a manner similar to the exchange reactions occurring with clay which are discussed below.

Nitrogen present in refuse organic matter is released in soluble form with microbial decomposition. In organic substances, nitrogen is in a chemically reduced state. With aerobic decomposition, the nitrogen is oxidized to nitrate ion. Under anaerobic conditions, nitrogen is released as ammonium ion. [3]

Aerobic: $2\ CH_3CHNH_2COOH + 7O_2 \longrightarrow 3CO_2 + 7H_2O + NO_3^-$ (1)

Anaerobic: $0.33\ C_4H_6ON + 0.073\ HCO_3^- + 0.64\ H_2O \longrightarrow 0.33\ NH_4^+ + 0.14\ CH_3COO^- + 0.13\ C_2H_5COO^- + 0.133\ C_3H_7COO^- + 0.193\ CO_2$ (2)

Anaerobic conditions are predominant in landfills. Thus, most nitrogen in leachate is present as ammonium. Nitrate which is formed aerobically may be reduced through denitrification to molecular nitrogen when it passes through anaerobic zones. The relatively small amount of nitrate produced, coupled with probability of denitrification explains the typically low nitrate concentration in leachate.

Nitrogen is released only if it is present in quantities exceeding the nutritional requirements of the microbial population effecting organic decomposition. The requisite amount of nitrogen can be expressed in relation to carbon. Carbon/nitrogen ratios up to about 10/1 will result in nitrogen release. Above that, most of the nitrogen will be fixed in microbial tissue. Fresh organic matter would be expected to release nitrogen during decomposition, whereas carbonized ash would not be.

Organic decomposition releases carbon dioxide in large amounts under aerobic conditions, and in smaller amounts under anaerobic conditions. The enrichment of the interstitial gas in refuse by carbon dioxide results in production of bicarbonate ion as follows:

$$CO_2 + H_2O \rightleftharpoons H_2CO_3 \quad (3)$$

$$H_2CO_3 \rightleftharpoons H^+ + HCO_3^- \quad (4)$$

$$HCO_3^- \rightleftharpoons H^+ + CO_3^{-2} \quad (5)$$

Only when the pH exceeds 9 does reaction (5) occur to a significant extent, (about 20)percent. [4] The production of carbonic acid (4) is proportional to the partial pressure of CO_2 in the atmosphere in contact with the water. The carbonic acid ionization to bicarbonate (4) is proportional to the carbonic acid concentration. Only a small fraction of the carbonic acid in the system is ionized.

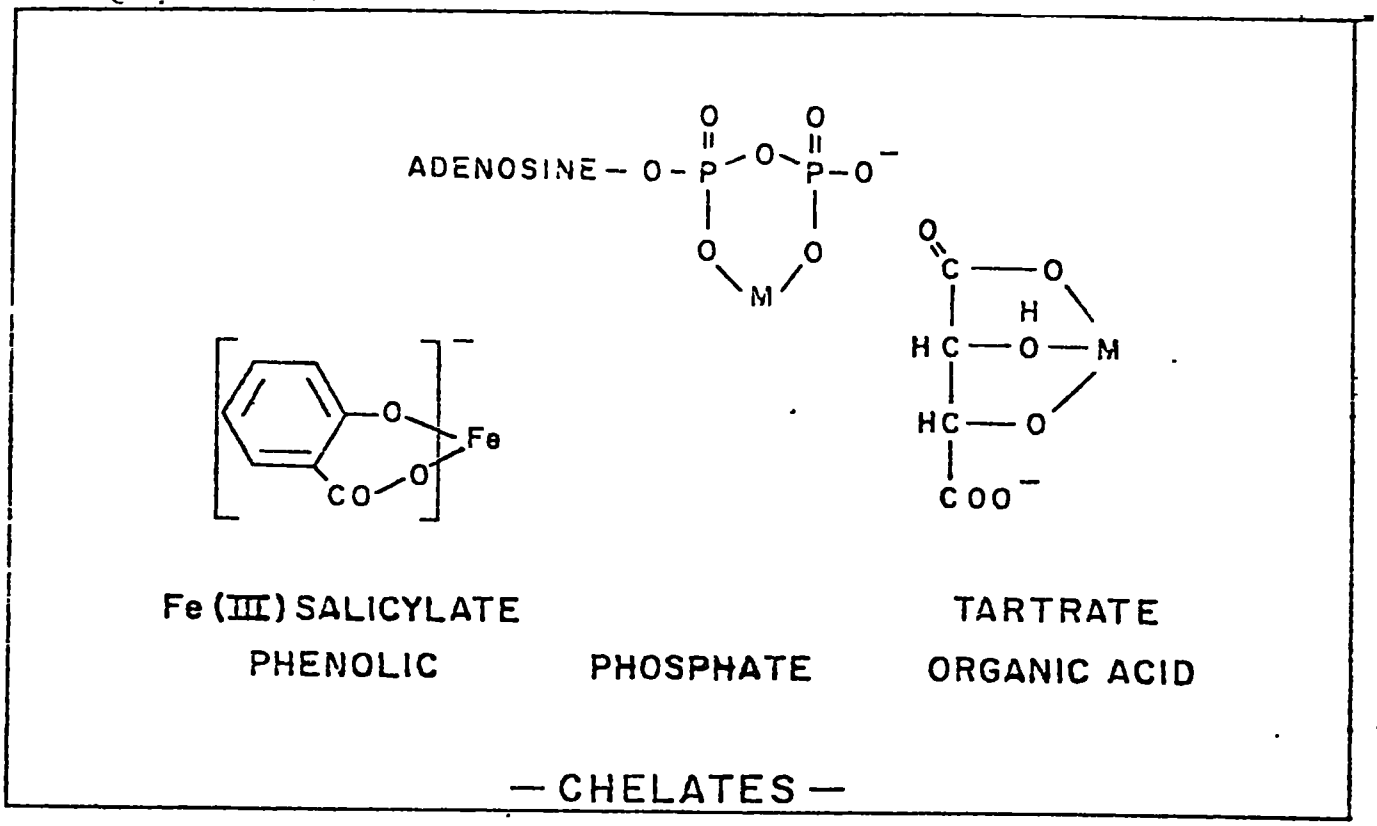
Bicarbonate is frequently a major anion in leachate. ecause of the reversible reactions (4) and (5), when present, bicarbonate acts as a buffer and tends to prevent large fluctuations in pH.

Other organic decomposition products include carboxylic acids (acetic, isobutyric),phenols (phenol, p-cresol), and amino acids (glycine, alanine) which can form ring complexes (chelates) with heavy metals, rendering them soluble and protected from adsorption.[5,6]Movement of heavy metals with percolate may be possible to a large degree because of such

complexes (Figure 3-8). The distance over which chelated metals move depends upon the chemical and biochemical activity encountered. Some chelates change ionic charge with changes in pH. Thus, if a change in pH in percolate occurs, the chelates may be adsorbed or otherwise deposited. Microbial decomposition of the organic portion of the molecule leaves the metal cation behind where it is prone to adsorption or precipitation. If the chelated metal encounters no conditions which affect it, it will be transported to the ground water.

Heavy metals in landfills are primarily in their metallic state and are not soluble . The exception is with deposition of soluble heavy metal salts either as solids or in solution. These may come from certain industrial activities such as electroplating or metal pickling. Most heavy metals occur in solution as cations (positively charged), but a few are usually present as anions (negatively charged). Those usually occurring as anions are chromium and vanadium. Included with heavy metals, but chemically somewhat different are arsenic, boron, and selenium which occur as anions.

A minor complement of heavy metals is present in the combustible (decomposable) fraction of urban refuse. In



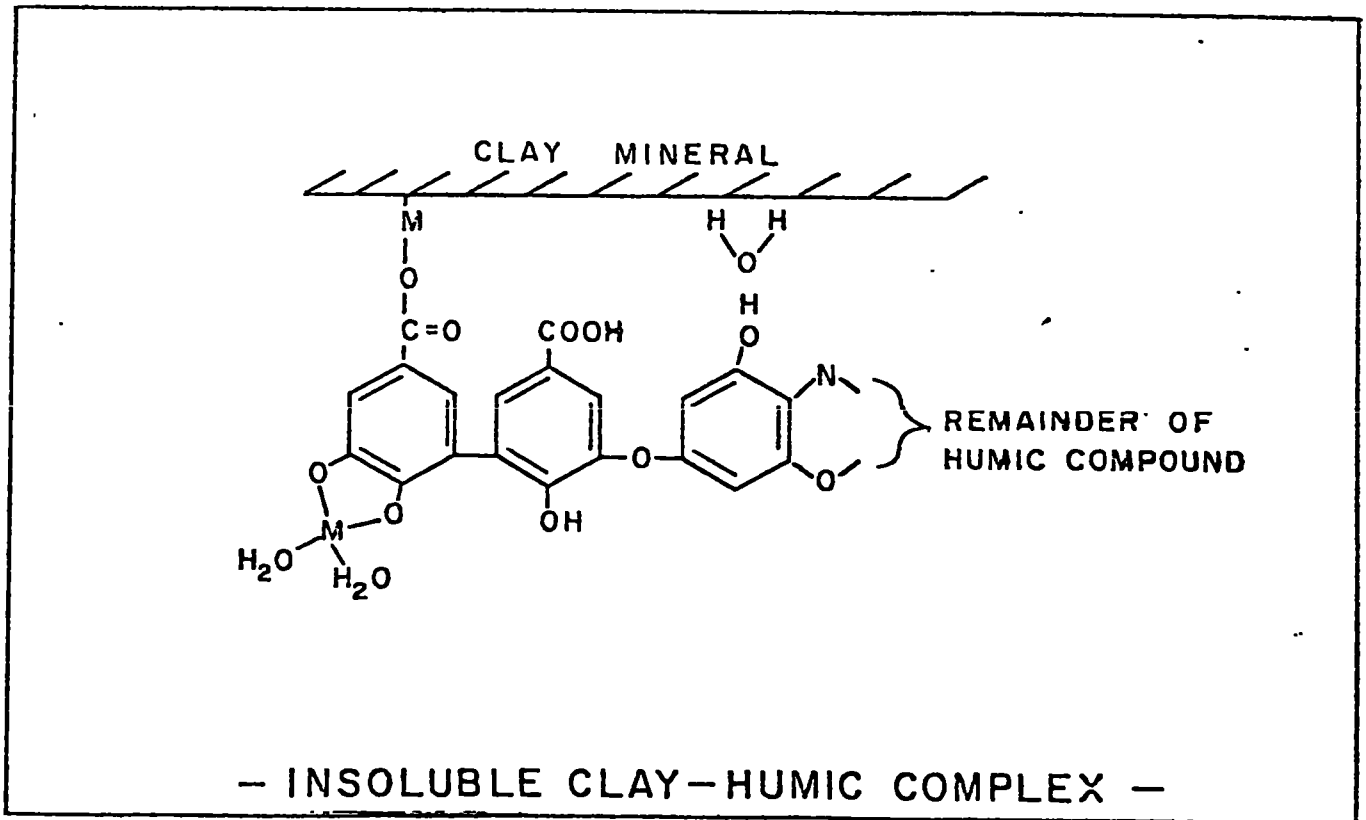


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Figure 3-8 Types of naturally occurring metal chelates and complexes. 7)

this usage, minor means concentrations in parts per million as contrasted with percentages as represented by metallic wastes. These heavy metals in the decomposable fraction are released when the organic matrix is decomposed. Table 3-10 lists metal concentrations taken from refuse sampled after segregation for use as fuel in an incinerator-electrical generator. Some of the combustibles are completely resistant to microbial attack (certain plastics) and others are only very slowly decomposed (certain plastic and rubber types). Therefore, metal concentrations may be higher in Table 3-10 representing total combustion than they would be for a biochemical decomposition which would be incomplete.

Iron and manganese are typically found in leachate in concentrations exceeding those of normal ground water.[9,10] The anaerobic percolate water can reduce these metals to lower valence states which are more soluble. The iron and manganese may be present as part of the refuse material, or may be part of the clay or hydrous oxide component in soil cover.

Sulfur under aerobic conditions is oxidized to the sulfate ion. Under anaerobic conditions, it is soluble as sulfide,

TABLE 3-10 INORGANIC ELEMENTS IN THE COMBUSTIBLE FRACTION OF URBAN REFUSE [8]

Major elements	Typical value, weight percent	Range weight percent
Aluminum [a]	1.1	0.4 - 1.6
Calcium	.45	.23 -1.0
Chlorine	.4	.3 - 1.5
Iron [a]	.18	.05- .65
Magnesium	.10	.05 -. 77
Phosphorus	.1	.07 -.7
Potassium	.07	.03-.20
Silicon	4.0	1.0 -.10
Sodium	.5	.15-.86
Sulfur	.2	.1 -.3
Titanium	.23	.07-.50
Zinc	.10	.04-.84
Minor and trace elements		
Antimony	45	21-77
Arsenic	--	2
Barium	50	35-79
Beryllium	--	1
Bismuth	22	6 -44
Boron	13	7 -70
Cadmium	15	3 -68
Chromium	30	10-175
Cobalt	7	2-17
Copper [a]	195	29-450
Germanium	--	4
Lead	230	110-1,300
Lithium	3	2-4
Manganese	85	50-480
Molybdenum	19	13-28
Nickel	16	4-49
Silver	4	1-16
Tantalum	--	4
Tin	50	33-95
Tungsten	--	20
Vanadium	13	7-70
Zirconium	10	1-70

[a] Present also in the metallic state

or may occur as hydrogen sulfide gas (H_2S) which has a rotten egg odor. Sulfate has been reported in leachate[9] and sulfide is probably present in some leachates, but its detection presents analytical problems. Moreover, many metals form insoluble sulfide salts which remove sulfide from solution.

Phosphorus is released by decomposition of organic matter. At the usual pH range of leachate, the $H_2PO_4^-$ and HPO_4^{-2} ions are predominant. As discussed below, soils have a high capacity for phosphate attenuation, whereas the refuse material does not. The clays and hydrous oxides responsible for attenuation comprise only a small fraction of the landfill mass. Phosphate can be and frequently is produced in substantial amounts in leachate. Fungaroli reported a maximum of 130 ppm in leachate from an experimental lysimeter.[11] His reported concentration is about as high as has been reported, but several others have reached several tens of ppm.[12] Effluents containing phosphate concentrations of this order of magnitude discharged into surface water would be expected to produce a rather severe eutrophication in the receiving waters. Were leachate to enter ground water directly, it would almost certainly contribute more phosphate than would percolate which has passed through soil and an unsaturated zone.

Water quality parameters which do not measure individual chemical species include biochemical oxygen demand (BOD), chemical oxygen demand (COD), total organic carbon (TOC), color, conductance, and turbidity. The refuse zone provides little, if any, attenuation of these characteristics; instead, it usually increases them. Bacteriological investigations of leachate are in progress at the University of Illinois and in the EPA. Fecal coliform and fecal streptococci have been observed in leachate, and poliovirus was reported in leachate from a simulated landfill.[12] The recent trend to use of disposable diapers has increased the source of enteric bacteria in solid waste. Another source in some areas is septage which may be disposed of on landfills with little or no treatment.[9] Sewage sludges from municipal waste-water treatment plants are frequently dried on sand beds and dumped in landfills also used for other municipal refuse.

Movement of bacteria and viruses within the landfill and through the unsaturated zone is dependent upon the porosity of refuse and underlying geologic formations. Refuse may offer many paths through which water can travel relatively unimpeded. If course sand and gravel or fractured rock underlie the refuse, percolating water may carry microorganisms with little or no attenuation except for natural die off. These conditions, judging from locations which have been studied, are the exception rather

than the rule.

3.5.2 UNSATURATED ZONE

As used herein, the unsaturated zone is defined as the area in soil or sediments between the bottom of the landfill deposits and the water table. The distance can vary between zero (refuse contacting ground water) to several hundred feet. This zone is below what is usually considered "topsoil" or the weathered, organic-matter-rich upper horizons of most soils. At most landfill sites, topsoil has been removed, and sometimes much subsoil also, prior to deposition of refuse. The porous materials comprising the subsoil are likely to be low in organic matter, have a sparse microbial population, and may vary in permeability over a wide range. For purposes of discussion, we will consider the unsaturated zone to be 20 to 200 feet thick. This range allows percolating water an opportunity to react chemically with its environment before reaching ground water. Percolating water has four options in passing through the unsaturated zone. It can move virtually unchanged, can show a net gain of solute, show a net loss of solute, or keep the same total ionic concentration with a net exchange of ions. Since few soils or sediments are chemically inert, changes in transported solute are to be expected.

Chemical activity in the unsaturated zone is primarily located at the surfaces of clay minerals and hydrous oxide coatings. Silts exhibit a small amount of chemical activity, and limited microbial activity may take place either from the indigenous population or that transported from refuse.

Clay minerals which occur in soils and sediments are small aluminosilicate crystals possessing a large specific surface area. The crystal structure is such that clay particles have a plate-like shape. Particle sizes fall in the range of true colloidal particles, <2 μm. The clay colloids carry a net-negative surface charge which results from internal atomic substitution and ionization of surface hydroxyl (OH) groups. The negative charge attracts cations from solution and they are adsorbed at the clay surface by weak chemical and electrostatic (van der Waals) forces. Cations can exchange on the clays and the cation exchange capacity (CEC) varies with clay type. The clay types in order of CEC are kaolinite < illite < montmorillonite. The latter clay has an expanding lattice which, when wet, allows cation exchange to occur on inner crystal surfaces. This increases the specific surface considerably. In the process of cation exchange, the affinity of the cation for the clay is proportional to cationic charge. Replacement on exchange sites is

accomplished on an equivalent charge basis, e.g. Ca^{+2} displaces 2 Na^{+}. Ions of higher charge usually displace those of lower charge in cation exchange reactions.

Cations will be removed from solution until either the cation exchange capacity is reached, or the limit of displacement reactions is reached. The limit of CEC can range from nearly zero to probably not more than 60 milliequivalents per 100 g soil. Solution concentrations, pH, and percolation rate affect the reactions quantitatively. Thus, no quantitative predictions about attenuation can be made without knowledge of specific site characteristics. It should be noted that adsorption is not a permanent fixation. Cations may be desorbed with changes in solution composition, pH, or oxidation-reduction (redox) potential.

Divalent and trivalent cations include most of the heavy metals. These are held more strongly than sodium, potassium, or ammonium on the cation exchange complex. Griffin and Shimp measured the attenuation by clay minerals of several components of leachate.[13] They rated attenuation as follows:

High	Hg, Pb, Zn, Cd
Moderate	Si, Mg, K, NH_4
Low	Na, Cl
None	Ca, Fe, Mn

The rating "none" is believed to be caused by desorption of calcium from clay exchange sites and dissolution of iron and manganese in clay crystals by leachate. Al, Cu, Ni, Cr, As, S, and PO_4 were too low in concentration to be rated.

Another source of cation adsorption and exchange is the coating of iron and manganese hydrous oxides frequently found on soil and sediment particles.[14] These coatings exist in amorphous and microcrystalline forms with specific surface areas of as much as 300 sq m/g. Cations adsorbed on the surfaces of newly formed hydrous oxides may, with time, be incorporated in their crystal structure. Cations thus incorporated are fixed, and no longer exchangeable.

Surface charges on hydrous oxides are produced by oxygen and hydroxyl groups. The surface charge is dependent on pH and redox conditions. Therefore, the CEC exhibited by hydrous oxides is pH and redox dependent. The CEC of hydrous oxides approximates that of illitic clays (10-40 meq/ 100 g). Because hydrous oxides are not a discrete fraction of soil as clays are, CEC calculations come from laboratory preparations.

Heavy metals are prone to sorption on hydrous oxide coatings in soil. The hydrous oxides are frequently cited as so

limiting metal solubility that agricultural deficiences of copper, zinc, and cobalt occur. [14] Attenuation of heavy metals present in leachate is desired. In locations virtually free of clay minerals, these coatings may be present on sand grains giving the sandy formation some ability to attenuate metallic ions.

Adsorption is only one mechanism for removing dissolved ions from solution. Changes in the geochemical environment can also affect solution equilibria. A transition from reducing conditions in the landfill to oxidizing conditions in the unsaturated zone can reduce the concentration of some redox-sensitive species in solution and change the chemical form of others. Iron and manganese will oxidize and precipitate from solution, for example. If porosity will allow bacterial movement, biochemical reactions involving leachate constituents can proceed. Sulfide and ammonium can be oxidized to sulfate and nitrate. Dissolved organic matter measured in terms of BOD and COD can be reduced through microbial decomposition. Some nutrient elements in the course of these reactions will be incorporated in bacterial tissue and thereby removed from solution until the bacterial cells die off. Conversion of ammonium to nitrate changes nitrogen from a subject to attenuation to a form which is not. Sulfide to sulfate

oxidation is not expected to be as significant. Sulfide can form insoluble precipitates with many of the heavy metals. For this reason, it may not be present in more than trace amounts in leachate. Microorganisms may also attack the organic ligands associated with chelated and complexed metals. Decomposition or absorption by microorganisms would remove the metals from leachate.

Phosphate reacts with a variety of soil components forming insoluble products. Calcium and phosphate react in solution to form hydroxyapatite [$Ca_5OH(PO_4)_3$] the least soluble phosphate compound known. Iron, aluminum, and manganese can also form virtually insoluble precipitates with phosphate. These reactions lead to a strong attenuation of phosphate when these metal ions are present in the unsaturated zone.

Bouwer reports that in the Flushing Meadows high-volume waste water recharge project, large amounts of phosphate have been fixed in the unsaturated zone by chemical precipitation.[15] Phosphate fixed in this way amounts to about 43,000 lb/acre (48,000 kg/ha) calculated as phosphorus, and it has enriched the unsaturated zone several tens of feet below the basin surface. This illustrates the potential for attenuation of chemical reactions as well as the more often considered colloid

surface reactions.

Although phosphate ions are negatively charged, they interact with clays and hydrous oxides forming insoluble complexes. These reactions may occur in either soils or subsoils. The phosphate complexes are so insoluble that phosphate added in fertilizer must be added in excess to compensate for fixation. The presence of clay or hydrous oxides in formations beneath landfills is valuable not only from the point of view of CEC, but also the fixation capacity for phosphate.

Carbonate also reacts with calcium, magnesium, and some heavy metals forming relatively insoluble compounds. The solubilities vary according to metal species and pH as shown in Table 3-11. Calcareous deposits in the unsaturated zone can be valuable in attenuating phosphate and heavy metals from leachate. Because carbonate neutralizes acids, BOD and COD present as organic acid may also be reduced. Some organic acids form insoluble salts with calcium, and organic bases are less soluble in alkaline solutions. Carbonate induced alkalinity may change solubilities of heavy-metal chelates and lead to deposition of heavy metals.

Redox potential considerations are particularly important in the unsaturated zone. Because of this, a brief discussion of

Table 3-11 CONCENTRATIONS OF METAL IONS FORMING CARBONATES OR IN EQUILIBRIUM WITH $CaCO_3$ (mg/l) 16)

pH	Ca^{2+}	Fe^{2+}	Cd^{2+}	Pb^{2+}
7.2	124	1.04	0.0035	0.0211
7.6	76	0.64	0.0021	0.0083
8.0	48	0.40	0.0013	0.0041
8.5	28	0.23	0.0008	0.0013

the concept of oxidation and reduction in water systems is included. Reduction is the gain of electrons by a chemical species; oxidation is the loss of electrons. Iron chemistry illustrates a common water component which is sensitive to redox conditions. It is reduced as follows:

$$Fe^{+3} \text{ (slightly soluble)} + e^- \longrightarrow Fe^{+2} \text{ (more soluble)} \quad (7)$$

$$Fe^{+2} + 2\bar{e} \longrightarrow Fe^0 \text{ (metallic)} \quad (8)$$

The oxidation state of iron or other major redox sensitive species in water in combination with the percentage saturation of dissolved oxygen determines the redox potential of water.

Redox potential in water is measured electrochemically with gold or platinum electrodes and a pH/millivolt meter. The voltage reading obtained for redox potential is termed Eh. Eh values range from over one volt for highly oxidized systems to negative voltage values for reduced systems.[4)]

Ground water frequently exists at a low Eh potential in comparison to surface water. The low Eh governs solubilities (iron, manganese), chemical species actually in solution (Fe^{+3}, $FeOH^{+2}$, FeO_4^{-2}), and governs certain geochemical transformations (nitrification, sulfate reduction). Because of these geochemical controls, it is important to determine Eh when geochemical interpretations must be made.

A comment on Eh determination is called for because, although useful, this measurement on ground water is difficult to make. Any exposure of the water to the atmosphere will instantaneously change the Eh value. This necessitates a closed system from the pump to sealed electrode holder. Water must also be pumped until the Eh stabilizes, which may require as much as several hours pumping time depending upon rate of discharge and subterranean conditions. An Eh determination made without proper care can be worse than useless because it will indicate conditions which, in fact, do not exist.

The unsaturated zone is influenced by the percolation of leachate into it and simultaneously influences the leachate. Water of low Eh first infiltrating into the unsaturated zone of high Eh will become more oxidized while simultaneously reducing substances in the unsaturated zone. A continued percolation of reduced water may convert what had been an oxidized system into a reduced one. Or the percolate may become oxidized if that capacity in the unsaturated zone is greater. The degree of influence of reduced leachate on the oxidized unsaturated zone and vice versa depends upon the reserves of material capable of oxidizing or reducing in the unsaturated zone and leachate. The greater the distance leachate travels between refuse and ground water, the better the chance that the entire path through the unsaturated zone

will not become reduced. Raising the Eh of leachate will tend to attenuate some components in solution at the point of exit of the refuse zone.

Leachate reaching ground water may as a result of the geochemical conditions en route be depleted in some constituents and enriched in others as dictated by the composition of the unsaturated zone and its overall affect on Eh and pH. Distance of travel, speed of percolation, flow-nonflow cycles, and leachate temperature are all parameters controlling leachate quality.

3.5.3 AQUIFER ZONE

Concepts useful for describing surface water pollution are generally not valid for ground water. Ground water movement is described by Darcy's Law which states that velocity is directly proportional to the permeability of the aquifer and the hydraulic gradient, and inversely proportional to the porosity. Ground-water flow velocities vary over a wide range, with 5 ft/yr to 5 ft/day being typical. Highly permeable outwash glacial deposits, fractured basalts and granites, and cavernous limestone aquifers allow very much higher velocities.

The generally slow velocity of ground water results in laminar flow which exhibits different characteristics of

mixing than does the turbulent flow usually associated with surface streams. A water of different chemical composition from ground water which is injected or percolated into ground water tends to maintain its integrity and is not diluted with the entire body of ground water. Instead, it moves with the ground-water flow as a plume undergoing minimal mixing. The plume shape is determined by the physical characteristics of the aquifer. Porous media give somewhat different shaped plumes from fractured rock or cavernous limestone. Figures____,____,____,____,and in Chapter 4 illustrate the paths of ground-water movement in various hydrologic regimes. It is obvious that plumes of leachate-enriched ground water in these environments would assume different shapes.

Other hydrologic conditions further influence plume shape. Hydraulic gradients going in more than one direction, such as occur if ground-water mounding occurs beneath a landfill, will spread leachate laterally, creating a plume wider than the areal extent of the landfill. A vertical gradient is less often encountered, but should it be present, leachate would follow ground-water flow downward as well as horizontally. Leachate may exert an influence on the shape of the plume of contamination it produces. Almost universally, leachate

temperature will be above ambient. It may be as much as 50° F above the ambient ground-water temperature. Leachate may also have a dissolved solid concentration sufficiently high to increase its density over that of ground water. These combined physical characteristics may significantly affect the way in which leachate interacts with ambient ground water. For example, one study in a highly permeable aquifer showed that leachate sinks directly to the bottom of the aquifer beneath the landfill.[9] No natural vertical gradient was measured where this phenomenon occurred.

Differential attenuation is defined as a reduction in concentration of a dissolved constituent with distance along the direction of water flow which is disproportional to changes in concentration of other constituents. Differential attenuation may result from chemical reactions which remove the constituent from solution or from self destruction. Apparent attenuation occurs from dilution by mixing with water of lower constituent concentration.

Dilution may take place in ground water in two ways. One is hydrodynamic dispersion, and the other is molecular diffusion. Microscopic dispersion is mixing caused by the tortuous flow of water around individual grains and through pores of various sizes in a porous aquifer. Macroscopic dispersion is mixing as water flows in and around heterogeneous geologic formations.

Molecular diffusion operates on a much more restricted scale. It is the diffusion of solute across a concentration gradient from stronger to weaker. Diffusion is seldom possible to measure in the field. There are mathematical formulas which describe dispersion.

In order to calculate a dispersion coefficient, an intensive investigation of the site over a period of time is necessary. Hydraulic gradient, porosity, concentration gradients in the plume, temperature, and measurement of solute movement are all factors entering into the formula. The time during which the plume has been in existence is also important. The extent of the dispersion is a function of time. Forecasting a future extent of the plume may require a mathematical modeling program in which dispersion is only one of the characteristics of the system.

Chemical interactions provide the greatest amount of differential attenuation in the aquifer zone. Hydrous oxides of iron, aluminum, and manganese or clay minerals present in aquifers attenuate cations in the same way that they do in soils or the unsaturated zone. Because hydrous oxide and clay colloids are in constant contact with water in the aquifer, it can be assumed that the exchange sites are saturated and essentially in equilibrium with the ambient ground water. Leachate-enriched ground water when contacting these colloids will initiate cation exchange which results in desorption of

cations which are less strongly held than those replacing them. In this way, hydrogen, sodium, calcium, and magnesium may be released into the aqueous phase by exchange with heavy metals and other cations in leachate. High hardness values associated with leachate plumes may be due in part to this ion exchange phenomenon.

Chemical precipitation in the aquifer is possible if the natural ground-water composition includes ions which form insoluble compounds with constituents in leachate. An example would be formation of hydroxyapatite with leachate phosphate and calcium in ground water. Other precipitation reactions may occur if geochemical conditions are encountered in the aquifer which lead to changes in redox potential or pH. Buffering reactions may change concentrations of bicarbonate, carbonate, ammonium, and sulfur (H_2S, HS^-). Reserves of hydrogen (acid) or hydroxyl (base) ions may be present in ground water if it has unusually high or low pH. Clays and hydrous oxides also are capable of releasing hydrogen ions from exchange sites for reaction with dissolved species.

The third means of attenuation in aquifers is that termed decay. Oxidation of organic compounds produces carbon dioxide and water and eliminates the compounds. Radioactive species undergo radioactive decay to stable daughter products. Some elements

decay in terms of seconds and others lose half of their activity in periods measured in thousands of years. Radioactive materials should not be present in municipal landfill leachate. Micro-organisms carried into the aquifer zone are deprived of a good nutrient supply and are subjected to a generally cooler temperature. This results in a lowering of biochemical activity, frequently to the point of cessation. Bacteria can quickly adapt to hostile conditions by encysting and ceasing activity. They may remain inactive, but viable in this form from days to weeks. Bacterial cells are attracted to inorganic colloid surfaces and are also subjected to physical filtration. These phenomena coupled with natural die off, tend to reduce bacterial numbers rather rapidly.

There are two additional complications in the interpretation of ground-water quality in leachate plumes. One is the variation in leachate concentration with time, and the other is the discontinuous recharge of leachate which occurs in most geographical regions.

Leachate production begins as soon as deposited refuse is wetted to field capacity. The lag time depends upon local climatic conditions and rate of refuse deposition. In an active landfill, older organic matter is stabilizing while

simultaneously new organic matter is beginning to ferment and produce stronger leachate. The net effect is an increasing leachate concentration from a given area, or an increasing areal contamination, or both as long as the landfill is active.

Leachate produced at the initiation of percolation through the landfill is less concentrated than that produced after several years' refuse accumulation. This leachate will be found at the distal end of the plume of leachate-contaminated ground water. The closer the sampling site to the landfill, the more concentrated should be the contaminated ground water. An increasingly concentrated leachate source in addition to the factors of dilution and attenuation must be considered in interpreting the results of sampling the plume. An erroneously high value for attenuation or dilution may be given if the variation in source strength is ignored.

The intermittent recharge occurring from most landfills also complicates interpretation of leachate-plume configuration. During summer months when evaporation frequently exceeds rainfall, little or no leachate may be produced. Ground water, however, moves under the landfill at a relatively steady rate. Thus, there will be variations in the volume and strength of leachate reaching ground water during the course of time. These variations will show in the leachate plume as variations in total solute concentration. A sample taken from the plume at any given time may represent a "high" or "low" in the

intermittent recharge pattern. One way to visualize this phenomenon would be to watch the response of a conductivity probe in a well screen over time. As leachate-enriched ground water moves past the point, conductivity will vary with changes in dissolved solids concentration. The variations may be noticeable only in time spans of weeks to months. Again, this complicates efforts to calculate values for dispersivity or dilution because concentrations vary from factors other than aquifer characteristics.

A generalized summary of the susceptibility of leachate constituents is provided in Table 3-12. The mechanism of attenuation which affects each constituent is listed for the zones through which leachate may pass. When data are summarized in this fashion, only the principal mechanisms can be cited. For example, no attenuation is listed for all of the constituents in the refuse zone. This is not really true as the previous discussion points out. However, quantification is impossible, and there is a net output of most of the constituents. Sulfate, nitrate, and ammonium are given biochemical conversion alternatives. These ions are subject to oxidation and reduction reactions which may convert or eliminate them. Heavy metals are also prone to one or more of the attenuation mechanisms, and may not be universally present in leachage. Biochemical reactions were not listed for the

Table 3-12 SUSCEPTIBILITY OF LEACHATE CONSTITUENTS TO DIFFERENTIAL ATTENUATION

Attenuated Constituent	Refuse Zone	Unsaturated Zone	Aquifer
Chloride	0	0	0
Sulfate	0-B	0-B	0
Sulfide	C	C-B	C
Phosphate	0	A-C	A-C
Nitrate	0-B	0	0
Ammonium	0-B	A-B	A
Sodium	0	0	0
Potassium	0	A	A
Calcium	0	A	A
Magnesium	0	A	A
Heavy metal anions (Cr, V. Se, B, As)	0-B	0-B	0
Heavy metal cations (Pb, Cu, Ni, Zn, Cd, Fe, Mn, Hg)	0-A-C	A-C	A-C
Organic nitrogen	0	B	0
COD	0	B	0
BOD	0	B	0
Volatile Acids	0	B	0
Phenols	0	A-B	0-A
MBAS	0	A-B	0-A

0 = no attenuation
A = adsorption
B = biochemical degradation on conversion
C = chemical precipitation

aquifer zone because biological activity is inhibited. In places, biological activity may be significant in the aquifer, but the amount and type cannot be predicted.

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