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Field Monitoring and Description of the Adams, MA Landfill Leachate Plume

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Technical Report

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Submitted to

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Division of Water Pollution Control
S. Russell Silva, Commissioner
Thomas C. McMahon, Director

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ABSTRACT

This research investigates the impact of the Adams Sanitary Landfill, Adams, MA, on the surrounding ground and surface water. The site was chosen because of a prior history of surface water contamination and what appeared to be a relatively straight forward regional groundwater flow pattern.

The first phase of study involved the installation of a piezometer well field and the concurrent research into investigations, studies and test borings done in and around the site. A total of 21 multiposition wells were installed consisting of 43 piezometers. On a monthly basis, the piezometers were sampled and analyzed and groundwater elevations were recorded. The data obtained from this work allowed us to identify the shape and extent of the leachate enriched groundwater plume and determine the flow patterns in the area.

Results from this study indicate that while leachate is being produced and ground and surface water contamination exists, the effects of geology, landforms, streams and human factors limit the extent of contamination. A small stream and the wetland through which it flows serve as discharge zones for the bulk of leachate enriched groundwater. Beneath the wetland, a thick, low permeability gray clay inhibits percolation of the waste and confines the zone of contamination to within 12 feet (3.7 m) of the ground surface.

The Adams landfill also illustrates two potential pitfalls of routine application of regulatory requirements: (1) the net result of covering the landfill with impervious material has been to create groundwater mounding within the landfill, and (2) the sharp decrease in groundwater pollution concentrations with distance from the landfill does not imply absence of significant pollution because of the strong vertical velocity gradients - gradients which would probably not be identified by a routine monitoring program.

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CHAPTER I

INTRODUCTION

Sanitary Landfills

Sanitary landfills have been identified as sources of both ground and surface water pollution. As early as 1925, Bailey (3) traced a manhole fire to the gases being produced at a nearby landfill. Besides gases, the various physical, chemical, and biological processes that take place in the refuse produce compounds that become dissolved or suspended in the water percolating through the waste generating a product which is known as leachate. The chemical characteristics and range of concentrations of leachate are compared with domestic wastewater in Table 1.

Statement of Problem

The intent of the sanitary landfill is to design, locate and operate a land waste disposal facility in such a manner as to minimize the seepage of leachate into the surrounding environment. Modifications on the design over the years reflect the inadequacies of many of the existing schemes. Unfortunately, the purchase of ideal sites, installation of expensive liners, purchase of expensive equipment and extensive operator training, which characterize an ideal design are rarely achieved in practice. The alternative to these solutions is the toleration of some degree of ground and surface water contamination, such as that which the town of Adams, MA is now facing. This report details our findings on the impact of the Adams Sanitary Landfill on its environment.

In Chapter 2 an overview of the various techniques and methodologies used in determining the extent of groundwater contamination is examined, and a review of the types of contaminants to look for in the water and the ones chosen in this study are detailed. The scope will then broaden to look at the various landfill studies which have been undertaken over the years, concluding with a general approach to monitoring and a description of groundwater flow.

Chapter 3 is presented to give the reader a broad history of the site on which the Adams landfill is located, accomplished by

Table 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 (μmhos)	--	--	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	3.7-8.5	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 Publiciation 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(2): 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 94th Annual Conference, June 17, 1974. Boston, Massachuetes. 23 p.

means of a basic geologic history and a more detailed site history, including waste placement. In Chapter 4 we introduce our step-by-step well installation procedure and the logic behind the location of wells, followed, in Chapter 5, with a description of how those wells were sampled and analyzed, and concluding with an overview of all of the additional tests that were performed at the site and in the laboratory.

The regional and local ground and surface water flow patterns are described in Chapter 6. The site is broken into three distinct zones, and the possible paths of a drop of water are tracked through each in order to set the stage for the production and eventual transport of leachate from the landfill into the ground and surface water. In Chapter 7 the environmental impact of that transport is outlined.

CHAPTER II

LITERATURE REVIEW

Studies of groundwater flow and the associated transport of leachate involve various methodologies and technologies. The local geologic setting, the goal of the study and financial constraints ultimately define the choices of methods and materials. When completed, the investigation should clearly describe the cause(s) of the problem, define its scope, and attempt to determine how the problem might vary over time. Over the years a variety of tools for obtaining and presenting this data have been developed and used to describe the behavior of sanitary landfills.

Groundwater Monitoring Techniques

As leachate analysis and groundwater monitoring techniques have improved, a number of landfill sites where ground and surface water contamination already exists have been uncovered. Growing public concern over the availability and quality of drinking water supplies led several states to require some form of groundwater monitoring at land disposal sites, but a 1980 survey by Clark and Sable (12) showed that implementation was proceeding slowly. One month after these findings were released, the EPA, under the groundwater monitoring subpart of the Resource Conservation and Recovery Act (RCRA), issued new guidelines to all landfill owners requiring them to implement, by November 19, 1981, a groundwater monitoring program capable of detecting the impact of that site on groundwater quality (17). Specifically, the ruling called for the installation of at least one upgradient well that would yield representative samples of the background water quality from the uppermost aquifer near the facility, and an additional three or more wells must then be installed downgradient of the refuse area, also for the purpose of detecting contamination in the uppermost aquifer. Each facility would then be required to develop a plan for sampling and analyzing the groundwater to establish the extent of the contamination. Installation of the piezometers at the Adams site for this research constitutes the type of system required by RCRA.

Beginning a groundwater monitoring program at any facility involves careful site evaluation prior to the installation of monitoring wells, using bedrock geology and topographic maps of the site, as well as any borings done in the area. Tinlin (61) provides an overview of approaches used in evaluating a site. In addition to maps, surficial features such as water bodies, rock outcrops or vegetation can be a useful and inexpensive means of determining certain gross information about a particular site.

A variety of surface geophysical methods have also been devised for determining subsurface information. Sandlein and Yazicigil (55) present a variety of these including seismic refraction, which can locate dipping bedrock layers, bedrock boundaries, and water table depths. Although the information obtained is valuable, much of the equipment described is suited to larger scale projects than the Adams investigation. Another geophysical technique gaining wide acceptance for New England soils is earth resistivity surveys. The contaminants in an aquifer will reduce the electrical resistivity of a saturated soil, so that measuring the resistivity across a site may likely locate the center of a leachate plume if the contrast between contaminated and uncontaminated water is high. Stollar and Roux (59) present four successful case histories of resistivity concluding that this method is both less costly and less time consuming than well installation, a conclusion also reached by Urish (65) while locating landfill plumes in New England glacial materials. Cartwright and McComes (8) also traced landfill leachate plumes using the resistivity equipment, but noted that uniform soils and water table elevations were necessary for truly accurate results. While tracing plumes is one application of a resistivity study, Kelly (32) has attempted to use this technique to evaluate hydraulic conductivities in New England glacial outwash material as well. Using the results from pumping tests, he has determined empirical relationships based on the specific soil conditions in a given area. Although resistivity was not used in the Adams study, it is apparent that it is a valuable plume tracing tool, particularly at large sites where time and money are of great concern.

For actually determining what contaminants are in a plume, however, the installation of monitoring wells is required. A variety of monitoring well designs available in a wide price range, are described and evaluated by the EPA (64), Everett (16) Johnson Division, UOP (23) and Campbell and Lehr (6).

After deciding on the particular type of well and its means of installation, one must then determine how they are to be sited. If resistivity studies have been done, that information may be used to locate the probable point of maximum contamination in the plume. Another method, which is less accurate than application of resistivity data, is to assume that the groundwater follows the general topographic contour of the surrounding landforms. Although this held true for the Adams site, it can also be entirely incorrect and is therefore not recommended. A more generalized approach to siting wells is presented by Caswell (9), who reviews horizontal and vertical flow, stratigraphy of soil units, seepage velocities, contaminant densities, and aquifer stresses. An effective 'trial and error' method of well placement is to pump and analyze each well as it is installed, providing data to sketch a sectional and plan view of the plume. This information can then be used to direct the placement of the succeeding wells.

While pumping and analyzing a monitoring well may seem to be a relatively straight forward procedure, several researchers have noted large scale discrepancies in groundwater quality while using different removal methods. The main point is that wells should be purged of water that has remained stagnant in the standpipe between sampling periods. Failure to do so can lead to variations in contaminant concentrations as has been noted by Schuller, Gibb, and Griffin (56) in successive pumpings from the same well. They determined that the removal of four to six well casing volumes of water were necessary before a representative sample could be obtained, however screens located in very fine grained soils are often unable to produce such volumes, and the water obtained from the first pumping must sometimes be used for analysis. Wilson and Rouse (67) however, note that this may not present such a large problem, showing that although purging is important, overpumping can actually induce mixing of formation water and alter groundwater quality measurements. They stress that the most important consideration is a clear understanding of the local hydrological regimes before any wells are pumped.

Although well purging is an important facet of a monitoring program, a more fundamental concern may lie in the well installation. The improper drilling or construction of wells has been examined by Fetter (19). Problem sources include drilling fluids used in the boring work, PVC well adhesives, incorrect screen placement or length, any other foreign substances introduced during the preliminary phases of work, and improper bentonite seal placement (to seal out surface water). The reliability of any data ultimately depends on the correct installation of any instrumentation.

Contaminant Parameters

Certain contaminant parameters must be established in order to identify the plume location both during the well installation and during the course of the monitoring program. The choice of the constituents to be monitored is in part determined by the nature of the source material itself. In a series of sanitary landfill investigations, Coe (13) identified the predominant materials that were present in the leachate, which included organic matter, chloride, sulfide, potassium, calcium, and sodium. Several researchers (1,4,7,39,64) recommend chloride as a particularly representative parameter in leachate enriched groundwater due to its conservative nature in a variety of geochemical environments and its ease of measurement. A portable kit can be used to analyze chloride concentration in the field, thereby reducing the chance that the concentration is altered by some external source. Because chloride does not readily form precipitates with the common cations present in leachate enriched groundwater, reductions in its concentration along the length and width of the plume can be attributed essentially to dispersion, diffusion, and changes in the source strength.

The use of a single index parameter, however, may not represent the trends of the contaminant migration. A study of septic tank plumes by Childs and Upchurch (11) provides an example where cross-sectional plumes of chloride, phosphorus and nitrate through the same aquifer were shown to be dramatically different. They concluded that these three parameters have different responses to infiltration rates, inhomogeneities in the formation, loading history and other aquifer characteristics, and that, despite research to the contrary, a fair amount of chloride may have been adsorbed onto the clay soils in their study area. Another factor to be considered when using a particular parameter is the potential presence of sources of contamination other than the one being studied. The effects of brine dumping and road salting can play a significant role in increasing chloride concentrations along with carbonate bedrock recharge. The latter is particularly problematic in the New England area, where road salting is commonly practiced. Jacobson and Langmuir (28) studied spring waters emanating from folded and faulted carbonates in Pennsylvania and determined that chloride was being added. In order to address this issue, the upgradient well required by RCRA should be monitored for all parameters being measured downgradient. In addition, consultation with town engineers, bedrock maps and other references for possible sources is recommended.

Another common landfill leachate parameter is specific conductance. Depending on the presence, mobility, valance and concentration of ions, an aqueous solution can carry an electric current, the numerical expression of which is the conductivity of the solution (58). It has been shown that inorganic acids, bases, and salts are good conductors. Because of their universal presence in leachate, specific conductance is often an excellent parameter for locating a plume. Specific conductance has the added advantage of being easy to measure in the field or laboratory using relatively inexpensive portable equipment. Jacobson and Langmuir (28) however, also suggested carbonate rocks as a source contributing to the specific conductance of a water. The aforementioned increase in chloride concentration in conjunction with a relatively long contact time in the carbonates was shown to cause an increase in the specific conductance at their site. The primary consideration then is whether there is a great enough difference between background and plume concentrations to warrant the use of a specific constituent as an indicator of the plume.

The third parameter used in this investigation and recommended by several researchers is hardness. LeGrand (39) has suggested that, as with chloride, hardness moves at about the same rate as the groundwater and may be an early indicator of a leachate plume. At Adams, however, this parameter is suspect in that several sources of hardness exist in materials other than the waste. For example, Jacobson and Langmuir (28) found that carbonate rocks also increased hardness in the surrounding waters. Their research showed that CO_2 was added as a gas to the spring waters during its downward diffusion through the soil, leading to a spring water which showed three times more capacity to hold hardness and alkalinity than did stream waters in the same area. Coe (13) noted that free CO_2 also increased in soils with increasing rainfall, thus increasing bicarbonate ion production and, therefore, water hardness as well. A more critical problem with hardness measurements at Adams relates directly to the landfill itself. After choosing this parameter, it was learned that limestone crusher waste was being used as cover material so that additional hardness, as a result of solubilized calcium from the limestone will enter the groundwater system. Because this takes place at the landfill, upgradient wells will not measure this contribution to the leachate. Nonetheless, while hardness concentrations do not solely represent those generated by the waste, they are still valid as a means of locating the plume.

In summary, many parameters can be used to identify the plume of leachate enriched groundwater emitting from a sanitary landfill, but these values should not be viewed as ends in themselves as many processes may alter or enhance their concentrations other than those taking place in the landfill. For example, the well documented attenuation process by certain soils may lower the concentration of an indicator contaminant as pointed out by Jennings (29), Fetter (18), and Cartwright, Griffin, and Gilkeson (7). Although the specific soils that Jennings and Tirsch (62) used in their study had a low attenuative capacity, they point out that given correct soil conditions, the attenuation can be significant.

It is evident that the complex character of soils and their reactions with leachates is very hard to describe at any site. Matis (42) notes that the microbial breakdown and oxidation of the waste may produce additional water soluble compounds, particularly in the humid eastern section of the United States. In addition, the solubility of municipal refuse components can change markedly as a result of pH changes, aeration, dilution, drying, wetting, freezing, and thawing. Fuller, Alesii, and Carter (20) note that the bulk of these solubility changes take place within three to five years of waste placement, but can occur for a longer period of time which can be particularly critical in a plume tracing study as the migration potential of the waste is directly proportional to its solubility.

A final point to consider is the nature of the waste or wastes being monitored. At most landfill investigations the general character of the plume is the most important question addressed. Several waste types, however, may not follow the flow pattern of the plume, or may not have their highest concentrations where the indicator parameter concentrations are the greatest. The controlling factor of the behavior of these substances is their density such as in the gasoline plume migration investigated by Kramer (35), who cautions that this substance may float and travel along the top of an aquifer. Pettyjohn (47) notes that dense fluids have an opposite effect, thus having the ability to sink to the confining layer at the bottom of an aquifer and miss detection by a monitoring system which is not screened through its entire thickness.

In sum, plume monitoring is as much art as it is science, and as such, a strong sense of what one wants to learn from the study should be established early in the planning process. With this framework in hand, intuition can be applied directly with the available technology to direct the study.

Landfill Studies

The literature abounds with landfill studies and simulations designed to predict landfill behavior, such as the procedures manual published by the EPA (64). For the particulars of evaluating groundwater flow at a given site, several sources are available including a manual published by the Department of the Interior (63), which details methods of investigating and quantifying the flow in aquifers from a field perspective.

Several simulated landfills have been constructed as a way to study them in a controlled environment. Fungaroli and Steiner

(21) designed a lysimeter to approximate a landfill in a temperate, humid climate and concluded simply that solid waste landfilled in this environment will produce leachate. A more detailed study was done by Qasim and Burchinal (49) who built a series of simulated landfills in order to make predictions about volumes of certain contaminants that can be expected in landfills over time. The major finding was that deeper fills are more environmentally advantageous, in that smaller concentrations of the parameters they monitored were produced with depth in landfills above the water table. As early as 1967 Anderson and Dornbush (1) agreed with this finding, saying that groundwater was affected by waste placed in sand and gravel near a high water table. Although not a simulation as such, Tenn, Haney and DeGeare (60) have developed a desk top method for predicting volumes of leachate produced at landfills, assuming the necessary data for a given site are available. Although limited by many simplifying assumptions, their method shows unquestionably that large volumes of leachate can be produced at landfills.

Studies done at active landfill sites are also quite common, such as the two landfills on Long Island, where Kimmmel and Braids (34) observed leachate plumes of 3200 and 1500 meters (10,500 and 4900 feet) downgradient of the fill. As at Adams, chloride was one main indicator chosen for identifying the plume movement. An important conclusion from this work was that the length and volume of the plume may be more closely related to the volume of the waste than the age of that particular landfill. Apgar and Langmuir (2) investigated a landfill in Pennsylvania that, as in Adams, rested on dolomite bedrock, but unlike our study, had a water table which was over 61 meters (200 feet) below the ground surface. Again, it was shown that concentration decreased significantly with depth in this unsaturated environment. Furthermore, Apgar and Langmuir state that increased waste saturation also increases the strength of the leachate produced. This was also observed by Kunkle and Shade (36). In their conclusions, they suggest that sulfate reduction might be one mechanism responsible for this occurrence.

In a field study that has many features similar to the Adams investigation, Zanone, Donaldson, and Grunwaldt (68) installed a series of nested wells in landfills in Anchorage, Alaska. They observed that localized severe groundwater pollution was taking place, but that a deeper aquifer was being protected by an intermediate clay layer that inhibited percolation of the leachate.

In general, Cameron (5) notes that the concentration of leachate at landfills is dependent on air temperature, the depth and age of the refuse, the amount of precipitation that falls on the site, and the amount of moisture in the waste. He concludes that the single most important factor in the generation of leachate is water infiltration through the waste, but cautions that the effects at any landfill are site specific and all aspects of the site should be fully investigated.

Groundwater Flow

Another important phase of a landfill investigation is defining the regional flow patterns and aquifer characteristics including the flow direction and velocity, the size and shape of the aquifer, and whether it is confined or unconfined. This information is vital, in that the very nature of contaminant migration is defined by the aquifer characteristics. Pettyjohn notes that in shallow aquifers, such as at the Adams site, this may be particularly critical. In two separate studies (47, 48) his research has shown that shallow and surficial aquifers are subject to sudden changes in groundwater quality as a result of flushing caused by recharge events. These events, rainfall and the subsequent runoff, can move the water soluble contaminants in the solid waste into the groundwater flow.

One means of identifying both flow direction and velocity is the use of groundwater tracers. Among the most popular choices cited by Davis, Thompson and Bentley (15) are bromide, chloride, rhodamine WT dye and fluorocarbons. By injecting one of these substances into a well and subsequently analyzing the water withdrawn from adjacent wells, flow characteristics such as velocity and direction can be determined. Keswick, Wang and Gerba (33) suggest that bacterial viruses are the best tracers because of their size, ease of assay, and lack of pathogenicity. The shortcomings of this method are generally due to incorrect tracer choice for given field conditions, insufficient tracer concentration at the point of injection, and a lack of understanding of the hydrology at the site prior to injection.

Groundwater flow and contaminant transport in and around wetlands such as those below the Adams landfill have been the subject of several studies. Motts and O'Brien (44) provide a good overview of the wetlands in Massachusetts, their formation, associated geological features, and how they are affected and in turn affect groundwater flow. Larson (37) and Reppert (51) also report on wetlands and their influence on groundwater recharge insisting the contribution is low and that these areas may in fact exist solely as discharge zones throughout most of the year. According to Saines (54) an area of discharge is one with increasing hydraulic head with depth, and information showing this to be the case in a wetland would help to explain its role in the regional groundwater flow patterns. In fact, it can be demonstrated that removal of water from the wetlands is the result of several mechanisms, some of which are seasonal in nature.

One seasonal mechanism aiding in water discharge is evapotranspiration. Motts and O'Brien (44) note that controversy still exists over whether climate or vegetation is the greatest driving force, but conclude that large volumes are nonetheless being removed during the summer months. Another factor governing flow at the Adams site is the storage of water. Williams (66) reports that water collected in areas surrounded by low permeability material may form groundwater mounds and it has been shown that conditions for this occurrence exist both under landfills and in wetlands. Existence of such a condition may greatly alter the normal groundwater flow patterns and affect interpretation of data, particularly under a landfill where the water has a greater volume to occupy and is readily absorbed by the waste. In the wetlands, the fine grained soils provide the absorptive capacity. The results of Williams' research led him to conclude that some wetlands act as groundwater sinks, while others may assume groundwater mound characteristics. As precipitation intensity and evapotranspiration rates increase, these closed basins may convert from discharge to recharge areas, and large ratios of surface drainage area to marsh area in regions of low relief may cause a reversal in the groundwater flow gradient. These changes may occur on a seasonal basis or react quickly to a sudden high volume storm event.

As a means of both quantifying and visualizing the flow patterns in an area Caswell (10) points to the use of multiposition piezometers, such as those installed at Adams. Piezometers are installed to determine the water level at a specific depth. When the level is converted to an elevation, one can use this value as a flow potential. By connecting equal potential points on a scaled cross-sectional drawing a flow net can be constructed to provide information on the groundwater flow direction. Using data on permeability and head, the flow net provides a rough estimate of groundwater discharge for the section.

Because the wetland groundwater flow conditions are variable, so then must be any leachate enriched water traveling with the mass. Consequently, monitoring a contaminant plume under these conditions may defy the use of the standard equations defining groundwater flow through porous media. Contaminant uptake also behaves differently in wetlands due to the nature of the fine grained soil through which it flows. Reppert (51) reports that a vehicle for pollutant reduction exists either in groundwater flow or by streams that flow through the wetland. The mechanisms by which reduction occurs in wetlands can be mechanical dispersion (e.g. 'filtering'), physical adsorption, chemical precipitation, ion exchange, or biological uptake. The ultimate mechanism(s) and reduction are dependent on the type of wetland and the nature of the vegetation.

One method for the removal of contaminants is the evapotranspiration process. Williams has shown that the loss of dissolved solids and hardness to plants during this process can be quite high. Kadlec (30) has also indicated the ability of wetland vegetation to remove heavy metals and chlorinated hydrocarbons. Pettyjohn (48), however, warns that if high evapotranspiration rates are occurring, increased salt concentrations could be drawn to the ground surface. If a large scale rainfall were to subsequently fall on the basin, this zone of contamination could be flushed back into the aquifer as a new high concentration contaminant mass. Additional contaminant sources may even be created by marshes and wetlands according to a study by Cook and Powers (14) conducted on artificially created marshes in New York State identifying excessive concentrations of iron and manganese which they believe were derived from the wetland plants and soils.

The conclusion from this discussion should be that groundwater flow in and around wetlands can be a complicated and variable process. Cyclic changes can occur due to temperature, rainfall, groundwater elevations, soil conditions, and several other factors. Pettyjohn (47,48) has found that confining beds, such as the one associated with the shallow wetland aquifer at Adams, could cause a perched mass to migrate laterally and eventually discharge via springs or seeps. Vertical flows may also occur as a result of soil permeabilities, contaminant density, hydraulic gradient, and other contributing factors. Finally, the gradient of flow may even reverse in response to the ever changing conditions in the wetland. Only through continued monitoring and investigation of all of the wetland dynamics can this process be understood.

Ultimately, the results of any field study are only as valuable as the data collected. One must be wary of rejecting that data which does not support the original hypothesis and anticipated patterns. Supporting data is also necessary for establishing the validity of trends of groundwater movement and contaminant migration. Additionally, the mere installation of three downgradient wells and the subsequent evaluation of their groundwater samples, will not accurately represent the condition existing at a given site. Only by using more of the tools discussed thus far can the true impact of a landfilling operation be evaluated.

CHAPTER III

BACKGROUND HISTORY: ADAMS SANITARY LANDFILL

The potential for leachate generation exists at any solid waste landfilling facility. The details of that generation, however, are site specific. The geologic setting of the site and the human influences surrounding the disposal of the waste, are factors playing major roles in leachate production and its subsequent migration. A thorough background history on these details will help to provide a clearer overall picture of the impact of that site on its surroundings.

Introduction

The research for this study took place at the Adams Sanitary Landfill, in Adams, Massachusetts (See Figure 1), a site owned and operated by the Town of Adams for approximately 40 years. The original intent of this work was to locate a site whose groundwater flow patterns were easily defined and where a strong potential for groundwater contamination existed. Information from state DEQE files confirmed that the latter requirement could be found at Adams.

The surficial features in the vicinity of the landfill, which are described in detail in a later section, provided what we believed to be a straight forward flow condition through the site. The landfill is located at the base of a relatively steep hillside, on the edge of the Hoosic River floodplain (See Fig 2). Between the landfill and a large grazing field that runs to the edge of the Hoosic River, is a small wetland. Running through the wetland is a small stream which is fed by a series of brooks that wind down the hillside. This stream drains to the north and eventually discharges into the Hoosic River. Initial observation of streams and landforms on the hillside allowed us to assume that the material was a relatively permeable stratified drift, and thus provided an area of groundwater recharge for the valley floodplain below.

For many of the early years of operation, the waste was merely dumped from the road above the landfill and allowed to collect in springs and seeps that emanated from the base of the slope below. Rather than covering this material, the common practice was to burn the waste. In more recent years, the Adams landfill has been cited for lack of suitable space to landfill,

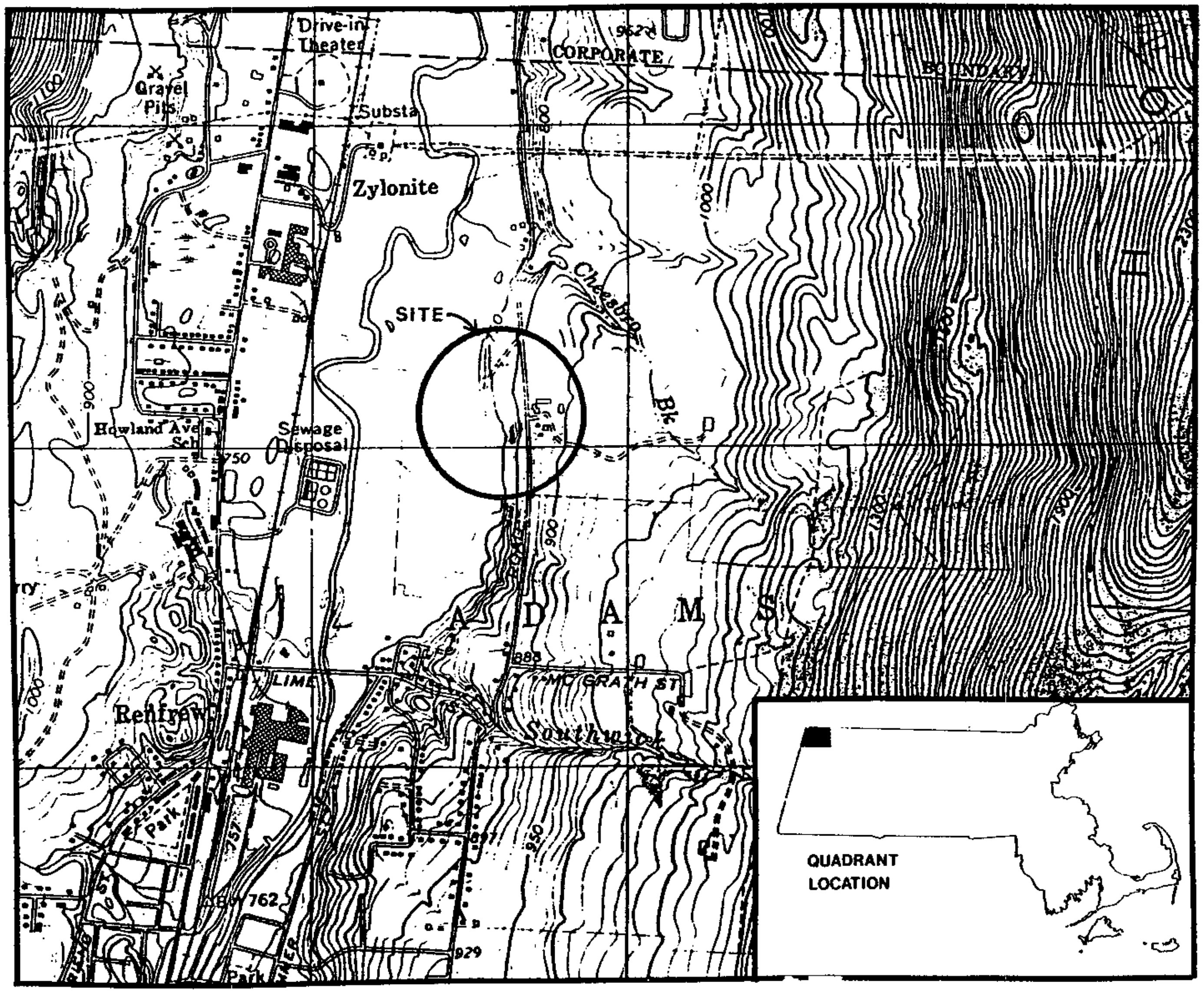


Figure 1. Adams, MA: Site Map

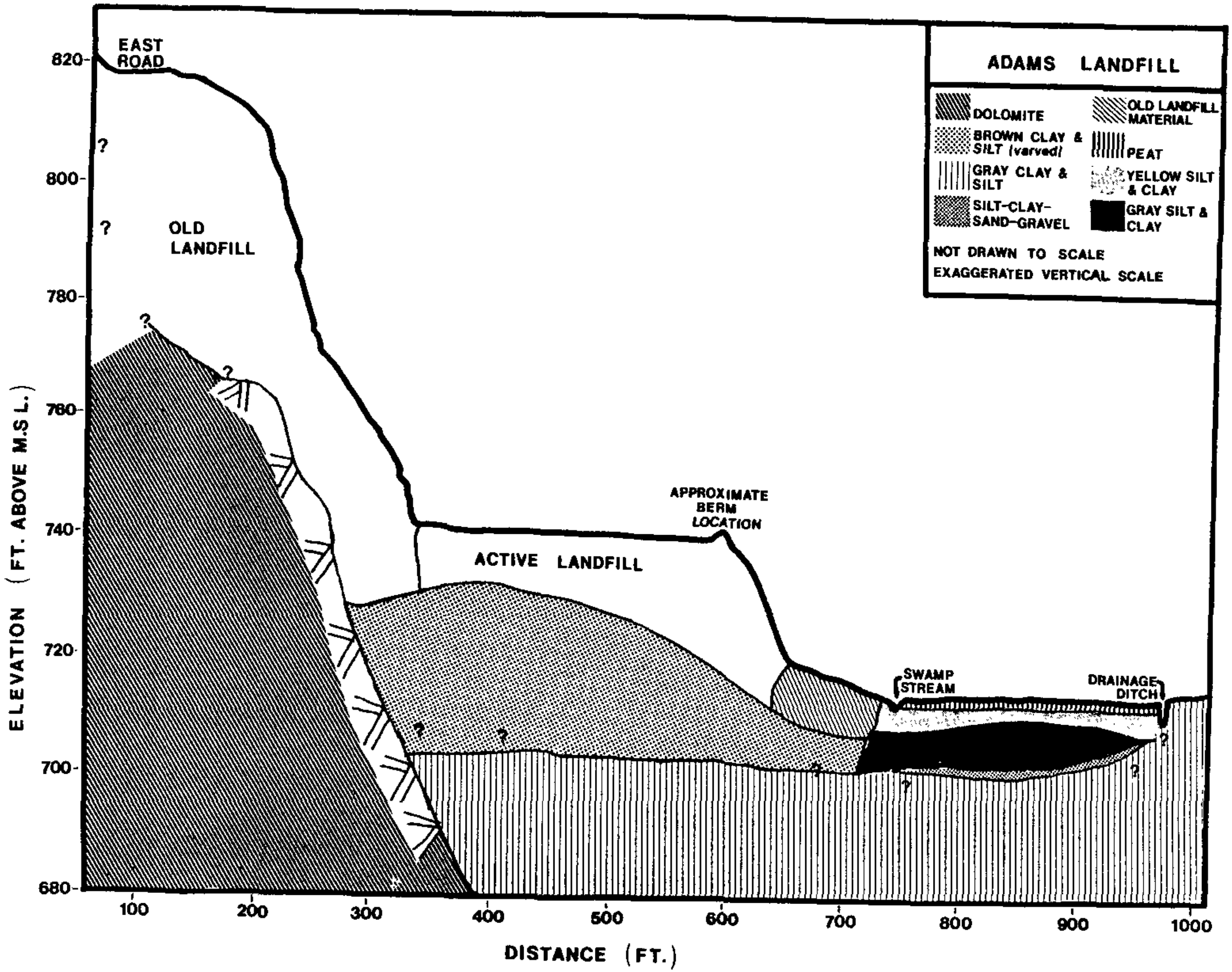


Figure 2. Adams Landfill: Section View

improper landfilling operations, objectionable appearance, public health hazards, and adverse effect on surface water quality (41). Additional evidence of potential groundwater pollution was obtained after observing leachate seeps and springs from the base of the fill during site visits.

In a study of refuse disposal sites on floodplains, Palmquist and Sandelein (46) define groundwater contamination enclaves as flame-like plumes that travel parallel to the groundwater flow lines in the downgradient flow direction when viewed in the cross section. The three-dimensional shape of a plume emanating from the landfill, therefore, was described as a tongue-like lobe whose leachate concentrations are highest along the central axis of that lobe. This description is not unique and finds widespread acceptance. The cross sections of plumes in the Kimmel and Braids study (29), and others follow this shape explicitly.

It has been shown by Gass (22) and others that surface features at a site do not necessarily describe the groundwater flow direction or availability. The steeply sloped valley floodplain features, however, do define a more invariable flow setting. Rahn (50) has already established that much of the water infiltrating steep hillsides in valley floodplains, moves down valley through the glacial drift to its discharge in the associated valley water bodies. In the case of Adams, these would be either the Hoosic River and/or the wetland and its stream. Because the landfill lies between the hillside recharge and the floodplain discharge, it was assumed that downvalley underflow must pass through the waste in the landfill. Thus, this water would solubilize components in the refuse introducing them into the groundwater flow and transporting them downgradient through the wetland aquifer, discharging the bulk into the Hoosic River. Although it is recognized that waste plumes may vary somewhat from the regional groundwater patterns, it was with this scenario in mind that the Adams site was chosen.

Geologic History

The landforms in the Hoosic River Valley are controlled chiefly by the bedrock in the area. Schists, gneisses and other metamorphic rocks comprise the hillside and upland formations, while carbonate rocks predominate in the valley. The Adams landfill, at the surficial contact of upland hills and valley bottom, bears on the Kitchen Brook dolomite unit.

The Kitchen Brook dolomite is approximately 1000 feet (305 meters) thick and consists of dolomite and differing percentages of quartz, mica, and feldspar. As with the abrupt change in the landforms, the geologic contact of the Kitchen Brook dolomite and the upland Dalton formation is a sharp lithologic break. A suspected thrust fault exists at the contact, and occurs approximately 1500 feet (457 meters) upgradient of the landfill.

The Kitchen Brook carbonate deposition took place during early Cambrian time in shallow seas that covered the area. Detrital quartz, whose origin is believed to be from the highlands to the north or west was deposited in conjunction with the deposition of the Kitchen Brook. Regional metamorphic periods followed, altering much of the highlands, but not affecting the valley carbonates to any great extent. Localized marble units, however, can be found throughout the region.

In more recent geologic time, periods of glaciation have encroached upon this area of the Berkshire Valley Lowland and produced the landforms that are now in place. The less resistant carbonates, including the Kitchen Brook dolomite, were scoured by glacial action, and the typical 'U-shaped' glacial valley was formed. The more resistant schists, gneisses, and quartzites underwent less deformation and scour, and thus form the surrounding highlands.

Surficial geology in the area is typical of the Berkshire Valley, with the highlands generally consisting of units of till and poorly to non-sorted stratified drift. The latter unit can have very high permeabilities and therefore may represent an area of groundwater recharge. Valley surficial geology is comprised of alluvium, alluvial fan deposits, and localized swamp deposits, comprised of alternating sands and silts with an occasional gravel stratum or lens. A thick clay layer deposited by glacial Lake Bascomb is also found across the valley bottom, and recent flooding events have added additional thin silt and clay layers to the valley soils (27).

The surficial geology in the study area at the base of the Adams landfill is representative of valley deposits, and as noted, this area bears on the Kitchen Brook dolomite. It is likely that glacial meltwaters solubilized an isolated section of the dolomite, forming a localized topographic depression. Glacial lake and floodplain clays subsequently filled this depression, as well as much of the immediate Hoosic River Valley. The blue-gray clay, which appears to become siltier with depth, is at least 6.1 meters (20 feet) thick beneath the study area. Test borings 610 meters (2000 feet) north of the landfill indicated that this unit was greater than 18.2 meters (60 feet) thick (57). Above the clay unit, borings indicate that a narrow, poorly sorted mixture of clay, silt, sand and gravel sized material was emplaced. A few feet to a few inches in thickness, this layer probably originated in the coarser, upland stratified drift sediments. The unit appears to pinch out towards the far side of the wetland, perhaps the result of lodging in the depression of the clay unit beneath it. Following this short-lived placement, low permeability, alluvial, yellow and gray silts and clays were laid in place. Poor drainage in this low lying area gave rise to the growth of swamp vegetation, whose subsequent decomposition accounts for the 0.3-0.6 m (one to two foot) thick peat unit at the ground surface.

The area underneath the present landfill is believed to be the contact between the alluvial valley soils and the stratified drift and dolomite of the highlands. Borings done prior to the placement of waste in the area indicate that the Kitchen Brook dolomite dips quite steeply from the hillside, west to the valley bottom, and is overlain by stratified drift and possibly a thin, low permeability till. The thin, poorly sorted stratum beneath the wetland appears to be hydraulically connected to the upland deposits, either drift or dolomite, and may provide a source of recharge to the landfill and/or wetland. Many springs and seeps have emanated from the base of the hillside, and it is reported that waste placement began in this discharge area. Additional soils and waste continued to be landfilled at this site in its 40 years of operation, bringing the site to its present elevation. A generalized cross section of the area described is shown in Figure 2.

Site History

The Adams landfill has been in operation at the present site for approximately 40 years. For the bulk of that time the facility was operated as an open dump with little or no cover material being used. Waste material dumped at the site has consisted of primarily municipal, light industrial, and demolition refuse, with the bulk being transported to the site by individual homeowners or employees. Old engineering reports document that much of this waste was being disposed of by end dumping from East Road to the base of the hill below. The bottom of this slope has been previously identified as a discharge zone for the upland groundwater in the form of seeps, springs and small streams. This practice, and the exposure of the waste to direct precipitation, served to produce large quantities of leachate which flowed directly into the wetland stream (swamp stream) or percolated through the soils and entered the groundwater flow regime.

On April 21, 1971, the regulations administered by the Massachusetts Department of Environmental Quality Engineering

(DEQE) (formerly Department of Public Health) concerning landfill standards were issued. These regulations were designed to prevent or minimize the occurrence of air and water pollution adjacent to waste disposal facilities. Despite certain modifications to the site the Department of Public Health and the Division of Water Pollution Control (DWPC) determined that the Adams landfill did not conform to the standards, and required that the town upgrade its solid waste disposal facility.

In September of 1974, Adams hired the firm of C. E. Maguire, Inc. to evaluate current practices and recommend remedial measures. As a result of their investigation, C. E. Maguire recommended that the existing landfill be closed, and be relocated immediately north of its present site. The problems at the old site which prompted this action have already been cited, the most critical of which, with respect to our research, was the adverse effect the landfill had on both surface and ground water quality.

Following the recommendations of C. E. Maguire, the town installed a crushed limestone earth berm at the existing site to separate the waste from the floodplain and wetland. Rather than entirely inhibit the flow of leachate from the face of the landfill, the berm was designed to allow that leachate which did pass to be filtered by the high pH calcium carbonate. C. E. Maguire's main intent was to reduce the BOD of that liquid which was already seeping from the face of the landfill. This procedure was followed by covering all of the exposed waste, grading the slope and seeding the site to minimize erosion, and the installation of a stormwater diversion system.

By 1977, Adams realized that the new landfill site was being filled at an alarming rate, and began searching for an alternative disposal location. After examining several sites, it was determined that better management and increased compaction from the purchase of new compacting equipment at the existing site, was the best alternative. At the recommendation of DEQE, a more experienced operator began managing what is now the present site of landfilling and the focus of this study.

CHAPTER IV

WELL INSTALLATION

One of the main objectives of the research at the Adams landfill was to detect and evaluate potential or existing ground and surface water degradation by landfill leachate. In order to accomplish this, a variety of tests had to be performed and various types of instrumentation installed. The data obtained provided a large portion of the information necessary to make a thorough evaluation of the impact of the site.

Preliminary Exploration

Given our assumptions concerning both groundwater flow direction and the source of contamination, we chose to begin installation of groundwater monitoring equipment while simultaneously researching any prior investigations done in and around the site. The first phase of our research was to try to identify the point of maximum contamination in the groundwater downgradient of the landfill. A rough estimate of the location of the contamination enclave would provide useful information for the siting of groundwater monitoring wells.

To obtain this data, several shallow hand augered holes, one inch in diameter, were made in an area downgradient from the landfill, in the wetland. After augering the holes groundwater, which had risen to the ground surface, was pumped out and sampled for specific conductance, temperature, and salinity. The results, which are shown in Figures 3, 4 and 5 indicate a general contamination trend and were used to site our first well. Drilling at well #1 began on November 15, 1981.

Wash Boring Installation

The site of our first well is along the swamp stream that meanders generally in a north-south direction through the swamp. It is also positioned about 45.7 m (150 feet) beyond the edge of active filling and within just a few feet of old fill material which at one time was being placed nearer to the wetland. Figure 6 shows the location of well #1 with respect to the surrounding soils. The site of this well lies on a line parallel to the assumed direction of flow, and through the approximate center of

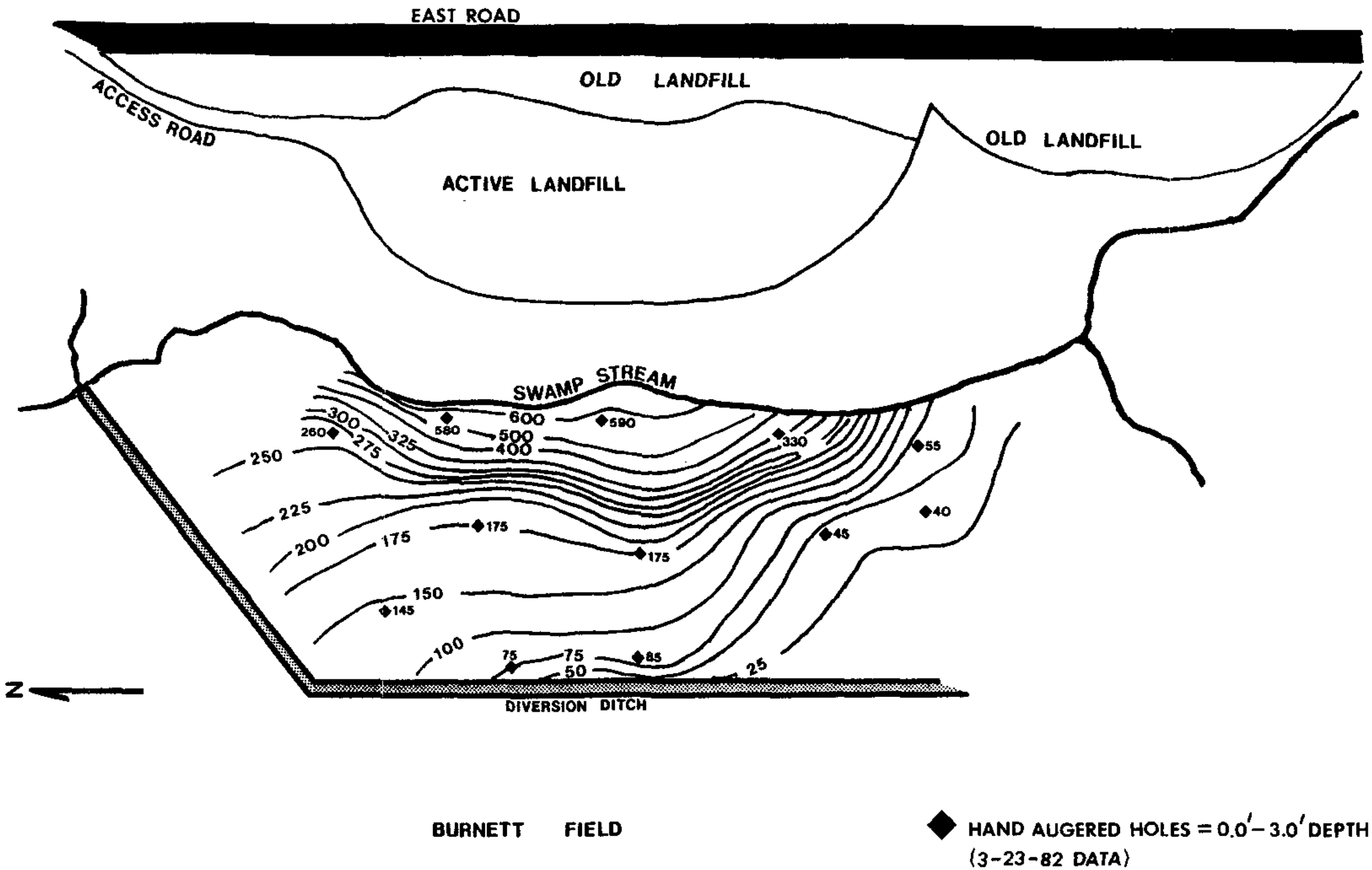


Figure 3. Adams Landfill: Plan View; Specific Conductance Contours

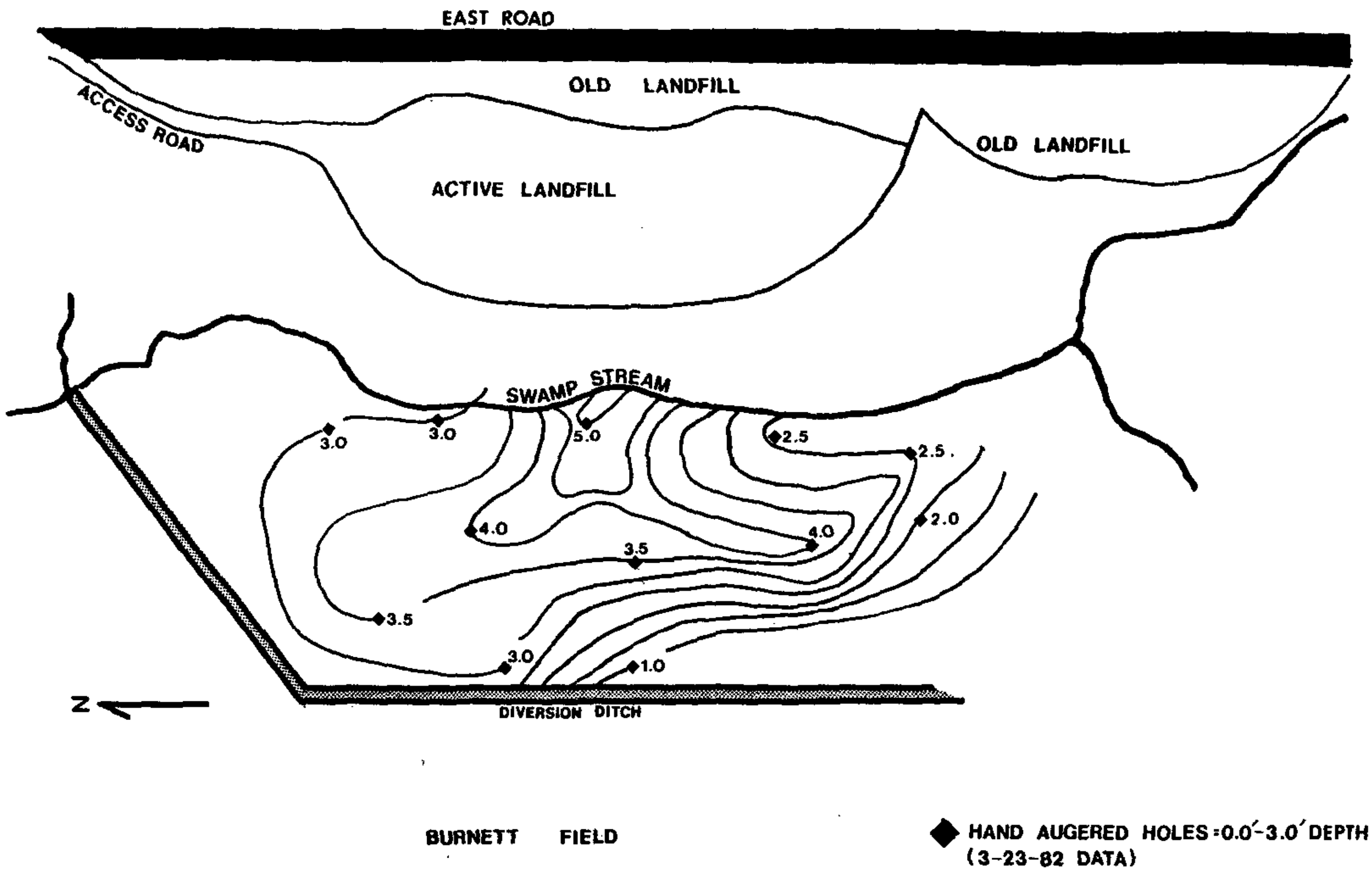


Figure 4. Adams Landfill: Plan View; Isotherms

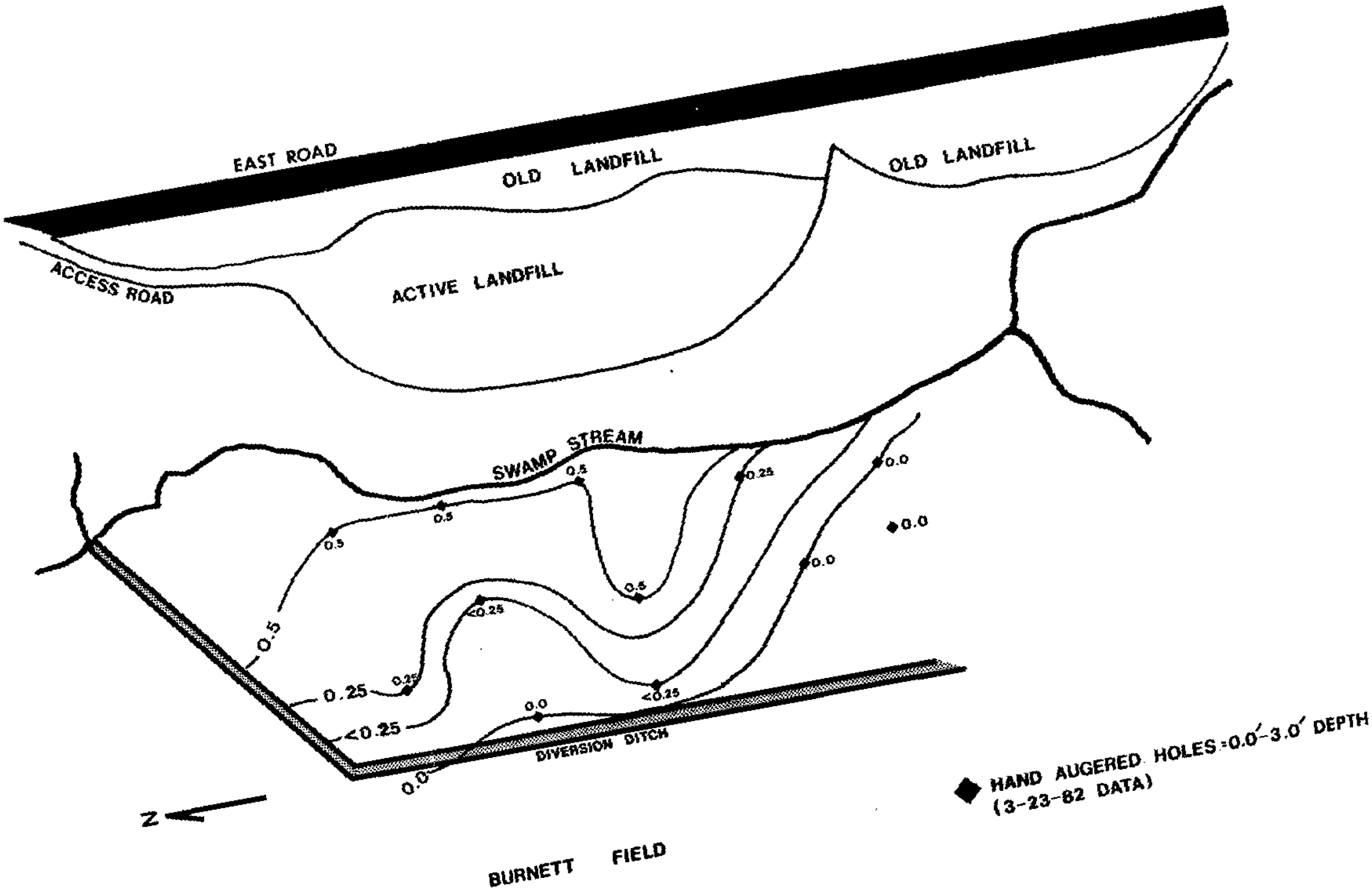


Figure 5. Adams Landfill: Plan View; Salinity Contours

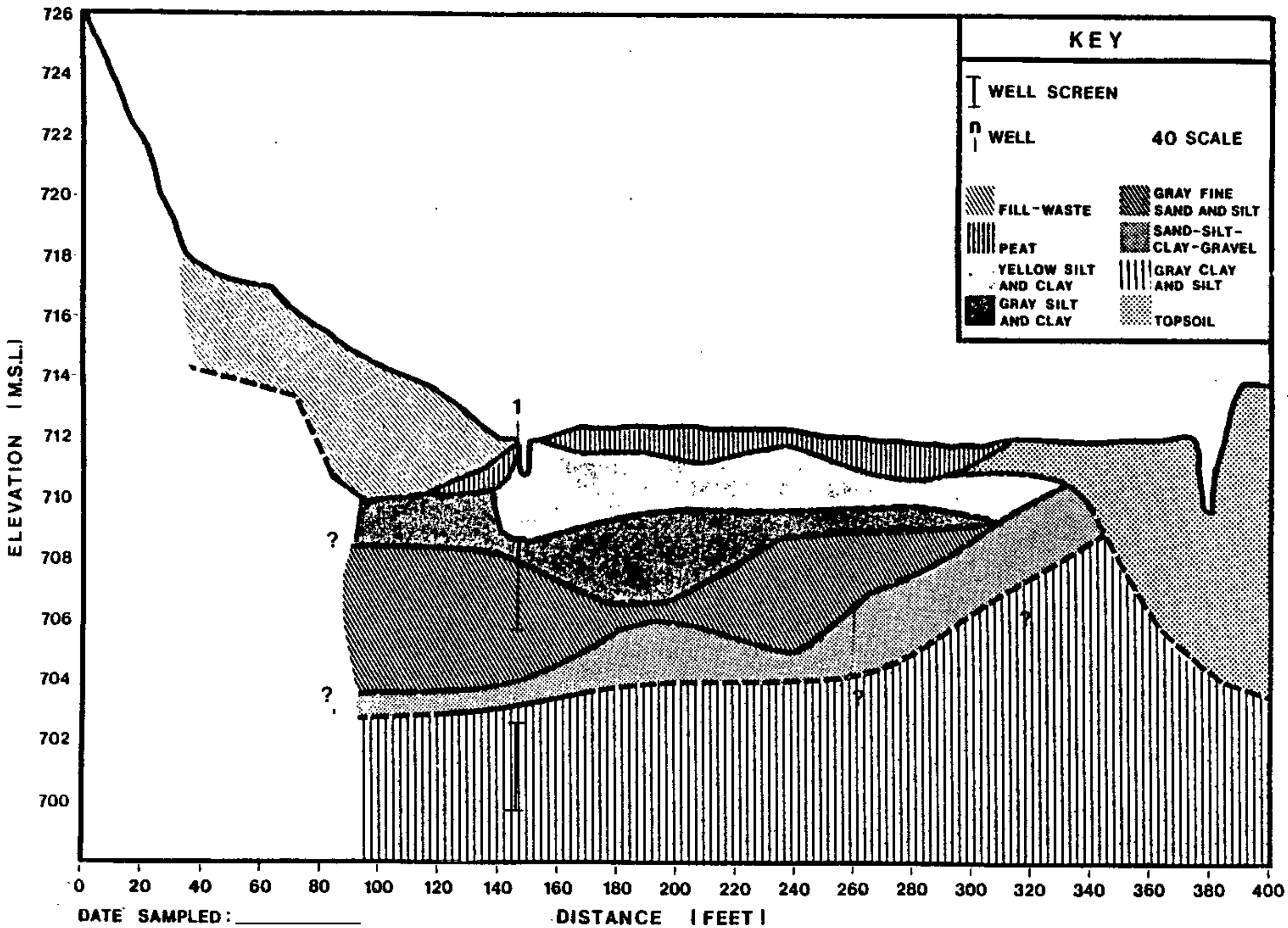


Figure 6. Complete Section View of Main Well Line (Well screens from 16-19 and 33.7-36.7 feet not shown. Gray silt and clay continued until 36.7 feet.)

the active filling area. Based on the results of the hand auger study, it was believed that it would detect the most central portion of the contamination enclave. Fuller, Alesii and Carter (20) also note that the bulk of leachate production occurs within three to five years of placement. Thus the leachate production would probably be greatest at the center of the active landfill. The well was also positioned just at the edge of the swamp stream, a known discharge zone for groundwater, and thus intended to intercept seepage flow migrating toward the stream. The location of well #1 is also at the approximate interface of old fill and waste, and the wetland soils, thereby intercepting a potentially concentrated leachate plume.

The installation of this well was accomplished by means of a wash boring drill rig. The rig consisted of a Milwaukee 5 HP engine with a rotary cat head and suction pump. This apparatus is mounted on a skid frame to facilitate movement. A pulley is suspended from a tripod attached to the rig, in order to hang casing driving weights.

The procedure for installing wells and sampling soil is fairly straight forward, as shown in Figure 7. A 136 kg (300 lb) weight with a hollow core is suspended from a rope which is passed over the pulley, and wrapped around the rotating cat head. A five foot length of hollow 6.35 cm (2.5 in) casing is pushed as far as possible by hand into the soil, and an additional special section of pipe is connected to the casing. This section, termed the drive head, has a larger 'donut' shaped piece that slides over its shaft. The hammer is then lowered over this same section and, with the aid of the cat head, drives the casing into the ground with a hammering action. The drive head and 300 lb hammer are removed, and water is pumped down the inside of the driven casing. This 'jetting' action removes the soil from the annulus of the casing to a depth equal to the depth of the pipe in the ground.

Sampling with a wash boring rig involves thinner pipes that fit inside the newly washed out casing. Attached to the end of one of these pipes is a 'split spoon' sampling device that is driven into the formation. The hammering procedure is the same as the casing driving procedure, but a 63.5 kg (140 lb) hammer is used in place of the 136 kg (300 lb) hammer. The sampler is driven 0.46 m (18 in) into the soil and removed, thus obtaining a sample from the desired depth. Although slightly disturbed by both the washing and the hammering, the sample is representative of the soils at that depth. A new piece of casing is then attached and the procedure is repeated.

Although the wash boring drilling technique is very effective when truck mounted, the skid mounted system was found to be very

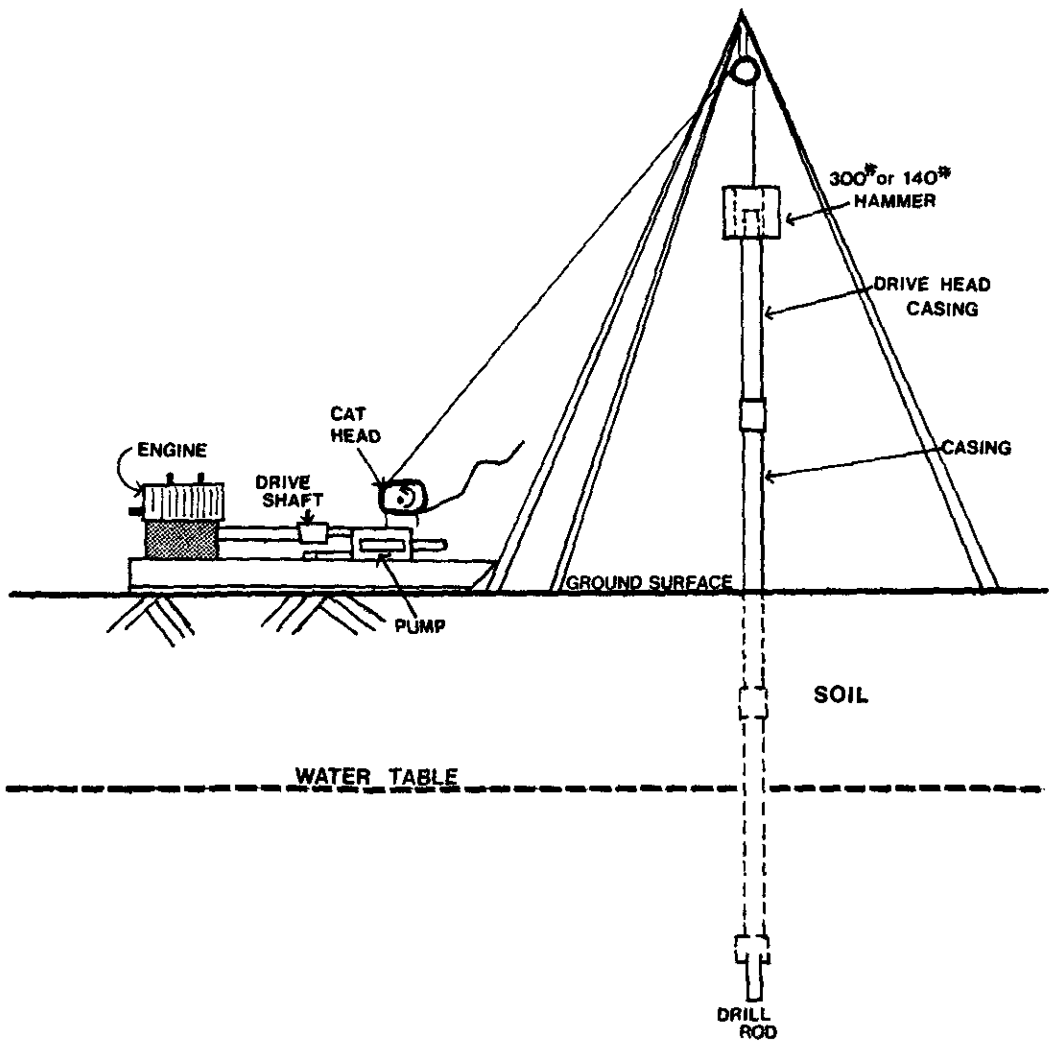


Figure 7. Wash Boring Drilling Rig

cumbersome. Movement of the rig at the Adams site had to be accomplished by placing it in the bucket of a front end loader. The organic soils found in the wetland were unable to support the weight of the loader and thus the location of well #1 was, in part, due to this limitation. The final reason for this well location is also a function of the drilling technique. Because a public water supply was unavailable, an alternate water source had to be located to provide wash water for the jetting process. Although the swamp stream is shallow, sufficient water was available there to accomplish this task.

Well #1 was drilled to a depth of 11.1 m (36.5 ft) with samples taken every five feet. Figure 8 is a well log of that boring. A total of four, 0.88 m (2.9 foot) slotted, PVC well screens attached to 1.3 cm (1/2 in) PVC pipe were installed at various depths in the hole. (PVC was chosen because of its ability to remain inert and durable under a variety of field conditions.) The procedure for installing the well screens was as follows. The screen was cemented to a 3.1 m (10 foot) length of PVC pipe and lowered into the hollow casing. Additional sections of PVC were attached until the well screen came to rest at the bottom of the hole, and a volume of Ottawa sand equal to approximately 1 m (3.0 ft) of casing was poured down the hole. The sand allows water moving through that soil unit to more easily enter the well screen. The size of sand chosen was meant to be small enough to filter out fines that might enter the screen, while large enough to not pass through the screen slots. Particle sizes, however, were smaller than anticipated and fines did plague the water removal process. The groundwater is confined at the site and thus flowed gently out of this well. Pouring sand through the flowing water proved to be both problematic and time consuming.

When the sand had come to rest around the well point and probes indicated the screen to be fully embedded, a five foot section of casing was pulled up and removed. The effect of this maneuver is to have the natural soils cave in around the Ottawa sand and hold it firmly in place. After that, bentonite pellets were poured down the casing to provide a vertical seal at the top of the well screen. This low permeability material isolates the well screen from other pressure environments and the resulting 'piezometer' provides a means of measuring pressure at the elevation in the aquifer where the screen is located, while simultaneously serving as a sampling location at a prescribed depth in the aquifer. Additional Ottawa sand was poured down the hole and the procedure was repeated for the remaining three piezometers in the same well. A final surface pack of bentonite was added to inhibit runoff from entering the higher permeability sand that had been packed around the piezometer nest. The

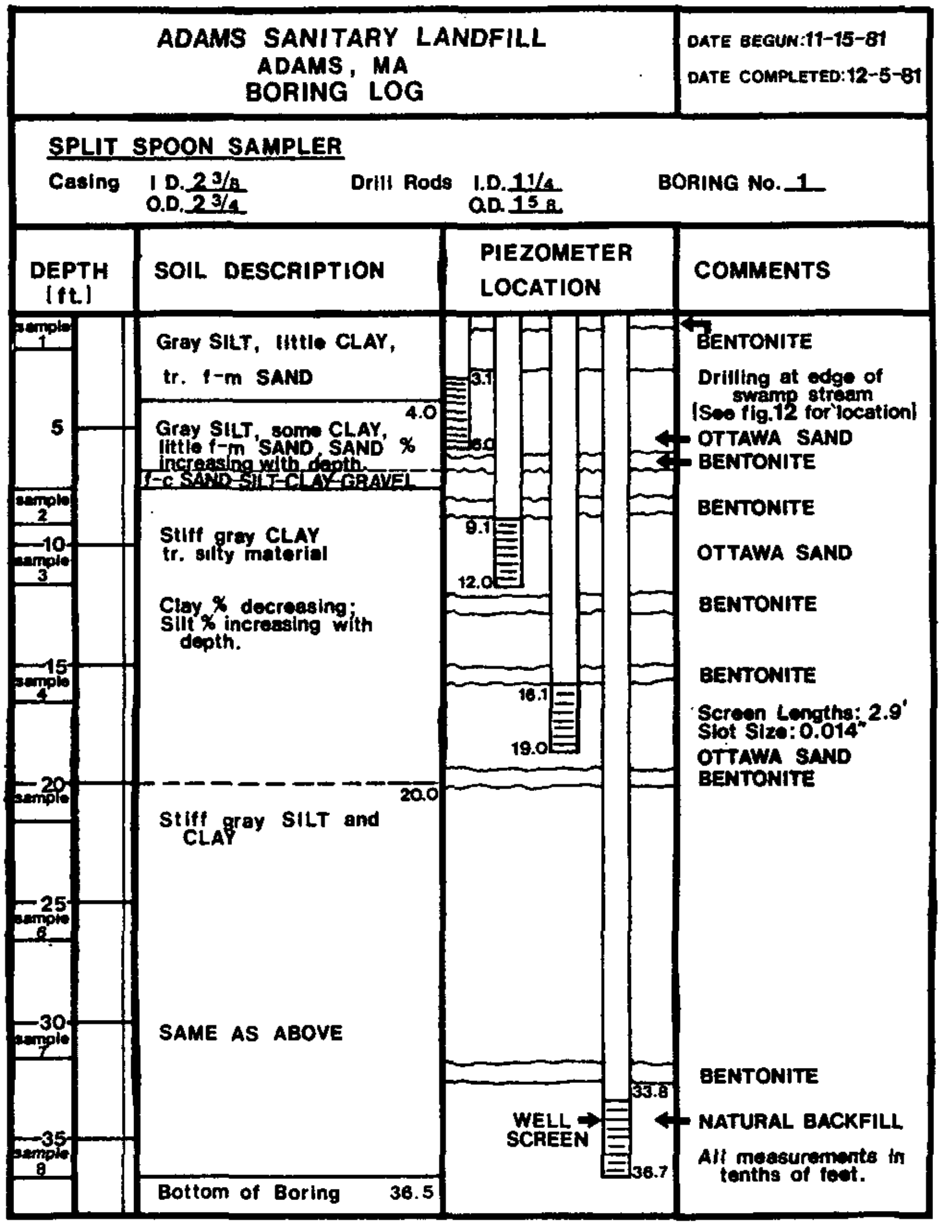


Figure 8. Adams Sanitary Landfill: Boring Log; Well #1 (Also showing screen locations for the four piezometers installed in this hole.)

completed well consists of four 1/2 inch PVC pipes, open to the air, and projecting from the ground surface, identified with an indelible marker for future sampling procedures.

Hand Augered Installation

The boring log from well #1 indicated that low permeability material was located at a depth of 2.3 m (7.5 feet) and continued until the bottom of the boring at 11.1 m (36.5 feet). The boring also established that the soil was saturated from the ground surface to at least this 7.5 foot depth, thus, if contaminants were migrating in the upper 2.3 m (7.5 feet) of soil, the low permeability material might provide an effective boundary layer to prevent the leachate enriched groundwater from penetrating any deeper. A conversation with geologist Jack McFadyen of Williams College, an investigator familiar with soils and soil exploration in the area, confirmed this hypothesis (43). Professor McFadyen described the entire area as being underlain with an impermeable gray clay unit which was the upper confining layer of a deeper artesian aquifer. Boring logs done elsewhere in the Hoosic Valley indicate that this unit may be in excess of 24.4 m (80 feet) thick.

On the basis of this information, the 2.54 cm (1.0 in) diameter hand auger, used in the groundwater quality investigation discussed previously, was now used to install well #2. If a shallow (4.6 m (<15 feet) below ground surface) well is to be installed in fine grained soil, the hand auger is recommended for several reasons. First, the necessary equipment used in this procedure is inexpensive relative to other drilling techniques. Secondly, installation of well points is both fast and easy. At Adams, for example, four wells with two well screens each were installed in under eight hours. Finally, sampling is done continuously as the hole is being augered and strata changes can be detected with a high degree of accuracy. Installation of the remaining 20 wells was accomplished with the use of the hand auger. The well design (i.e. sand, bentonite, and well screen placement) was identical to the procedure described for well #1.

Well Placement

Owing to the ease of installation, the location of the majority of wells proceeded on a trial and error basis. In other words, as each was installed, water was pumped from the well and sampled for the three leachate parameters, the results directing the placement of the subsequent wells. Wells two through eight were installed in order to locate the extent of groundwater contamination parallel to the assumed direction of flow. Figure 9 is a sectional view of the location of these wells.

Well #2 was placed across the swamp stream from well #1 in an attempt to determine what percentage of groundwater, if any, was being discharged into the swamp stream. A significant decrease in the concentration of a conservative parameter such as chloride would be a possible indication that this was occurring. The placement of well #3 was due in part to logistics. A 'worst case' well was sought in order to evaluate the highest leachate concentrations entering the aquifer beyond the landfill. Well #3 was as close to the active filling area as the landfill operator would allow. With wells one, two, and three in place and analyzed, it was clear that a reduction in concentration was taking place in the assumed direction of groundwater flow. In addition, it also appeared that the concentrations were decreasing with depth as observed in the multiposition piezometers in wells 1 and 2. Chloride concentration data from just these three wells is shown in Figure 10.

The field adjacent to the wetland (see Figure 11) is presently being used for both crop production and cattle grazing. Because machinery must periodically harvest hay, no wells were permitted in this field and the far edge of the wetland, at the diversion ditch, is thus designated as another boundary of the study area. With this in mind, well #4 was located on the far side of the wetland. The installation of this single piezometer proved to be the most difficult in the well field. A thin unit of coarse material, previously identified in well #2 at a depth of approximately 2.3 m (7.5 feet) now appeared at 1.1 m (3.5 feet). While the auger is an excellent tool for fine grained soils, material of coarse sand size and larger (>2 mm) is difficult to auger through. In addition, the action of scraping the auger tip against these materials tends to dull the auger screw and decrease its effectiveness. Despite the presence of a fine grained matrix in this coarse unit, it was assumed that groundwater could travel at the greatest velocity through this zone. Consequently, a single well screen was positioned in this stratum. Results of analyses done on the water pumped from this well showed it to be of significantly better quality than any of the previously installed wells.

Wells 5 through 8, which continue to parallel the flow direction, were installed in a similar manner. A cross section similar to the one shown in Figure 10 of the aquifer was sketched. Concentrations of the various parameters were placed at the position of the well screen in the aquifer as the well points were

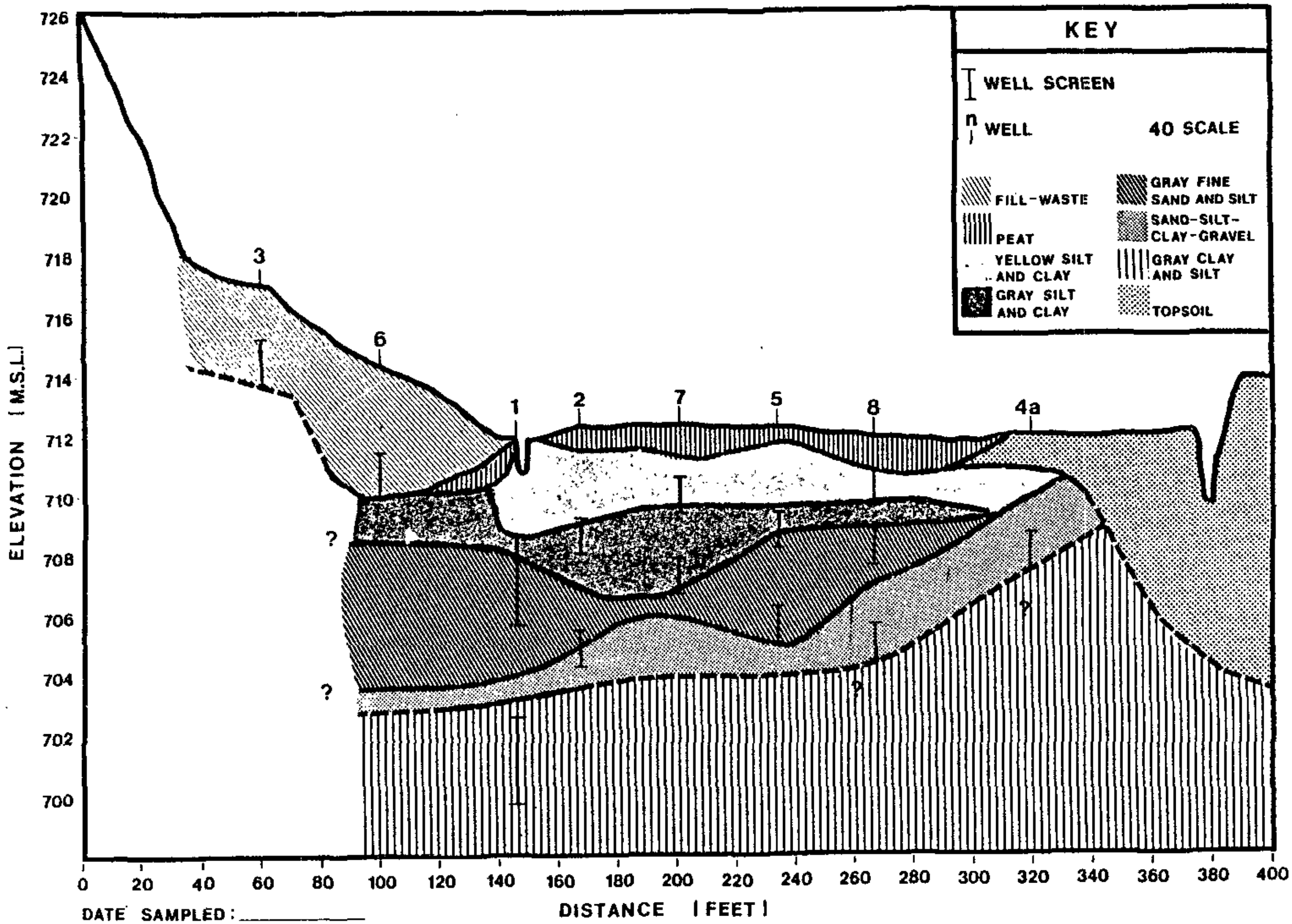


Figure 9. Well Location: Section View along the Central Line of Wells

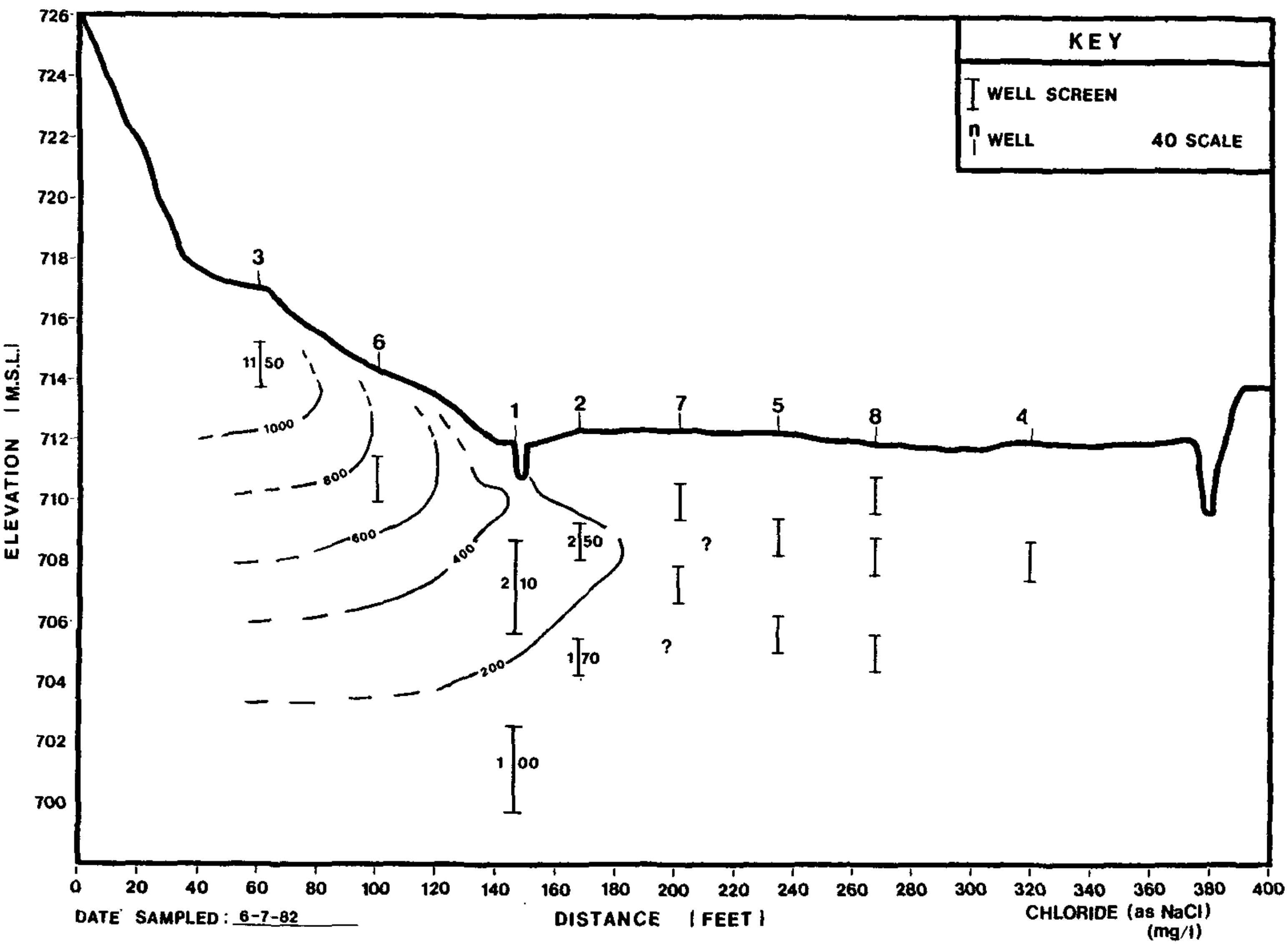


Figure 10. Concentration Data and Plume Location: Wells 1, 2 and 3

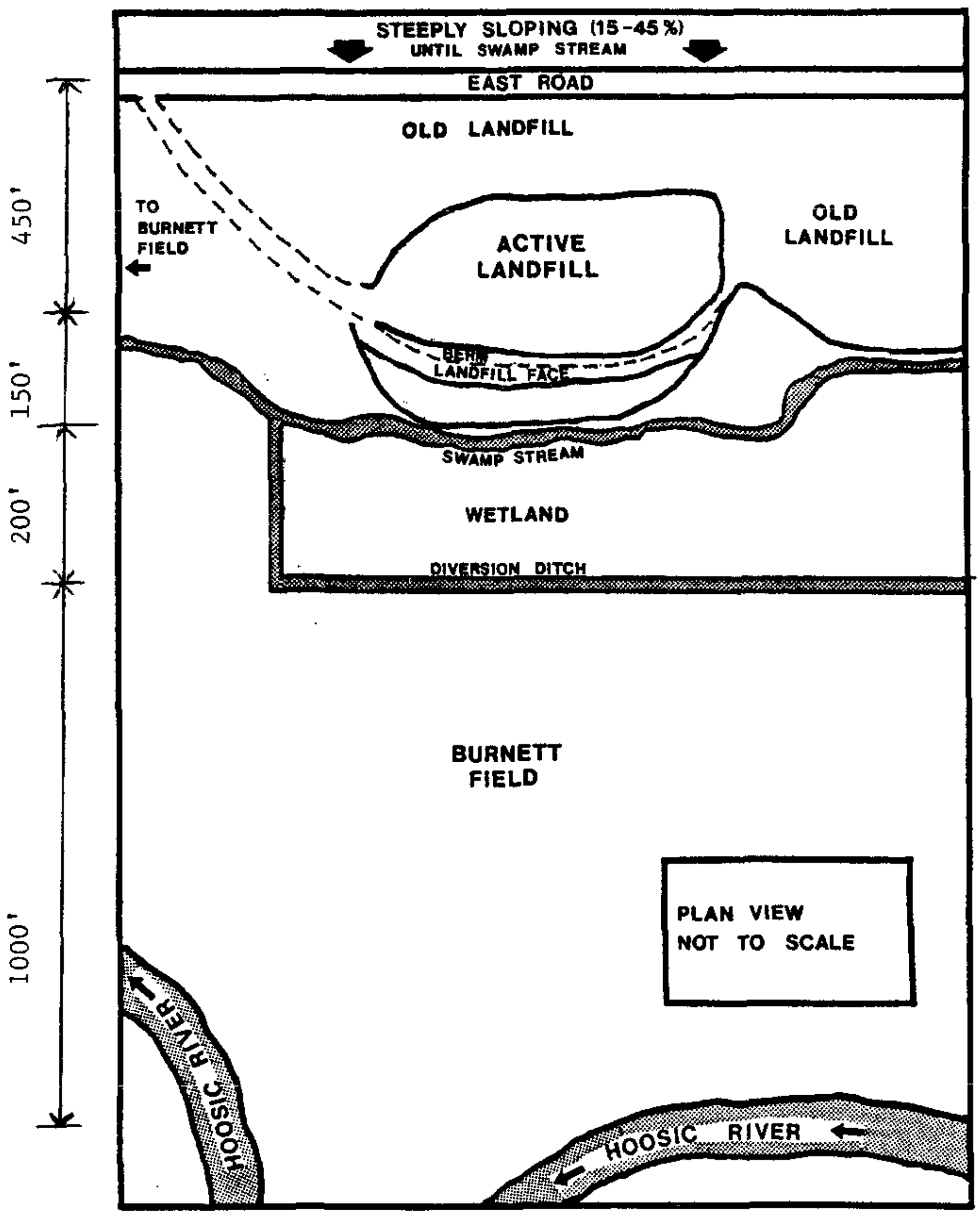


Figure 11. Study Location: Plan View

installed and sampled. From this representation of contaminant concentrations, the gaps in concentration knowledge were identified and wells were installed to fill those gaps. An interesting observation made during this phase of installation was that while wells 5, 6 and 7 showed decreases in concentration with increasing distance from the landfill, well #8, located further than these others from the source, did not. On the basis of site information at that time, no explanation for this discrepancy was immediately apparent.

The next distinct phase of installation involved wells 9 through 18. The general intent of the placement of these wells (shown in Figure 12) was to define the areal extent of the leachate plume, and to determine if wells 1 through 8 did intercept the center line of maximum concentration in that plume. The design of this well field was also intended to establish an areal grid that might be used later in modeling work.

Wells 9 through 12 were augered approximately 50 and 100 feet north and south of well #6 and in such a manner as to create a line perpendicular to the general flow direction. Again, the wells were pumped and analyzed after installation. Now, however, plan view sketches were drawn to identify the width of the plume, in addition to cross sections of the new line, examples of which are shown in Figures 13 through 15. Wells 13 through 16 were placed in a similar manner to 9 through 12, 50 and 100 feet north and south of well #5. Here, the intention was to identify contaminant movement in the plan view with respect to the parameter values obtained from wells 9 through 12. On the basis of data from wells 13 through 16, it was established that contamination was indeed moving from line 9 through 12 to line 13 through 16. Wells 17 and 18 were located 100 feet north and south of well #8 after viewing the information from the previous well lines. Here again, the intent was to locate the furthest areal extent of contamination migration while staying within the bounds of the study area.

Values obtained in any plume study are only valuable when viewed in the light of background groundwater quality. Thus, concentrations viewed as low values in a plume may in fact represent average, 'natural' groundwater parameter concentrations. It was with this logic that one upgradient well was required in the RCRA specifications regarding groundwater monitoring in the vicinity of a landfill. At Adams, the values obtained at well #4, as noted, represented the lowest concentrations obtained. In order to determine if these reflect leachate enriched groundwater or natural background values, well #19 was installed.

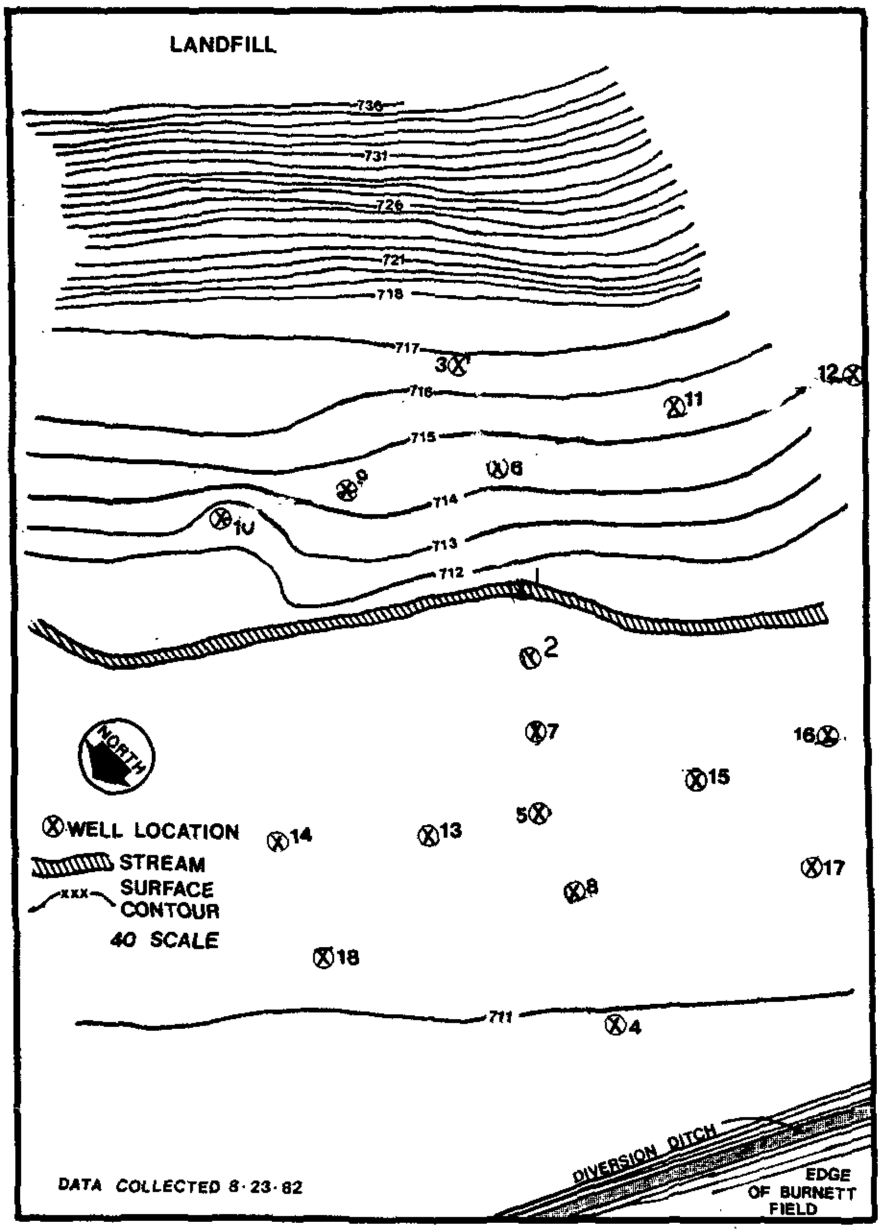


Figure 12. Adams Landfill: Surface Contours and Well Locations

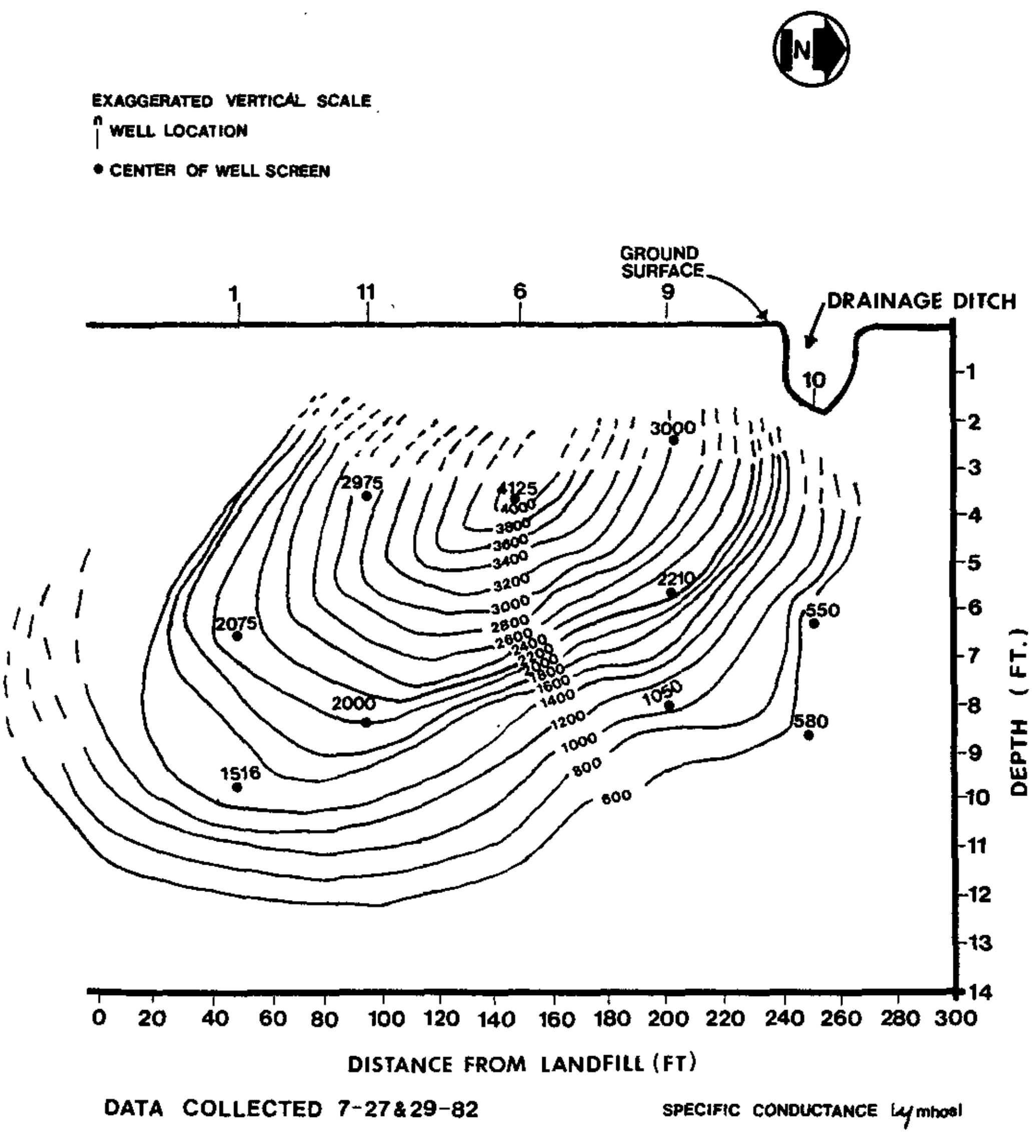


Figure 13. Adams Landfill: Specific Conductance Concentration Contours

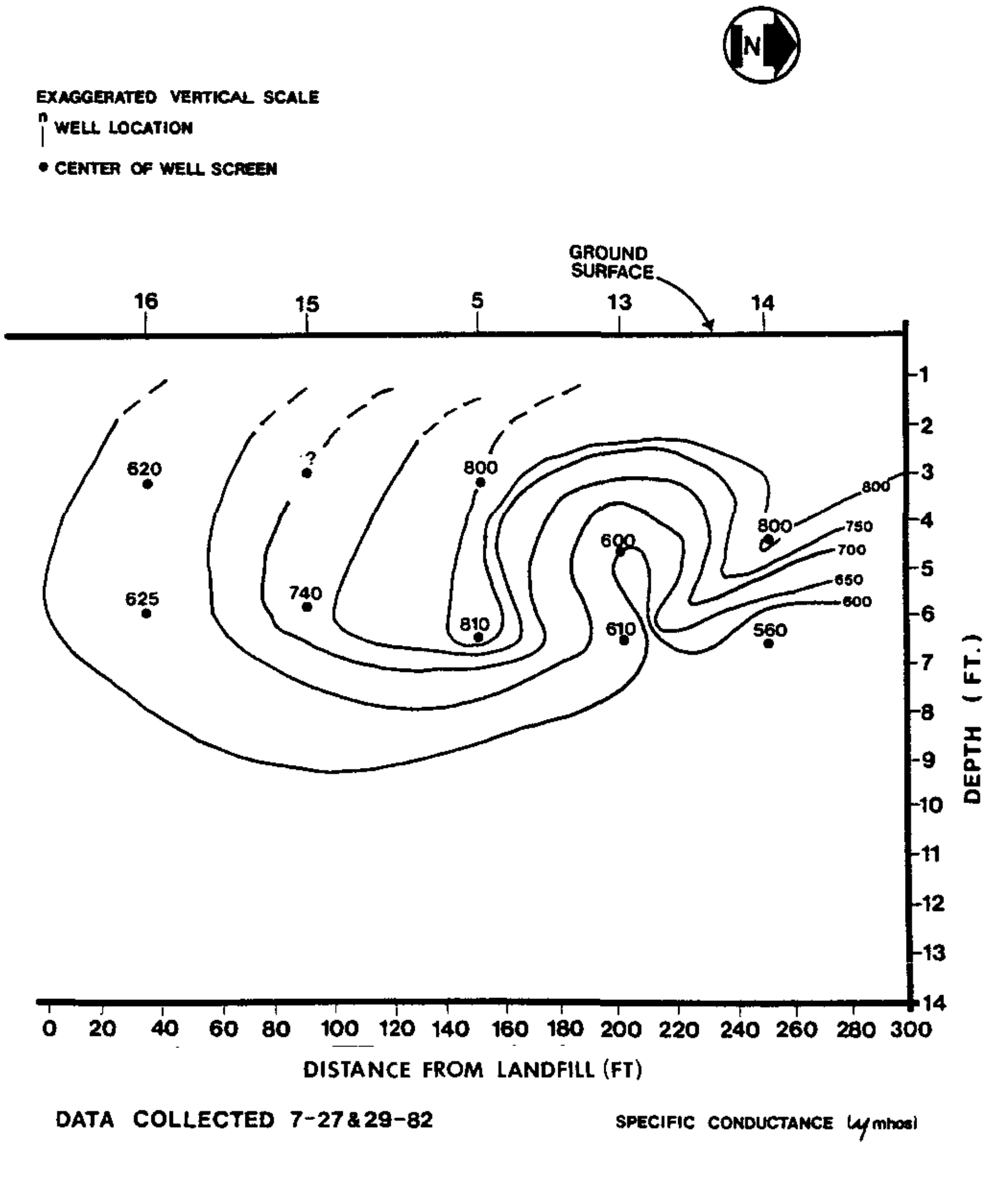


Figure 14. Adams Landfill: Specific Conductance Concentration Contours

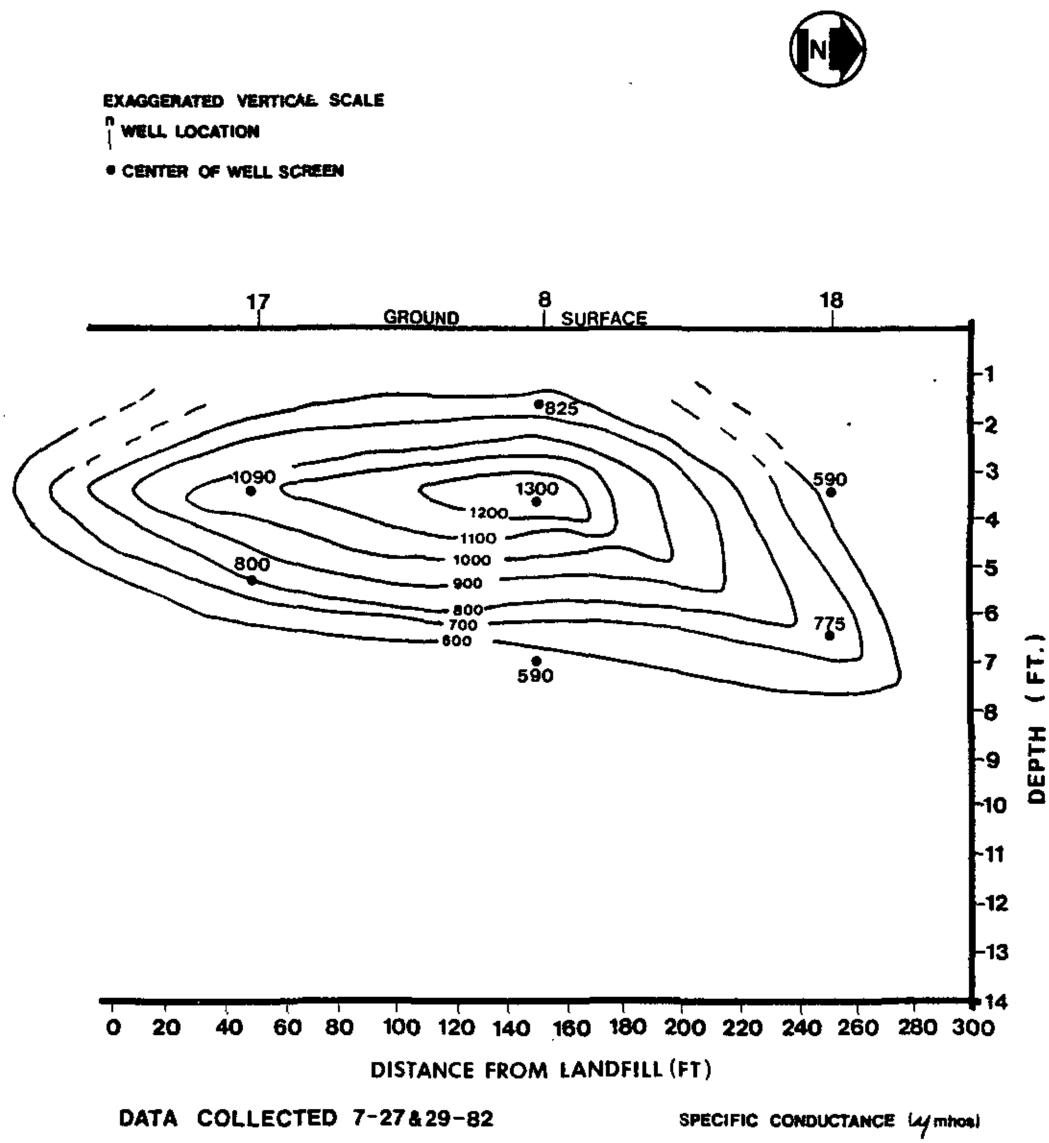


Figure 15. Adams Landfill: Specific Conductance Concentration Contours

Well #19 was located approximately 1/4 mile (0.4 km) south of the well field, at the far edge of the marsh. This placement was chosen on the basis of a number of factors. The initial intent was to place a well, as prescribed by RCRA, upgradient from the landfill. The coarse nature of the upgradient stratified drift and the depth to groundwater, however, made installation impossible with the hand auger. In the wetlands, as noted, the installation procedure is much simpler. Secondly, although the groundwater elevation is approximately the same as that of the well field, the site of well #19 did not appear to be in line with potential plume migration. Finally, because the majority of wells are located in the wetland, an 'uncontaminated' well in the same geological setting seemed more representative of background water quality. In the final analysis, however, it was hard to obtain groundwater samples at well #19 and it was essentially abandoned.

For a proper site evaluation, a true upgradient well had to be located. In a conversation with Douglas Burnett, owner of the grazing fields beyond the wetland, it was learned that an observation well was installed approximately 500 feet directly upgradient from the landfill in the stratified drift. (A large field owned by Burnett had been identified as an alternative site for the Adams landfill when the south section was closed in 1975. In order to determine the impact of landfilling at this site on the groundwater, the well was installed.) Mr. Burnett permitted access to the well and it was pumped and analyzed.

The final three wells, 20 through 22, were installed in the original line of wells 1 through 8. Since the cross-sectional profile along the line of highest contamination was the primary concern, these wells were installed to fine tune the data collected thus far. The positions of these wells relative to wells 1 through 8 are shown in Figure 16. Well #20 was located on the far side of the diversion ditch at the edge of the study area, such that both the extent of the plume and the effect of the diversion ditch on that plume could be considered. Wells 21 and 22, located several feet on either side of the swamp stream, were installed in an attempt to clear up confusion about the existing plume cross section, and completed the well installation program. In all, a total of 21 wells consisting of 43 piezometers were placed in the well field.

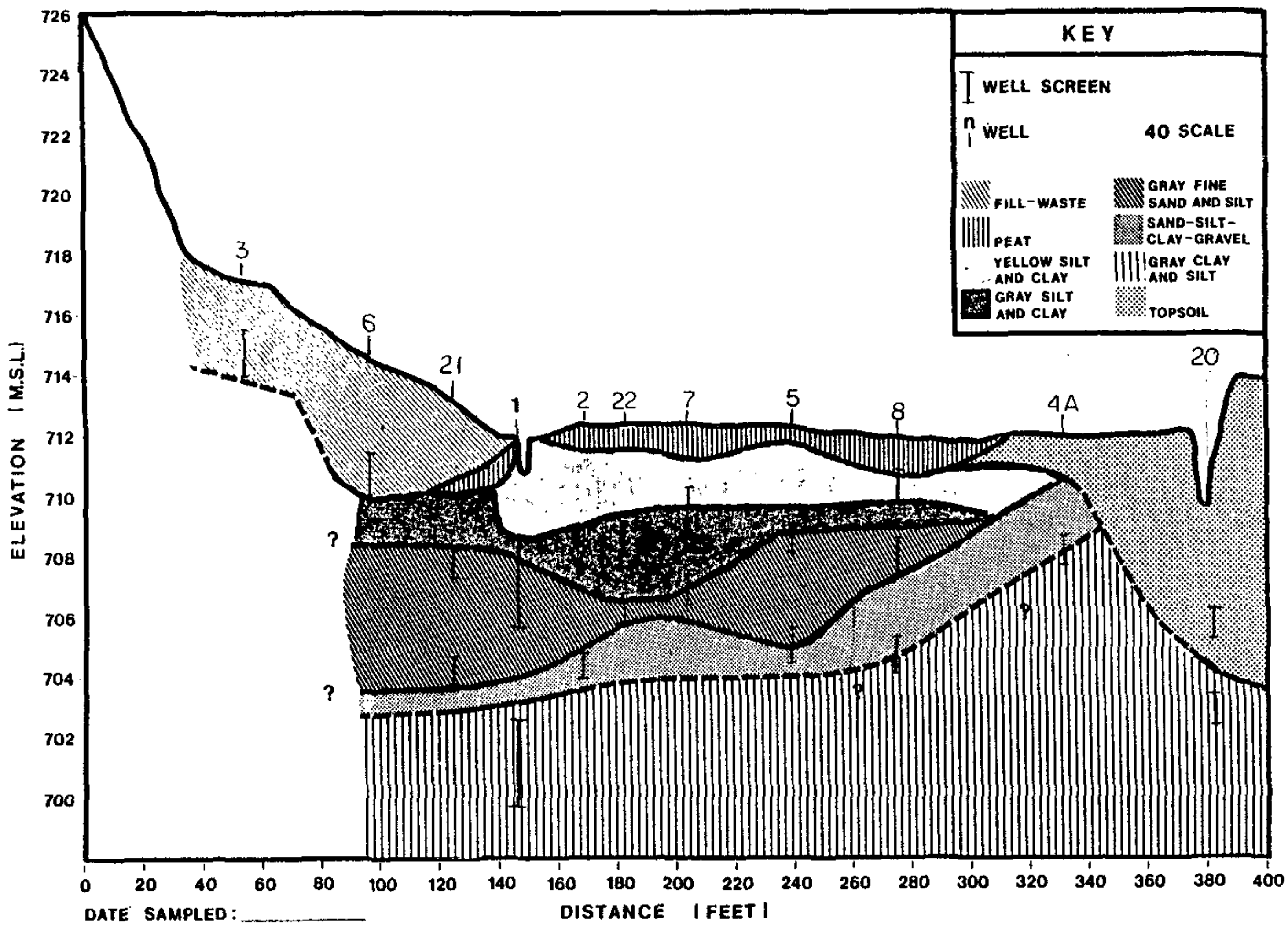


Figure 16. Section View: Main Well Line

CHAPTER V

SAMPLING, ANALYSIS AND TESTING

Sampling Procedures

Throughout the well installation program, as mentioned, the wells were pumped and analyzed for three contaminant parameters. Pumping was accomplished with an inexpensive, portable, hand operated vacuum pump system. A length of polyethylene tubing, connected to a vacuum flask was lowered down each 1.3 cm (one-half inch) PVC piezometer tube. The Guzzler hand pump is connected to the same flask and pumped to evacuate a groundwater sample from the piezometer tube.

Although the importance of purging a well is stressed by researchers (52,56,67), this could not be accomplished at the majority of wells. The permeability of the soils in which many of the well screens were located, was on the order of 10^{-6} to 10^{-7} cm/sec, therefore, pumping the four to six well pipe volumes suggested (for instance by Schuller, Gibb and Griffin (56)), plus the sample to be analyzed, would take several hours. For the sake of expediency, the standing water in the piezometer tube was used for the sample. In an independent test on a well which did yield the necessary well volumes, values for specific conductance, chloride, and hardness changed very little from volume to volume. On the basis of this test, and the factor of time, the water that was first pumped from the well constituted the sample. It should also be noted that the object of sampling and analysis in this study was to identify plume trends, and not to determine exact concentrations of leachate constituents found in the groundwater.

Chemical Analysis

Given the sampling objective and the limitations on sample collection, it was decided to analyze samples in the field insofar as possible. For the analysis of chloride (as NaCl) and hardness concentration, a Hach field testing kit was used. The measurement of specific conductance (and temperature and salinity when measured) was accomplished with a portable YSI Model 33 S-C-T meter. The selection of both analysis methods was in part determined by the ease of on site measurement, since achieving extreme accuracy was unnecessary.

In the beginning of the study, when few wells were in place, both the Hach kit and the conductance probe were used in the field, and the sample was withdrawn from the piezometer into a 1000 ml flask. The probe was lowered into the flask and readings of conductance, temperature and salinity were recorded. A portion of the sample was then poured off and used for the hardness and chloride titrations. The total time required for sampling and analyzing the water from each piezometer was approximately 15-20 minutes. As the number of wells increased, sampling and analysis the entire well field in the field was no longer feasible. Instead, all samples were stored in glass jars and returned to the lab for analysis. All analyses were completed within 24 hours of collection.

Although the ideal procedure is to sample the water immediately upon withdrawing it from the well, laboratory analysis had certain advantages over testing the water in-situ. First, all glassware could be thoroughly cleaned between analyses, thereby not allowing the residue in a sample bottle to alter the results of the subsequent analyses. Secondly, the samples could be filtered in the lab prior to analysis. Because a majority of the well screens were located in fine grained material, fines would enter the sample bottle when pumped. Filtering the water made titrations much easier, and thus improved the accuracy of the results.

In addition to groundwater samples, a number of surface water samples were taken, particularly in the swamp stream. In fact, the Adams site has a history of ground and surface water sampling that begins as early as 1972. C. E. Maguire (41) reported that a total of 51 samples were collected and analyzed for a variety of parameters and by several different researchers. Realizing the limitations of such a broad based and uncoordinated sampling effort, C. E. Maguire evaluated the data in order to reveal trends detected at the site. Their results, labelled 'Leachate Test Evaluation, Range of Values', are presented in Table 2. The Maguire report concluded that surface and groundwater entering the disposal area receives large quantities of dissolved solids, but as the water left the study area, a trend of gradual attenuation was taking place. The recommendations from this report stated that, 'excessive pollutants are emerging from the landfill base and will require certain site modifications to reduce pollutants to acceptable levels.' In general, results from our own surface water sampling conformed with those reported by C. E. Maguire. As the stream flowed down the hillside to the south of the study area, no significant changes in concentration were detected. As it entered the wetland, however, increases in the three parameters used in the groundwater analysis were observed, appearing to be affected most strongly by the many small rivulets of leachate that

TABLE 2

Leachate Test Evaluation (Range of Values)

	Surface Water Before Landfill Contact		Surface Water(Leachate) Along Landfill Perimeter		Surface Water After Leaving Landfill Area (in Meadow)		Common Permissible Levels
Chloride	6.9-31.0	(2)	54-150	(7)	28.5	(1)	250
Hardness	78-192	(2)	173-460	(4)	208	(1)	250
Iron	0-0.2	(8)	0.3-260	(33)	.0-.8	(6)	0.3
Manganese	0-.5	(8)	0-3.6	(33)	.1-.8	(6)	.05
Sulfates	16-80	(7)	76-470	(29)	33-42	(6)	250
pH	7.3-7.7	(8)	6.6-7.9	(36)	7.1-7.7	(6)	6.0-8.5
BOD	4.5-8.0	(2)	8.7-288	(7)	5.8	(1)	4.0
COD	22	(1)	22-101.2	(7)	39.6	(1)	---
Ammonia	.2	(1)	.2-2.2	(6)	.48	(1)	.05
Nitrates	0-.75	(2)	.18-14.8	(4)	.25	(1)	45

Note: 1) all values in mg/L except pH.
2) ( ) = Number of Tests

Source: Reference (41).

can be seen coming from the base of the landfill and draining directly into the stream. The concentrations then decrease as the swamp stream leaves the influence of the landfill and passes through a series of excavated canals on the Burnett property to the north.

Additional Testing

In order to fully evaluate the groundwater flow conditions and the extent of groundwater contamination in the study area, several additional tests were performed, including analyses of the wetland soils. The velocity, direction, and general movement of the groundwater, and the associated contamination enclave depend in part on the properties of the soils through which the water moves.

Soil samples were routinely removed and classified by visual inspection during both the wash boring and hand auger well installations. The resulting soil descriptions were recorded on boring logs for each well and subsequently used to compile cross-sectional sketches of the site. These boring logs, and those obtained from other drillers who had worked in the area, allowed us to graphically piece together the approximate soil conditions across the site. Examples of some of the commercial boring logs are presented in Figures 17 through 19.

A series of soil samples were also removed from the wetland and brought into the lab for analysis. The soils were sampled either in the split spoon sampler, with the hand auger, or with a hand shovel. The samples were dried and sieved in order to determine various soil properties, which are listed in Table 3.

Permeability tests were run on four of the soils. Due to the low hydraulic conductivity values anticipated, the tests were run on a falling head permeameter. The test apparatus is shown in Figure 20 and the results of the test are listed in Table 4. For comparison, it is noted that a value of 10^{-7} cm/sec is generally considered an 'impermeable' soil. Using these values and the potentiometric surface measurements, estimates of both groundwater velocity and discharge can be made.

In an attempt to establish a fluid mass balance at the site, estimates of stream flow were made, although the limited flow and depth of the swamp stream made the use of a conventional flow meter or a weir impractical. Instead, rough estimates were made by floating a ping-pong ball along stretches of the stream, BORING CONTRACTOR: Soils Engineering, Inc. Charlestown, N. H.

LOG PREPARED BY: CONTR. ____ CEM x

CE MAGUIRE, INC.
ARCHITECTS-ENGINEERS-PLANNERS
BORING LOG

TOWN, STATE: Adams, Massachusetts
PROJECT NAME: Sanitary Landfill
CEM NO. 2555 OFFICE: Providence

SHEET 1 OF 1
LOCATION:
HOLE NO: BH-3
BORING TYPE: H.S. Auger
LINE & STA.:
OFFSET:

GROUND WATER OBSERVATIONS
AT 34.7 FT. AFTER 3 DAYS
AT 35 FT. AFTER 10 DAYS

	AUGER	CASING	SAMPLER	CORE BAR.
TYPE	H.S.		S/S	
SIZE, I.D.	4"		1 1/2"	
HAMMER WT.			140	BIT.
HAMMER FALL			30	

SURFACE ELEV. 787
DATE STARTED-FINISHED: 5/9/75
BORING FOREMAN: M. Domingue
INSPECTOR: G. Sauter
SOILS ENGR.:

LOCATION OF BORING: See Plan PVC Observation Pipe Installed (40')

DEPTH BELOW SURFACE	CASING BLOWS PER FOOT	SAMPLE DEPTH FROM – TO	TYPE OF SAMPLE	BLOWS PER 6" ON SAMPLER 0-6	6-12	12-18	STRATA CHANGE DEPTH ELEV.	FIELD IDENTIFICATION OF SOIL & ROCK INCL. COLOR, LOSS OF WASH WATER, JOINTS IN ROCK, ETC.	SAMPLE NO.	PEN.	REC.
		0 - 1.5	D	1	1	2	1.5	Brown SILT LOAM	1	18	
							785.5				
5		5 - 6.5	D	4	8	16		Brown fine to coarse SAND, some silt	2	18	
10		10 - 11.4	D	12	23	46			3	17	
							13.0	BOULDERS			
							774				
15		15 - 16.5	D	53	54	64		Brown fine to coarse SAND and GRAVEL, some silt	4	18	
20								BOULDERS			
25		25 - 26.5	D	81				Dark brown fine to coarse SAND, little gravel, some silt,	5	6	
30		30 - 30.5	D	93				COBBLES and BOULDERS	6	6	
35		35 - 36.5	D	65	153			Dark brown fine to coarse SAND and SILT	7	12	
							40.0				
40		40					747	Refusal at 40.0'			

GROUND SURFACE TO ____ FT., USED ____" CASING: THEN ____" CASING TO ____ FT.
D=DRY W=WASHED C=CORED P=PIT A=AUGER V=VANE TEST
UP=UNDISTURBED, PISTON UB=UNDISTURBED, BALL CHECK OER=OPEN END ROD
PROPORTIONS USED: TRACE=0-10%, LITTLE=10-20%, SOME=20-35%, AND 35-50%

FOOTAGE IN EARTH	40.0
FOOTAGE IN ROCK	0
NO. OF SAMPLES	7
HOLE NO. BH-3 TYPE	HSA

Figure 17. Boring Log: Elevation 787

BORING CONTRACTOR: Soils Engineering, Inc. Charlestown, N.H.	**CE MAGUIRE, INC.** ARCHITECTS-ENGINEERS-PLANNERS **BORING LOG**	SHEET 1 OF 2
	TOWN, STATE: Adams, Massachusetts	LOCATION:
LOG PREPARED BY: CONTR. ___ CEM X	PROJECT NAME: Sanitary Landfill	HOLE NO: BH-4
	CEM NO. 2555 OFFICE: Providence	BORING TYPE: H.S. Auger
		LINE & STA.:
		OFFSET:

GROUND WATER OBSERVATIONS

AT 30 FT. AFTER 24 HOURS

30.2 DAYS

AT ___ FT. AFTER 7 ~~HOURS~~

	AUGER	CASING	SAMPLER	CORE BAR.
TYPE	H.S.		S/S	
SIZE, I.D.	4"		$1\frac{1}{2}$"	
HAMMER WT.			140	BIT.
HAMMER FALL			30	

SURFACE ELEV. 769

DATE STARTED-FINISHED: 5/12/75

BORING FOREMAN: M. Domingue

INSPECTOR: G. Sauter

SOILS ENGR.:

LOCATION OF BORING: See Plan

DEPTH BELOW SURFACE	CASING BLOWS PER FOOT	SAMPLE DEPTH FROM – TO	TYPE OF SAMPLE	BLOWS PER 6" ON SAMPLER FROM – TO 0-6	6-12	12-18	STRATA CHANGE DEPTH ELEV.	FIELD IDENTIFICATION OF SOIL & ROCK INCL. COLOR, LOSS OF WASH WATER, JOINTS IN ROCK, ETC.	SAMPLE NO.	PEN	REC.
		0 - 1.5	D	1	1	3	2.0 767	Brown SILT, some sand, trace gravel	1	18	
5		5 - 6.5	D	6	4	6	7.5 761.5	Brown fine to coarse SAND, little gravel w/silt lenses	2	18	
10		10 - 11.5	D	10	12	11	13.0 756	Brown fine to coarse SAND and SILT, trace gravel	3	18	
15		15 - 16.5	D	41	98	25		Brown fine to coarse SAND, some gravel and silt, BOULDERS	4	18	
20		20 - 21.5	D	35	22	31		Brown fine to coarse SAND, some silt, trace gravel	5	18	
25		25 - 26.5	D	49	29	80	25.0 744	Brown fine SAND and SILT BOULDERS	6	18	
30		30 - 30.5	D	100			32.0 737		7	6	
35		35 - 36	D	100	91			Brown fine to coarse SAND and SILT, trace gravel	8	12	
40											

GROUND SURFACE TO ___ FT., USED ___" CASING: THEN ___" CASING TO ___ FT.

D=DRY W=WASHED C=CORED P=PIT A=AUGER V=VANE TEST

UP=UNDISTURBED, PISTON UB=UNDISTURBED, BALL CHECK OER=OPEN END ROD

PROPORTIONS USED: TRACE=0-10%, LITTLE=10-20%, SOME=20-35%, AND 35-50%

FOOTAGE IN EARTH	41.0
FOOTAGE IN ROCK	0
NO. OF SAMPLES	9
HOLE NO. BH-4 TYPE	HSA

Figure 18a. Boring Log: Elevation 769

BORING CONTRACTOR: Soils Engineering, Inc. Charlestown, N.H.	CE MAGUIRE, INC. ARCHITECTS-ENGINEERS-PLANNERS BORING LOG	SHEET 2 OF 2
LOG PREPARED BY: CONTR. ___ CEM x	TOWN, STATE: Adams, Massachusetts	LOCATION:
	PROJECT NAME: Sanitary Landfill	HOLE NO: BH-4
	CEM NO. 2555 OFFICE: Providence	BORING TYPE:
		LINE & STA.:
		OFFSET:

GROUND WATER OBSERVATIONS		AUGER	CASING	SAMPLER	CORE BAR.	
AT ___ FT. AFTER ___ HOURS	TYPE	H.S.		S/S		SURFACE ELEV. 769
	SIZE, I.D.	4"		1 ½"		DATE STARTED-FINISHED: 5/12/75
AT ___ FT. AFTER ___ HOURS	HAMMER WT.			140	BIT.	BORING FOREMAN: M. Domingue
	HAMMER FALL			30		INSPECTOR: G. Sauter
						SOILS ENGR.:

LOCATION OF BORING:

DEPTH BELOW SURFACE	CASING BLOWS PER FOOT	SAMPLE DEPTH FROM - TO	TYPE OF SAMPLE	BLOWS PER 6" ON SAMPLER FROM - TO 0-6	6-12	12-18	STRATA CHANGE DEPTH ELEV.	FIELD IDENTIFICATION OF SOIL & ROCK INCL. COLOR, LOSS OF WASH WATER, JOINTS IN ROCK, ETC.	SAMPLE NO.	PEN.	REC.
40		40 - 41	D	152	162			Brown fine to coarse SAND and SILT, little gravel	9	12	
								Bottom of Boring 41.0'			
								PVC Observation Pipe Installed (40')			

GROUND SURFACE TO ___ FT., USED ___" CASING: THEN ___" CASING TO ___ FT.
D=DRY W=WASHED C=CORED P=PIT A=AUGER V=VANE TEST
UP=UNDISTURBED, PISTON UB=UNDISTURBED, BALL CHECK OER=OPEN END ROD
PROPORTIONS USED: TRACE=0-10%, LITTLE=10-20%, SOME=20-35%, AND 35-50%

FOOTAGE IN EARTH	41.0
FOOTAGE IN ROCK	0
NO. OF SAMPLES	9
HOLE NO. BH-4 TYPE	HSA

Figure 18b. Boring Log: Elevation 769

BORING CONTRACTOR: Soils Engineering, Inc. Charlestown, N.H. LOG PREPARED BY: CONTR. ____ CEM X	CE MAGUIRE, INC. ARCHITECTS-ENGINEERS-PLANNERS BORING LOG TOWN, STATE: Adams, Massachusetts PROJECT NAME: Sanitary Landfill CEM NO. 2555 OFFICE: Providence	SHEET 1 OF 1 LOCATION: ____ HOLE NO: BH-6 BORING TYPE: H.S. Auger LINE & STA.: ____ OFFSET: ____

GROUND WATER OBSERVATIONS		AUGER	CASING	SAMPLER	CORE BAR	
AT * FT. AFTER ____ HOURS	TYPE	H.S.		S/S		SURFACE ELEV. 739
AT 12.4 FT. AFTER 6 DAYS	SIZE, I.D.	4"		1 ½"		DATE STARTED-FINISHED: 5/13/75
* No water observed	HAMMER WT.			140	BIT.	BORING FOREMAN: M. Domingue
	HAMMER FALL			30		INSPECTOR: G. Sauter
						SOILS ENGR.:

LOCATION OF BORING: See Plan

DEPTH BELOW SURFACE	CASING BLOWS PER FOOT	SAMPLE DEPTH FROM - TO	TYPE OF SAMPLE	BLOWS PER 6" ON SAMPLER FROM — TO 0-6	6-12	12-18	STRATA CHANGE DEPTH ELEV.	FIELD IDENTIFICATION OF SOIL & ROCK INCL. COLOR, LOSS OF WASH WATER, JOINTS IN ROCK, ETC.	SAMPLE NO.	PEN.	REC.
		0 - 1.5	D	1	2	1		Brown fine to medium SAND and SILT, little gravel	1	18	
5		5 - 6.5	D	12	12	12		Brown fine SAND and SILT	2	18	
10		10 - 11.5	D	71	61	30		Brown fine SAND, some gravel	3	18	
15		15 - 15.8	D	68	723			Brown fine to coarse SAND, SILT and GRAVEL	4	18	
							19.5				
20							719.5	Refusal at 19.5' PVC Observation Pipe Installed (18')			

GROUND SURFACE TO ____ FT., USED ____" CASING: THEN ____" CASING TO ____ FT.

D=DRY W=WASHED C=CORED P=PIT A=AUGER V=VANE TEST

UP=UNDISTURBED, PISTON UB=UNDISTURBED, BALL CHECK OER=OPEN END ROD

PROPORTIONS USED: TRACE=0-10%, LITTLE=10-20%, SOME=20-35%, AND 35-50%

FOOTAGE IN EARTH	19.5
FOOTAGE IN ROCK	0
NO. OF SAMPLES	4
HOLE NO. BH-6	TYPE HSA

Figure 19. Boring Log: Elevation 739

TABLE 3

Soil Characteristics*

	Peat	Yellow Silt & Clay	Gray Silt & Clay
Water content	126%	42%	40%
Porosity	78%	61%	64%
Void Ratio	3.58	1.7	1.75
Degree of Saturation	91%	65%	57%
Wet density	78.4 lb/ft^3	84.3 lb/ft^3	81.1 lb/ft^3

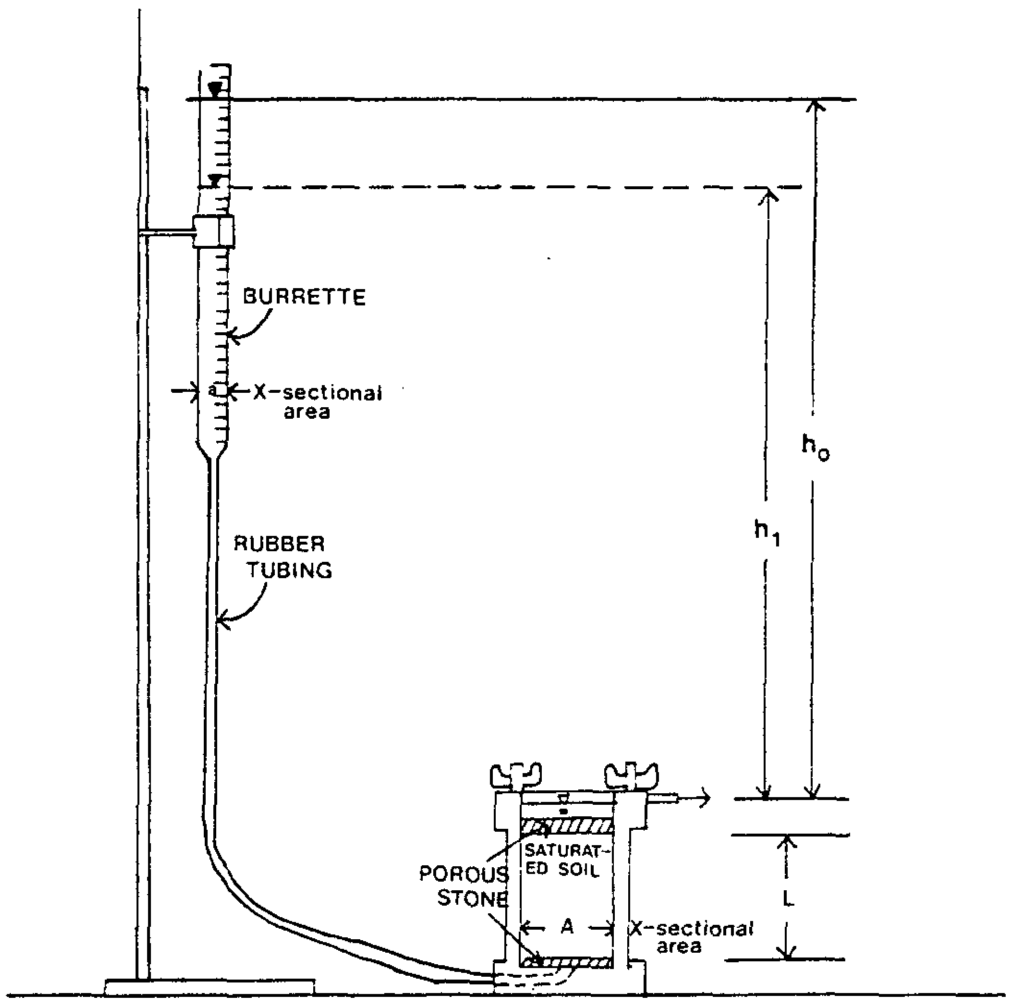
*Soils hand sampled 6/9/83.

TABLE 4

Results of Falling Head Permeameter Test

Soil	Permeability
Peat	$k = 2.5 \times 10^{-6}$ cm/sec
Yellow silt and clay	$k = 6.2 \times 10^{-7}$ cm/sec
Gray silt and clay	$k = 2.9 \times 10^{6}$ cm/sec

NO SCALE



t_o = initial time
t_1 = time of measuring height h_1

Permeability $= K = \frac{al}{A(t_1 - t\ )} \ln \left(\frac{h_o}{h_1}\right)$

Figure 20. Falling Head Permeameter Apparatus

diversion ditch and several of the larger leachate streams, and measuring its time of travel. While the values recorded on the given dates are significant, the diversion ditch and many of the leachate streams flow almost directly in response to rainfall and the subsequent runoff. Two weeks after measuring the diversion ditch velocity (see Table 5) a return visit found the water to be stagnant and several of the leachate streams had slowed considerably.

One additional test conducted at the site was an estimate of groundwater seepage into the swamp stream and the diversion ditch. The device, known as a seepage meter, is constructed by cutting off the top 15.2 cm (six inches) of a 208 L (55 gallon) drum. A rubber stopper with a narrow glass tube through it is positioned in a hole on the top of the drum, and a balloon is fastened to the protruding glass tubing. The meter (drum) (with the balloon detached) is pressed into the bottom of the stream until water displaces all of the air space and rises up the glass tubing. The balloon is then attached to the tubing and the time is recorded. After several hours (depending on the permeability of the soil and other factors) the balloon is removed and the amount of water it contains is measured. Lee (38) provides a detailed outline of the necessary equipment and installation procedures. The results provide a rough estimate of the discharge of groundwater into the stream bottom per unit area, a value which can then be applied to a reach of the stream equal to the width of the control volume being studied. For our work, we chose a distance of 100 ft (30.5 m) on either side of the well line parallel to the groundwater flow direction, and the results of the test were surprising. The volume entering the swamp stream across our 61 m (200 ft) control volume was considerably less than anticipated ((.0035 L/sec) (80 gal/day)) while that water entering the diversion ditch was almost negligible. It is assumed that these values, as with stream flows are highly dependent on groundwater elevations, and therefore rainfall and runoff as well.

Another task accomplished at the site was a survey of all well points and pertinent ground elevations using a transit. Not only did this allow us to accurately draw sketches of the site, but it gave us a reference point from which to measure groundwater elevations. During sampling, a tape was lowered down each piezometer tube and the groundwater elevation for that day was measured. Because each piezometer is isolated in its own pressure environment with the bentonite seal, the elevation measured represents a flow potential at that particular point in the aquifer. By plotting these points on a scaled cross-sectional drawing, flow lines can then be drawn parallel to lines of equal flow potential thus forming a flow net. These nets provide a good estimate of groundwater flow direction on that date. Given values

TABLE 5

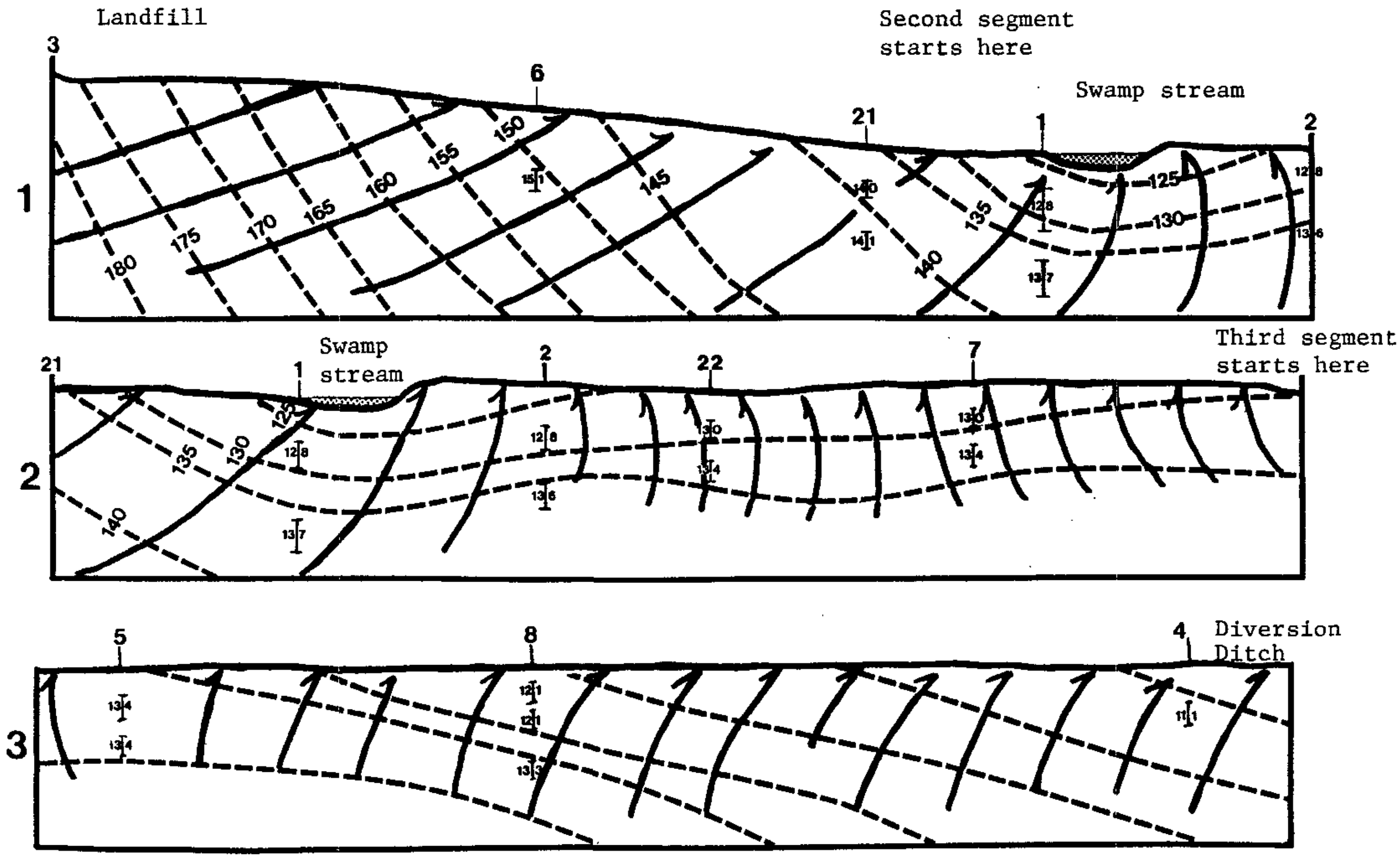
Swamp Stream and Diversion Ditch Discharge Measurements

Date	Location	Discharge (ft/sec)
5/25/83	Swamp Stream Near Well #1 (See Fig. 12)	0.61
6/9/83	Swamp Stream Near Well #1	0.61
6/9/83	Diversion Ditch Near Well #20	0.53
7/9/83	Swamp Stream Near Well #12	0.01
7/9/83 (1)	Swamp Stream Near Well #1	0.34
7/9/83	Swamp Stream Near Well #9	0.31
7/9/83	Diversion Ditch Near Well #20	Negligible flow

NOTES:

1. Several leachate streams drain into the swamp stream near this measurement.

of head change and soil permeability, one may then calculate a value for groundwater flow direction on that date. Groundwater discharge values were calculated from the flow net, shown in 3 sections in Figure 21. The estimates from the flow nets were consistent with those given by the seepage meters.



DATE MEASURED: 6-9-83

Figure 21. Adams Landfill: Flow Net Analysis (Flow net for entire site divided into three segments for presentation.)

CHAPTER VI

GROUNDWATER FLOW

A clear assessment of the ground and surface water flow characteristics are fundamental to evaluating the impact of a landfill on its surroundings.

In the preceeding chapter, it was established that a clear picture of the soil conditions in the wetland was obtained during our well installation. In addition to these well logs, several local drillers provided copies of borings they had done in the area as well as descriptions of material they had encountered. When the pieces of these drilling records were pieced together, a more regional subsurface profile was in hand. By following the path of a drop of rainwater through the particular profile of strata found around the Adams landfill, one can more fully understand how both the production of leachate and its subsequent transport takes place. In order to accomplish this, the site will be divided into three distinct sections: the hillside, the landfill, and the wetland.

Hillside Flow Patterns

The surficial material on the hillside upgradient of the landfill was deposited directly by a glacial ice mass which advanced generally in a northeast to southwest direction (53). It is identified as consisting of lenses of gravel, sand, silt and clay with a few bodies of stratified sand and gravel. In a visual inspection, we have identified the zone directly above the landfill to be a stratified drift, an area several researchers point to as a strong candidate for a zone of recharge (22,44,50). Although the general upland topography within the Hoosic River watershed is one of steeply sloping hillsides, the area just above the landfill has been cleared and leveled for farming, enabling a large percentage of rain falling on this surface to infiltrate rather than going to runoff. Water which does go to runoff, makes its way into a number of small, slowly flowing brooks, some of which flow directly into the swamp stream below.

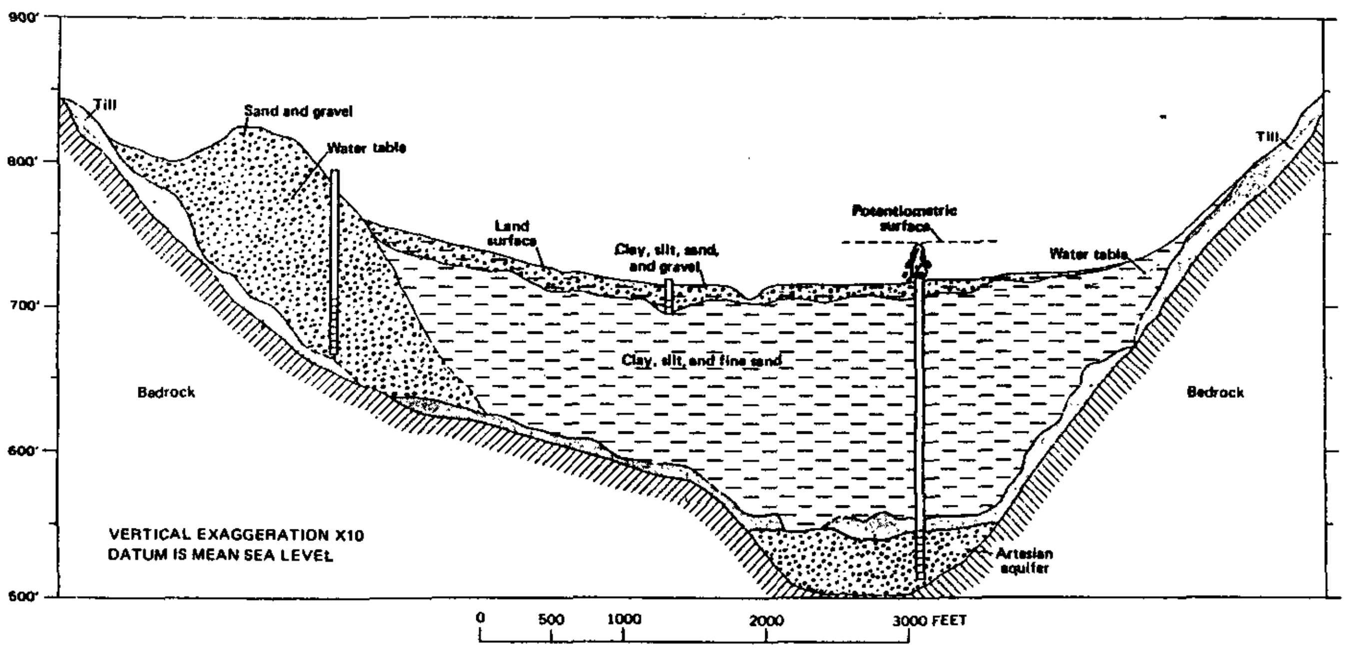
Beneath the thick layer of stratified drift is a locally discontinuous, low permeability till. This till, as with most found in the New England soils, has been tightly compacted by glacial action and can generally be considered an aquitard. The last distinct unit on the hillside is the fine to medium grained Kitchen Brook dolomite (26), which lies beneath the till or the drift when the till is absent. The dolomite, which has been described in detail previously, can be seen to outcrop in several fields and many of the hillside brooks. This Cambrian unit slopes steeply across the Hoosic Valley to abut the Clarendon Springs dolomite. The Pfizer Chemical Company has at least seven production wells set in the dolomite leading to the conclusion that a large percentage of the infiltrating water may work its way into the joints and fissures of this formation. One researcher also notes the existence of a large sand and gravel aquifer in the valley bottom above the carbonate (25). Figure 22 is a generalized cross section of the Hoosic River Valley, as presented by the U.S.G.S., showing these units.

With this geologic framework, we are now able to trace the movement of water falling on the hillside. One fate of the rainwater is runoff, although the majority of the water appears to infiltrate the highly permeable glacial material and migrate downward. A certain percentage of the water will also go to evapotranspiration, but little attention has been focused on this often elusive measurement as evapotranspiration on the hillside does not affect the movement of the leachate plume.

Water which moves from the hillside to the floodplain via streamflow does so quickly and without an effect on leachate production, while infiltrating water follows a more circuitous route and plays a major role in the generation of a plume. Hansen, Gay and Toler (25) report that unconsolidated deposits in the Hoosic River Valley may be hydraulically connected to the carbonate bedrock aquifer beneath and alongside them. It is our belief that this condition exists upgradient of the landfill and accounts for the transport of a large volume of water that falls on the basin. Again, this theory finds support in the Pfizer records which indicate pumpage yields of up to 1000 gal/min from wells located in the carbonate unit (24).

The firm of Soil Engineering Incorporated, under contract to C. E. Maguire, was hired in May of 1975 to drill test borings in the site of what is now the active landfill. These logs, and those of other drillers, indicate that the dolomite dips steeply just above the landfill, and show that this unit cuts across the zone of active filling. Figure 23 shows the approximate location of the dolomite, and its position relative to the other soils. In light of its water transmitting capabilities and its position with respect to the landfill, it is clear that groundwater may flow directly from the dolomite into the waste.

The last flow environment on the hillside is the stratified drift. In Figure 23, it too, is shown to be in direct contact with the landfill material. It would appear that water infiltrating this unit may also contribute to the flow of groundwater through the landfill below. Clearly, the point to be made is that water is draining from the upgradient strata directly into the refuse below. C. E. Maguire (41) noted that prior to the



WATER IN SAND AND GRAVEL AQUIFERS OF THE HOOSIC RIVER VALLEY MAY BE UNDER EITHER WATER-TABLE OR ARTESIAN CONDITIONS.

Some unconsolidated deposits may be hydraulically connected to the carbonate bedrock aquifer that underlies most of the valley, which may result in artesian conditions. (See diagram showing occurrence of water in bedrock.) At a given location the unconsolidated units shown are not always present or in the same sequence shown by the generalized cross section.

SOURCE: Ref. (35)

Interior—Geological Survey, Washington, D.C.—1973—W72319

Figure 22. Generalized Section View: Hoosic River Valley

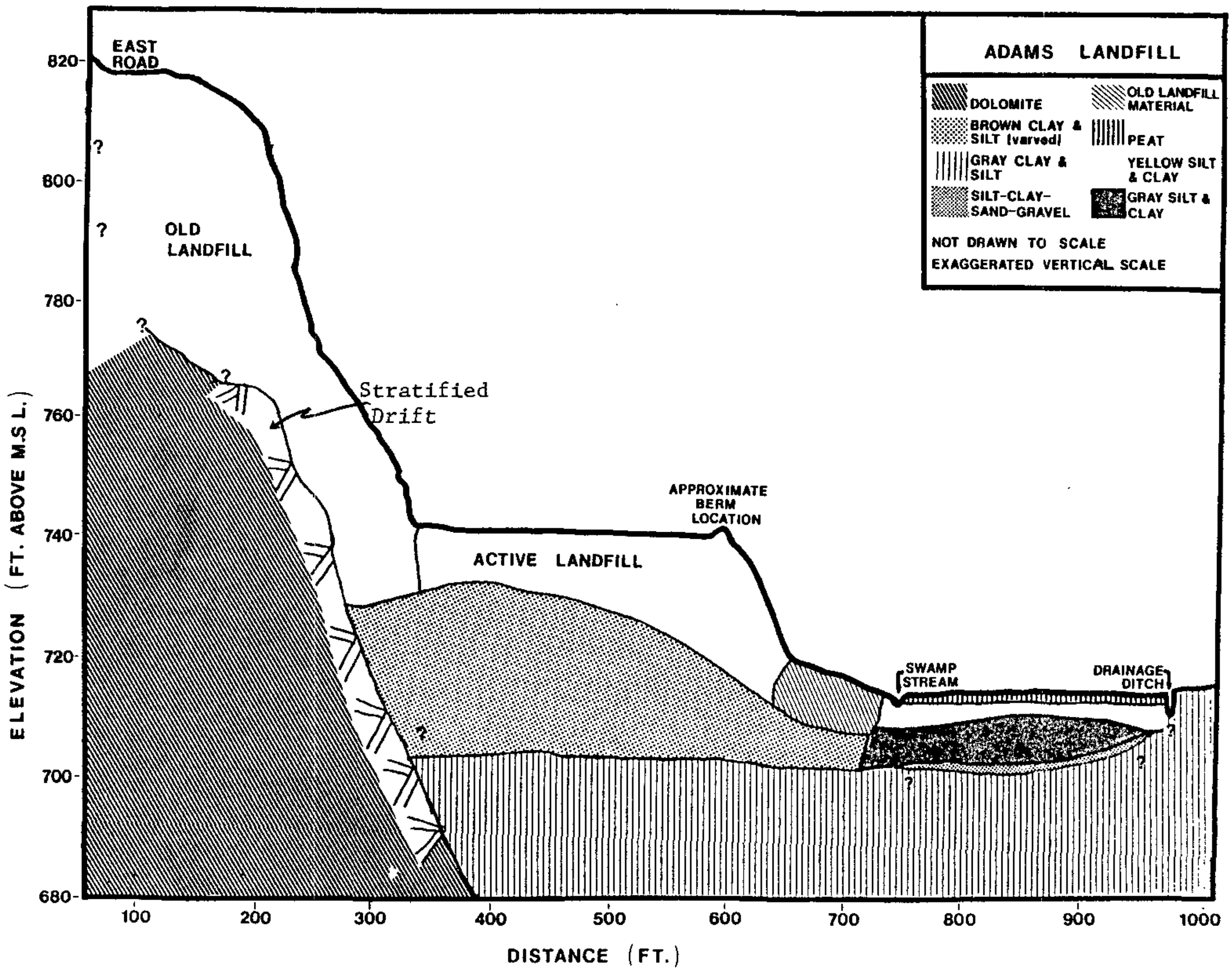


Figure 23. Adams Landfill: Section View

placement of waste, the present landfilling site was a point of groundwater discharge. The Maguire report went on to observe "...the refuse is located in an area where much of the refuse is saturated from springs (and) buried brooks."

Evidence exists to suggest that the majority of seeps and springs emanating from the hillside into the waste find their origin in the dolomite. By back calculating the potentiometric surface data from the wetland, upgradient through the landfill, we found that it intercepted the carbonate unit at an elevation consistent with groundwater data from the C. E. Maguire test borings. (See Figure 24). Groundwater data from the Soil Engineering, Inc. observation wells also indicate that the groundwater elevation is within the dolomite.

While it seems clear that the source of most of the water entering the landfill is the dolomite, the Pfizer pumpage data suggests that much of that water entering the dolomite probably infiltrates deeper into the unit. The generalized cross section view of the valley (See Figure 22) shows that this water mass can follow two individual paths of flow. Water that enters the ground nearer to the landfill will percolate through the drift and enter the carbonate. Depending on its point of entry, it will then either discharge directly into the waste, or just below it into a sand, silt, clay and gravel mixture. That water which recharges the dolomite further up on the hillside, will more likely continue its percolation to recharge the deeper artesian aquifer below.

Landfill Flow Patterns

The next major zone of concern is the landfill. Because of the restrictions on drilling through the active fill area, recreating the geology in the fill was dependent on information gathered from other sources. As noted, prior to waste placement, springs and seeps drained from the hillside strata. This water then flowed over low permeability silts and clays, into a natural depression in the floodplain. The wetland is what is left of that depression.

When refuse disposal began, some 40 years ago, the waste was either covered with sand and gravel or not covered at all, thus, water falling on the landfill infiltrated through the waste, solubilizing the contaminants. The leachate produced from this reaction moved down through the waste until it reached the low permeability silts and clays. From this point, much of the contamination joined with the water already entering from the face of the hillside and moved as small leachate streams toward the swamp stream. The remaining portion of the water remained either bound in the saturated waste or percolated through the silts and

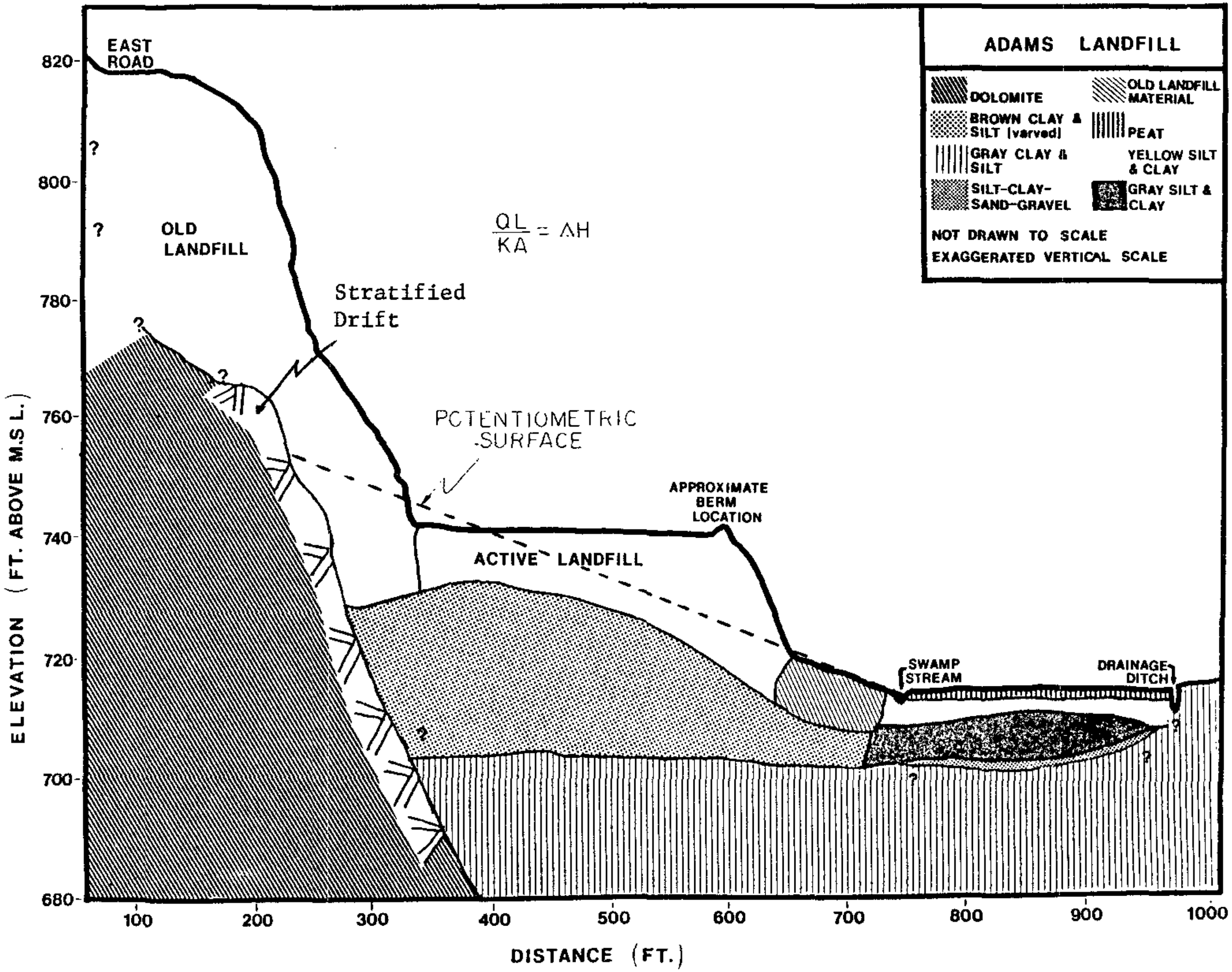


Figure 24. Theoretical Groundwater Profile

clays below. C. E. Maguire reports seeing large masses of saturated waste projecting from the hillside and also confirms the surface water contamination problem which resulted from the discharge of leachate streams into the swamp stream.

In an effort to control this surface water contamination, DEQE required the operator to apply a daily cover of low permeability limestone crusher waste to the refuse. The desired effects of this plan were to encapsulate the waste in this low permeability cell and reduce the infiltration of rain falling on the surface of the site. That water which did pass through the waste was then to have been 'treated', to a certain extent, by the high pH of the limestone. The goal of reduced infiltration was achieved. The entire plan was unattainable, however, due to the seeps and springs which continued to discharge into the waste. While covering the landfill inhibits the infiltration of rainfall, it has no effect on the subsurface flow which will continue to enter the landfill. Instead, the effect of the low permeability cover is to change the flow pattern of the water leaving the landfill. Prior to the placement of the cover the water flowed through both the waste and the silt and clay until it broke out of this material as a seep draining into the swamp stream. Additional waste placed on site eventually attained an elevation above the groundwater flow and became solubilized only as a result of percolating rainwater falling directly on the fill. When an impermeable cover, which extends from the hillside to near the edge of the wetland, was put in place, the flow patterns were greatly altered. Instead of passing easily to the wetland below, (it is believed) the water backed up under the cover material and formed a groundwater mound. The effect of such a situation in the landfill is to increase the size of the zone of saturation and thus the production of leachate. During periods of high discharge from the hillside into the landfill, strong pressure is also able to build up in the waste mass. When this is accompanied by cover erosion from runoff streams, the pressurized leachate enriched groundwater is able to break through the thinner cover barrier, resulting in large volumes of leachate pouring down the face of the landfill directly into the swamp stream.

A review of the landfill flow environment provides a background for contamination entering the wetland below. Groundwater enters the landfill or the silts and clays beneath it, directly from the carbonate. Downward percolation of this waste is largely confined within these two units by a thick gray clay which lies below the silty soil. On the top and sides, an engineered clay layer covers the waste and extends from hillside to wetland. While this clay layer decreases infiltration, it also serves as an internal flow inhibitor which leads to groundwater mounding in the waste. Water leaving this zone does so either as runoff from the surface of the landfill, as groundwater flow through the saturated silt below the waste, or as new seeps and springs of leachate from the face of the landfill. During periods of extensive rainfall and the associated high groundwater elevations, the combination of internal water pressure in the landfill and erosion of the cover material can lead to breaks in the leading edge of the landfill that are evidenced by leachate streams draining towards the wetland. A few of these seeps have been observed to be flowing constantly during the course of this research.

Wetland Flow Patterns

The final zone of concern is the wetland. By far, the majority of data gathered has come from this area because of its accessibility and our original assumption that leachate was discharging into it. Prior to 1975 the landfilling operation at Adams was a fairly disorganized process. Consequently waste placement took place in several areas, including parts of the wetland. In fact, waste disposal at one time came to within approximately 6.1 m (20 feet) of the swamp stream. Borings for wells 3, 6, 9, 11, and 12 all encountered the remnants of prior filling activity. The most striking example of filling activity in the wetland was observed in well #12. After augering through 1.2 m (four feet) of trash, roots, cinders, and other fill material, we encountered 1.2 m (3.9 feet) of peat from the original surface of the wetland.

The peat seen in well #12 is the surface soil found across most of the wetland. The exceptions to this are found in the fill encountered in the wells just mentioned, and in well #4 where the peat has taken on more of the character of a topsoil. The significance of this observation will be presented as the flow through the wetland is more fully evaluated. Permeameter and soil tests done on the peat (see Tables 3 and 4 and Figure 25) indicate that it has a very high natural water content and a low permeability. The large amount of water that it can hold (>125 percent) supports the dense cover of wetland vegetation.

Beneath the peat is a unit of very low permeability yellow silt and clay. Permeameter tests on this material yielded permeabilities of less than 10^{-7} cm/sec; essentially impermeable. This stratum is found only between the swamp stream and the diversion ditch and ranges in thickness from 0.7 m (1.4 feet) at well #13 to a trace at well #8. The next unit encountered is a gray silt and clay which has permeabilities comparable to those measured in the peat. Although this material is of low permeability at the yellow-gray silt and clay interface, test borings indicate that it does become somewhat coarser with depth. This trend continues for 0.6-1.2 m (two to four feet) until what

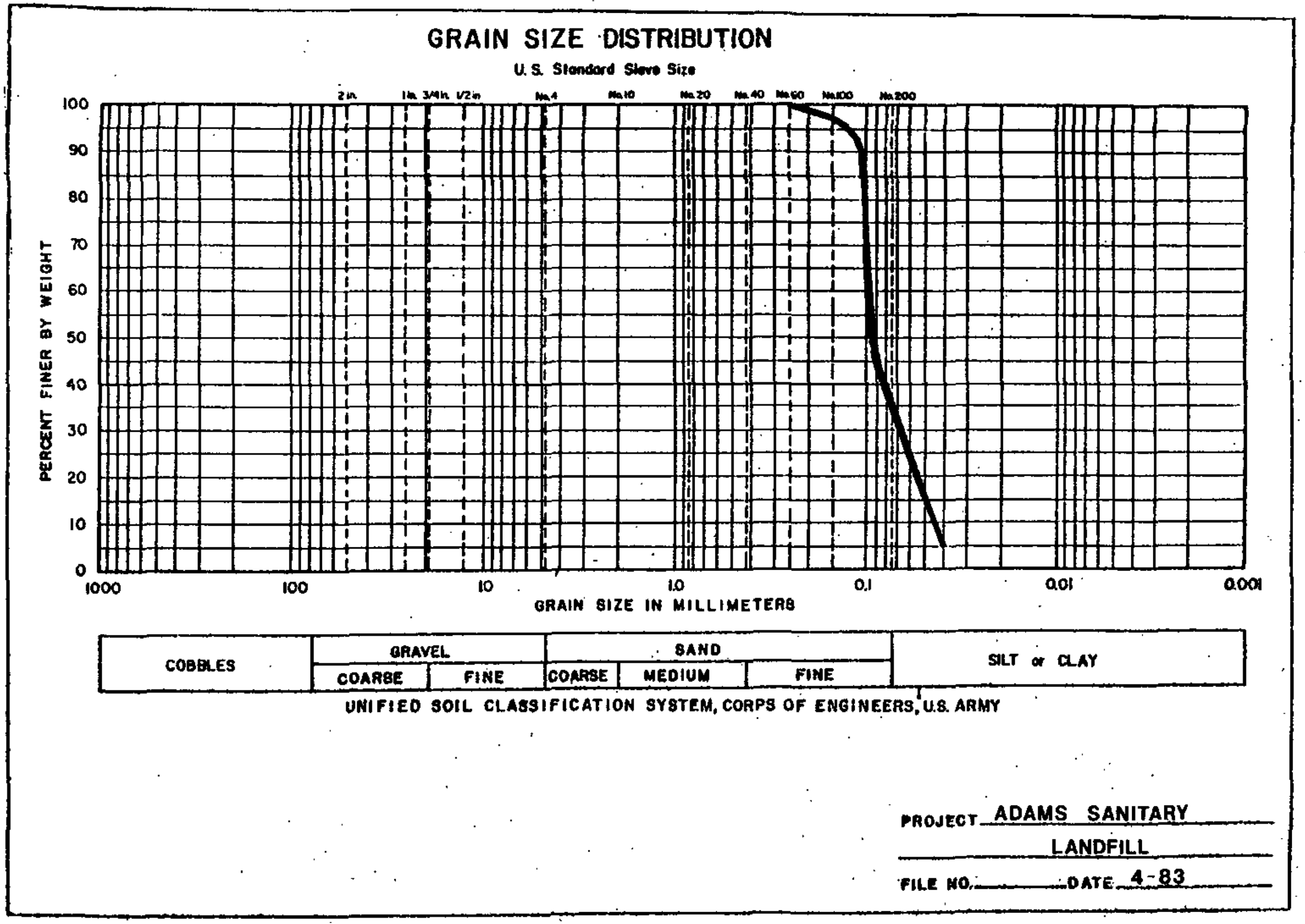


Figure 25. Grain Size Distribution of the Peat Layer

appears to be a mixture of clay, silt, sand and gravel sizes are encountered. The exact description of the soil is unclear, as the coarse nature made removal of a representative sample impossible. While probing with a sand sampling attachment, however, it was clear that this zone was coarser than any of the others found in the wetland. Because of the difficulty in removing a sample, no permeameter test was done on material recovered from this stratum. Owing to the percentage of fines recovered, it appears that while this zone will transmit water better than the others encountered, estimates on its permeability would lie in the range of 10^{-4} to 10^{-5} cm/sec. In addition, its thickness, which ranges from 0.6-0.9 m (two to three feet) also inhibits the movement of large volumes of water.

The lower boundary of the wetland study area is the thick, gray, lacustrine clay which was described earlier in the landfill section. In the cross section of soils found in the wetland (see Figure 26) the concave shape of the gray clay can be seen clearly. As noted earlier, the existence of this depression provided an area for the deposition of the alluvial deposits found above it. This depositional sequence, in conjunction with a constant drainage from the hillside brooks and seeps, led to the formation of the present wetland. Thus, any leachate generated by the hillside flow must probably work its way into the wetland ecosystem.

With a geologic base established, the mass balance of flow in and out of this discharge environment can now be evaluated. There are three main surface water bodies flowing into and out of the wetland. The first are the several seeps and springs of highly contaminated water draining from the landfill to the swamp stream. One estimate of the flow in one of these many streams showed a discharge of nearly 0.57 L/sec (13,000 gal/day) and, on that day, represented >1 percent of the total flow of the swamp stream into which it drained. Other leachate stream discharge values are listed in Table 6. The second body of water, whose flow is governed in part by the discharge from the leachate streams, is the swamp stream which has a known range of flow of from 2.8-17.0 L/sec (0.1 to 0.6 ft^3/sec). A number of smaller brooks originating on the hillside empty into this stream in the floodplain just to the south of the landfill. Once in the study area, the stream winds slowly northward through a narrow (0.9-2.1 m (3 to 7 foot wide)) channel until it breaks up into a series of machine dug canals on the Burnett property. These canals were excavated at the same time as the diversion ditch, the third major water body in the wetland. In the years past, spring runoff from the landfill occasionally spread a layer of iron-red leachate enriched water across the grazing field west of the wetland. The canals and the diversion ditch were put in place in an effort to

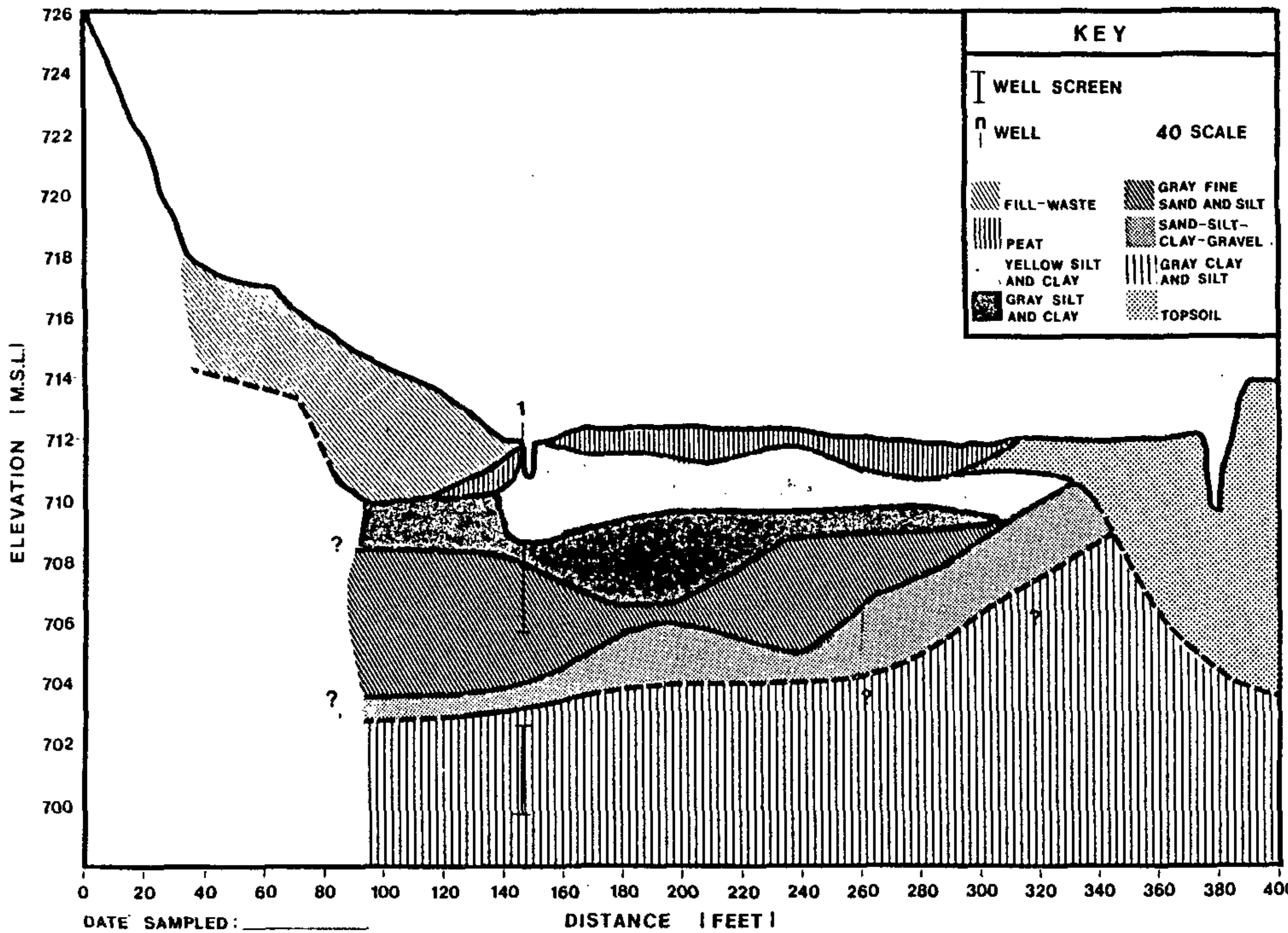


Figure 26. Complete Section View of Main Well Line

TABLE 6

Leachate Streams[1]: Discharge and Concentration Values

Date	Discharge (gal/day)	Specific Conductance (µmhos/cm)	Chloride (as NaCl) (mg/L)	Hardness (mg/L)
5/7/83	--	7800	1900	1150
5/25/83	4150	--	--	--
6/9/83	--	2300	450	700
7/9/83	6500	--	--	--
7/9/83	12900	--	--	--
7/9/83	6500	--	--	--
7/10/83	--	4900	1000	580

NOTES:

1. Leachate streams are defined here as flowing bodies of malodorous and discolored liquid draining from the face of the landfill, into the swamp stream. The streams above represent those believed to have the greatest discharge on that given date, but do not constitute the total number of leachate streams flowing on that date. The approximate locations of the seeps are shown in Figure 27.

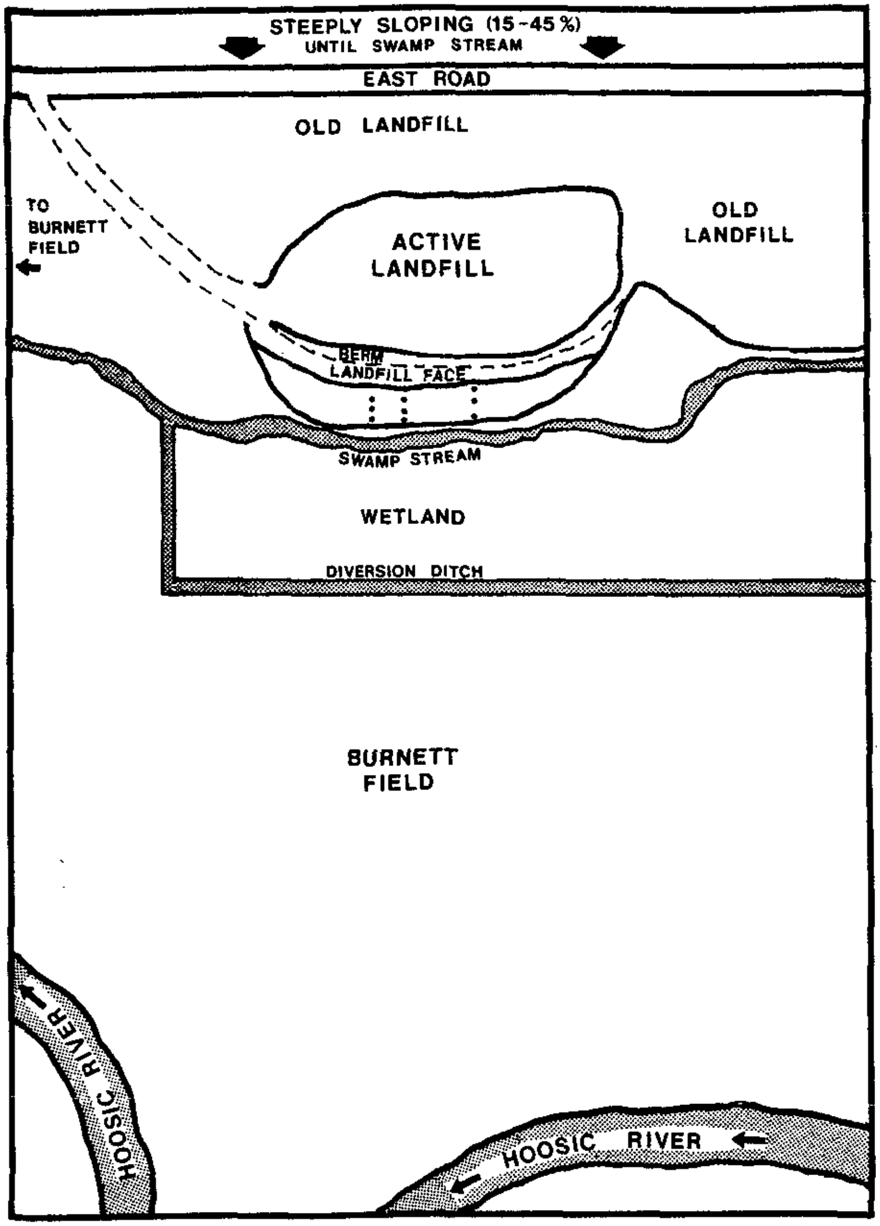
divert this seasonal event. The excavated diversion ditch and the swamp stream diverge at a point approximately 183 m (600 feet) south of well #1 and converge again where the swamp stream empties onto the Burnett property. On the same date that the 17.0 L/sec (0.6 ft^3/sec) flow was measured in the swamp stream, a similar value was recorded in the diversion ditch. Two weeks later, however, while the swamp stream flow had slowed considerably, the flow had completely stopped in the diversion ditch leading to the conclusion that the diversion ditch serves only to remove runoff and, to a much smaller extent, act as a discharge point for groundwater.

Another stream channel, which may or may not be of human origin, drains surface runoff from the swamp stream into the diversion ditch just north of well #4. No velocity measurements were made on this channel, which also seems to flow only in response to a storm event. The locations of all water channels at the site can be seen in Figure 27.

The subsurface flow conditions are less clear than the surface flows due to the incomplete geological information upgradient of the wetland, but in general this wetland is a zone of discharge for the aquifer that lies beneath. Motts and O'Brien (44) point out that in many wetlands, this role may in fact be reversed in late summer, when the peat, with its high degree of saturation, can provide recharge water to a deeper aquifer. The thickness of the peat at Adams, however, probably does not lend itself to providing substantial recharge and this contribution has been discounted.

The contribution of precipitation to the Hoosic River Valley during the study can be shown graphically in Figure 28. The trends of groundwater elevation vs. time in the well field piezometers follow closely the precipitation data from water years 1982-83. While studying wetland basins in Eastern Massachusetts, O'Brien (45) also noted rapid groundwater rise in response to precipitation. In addition, the rise was shown to be in near synchronization with stream levels, an observation also made at Adams. This response led O'Brien to conclude that a close coupling between groundwater and wetland exists. We have reached this same conclusion.

The groundwater response leads to the second contributor to wetland recharge, that of baseflow. Groundwater moving into the wetland is under artesian conditions in the wetland. Groundwater elevations in the piezometers are, in many cases, above the ground surface as evidence of this fact. Additionally, hand-excavated holes through the peat will yield water, which rises quickly in the hole and discharges to the ground surface. By definition, an artesian or confined aquifer is one in which the formation



Legend
...Leachate seeps

Figure 27. Study Location; Plan View

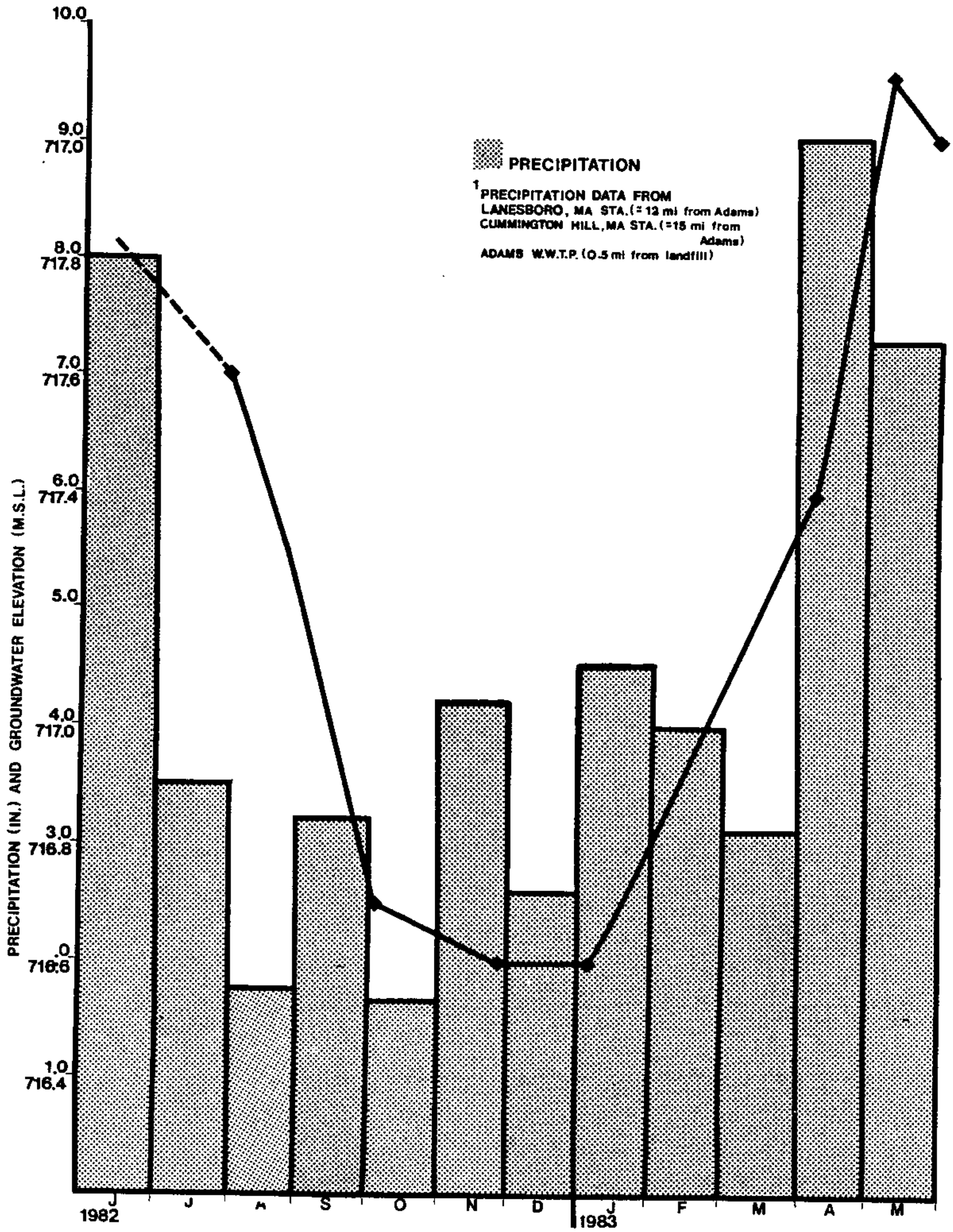


Figure 28. Precipitation (1982-83) vs Groundwater Elevation (Well #3)

transmitting the groundwater must be confined above and below by material with lower permeability. Figure 29 is a schematic showing the difference between confined and unconfined aquifers. It is our belief that the thin unit of fine and coarse material described earlier might provide hydraulic connection between the hillside and the wetland and give rise to the artesian condition seen.

In this scenario, water from the carbonate enters this unit after passing through or beneath the landfill. The flow continues its downgradient migration until a point approximately beneath the swamp stream. At this spot in the aquifer the groundwater changes from a generally horizontal pattern of flow to a more vertical trend. The flow net in Figure 21 and the vertical gradients measured in the nested piezometers, which are an order of magnitude greater than the horizontal gradients, attest to this observation. The reason for this change, however, is not as apparent as its actual occurrence, but borings at wells #8 and #4 may provide a clue. In well #8 the thin, coarser unit described earlier is approximately 0.8 m (2.5) feet thick and located at a depth of 1.5 m (5.0 feet). In well #4 the coarse soil is encountered at 1.4 m (4.5 feet), but now the percentage of fines in the soil matrix has increased considerably. Our belief is that the coarse unit, as found in well #8, becomes much more fine grained between these two wells. The more fine grained material now acts as an aquitard, preventing further significant horizontal flow. With an impermeable gray clay below, and a decrease in the permeability of the coarse unit, the natural flow direction for the water, under a strong gradient from the hillside, is to move in the direction of least resistance. Gradient measurements at almost all the wetland piezometers show this direction to be upward.

A striking example of this flow pattern can be seen on the ground surface between wells #8 and #4. At well #8 the ground is saturated, has occasional standing water, and is covered with lush wetland phreatophytes. At well #4, some 12.2 m (40 feet) away, the ground is dry and firm and the vegetative cover is a short dry grass. A distinct vegetative interface can be identified between the two wells.

The third contribution of water moving through the wetland is runoff. This includes the water draining into the swamp stream from the hillside breaks, and the leachate springs emanating from the base of the landfill. We also noted the possible surface runoff from the wetland itself into the diversion ditch. A fourth contribution to runoff is that water which drains from the surface of the active landfill. This flow generally works its way down the landfill face and into the channels carved by the leachate springs. The effect of this erosional process on the creation of new seeps has already been discussed.

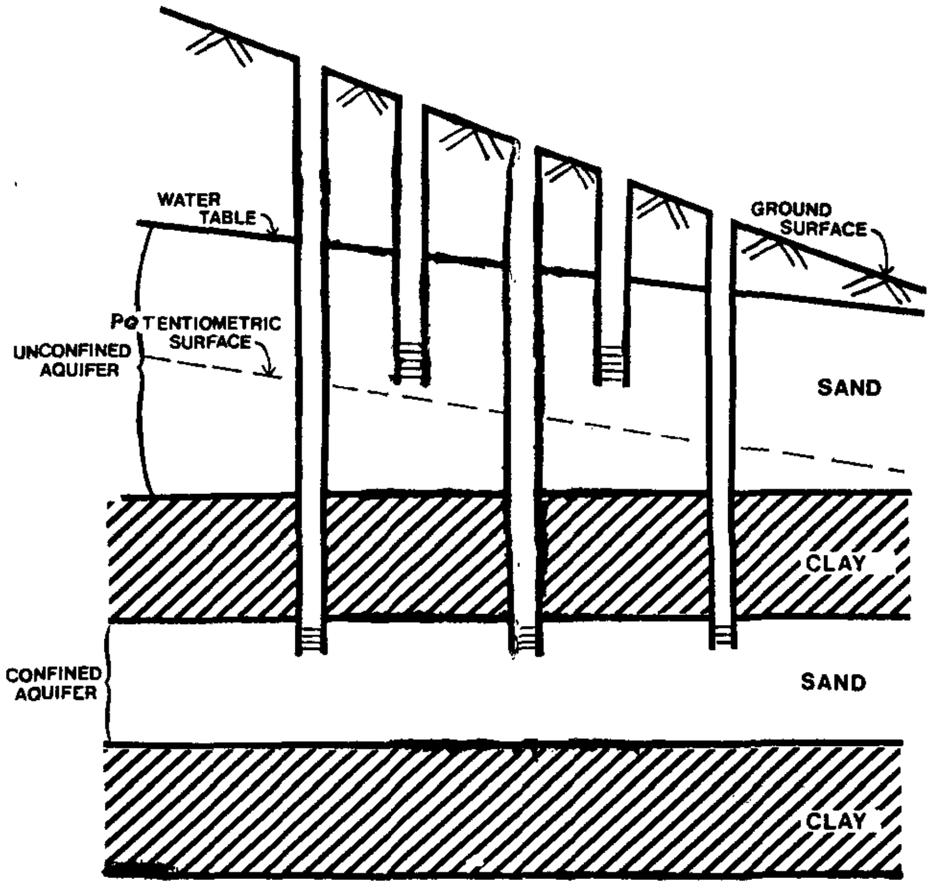


Figure 29. Confined and Unconfined Aquifers

Groundwater flowing from the hillside has been shown to change from a generally horizontal flow pattern to a more vertical flow in the vicinity of the wetland. It would seem likely that significant volumes would move toward the point of lowest head, a free water surface, and discharge. This appears to be particularly true for the swamp stream. Using the flow net shown in Figure 21 we can estimate a seepage over a given cross-sectional area which includes the swamp stream, and determine that as much as 25 percent of the groundwater moving toward the wetland will discharge into the swamp stream. In the diversion ditch, however, both flow net analysis and seepage meter tests indicated a significantly smaller percentage of groundwater discharging. It is clear that the bulk of groundwater in this zone discharges to either the ground surface or the swamp stream, and very little reaches as far as the diversion ditch.

As an example of this, the results of seepage meter tests in both bodies of water can be compared. When discharge into the diversion ditch over a 61 m (200 foot) length (the width of our control volume) is evaluated, it accounts for only 6.6×10^{-6} L/sec (0.15 gal/day) or less than 7.6 L/day over the 55.7 m^2 (600 ft^2) of diversion ditch. For a similar reach length of swamp stream (200 ft) the contribution amounted to 302.4 L/day (80 gal/day) over the 130 m^2 (1400 ft^2) of stream bottom. Thus, much more water discharges to the swamp stream than to the diversion ditch. It should be noted, though, that the total flow in the swamp stream on that same day (7.6×10^6 L/day); 200,000 gal/day) indicates that, overall, runoff provides a significantly greater portion of the streamflow. Therefore it appears that the diversion ditch functions simply to handle excess runoff in the wetland and not as a primary zone of either recharge or discharge.

The volume of groundwater which does not discharge into the swamp stream continues its migration through the wetland soils. A small percentage of this groundwater will probably pass through the wetland aquifer and continue its travel beneath the grazing field until it discharges into the Hoosic River, some 305 m (1000 feet) beyond. Most of the water which does not discharge to the swamp stream, however, discharges over the entire surface of the wetland. In addition to the flow net data, support for this hypothesis lies in the very nature of the wetland and its role in the hydrologic cycle. Saines (54), Williams (66), Motts and O'Brien (44), and others unanimously agree that wetlands represent zones of discharge for the majority of the year. While Williams points out that this may change during August and September in temperate climates, the bulk of the water entering this environment is discharged.

The water discharged to the ground surface in the wetland is driven during much of the year by the mechanism of evapotranspiration. Linsley, Kohler and Paulus (40), for example, point out that in a zone densely covered with phreatophytes, this can amount to 10^6 $M^3/km^2/yr$. A value of this magnitude in the Adams wetland would account for more than one-third of the entire aquifer volume. Given the small groundwater velocities through this zone, evapotranspiration could account for a large flow out of the wetland control volume. Tenn <u>et al</u>. (60) and Karpl (31) have noted similar large values for evapotranspiration from the surface of bogs and marshes. Values from their research are presented in Table 7.

Returning to the question of only 25 percent of the groundwater discharging into the stream bottom, we thus turn to the evapotranspiration in the wetland. Whereas groundwater may only move a few centimeters per year through the silt and clay, phreatophyte roots penetrating this stratum provide a channel of escape and become the zone of least resistance to groundwater flow rather than the water surface in the stream.

In conclusion, it appears that the bulk of the water falling on the three zones discussed thus far either percolates into the carbonate to recharge the deeper artesian aquifer or makes its way into the streams via surface runoff. That percentage which does enter the landfill, solubilizes the waste and moves as groundwater flow to discharge into the swamp stream or wetland surface. The other path of contaminated groundwater discharge is via the seeps and streams from the face of the landfill which are the result of groundwater mounding occurring beneath its surface. Fresh water from the carbonate may also be flowing beneath the landfill and serving to dilute the leachate enriched groundwater as it moves to discharge to the surface of the wetland. The extent of contamination as a result of this flow scenario can now be evaluated.

TABLE 7

Approximate Annual Consumption of Water by Plants

Types of plants	Inches of Water	mm of Water
Coniferous trees	4-9	100-360
Deciduous trees	7-10	180-250
Clover and alfalfa	2.5+	60+
Wheat	20-22	510-560
Meadow grass	22-60	560-1500
Lucern grass	26-65	660-1650

Source: Ref. (60).

Approximate Annual Consumption of Water at Various Sites

Type of Site	Inches of Water	mm of Water
British Bog	45	1200
Wyoming Bog	40	965
Minnesota Bog	30-60	715-1565
Grass Marsh	95	2375

Source: Ref. (31).

CHAPTER VII

WATER QUALITY

The preceeding chapter presented a generalized approach to the regional flow patterns in and around the Adams landfill. While Pettyjohn (47), Kramer (35), and others note that the density of the particular contaminant may cause it to sink or float in an aquifer, this study involves conservative parameters that are likely to follow the regional flow. Additionally, the thickness of the aquifer (3.1-3.7 m) (10-12 feet) is such that a strong density gradient is unlikely.

Groundwater Monitoring

The brunt of the monitoring effort involved groundwater rather than surface water, with the majority of sampling done in the main well line running perpendicular to the landfill. Sampling was initiated in March 1982 and continued monthly until July 1983. On a less regular basis, surface water samples were taken from the swamp stream, diversion ditch, and a number of other locations around the study area. The purpose of this sampling program was to identify the location and extent of the leachate plume, monitor its movement over time, and determine the impact of the landfill on the surface and ground water quality at the site.

In all samples, the three parameters, chloride, hardness, and specific conductance were measured. Additional parameters were occasionally measured during this study. Some samples were also analyzed by MDWPC and DEQE. Their data confirmed the contaminant trends established with the primary parameters, defining spatial variations of different indicators in the same plume.

As each block of data was received, the parameter concentrations were plotted in cross-sectional sketches along the main well line (see Figure 30). The data was also plotted on graphs that were designed to show trends in a given parameter over time for each individual well. Plan view maps were drawn for the study area showing changes in concentration as the plume moved across the wetland. These figures and graphs, in conjunction with the permeability tests done on the wetland soils, provided more information to support the explanation of the flow patterns in the wetland aquifer presented above and the movement of the associated leachate plume. By themselves, the parameter measurements of surface and groundwater quality did not define the extent of the groundwater plume. Rather, this information, observations made

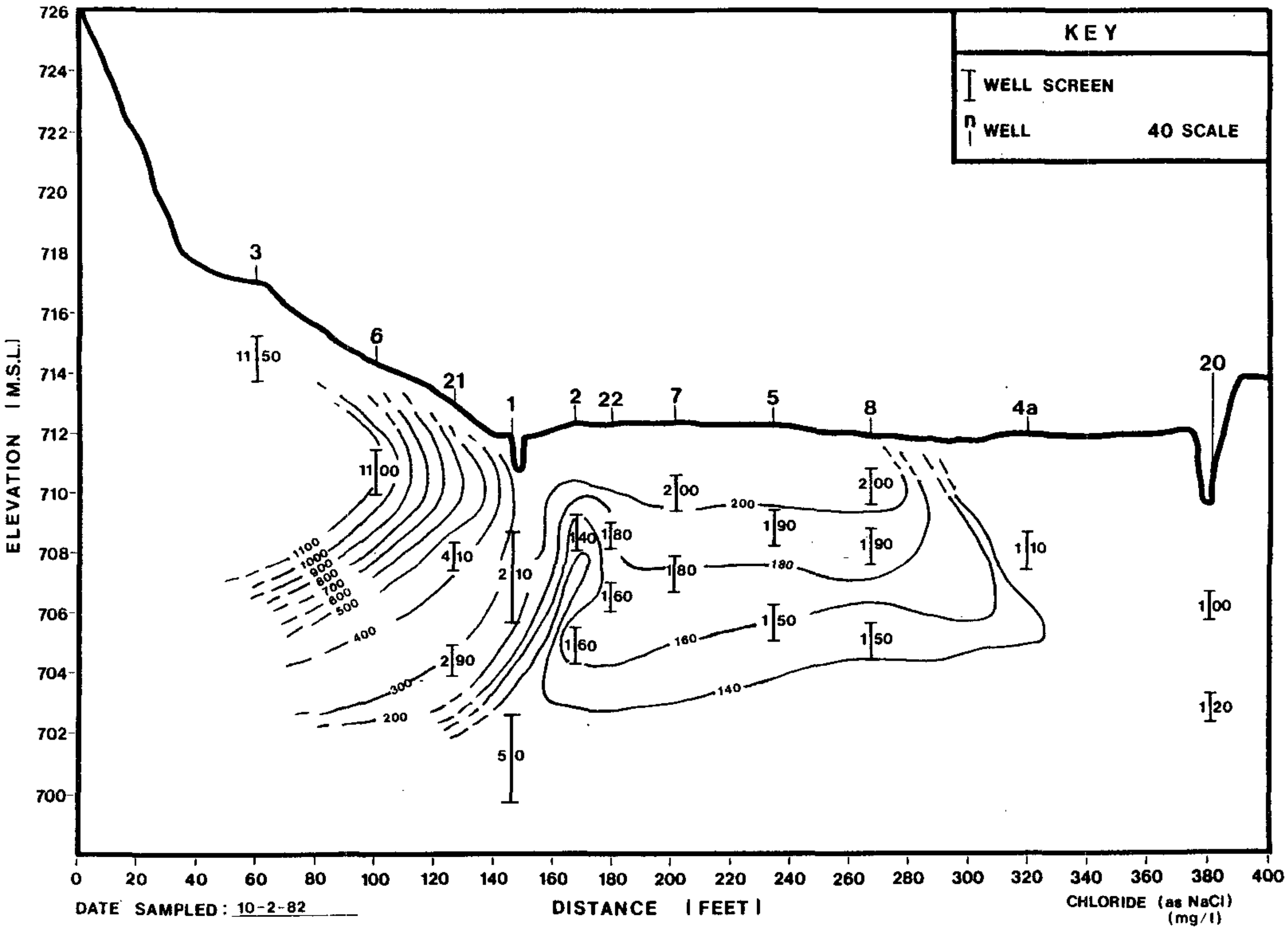


Figure 30. Profile of Chloride Concentrations along Main Well Line

during site inspections, soil and rock geology, and the other evidence gleaned during the course of the study were used to define the plume.

In general, the leachate being produced by the landfill is moving via surface and groundwater towards the wetland well field. It was also found that leachate migration originating as a surface flow is of significantly more deleterious quality than any measurement made on groundwater quality. On May 7, 1983, for example, a surface water quality measurement from a leachate stream discharging from the face of the landfill recorded specific conductance and chloride values of 7800 µmhos/cm and 1900 mg/L respectively, while on that same day, the highest values recorded in any well came from #3-1 (i.e., the top screen of well #3 - 3 to 6 feet deep), with readings of 3300 µmhos/cm and 700 mg/L. It should be noted that the surface values in question here are from leachate streams only, and do not include measurements made in either the swamp stream or the diversion ditch.

Monitoring wells #3 and #6, (see Figure 30) located in old fill and refuse, show high concentrations beyond the zone of active filling, although slight decreases in concentration are observed in the direction of flow. Between wells #6 and 21 a strong decrease in concentration is detected. As the migrating plume leaves the old fill material and the zone of surface runoff, and enters the saturated wetland soils, a large amount of dilution appears to be taking place. In addition, well #21 lies within 6.2 m (20 feet) of the swamp stream, where a significant volume of groundwater is discharged, changing from a horizontal movement to a vertical one. Also, it appears that freshwater is discharging from the thin coarser unit below and plays a large role in diluting the contaminants in well #21.

Beyond well #21 the next well in the direction of flow is the deep well #1. The water quality in the deep piezometers 1-1 and 1-2 was found early on to be of significantly better quality than any other piezometer samples at the site. This knowledge served to confirm the theory that the thick clay unit beneath the site, while having the ability to transmit some water, effectively seals any deeper aquifers from contamination from the landfill. In fact, the water quality in these well points was found to be of better quality than the water taken from the upgradient background well.

Piezometer 1-4, however, is located in the shallow aquifer and has leachate enriched groundwater being pumped from it. The quality of the water being withdrawn has been shown to define somewhat of a groundwater divide. The concentrations are less than those found in well #21, yet are of a similar order of magnitude to those wells located in the wetland, a trend which has also been seen to shift as well, with concentrations in 1-4 occasionally reflecting more the quality of the near landfill wells. Again, dilutional effects of fresh water moving to discharge into the stream and a shift in flow patterns may account for this unique condition.

Continuing along this well line in the direction of flow, the piezometers in wells #2, 22, 7 and 5 have consistently remained at about the same concentrations throughout the course of the study. Because the parameters monitored are conservative substances and because a large volume of water is being discharged via evapotranspiration on the ground surface above these wells, this would be expected to be the case. In addition, groundwater velocities through the wetland soils are such that appreciable changes in concentration through the zone would not be expected during a study of this duration.

An unusual observation is made at the next well in the line, well #8. The center piezometer on this three position well, 8-2 has had, until just recently, higher concentrations of all parameters than the wells immediately upgradient from it. While this may seem unlikely given the flow pattern postulated thus far, two reasonable explanations exist. It has been shown, that the swamp stream is one point of discharge for subsurface contaminants moving downgradient from the landfill. If, during a period of drought, the ground water levels were to fall below the bottom of the stream, where then would the contaminants go? Logically, the leachate plume would continue moving towards the next available discharge point, the wetland plants, and in so doing, entirely bypass the stream, enter the wetland, and move to discharge at the ground surface or continue on toward the Hoosic River. If the water table were then to rise, part of the plume would again be discharging a percentage of its flow to the stream, decreasing the concentration of leachate entering the wetland soils. What was once a highly concentrated plume entering the wetland, is now a segregated slug of leachate enriched groundwater with concentrations higher than that water which follows it. Figure 31 illustrates this hypothesis.

The second explanation, although less interesting, may be more reasonable. On May 7, 1983, piezometer 8-1 (see Figure 30) was sampled and analyzed, and the results indicated surprisingly high values (Spec. cond = 19,000 µmhos/cm, pH = 1.8). While the possibility of something being buried at this location in the wetland seems remote, it would explain the "pocket" of contamination that has been seen at this well for the duration of the monitoring period. It is curious to note, however, that on the day that those high values were recorded, piezometers 8-2 and 8-3 showed decreases in their concentration from the previous reading. As with the anomaly in well #1-3, more research would be needed at this specific site in order to clear up this

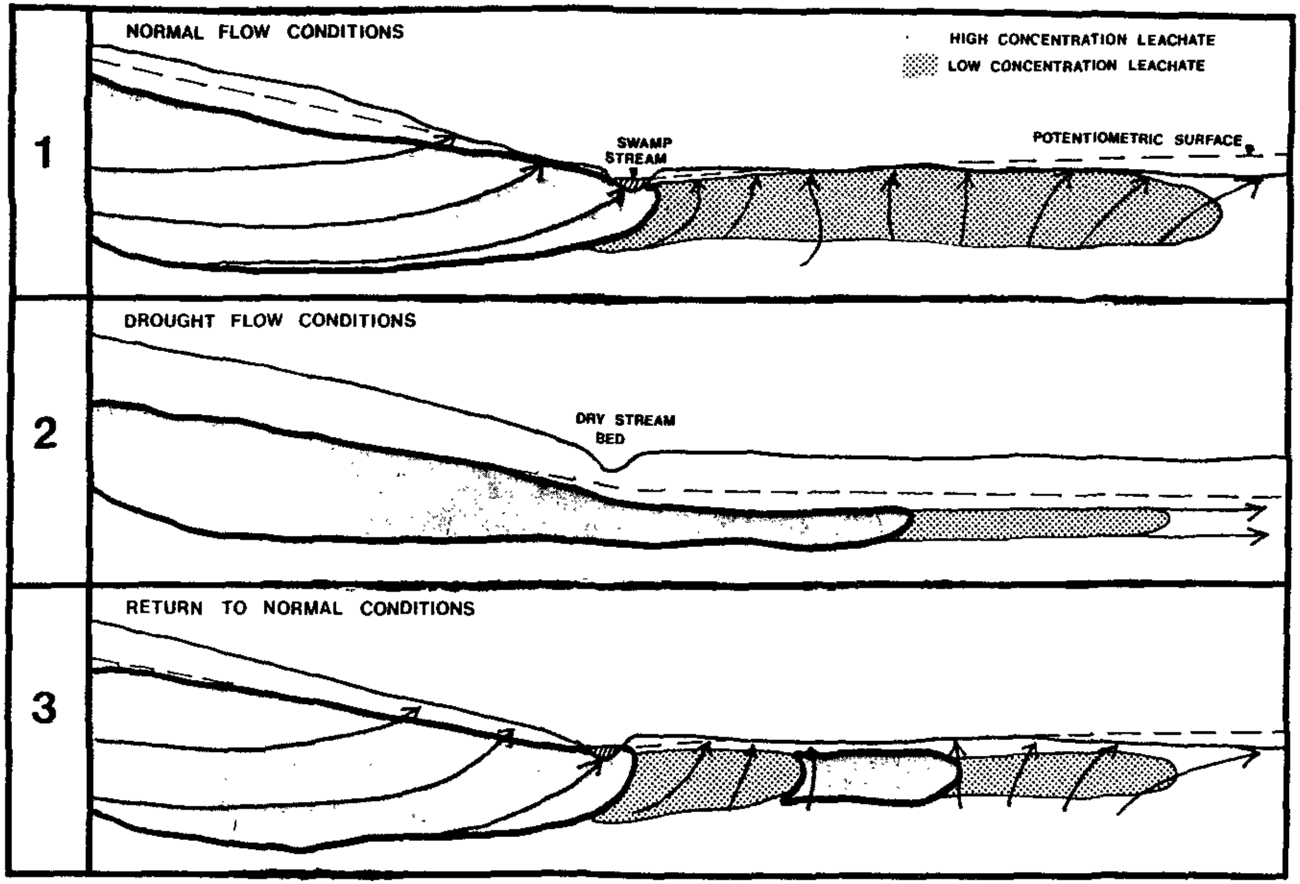


Figure 31. Possible Explanation for the Presence of an Isolated Zone of High Concentration

discrepancy. Owing to the values recorded, though, extreme caution is suggested to anyone investigating this problem.

The vegetation divide that has been previously discussed is beyond well #8. The concentrations in the wells beyond this point are less than those found in the central section of the wetland, which conforms with the flow patterns previously described. While decreases in concentration are observed, it appears that a significant volume of the flow is still being directed horizontally towards the grazing field, as evidenced by the head gradients in the piezometers. Wells #4 and 20 have shown concentrations somewhat higher than the background well as still further proof.

Surface Water Monitoring

As previously mentioned, the surface water quality at the Adams landfill has been the object of investigation on several previous occasions. The severe, malodorous runoff from both the landfill surface and face which drains into the swamp stream has led previous researchers at the site to undertake their studies. While groundwater monitoring was the primary focus of our water quality study, such a large volume of runoff cannot be overlooked in a comprehensive field evaluation.

Previous studies have indicated that a fairly good quality water is entering the wetland from the brooks on the hill. That quality progressively degrades as the swamp stream meanders through the site, only to improve in quality again as it empties into the canals in the Burnett field north of the site. C. E. Maguire (41) states that, 'Water flowing out of the site area by one brook (the swamp stream) through a meadow (the Burnett field) shows a trend in the gradual attenuation of the dissolved solids as the water flows toward the Hoosic River.' The figures (Figures 32 and 33) prepared by the various investigators show this trend and point to a decrease as the stream leaves the wetland.

Contrary to this finding, however, our results (see Figure 34 and Table 8) show a continuing increase in concentration as the stream enters the north Burnett field. It is not until well across this field, after passing through the series of canals, that the concentration begins to decrease significantly. In general, it is clearly shown that deleterious stream water quality is the direct result of both runoff from the face of the landfill and the groundwater discharge into the bottom of the swamp stream. While our findings indicate a longer stream processing time to reduce the concentrations to near background water quality than previously reported, the measurements made show that a relatively good quality water is eventually discharged into the Hoosic River.

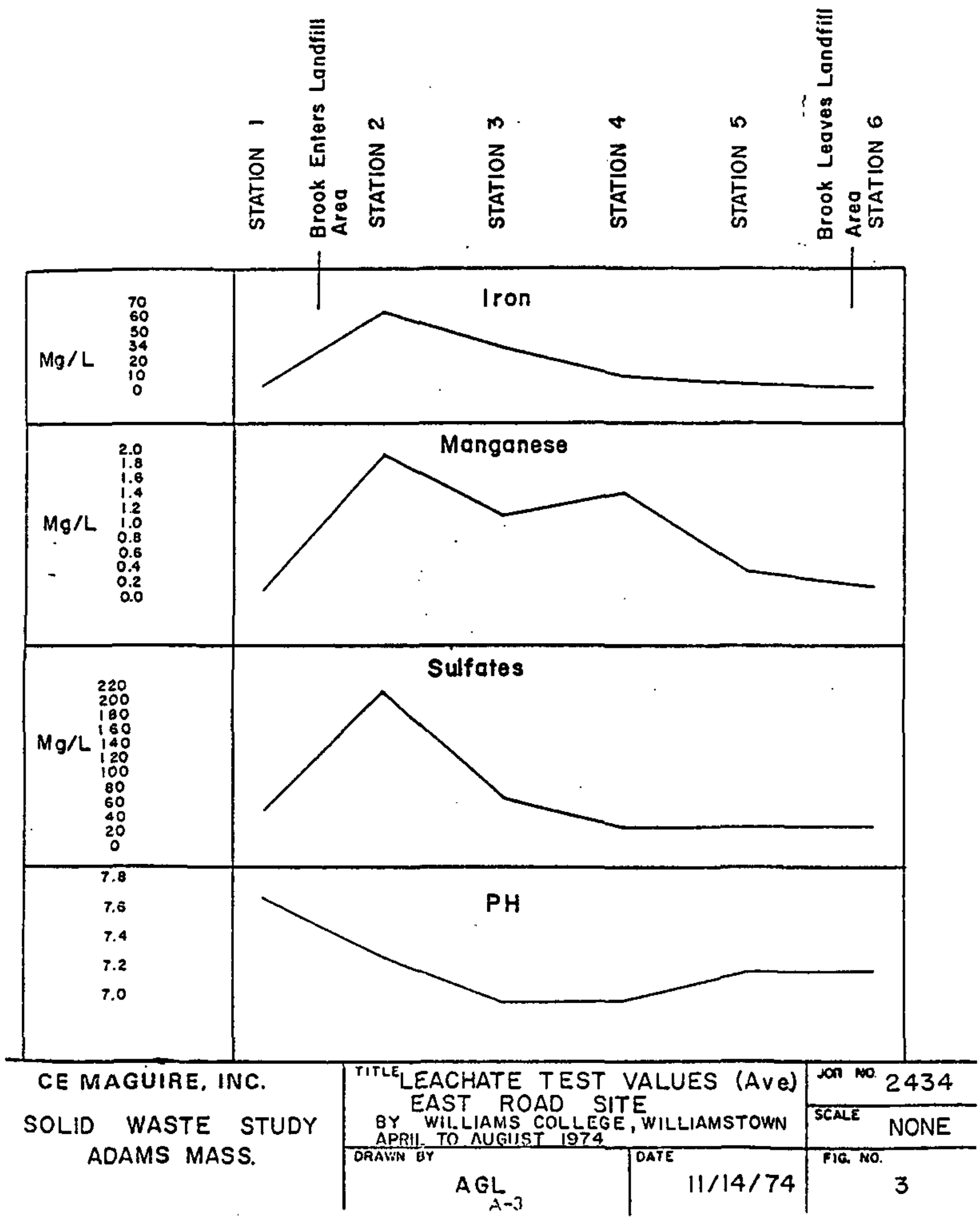


Figure 32. Leachate Test Values (Williams College Study)

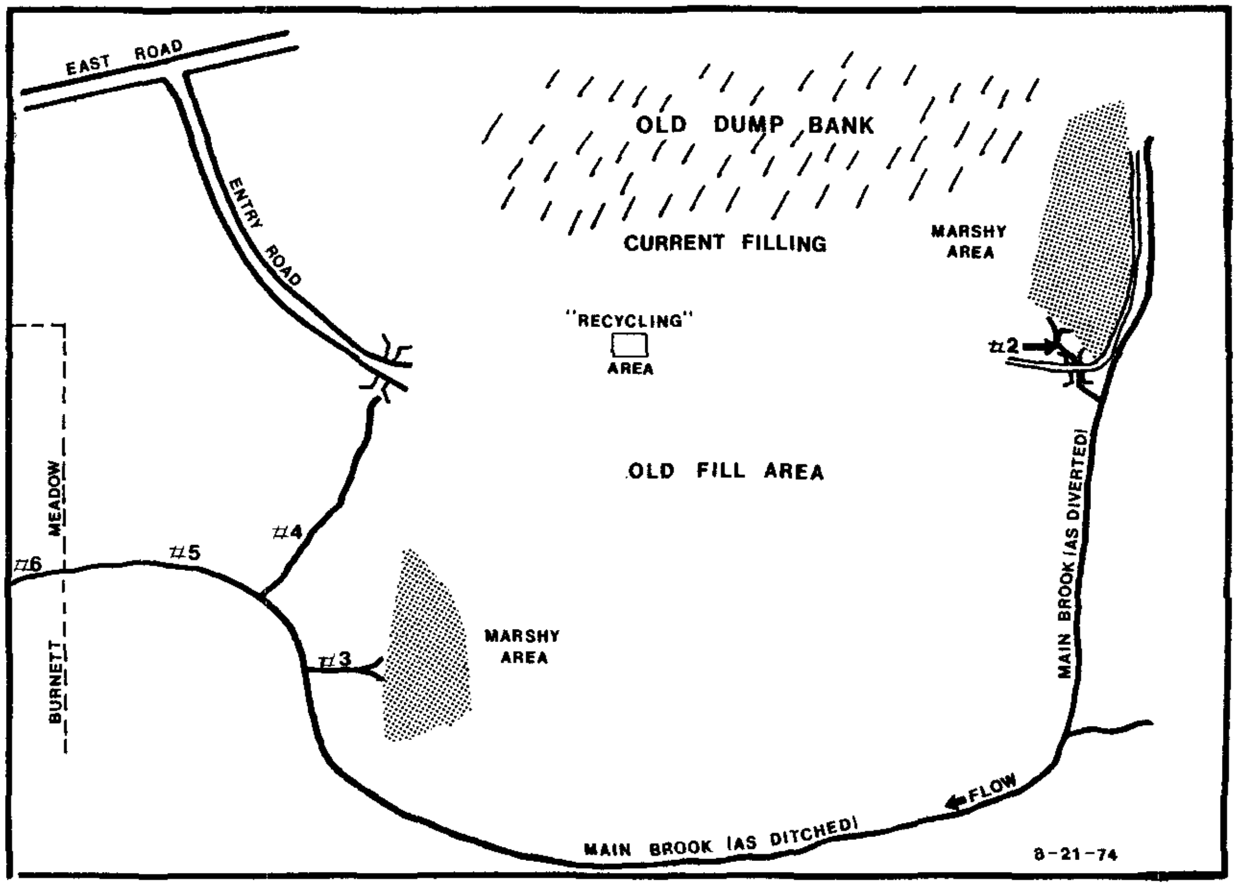


Figure 33. Williams College Study: Plan View

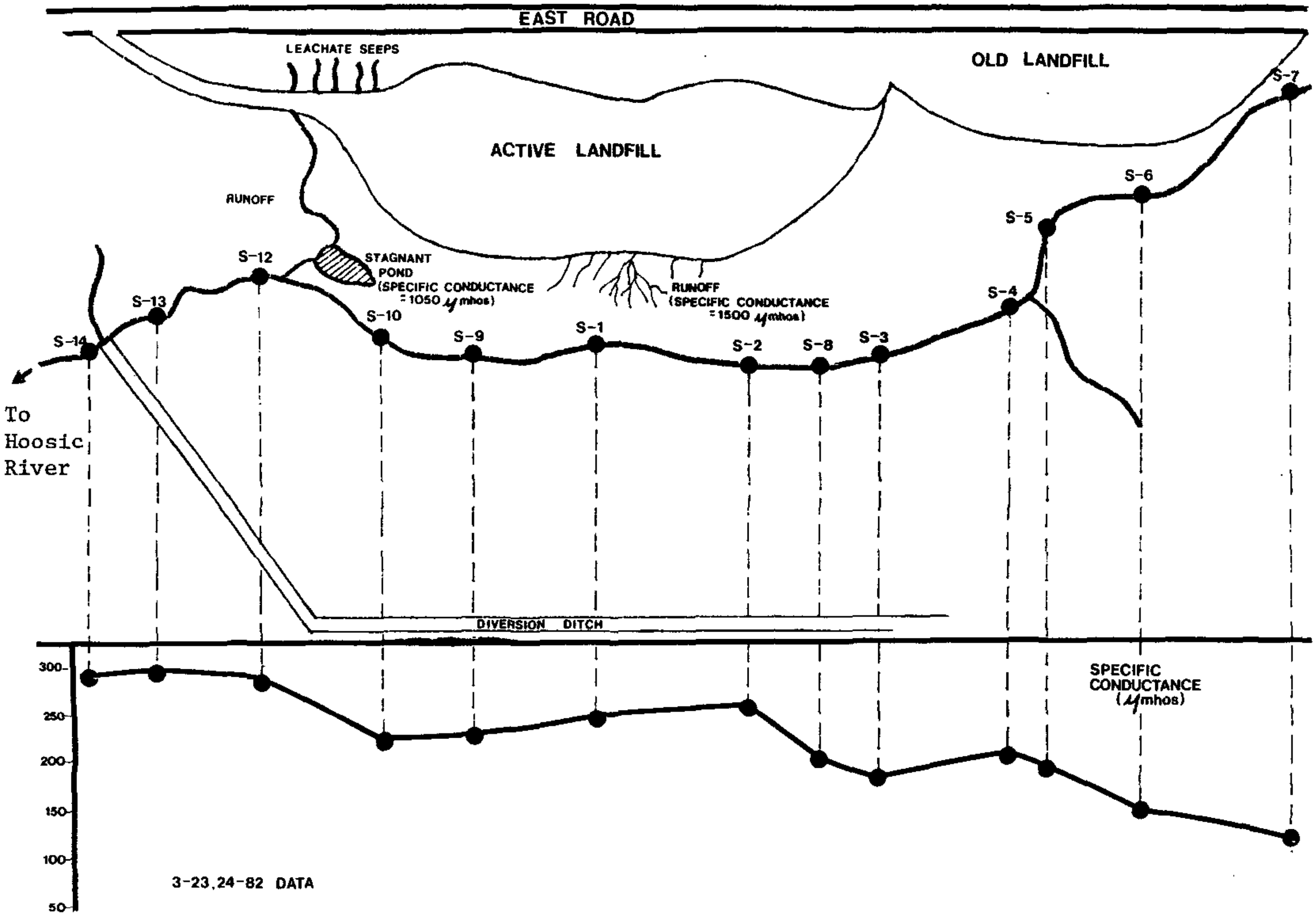


Figure 34. Adams Landfill: Specific Conductance in Swamp Stream

TABLE 8

Swamp Stream and Diversion Ditch Concentration Data

Date	Location	Specific Conductance (μmhos/cm)	Chloride (as NaCl) (mg/L)	Hardness (mg/L)
3/13/82	S1[1]	430	-	-
	S2	380	-	-
	S3	185	-	-
	S4	205	-	-
	S5	200	-	-
	S6	170	-	-
	S7	115	-	-
	S8	350	-	-
	S9	365	-	-
	S10	365	-	-
	S11	225	-	-
	S1	250	-	-
	S2	260	-	-
	S3	180	-	-
	S4	210	-	-
	S5	195	-	-
	S6	150	-	-
	S7	115	-	-
	S8	210	-	-
	S9	225	-	-
	S10	220	-	-
	S11	1050	-	-
4/7/83	Swamp Stream 8.0' upstream of well #1	625	90	255
5/7/83	Swamp Stream at Well #1	650	100	270
5/7/83	Swamp Stream 10.0' upstream of well #1[2]	1100	190	270
5/7/83	Diversion ditch at well #20	350	50	200

TABLE 8, Continued

Date	Location	Specific Conductance (µmhos/cm)	Chloride (as NaCl) (mg/L)	Hardness (mg/L)
5/7/83	Burnett Property half way between wetland & Hoosic River	550	90	260
6/9/83	Swamp Stream at well #1	1625	250	615
6/9/83	Diversion Ditch at Well #20	500	60	240
6/9/83	Diversion Ditch N.W. corner of wetland	500	75	240
6/24/83	Swamp Stream[3]	300	40	200
6/24/83	Swamp Stream[4]	950	240	340
7/9/83	Swamp Stream near Well #12	625	80	360
7/9/83	Swamp Stream at well #1	806	110	390
7/9/83	Swamp Stream near well #9	750	110	390

NOTES:

1. See Figure 34 for the location of these samples.
2. A leachate stream drains into the swamp stream at this location.
3. A small stream on the hillside; east side of East Road and draining into the swamp stream in the valley below. This water represents 'Background' stream water quality.
4. Where the swamp stream leaves the wetland and enters the excavated canals in the North Burnett field.

The reasons for the degradation of surface water quality may be due to the general impression that the overall site conditions have deteriorated in the 21 months of this study. The flora and the fauna observed in and around the stream during the first summer of study had either disappeared or were less plentiful in the second summer of research. The odors and number and general appearance of leachate seeps also appeared to have taken a turn for the worse. With greater volumes of contaminated discharge draining into the swamp stream, a longer detention time is necessary for the stream to recover.

It should be pointed out that when research began at the site, the water levels were much lower than at any time during the study. For example, drilling at well #1 was done on dry ground 0.3-0.6 cm (1-2 feet) from the swamp stream. .Following a heavy runoff period in the spring of 1982, the swamp stream rose to encompass the well, and never completely receded. Higher than average precipitation for water year 1982-83 also accounts for the rising groundwater elevations. The unusually high rainfall values are evident when compared to average precipitation for the region (see Figure 35). These higher ground and surface water levels may effectively increase the production and subsequent transport of leachate and provide an explanation for the site deterioration.

The water in the diversion ditch, when it is flowing, is usually of just slightly better quality than the water pumped from well #20, and leads to an interesting conclusion. The ground water which does not discharge into the swamp stream or discharge via evapotranspiration from the surface of the wetland is probably discharged into the diversion ditch. This would again confirm that the wetland area is a discharge zone and that the groundwater and surface water contamination is contained almost entirely in the wetland soils and/or discharged to its streams.

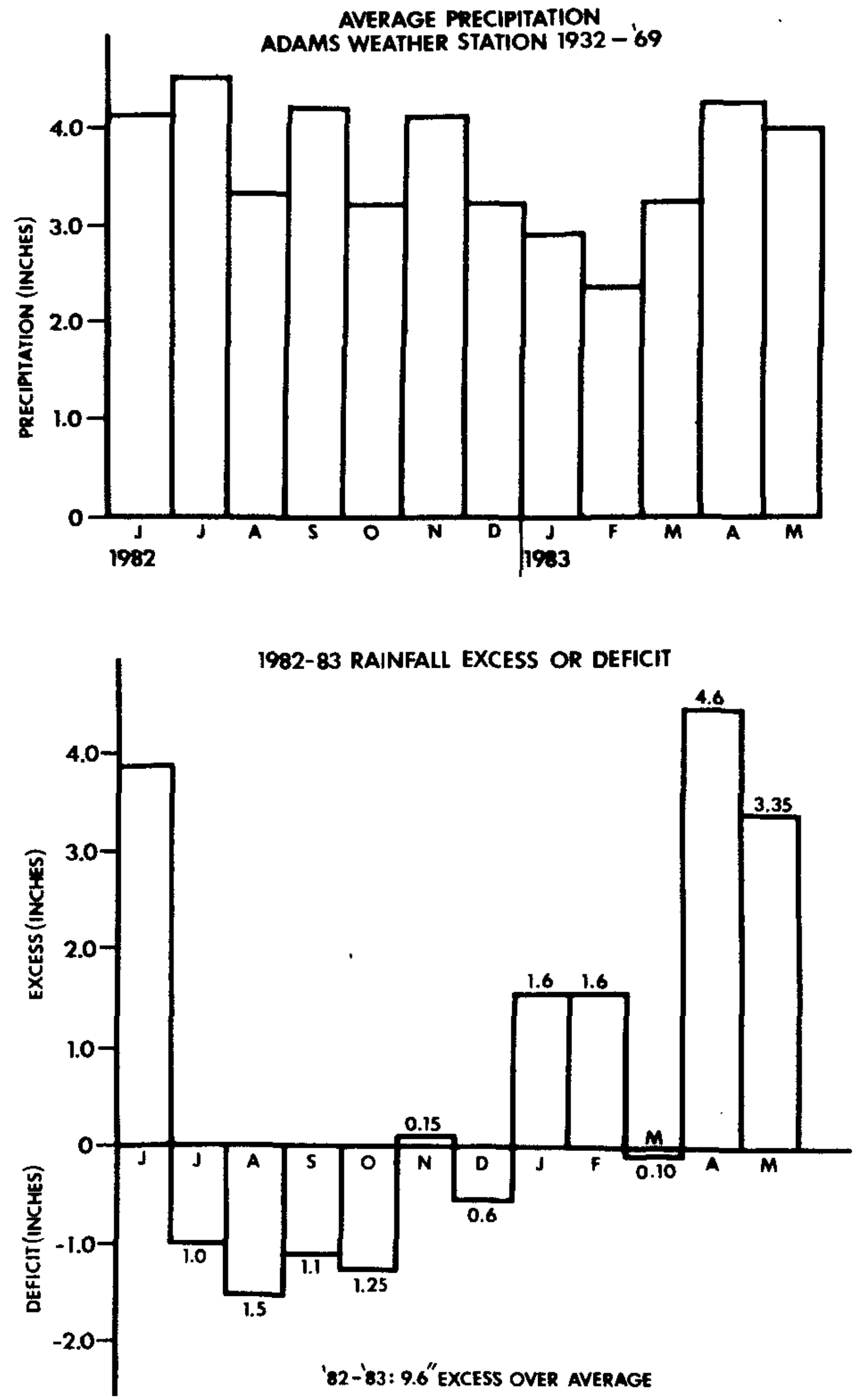


Figure 35. Average Precipitation at Adams (1932-69) and Rainfall Excess or Deficit 1982-83 vs. 1932-69 Average

CHAPTER VIII

CONCLUSIONS

It is clear that leachate is being produced by the Adams landfill. These pollutants are then transported either as surface runoff or groundwater flow into the wetland environment below the landfill.

Summary

The regional groundwater flow pattern, as anticipated, moves from the upland hillside to discharge in the valley bottom. While this affects the production of leachate by having groundwater discharging directly into the waste, the majority of infiltrating water continues its downward percolation through the dolomite unit. That water which does pass through the refuse enters the landfill horizontally from the carbonate unit. Rather than passing directly through, however, the low permeability cover material on the landfill sides inhibits the flow, leading to a groundwater mounding situation and the subsequent saturation of greater volumes of waste.

Pressures induced by the presence of the mound lead to the existence of seeps and springs from the base of the landfill. These, in conjunction with leachate enriched groundwater constitute the two main sources of water contamination to the wetland below. Approximately 25 percent of the groundwater flow and virtually all of the surface seeps find their way into the swamp stream which meanders along the edge of the wetland. By the mechanisms of dilution and attenuation this leachate is reduced in strength by the swamp stream. Following its discharge into the stream it flows through a series of excavated canals during which time concentrations are further reduced.

That percentage of leachate enriched groundwater which migrates beneath the swamp stream is believed to discharge via evapotranspiration in the wetland. Both flow net analysis and prior research bear out this finding. Any contaminated groundwater continuing through the wetland soils either discharges into the diversion ditch (which will eventually rejoin the swamp stream) or continues on beneath the grazing field, eventually reaching the Hoosic River. It appears that the contribution of contaminated groundwater to the Hoosic River is virtually negligible.

Thus, while site conditions appear to have deteriorated during the 21 months of research, significant groundwater contamination is not occurring as a result of the Adams landfill. For the parameters studied, the strongly contaminated water is 'treated' either by dilution in the swamp stream or evapotranspiration in the wetland. Due to the shallow nature of the aquifer involved and the close proximity of the wetland and swamp stream with respect to the wetland, a very small area, beyond the landfill itself, appears to be affected.

During the installation of the monitoring wells, an interesting observation was made. The RCRA regulations indicate that a minimum of three downgradient wells must be located in order to detect contamination in the uppermost aquifer. Though there is nothing inherently wrong with this requirement, it is imperative that careful and detailed study precede the actual well installation. The situation at Adams illustrates the pitfalls of failing to do site specific evaluations. As one moves from well #7 to well #5 in the cross section, it would appear that concentrations, which had once been decreasing in a direction consistent with the assumed direction of flow, were now increasing. If only these two wells had been installed, contaminant data might have indicated a completely different flow direction or contaminant migration trend. Depending on the well placement, the results might have indicated a different groundwater flow direction, missed the influence of the swamp stream, or missed the plume altogether if located beyond the wetland. The ramifications of this type of miscalculation are evident. Clearly, all avenues of input into the overall picture of a site evaluation must be covered and analyzed.

The research at the Adams site has shown the intricacies that can exist when dealing with a site evaluation. Flow net evaluations have shown how patterns of discharge change dramatically. Seasonal effects, particularly in a wetland environment can govern the reversal of flow gradients. Temperature, rainfall, vegetative cover and countless other variables play significant roles in the evaluation of the behavior of a given site.

Recommendations for Future Research

As with any study of this magnitude, the site abounds with possibilities for future research. The most likely follow-up work would be to continue the monitoring program in order to continue following the behavior of this particular plume. The majority of the plume is believed to be discharging primarily to the swamp stream and wetland surface. A more detailed fluid mass balance should be conducted to refine the discharge to each zone.

The effects of evapotranspiration and its contribution in a zone of discharge remain a gray area in most studies of this kind. At a site such as Adams where the contribution appears to be substantial, a study on the mechanisms of evapotranspiration and its seasonal effects on the overall flow characteristics would be useful. The wetland plants have been shown in laboratory studies to reduce the concentrations of contaminants in the water which they utilize. The wetland would provide a good field setting for analyzing such mechanisms. Treatment also seems to be taking place in the swamp stream. While dilution and attenuation are thought to be the processes effecting concentration decreases in the stream, the organisms in the leachate may also provide some biological treatment.

Additional upgradient wells should be installed to improve the understanding of the relationship between upgradient recharge and flow through and under the landfill.

Other parameters should be studied. The parameters used here were chosen for their convenience and utility in defining the plume. They are not significant pollutants when compared with the metals and toxic organics often found in landfill leachate.

The 21 wells will remain in the wetland for future research. Studies are encouraged on these and many of the other possibilities that can be found at a sanitary landfill.

BIBLIOGRAPHY

1. Anderson, John R. and Dornbush, James N., 'Influence of Sanitary Landfills on Ground Water Quality.' Jour. American Water Works Assn., Vol. 59, No. 4, pp. 457-70 (1967).

2. Apgar, Michael A. and Langmuir, Donald., 'Ground Water Pollution Potential of a Landfill Above the Water Table.' Groundwater Vol. 9, No. 6, pp. 76-94 (1971).

3. Bailey, L. C. 'Gas Fire in Sewer Manhole Traced to Sanitary Fill Operation.' Abstract. Engineering News Record, Vol. 139, p. 863, (1925).

4. Brunner, Dirk R. and Keller, Daniel J., 'Sanitary Landfill Design and Operation.' U. S. Environmental Protection Agency. Report No. SW-657S (1972).

5. Cameron, Robert D., 'The Effects of Solid Waste Landfill Leachates on Receiving Waters.' Journ. American Water Works Assn. Vol. 70, No. 3, pp. 173-76 (1978).

6. Campbell, M. D. and Lehr, J. H., Water Well Technology McGraw-Hill, Inc., New York, New York (1978).

7. Cartwright, Keros, Griffin, Robert A., and Gilkeson, Robert H., 'Migration of Landfill Leachate Through Glacial Tills.' Groundwater Vol. 15, No. 4, pp. 294-305. (1977).

8. Cartwright, Keros and McComas, Murray R., 'Geophysical Surveys in the Vicinity of Sanitary Landfills in Northeastern Illinois.' Groundwater Vol. 6, No. 5, pp. 23-30 (1968).

9. Caswell, Brad. 'Locating Monitoring Wells'. Water Well Journal Vol. 36, No. 5, pp. 42-3 (1982).

10. Caswell, Brad "Lessons from Piezometers." Water Well Journal Vol. 37, No. 7, pp 30-1 (1983).

11. Childs, K. E., Upchurch, S. B., and Ellis, B. 'Sampling of Variable Waste Migration Patterns in Groundwater.' Groundwater Vol. 12, No. 6, pp. 369-77 (1974).

12. Clark,Thomas P., and Sabel, Gretchen V. 'Requirements of State Regulatory Agencies for Monitoring Groundwater Quality at Waste Disposal Sites.' Groundwater Vol. 18, No. 2, pp. 168-74 (1980).

13. Coe, Jack J. 'Effect of Solid Waste Disposal on Groundwater Quality.' Journ. American Water Works Assn. Vol. 62, No. 12, pp. 776-83 (1970).

14. Cook and Powers 'Early Biochemical Changes in Soils and Waters of Artificially Created Marshes in New York' (1958) in Motts, Ward S., and O'Brien, Arnold L. 'Geology and Hydrology of Wetlands in Massachusetts.' Water Resources Research Center. University of Massachusetts/Amherst. Publication No. 123 (1981).

15. Davis, Stanley N., Thompson, Glenn M., Bentley, Harold W., and Stiles, Gary. 'Groundwater Tracers-A Short Review.' Groundwater Vol. 18, No. 1, pp. 14-23. (1980).

16. Everett, L. G., Ground Water Monitoring. General Electric, Co., Business Growth Services, Schenectady, New York, 440 pages (1980).

17. Federal Register. Rules and Regulations. Subpart F-Groundwater Monitoring. Resource Conservation and Recovery Act (RCRA) Vol. 45, No. 98, May 19, (1980).

18. Fetter, C. W. Jr., 'Attenuation of Waste Water Elutriated Through Glacial Outwash.' Groundwater Vol. 15, No. 5, pp. 365-71 (1977).

19. Fetter, C. W., Jr., 'Potential Sources of Contamination in Groundwater Monitoring.' Ground Water Monitoring Review Vol. 3, No. 2, pp. 60-64 (1983).

20. Fuller, W. H., Alesii, B. A., and Carter, G. E., 'Behavior of Municipal Solid Waste Leachate. I. Composition Variations.' J. of Env. Sci. and Health. Vol. A14, No. 6, pp. 461-85 (1979).

21. Fungaroli, A. A. and Steiner, R. L. 'Laboratory Study of the Behavior of a Sanitary Landfill.' Journ. of Water Pollution Control Federation. Vol. 43, No. 2, pp. 252-67 (1971).

22. Gass, Tyler, 'Glaciation and Groundwater.' Water Well Journal Vol. 35, No. 12, pp. 52-4 (1981).

23. Ground Water and Wells. Published by Johnson Division, UOP, Inc., Saint Paul, Minnesota (1975).

24. Hansen, Bruce P., Gay, Frederick B., and Toler, L. G., 'Hydrologic Data of the Hoosic River Basin, Massachusetts.' Massachusetts Hydrologic-Data Report No. 15 Prepared by

Commonwealth of Massachusetts Water Resources Commission (1974).

25. Hansen, Bruce P., Toler, L. G., and Gay, Frederick B., 'Hydrology and Water Resources of the Hoosic River Basin, Massachusetts.' Hydrologic Investigations Atlas HA-481. U. S. Geological Survey, Washington, D.C. (1973).

26. Herz, Norman, 'Bedrock Geology of the North Adams Quadrangle, Massachusetts-Vermont.' Map GQ-139 U. S. Geological Survey, Washington, D. C., (1961).

27. Holmes, G. William. 'Preliminary Materials Map of the North Adams Quadrangle, Massachusetts-Vermont.' U.S.G.S. open-File Report No. 680133. U. S. Geological Survey, Denver, Colorado, (1965).

28. Jacobson, Roger L., and Langmuir, Donald. 'The Chemical History of Some Spring Waters in Carbonate Rocks.' Groundwater Vol. 8, No. 3, pp. 5-9 (1970).

29. Jennings, Aaron A., ed., 'Sanitary Landfill Leachate Collection and Treatment.' Proceedings from the 1977 Symposium, University of Massachusetts/Amherst, EVE Report No. 67-81-2, (1981).

30. Kadlec, R. H. 'Water Quality Maintenance Values of Wetlands.' in Scientists Report: National Symposium on Wetlands, Lake Buena Vista, Fla., pp. 80-99, Nov. 6-9, (1978).

31. Kappel, William - Personal Communication, Values of Evapotranspiration. U. S. Geological Survey, Ithaca, NY August 2, (1983).

32. Kelley, William E., 'Geoelectric Sounding for Estimating Aquifer Hydraulic Conductivity'. Groundwater Vol. 15, No. 6, pp 420-25 (1977).

33. Keswick, Bruce H., Wang, De-Shin, and Gerba, Charles P., 'The Use of Microorganisms as Ground Water Tracers. A Review.' Groundwater Vol. 20, No. 2, pp. 142-49 (1982).

34. Kimmel, Grant A., and Braids, Olin C. 'Leachate Plumes in a Highly Permeable Aquifer.' Groundwater Vol. 12, No. 6, pp 388-93. (1974).

35. Kramer, William H., 'Groundwater Pollution from Gasoline.' Ground Water Monitoring Review Vol. 2, No. 2, pp. 18-22 (1982).

36. Kunkle, G. R. and Shade, J.W. 'Monitoring Ground Water Quality Near a Sanitary Landfill.' Groundwater Vol. 14, No. 1, pp. 11-20 (1976).

37. Larson, Joseph S., ed., 'Models for Assessment of Freshwater Wetlands.' Water Resources Research Center. University of Massachusetts/Amherst, Publication No. 32; Completion Report FY-76-5 (1976).

38. Lee, D.R. 'The Role of Groundwater in Eutrophication of a Lake in Glacial Outwash Terrain.' Int. J. Speleol. Vol. 8, pp. 117-26 (1976).

39. LeGrand, Harry E., 'Monitoring Changes in Quality of Groundwater.' Groundwater Vol. 6, No. 3, pp. 14-18 (1968).

40. Linsley, Ray K., Jr., Kohler, Max A., and Paulhus, Joseph L., Hydrology for Engineers, 2nd ed., McGraw-Hill, Inc., New York, New York (1978).

41. Maguire, C.E., Inc., 'Report on Design and Procedure for Closing Existing Refuse Disposal Area for the Town of Adams, Massachusetts.' C. E. Maguire, Inc., December (1975).

42. Matis, John, 'Solid Waste Disposal and Groundwater', Groundwater Vol. 9, No. 5, pp. 2-3 (1971).

43. Jack McFadyen - Personal Communication, Regarding Adams Geology, January (1982).

44. Motts, Ward S. and O'Brien, Arnold L., 'Geology and Hydrology of Wetlands in Massachusetts.' Water Resources Research Center. U of Mass/Amhest. Publication 123 (1981).

45. O'Brien, Arnold L., 'The Role of Groundwater in Stream Discharges from Two Small Wetland Controlled Basins in Eastern Massachusetts.' Groundwater Vol. 18, No. 4, pp. 359-65 (1980).

46. Palmquist, Robert and Sandlein, Lyle V.A., 'The Configuration of Contamination Enclaves from Refuse Disposal Sites on Floodplains.' Groundwater Vol. 13, No. 2, pp. 167-81 (1975).

47. Pettyjohn, Wayne A., 'Cause and Effect of Cyclic Changes in Groundwater Quality.' Ground Water Monitoring Review Vol. 2, No. 1, pp. 43-49 (1982).

48. Pettyjohn, Wayne A., 'Monitoring Cyclic Fluctuations in Groundwater Quality.' Groundwater Vol. 14, No. 6, pp. 472-80 (1976).

49. Qasim, Syed R. and Burchinal, Jerry C. 'Leaching from Simulated Landfills.' Journn. of Water Pollution Control Federation Vol. 42, No. 3, pp. 371-79 (1970).

50. Rahn, Perry H. 'The Hydrology of an Induced Streambed Infiltration Area.' iGroundwater Vol. 6, No. 3, pp. 21-32 (1968).

51. Reppert, Richard T., et al. 'Wetland Values. Concepts and Methods for Wetlands Evaluation'. U.S. Army Corps of Engineers. Research Report 79-R1 (1979).

52. Rubin, Michel J. L., Dytynyshyn, David J. and Sweeney, Stewart J. 'Two Gas-Drive Sampling Devices.' Ground Water Monitoring Review Vol. 2, No. 1, pp. 63-66 (1982).

53. Rushton, K. R. and Wedderburn, L. A., 'Aquifers Changing Between the Confined and Unconfined State.' Groundwater Vol. 9, No. 5, pp. 30-39 (1971).

54. Saines, Marvin, 'Errors in Interpretation of Ground Water Level Data.' Ground Water Monitoring Review Vol. 1, No. 1, pp. 56-61 (1981).

55. Sandlein, Lyle V. A., and Yazicigil, Hasan, 'Surface Geophysical Methods for Groundwater Monitoring. Part I' Ground Water Monitoring Review Vol. 1, No. 3, pp. 42-46 (1981).

56. Schuller, Rudolph M., Gibb, James P., and Griffin, Robert A. 'Recommended Sampling Procedures for Monitoring Wells.' Ground Water Monitoring Review Vol. 1, No. 1, pp. 42-46 (1981).

57. Slattery, Joseph - Personal Communication Regarding Thickness of Clay Unit, Hoosic River Valley. November, (1982).

58. Standard Methods for the Examination of Water and Wastewater 15th Edition, American Public Health Association, Washington D.C. (1980).

59. Stoller, Robert L., and Roux, Paul, 'Earth Resistivity Surveys - A Method for Defining Groundwater Contamination.' Groundwater Vol. 13, No. 2, pp. 145-50 (1975).

60. Tenn, D. G., Haney, K. J. and DeGeare, T. V. 'Use of the Water Balance Method for Predicting Leachate Generation from Solid Waste Disposal Sites.' U.S. Environmental Protection Agency, Office of Solid Waste Management Programs, SW-168, Washington, D.C. (1975). in Vesilind, P. Aarne, and Rimer,

Alan E., Unit Operations in Resource Recovery Engineering. Prentice-Hall, Inc., Englewood Cliffs, NJ, (1981).

61. Tinlin, Richard M., 'A Methodology for Monitoring Ground Water Quality Degradation.' Ground Water Monitoring Review Vol. 1, No. 2, pp. 27-32 (1981).

62. Tirsch, Franklin, S. and Jennings, Aaron A. 'Leachate Reactions with Soils Under Anaerobic Conditions.' Env. Eng. Program, Department of Civil Engineering, University of Massachusetts/Amherst. Report No. Env. E. 76-78-3 (1978).

63. U. S. Department of the Interior. Ground Water Manual. A Water Resources Technical Publication. U. S. Government Printing Office, Denver, Colorado (1981).

64. U. S. Environmental Protection Agency. 'Procedures Manual for Ground Water Monitoring at Solid Waste Disposal Facilities.' Report No. EPA/530-SW-611. Washington, D. C. (1977).

65. Urish, Daniel W. 'The Practical Application of Surface Electrical Resistivity to Detection of Ground Water Pollution.' Groundwater Vol. 21, No. 2, pp. 144-52 (1983).

66. Williams, R. E., 'Flow of Groundwater Adjacent to Small Closed Basins in Glacial Till.' Water Resources Research. Vol. 4, No. 4, pp. 777-83 (1968).

67. Wilson, Lee C., and Rouse, Jim V. 'Variations in Water Quality During Initial Pumping of Monitoring Wells.' Ground Water Monitoring Review Vol. 3, No. 1, pp. 103-09 (1983).

68. Zanone, Donaldson and Grunwaldt. 'Ground Water Quality Beneath Solid Waste Disposal Sites at Anchorage, Alaska' Groundwater Vol. 13, No. 2, pp. 182-90 (1975).

SUPPLEMENTARY BIBLIOGRAPHY OF RELATED MATERIALS

1. Baedecker, Mary Jo and Back, William., 'Hydrogeological Processes and Chemical Reactions at a Landfill.' Groundwater Vol. 17, No. 2, pp. 429-37. (1979).

2. Barvenick, Matthew and Cadwgan, Richard. 'Multilevel Groundwater Sampling Systems.' BARCAD: Informational Brochure. 14 pages.

3. DeWalle, F. B. and Chian, E.S.K., 'Solid Wastes and Water Quality.' Literature Review. Journ. Water Pollution Control Federation. Vol. 51, No. 6 pp 1402-10. (1979).

4. Fetter, C. W., Jr., 'Determination of the Direction of Groundwater Flow.' Ground Water Monitoring Review. Vol. 1, No. 3, pp. 28-31 (1981).

5. Fungaroli, A. A. and Steiner, R. Lee, 'Investigation of Sanitary Landfill Behavior. Vol. I., Final Report.' Report No. EPA-600/2-79-053a, U. S. Environmental Protection Agency (1979).

6. Giddings, M. Todd, Jr., 'The Lycoming County, Pennsylvania Sanitary Landfill: State-of-the-Art in Groundwater Protection. Groundwater Vol. 15, No. 1, pp. 5-14 (1977).

7. Gundlach, David L., 'Some Aspects of Approximating Aquifer Discharge.' Groundwater Vol. 12, No. 3, pp. 135-39 (1974).

8. Hansen, Edward A., and Harris, Alfred Ray, 'A Groundwater Profile Sampler.' Water Resources Research Vol. 10, No. 2, p. 375 (1974).

9. Maslansky, Steven P., 'Well Drilling and Hazardous Material Sites.' Water Well Journal Vol. 37, No. 4, pp. 46-50 (1983).

10. Miller, David W., 'Guidelines for Developing a State-wide Ground-water Monitoring Program.' Ground Water Monitoring Review Vol. 1, No. 1, pp. 32-3 (1981).

11. Morrison, Robert D., and Brewer, Paul E. 'Air-lift Samplers for Zone-of-Saturation Monitoring.' Ground Water Monitoring Review Vol. 1, No. 1, pp 52-5, (1981).

12. Pickins, J. F., Cherry, J. A., Coupland, R. M., Grisak, G. E., Merritt, W. F., and Risto, B. A. 'A Multilevel Device for

Groundwater Sampling.' Ground Water Monitoring Review Vol. 1, No. 1, pp. 48-51 (1981).

13. Sgambat, Jeffrey P. and Stedinger, Jery R., 'Confidence in Groundwater Monitoring.' Ground Water Monitoring Review Vol. 1, No. 1, pp. 62-69 (1981).

14. Weist, William G., Jr., and Pettijohn, Robert A., 'Investigating Ground Water Pollution from Indianapolis' Landfills. - The Lessons Learned.' Groundwater Vol. 13, No. 2, pp. 191-96 (1975).