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Solid Waste Disposal By Land Burial In Southern Indiana

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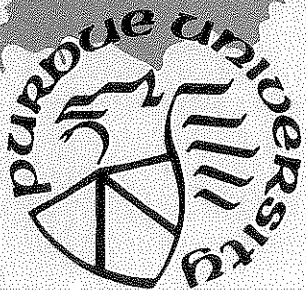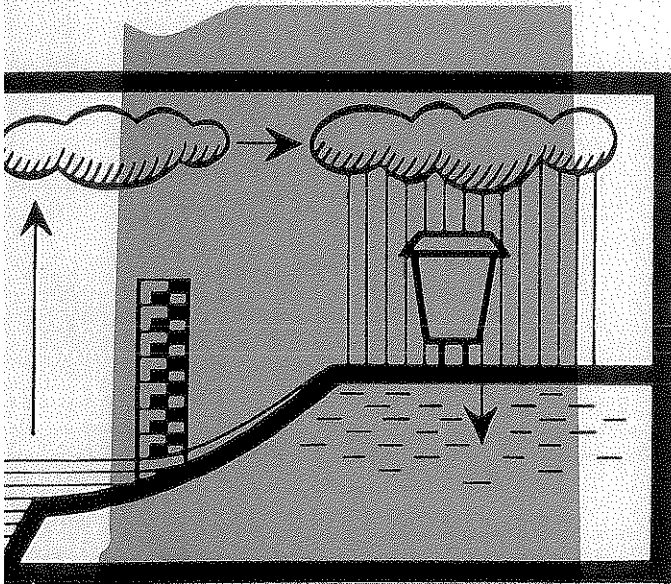http://docs.lib.purdue.edu/watertech/44

SOLID WASTE DISPOSAL BY LAND BURIAL IN SOUTHERN INDIANA

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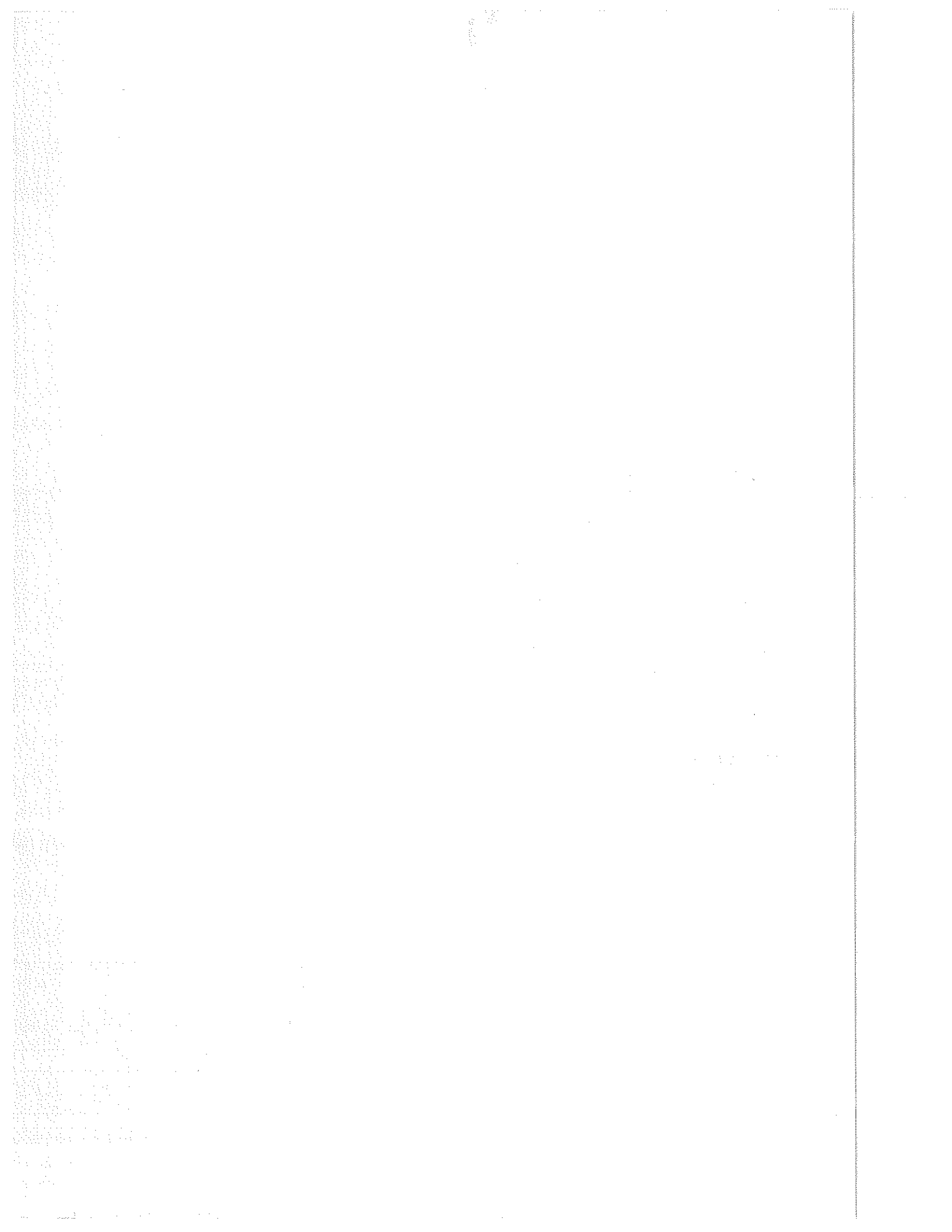
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PURDUE UNIVERSITY
WATER RESOURCES RESEARCH CENTER
In Cooperation With
INDIANA UNIVERSITY
WATER RESOURCES RESEARCH CENTER

SOLID WASTE DISPOSAL BY LAND BURIAL IN SOUTHERN INDIANA

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With an additional chapter

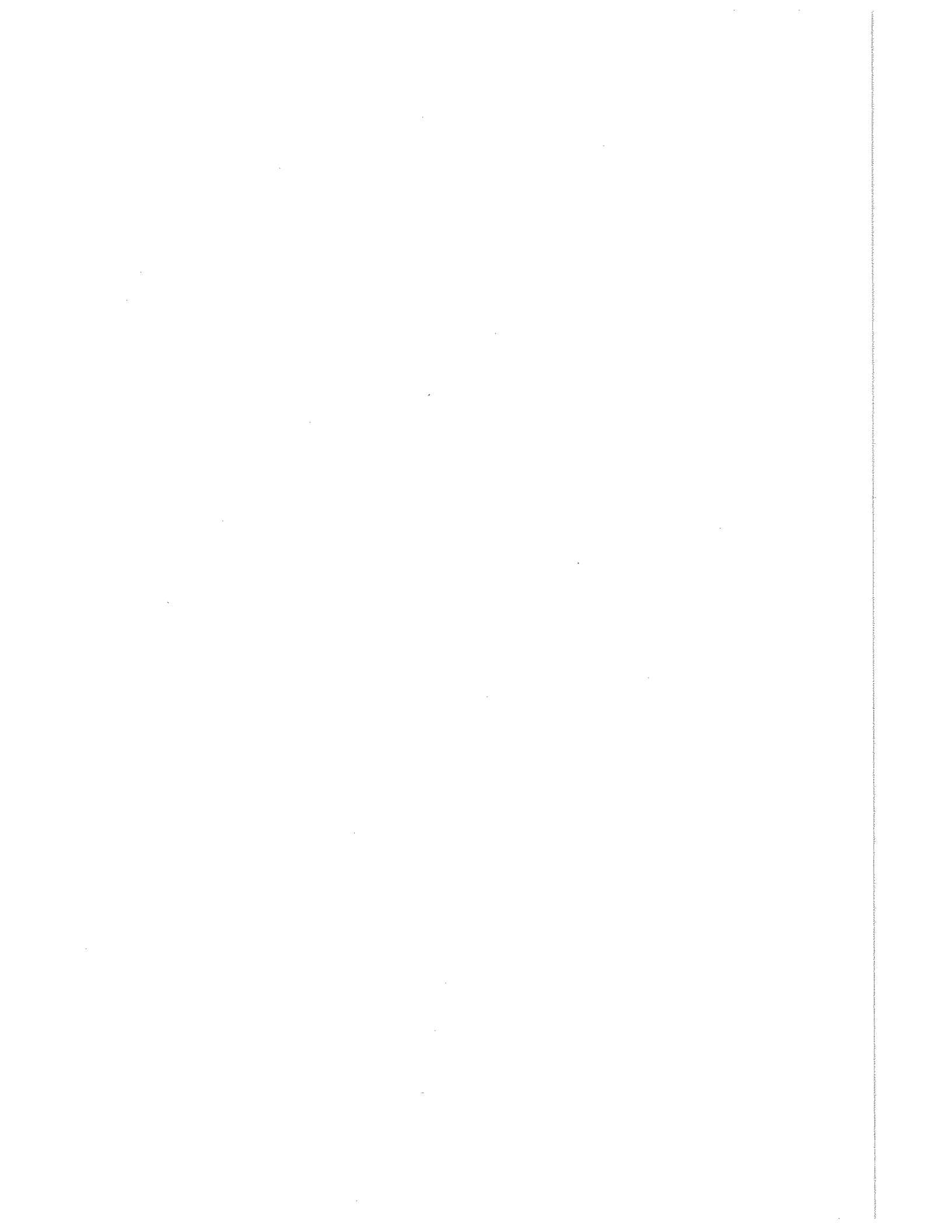CARBON IN GROUND WATER AT THE COLUMBUS, INDIANA LANDFILL

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Contents

Introduction

In 1969 the Indiana State Legislature passed the Refuse Disposal Act which requires that solid waste be disposed of by incineration or by burial in "sanitary landfills" (Indiana State Board of Health, 1969). This report deals with solid waste disposal by land burial at selected landfills in southern Indiana. Priority was given to locations where studies could be made prior to, during, and after landfilling operations, so that "before" and "after" effects could be determined.

The objectives of the study are: (1) to define the field models of the geohydrologic systems of the differing terrains of the landfills, (2) to determine the ground- and surface-water movement, (3) to analyze the original water quality and any changes caused by landfilling, (4) to establish criteria for protection of the water resources that may be affected by landfilling, and (5) to establish guidelines relating to emplacement of landfills in different terrains.

Three case studies are given in this report: (1) Monroe County Landfill near Bloomington, (2) Brown County Landfill near Helmsburg, and (3) Municipal Landfill in Columbus. The first two landfills continue in operation, so discussion of them concerns only the "before" and "during" phases. The landfill at Columbus was discontinued for economic reasons, so the "after" phase will be examined there. However, the volume of waste that was buried is only a minor part of the volume originally planned for burial.

A progress report is also presented from a cooperative study on the organic compounds and carbon in ground water from landfill effluent.
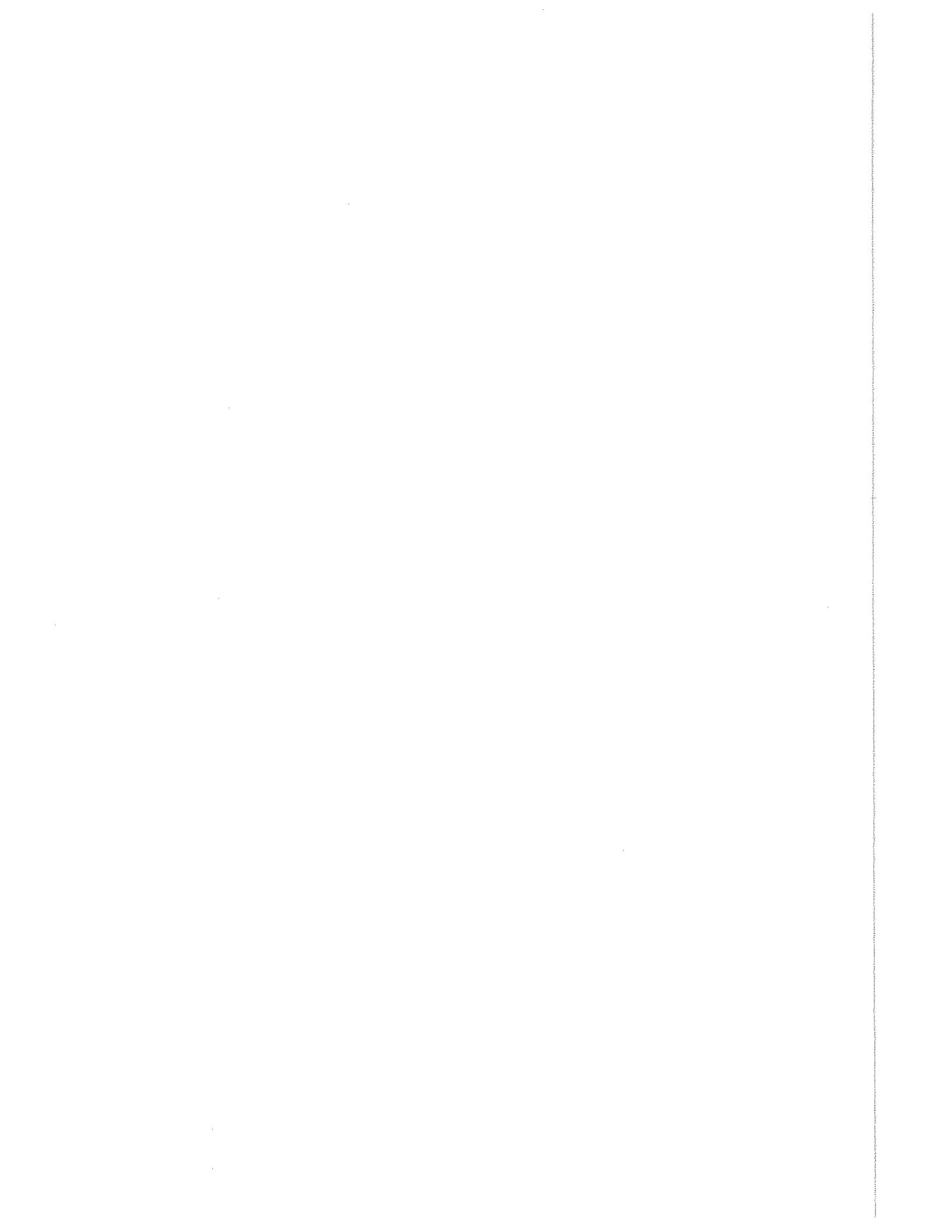
Monroe County Landfill

Topography. The Monroe County landfill is located along Anderson

Road about 8 miles northeast of Bloomington in the $NW\frac{1}{4}$, section 25,

T. 10N., R.1W. The site covers 160 acres on the north valley slope of

Beanblossom Creek in dissected terrain consisting of stepped topographic

levels ascending interfluves between tributaries of the main stream.

Several tributary streams originate within the landfill site and

descend between the interfluves to a large artificial drainage ditch

that flows on the north side of Anderson Road. The ditch then passes

under the road through a culvert near site 8 (Fig. 1) and continues

half a mile southward where it drains into Beanblossom Creek. The

longitudinal profiles of the tributaries have 6 percent slopes in the

upper reaches decreasing to about one percent in the lower reaches

toward the main valley of Beanblossom Creek.

Quite steep hillslopes (up to 8 percent) descend to the broad

alluvial valley and to tributary streams. Elevation of the area

ranges from 602 feet in the alluvial valley to nearly 714 feet in

the highest part of the uplands.

Stratigraphy and Surficial Deposits. The surficial and sub-

surface materials of the original area were studied at 12 drill sites

where continuous cores were taken from different topographic posi-

tions (Fig. 1).

The bedrock is siltstone of the Borden group (Wier and Gray,

1961). Depths to bedrock, determined during drilling, indicate that

the bedrock surface is very undulatory and is higher under the inter-

fluves than beneath tributary valleys and the main valley. Under

the uplands (sites 1 and 2), bedrock is at an elevation of at

MONROE COUNTY LANDFILL

0 feet   500

Figure 1.   BASE MAP OF MONROE COUNTY LANDFILL

▣   SURFACE SAMPLE SITE

△   RAIN GAUGE

⊕   WATER QUALITY/
     WATER LEVEL WELL

⊙   WATER LEVEL WELL

+   CONTINUOUS CORE

—— LEVEL OF VALLEY
     FILL SEDIMENTS
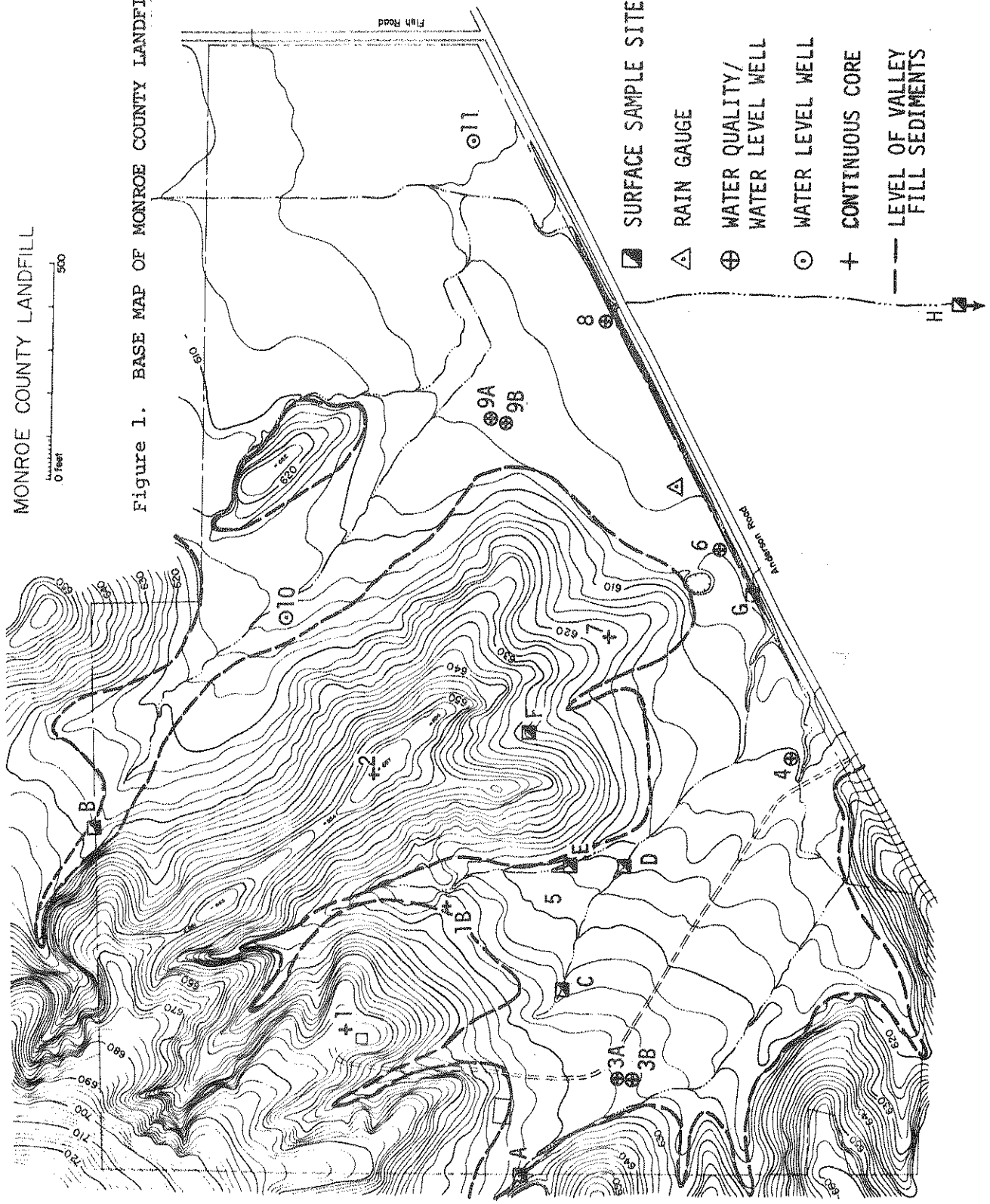
least 610 feet. In a tributary valley (sites 3 and 5), bedrock is at an elevation of about 570 feet. Under the main valley (sites 6, 9, and 11), the bedrock surface descends to an elevation of less than 570 feet and is less than 560 feet at site 8. Although the surface topography shows that the terrain is dissected by the smaller tributary streams descending to the main valley, the bedrock surface was previously incised.

A weathered zone occurs above bedrock at all drill sites. The zone is gray and contains fragments of bedrock. Textural data (Table 1) indicate that the material is mostly silt (50 to 60 percent) and sand (30 percent). Above this weathered zone at upland sites 1 and 2, another sediment, 7 to 11 feet thick, contains granite pebbles and cobbles. This unit is probably glacial till of Illinoian age. This till is about 5 miles south of the recognized Illinoian drift border in south-central Indiana (Wayne, 1958).

Two thick beds of silt and clay are above the weathered bedrock or till at all drill sites. One bed of probably Illinoian age is at least 19 feet thick under the uplands (site 1), and the other bed of Wisconsinan and Holocene age is more than 36 feet thick in the valley (site 8). These fine-textured sediments may be lacustrine in origin (Pratt, 1960) and are mostly silt (50 to 64 percent) in the upper part of the bed and are alternating layers of silt and clay in the lower part of the bed. Where silt and clay are interbedded, the silt layers contain from 69 to 83 percent silt in contrast to 34 to 41 percent silt in the clay layers.

At three valley sites (3, 9, and 10), beds of alluvial gravel are above or are interbedded with the silt and clay. These gravels

range in thickness from less than 2 feet (site 9) to more than 7 feet

(sites 3 and 10).

TABLE 1. Physical Properties of Sediments at Sites 1 and 3

Depth (ft)	Material	Sand* (%)	Silt* (%)	Clay* (%)
Site 1:				
0-7	silt	1.1	85.2	13.7
7-24	silt	4.3	69.4	26.3
	clay	3.8	34.7	61.5
24-31	till	-	-	-
31-33	weathered bedrock	29.2	56.9	13.9
Site 3:				
0-4	silt	2.3	85.8	11.8
4-7	gravel	22.4	61.0	14.6
7-9	silt	7.7	54.8	38.5
9-15	gravel	23.5	51.2	15.2
15-29	silt	8.0	74.9	17.1
	clay	8.7	40.6	50.7
29-31	weathered bedrock	33.0	54.9	12.1

*Particle-size analysis based on all materials less than 2 mm diameter.

On the uplands the surface material contains more than 85 percent

silt (Table 1) and is Wisconsinan loess (Caldwell and White, 1956). A

representative stratigraphic section on the uplands (site 1) has 7 feet

of loess containing a fairly well-developed soil overlying 17 feet of

silt and clay, and in turn 7 feet of glacial till and 2 feet of weathered

bedrock (Fig. 2). Depth to bedrock is 33 feet.

In a drainageway (site 3) a representative stratigraphic section
has 3 feet of alluvial silt containing a less-developed soil overlying
3 feet of gravel, 2 feet of silt, another 6 feet of gravel, 3 more
feet of silt, 12 feet of interbedded silt and clay, and 2 feet of
weathered bedrock (Fig. 2). Depth to bedrock is 31 feet.

Figure 2. GENERALIZED SECTION DESCRIPTIONS

Site 1
(Elev. 664')

```
__0
          silt loam

__10
          silts

          and

          clays
__20

          silty clay
          granite pebs
          (till)
__30
         wthd. bdrk.

          bedrock
```

Site 3
(Elev. 627')

```
__0
          silt loam
          gravelly
            loam
          silty clay
__10
          gravelly
            loam

          silty clay
__20
          silts

          and

          clays
__30     wthd. bdrk.

          bedrock
```

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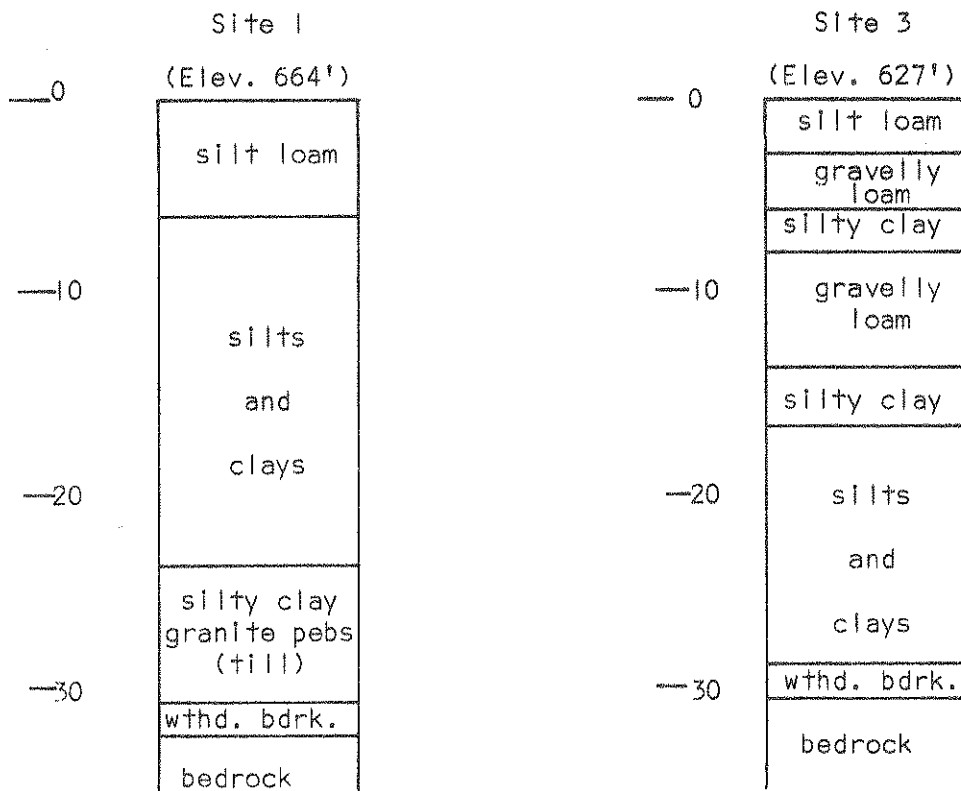
The boundary between upland and valley-fill sediments was delineated in field mapping. Valley-fill deposits occupy 55 percent of the surface area of the landfill site (Fig. 1).

Prior to landfill operations, the upland soils were Cincinnati and Ava formed in thin loess over a differing substratum (Agric. Exper. Sta./ Coop. Exten. Serv., Purdue Univ., 1971). The soils in the alluvium are the Wakeland-Stendel-Haymond-Bartle association.

Hydrology. Annual precipitation in the area is about 44 inches with most rain occurring in spring and least in fall. May and June are generally the wettest months with October being the driest month (Table 2).

TABLE 2.   Average Monthly Precipitation, Bloomington, Indiana*

Month	Amount (inches)
January	3.87
February	2.77
March	3.96
April	3.91
May	4.48
June	5.09
July	3.50
August	3.33
September	3.76
October	2.68
November	3.49
December	3.13

*Climatological Data (Indiana), vol. 78, no. 13, p. 2.

Prior to landfilling the hydrologic system at the Monroe County landfill could be separated into three components: (1) the ground water in the main saturated zone, (2) isolated zones of subsurface

water perched above relatively impermeable sedimentary beds, and
(3) surface water in the intermittent streams.

The ground water in the main saturated zone occurs within and
above Borden siltstone. Elevation of the ground-water table at each
site was measured during drilling in June, 1971. The ground water
was considerably higher under the uplands than under the descending
hillslopes and floodplain. A map of the water table, based on static
levels of water in drill holes, shows that the ground-water surface
generally parallels the slope of the surface topography (Fig. 3,
cf. Fig. 1).

From Darcy's law the flow of a liquid through a porous medium
is in the direction of and proportional to the hydraulic gradient and
also proportional to the rate of permeability (Hillel, 1971):

$$V = k \frac{dh}{dl}$$

where V is the velocity of flow in inches per hour; k is the per-
meability or hydraulic conductivity in inches per hour; and dh/dl
is the hydraulic gradient in feet per feet. True Darcian flow may
not occur in fine-textured materials (Hillel, 1971; Olsen, 1965)
because of interference by clay particles and their adsorptive
fields that attract water causing it to be more static and not
as free for movement (Swartzendruber, 1962).

However, approximate flow velocities may be calculated by
using Darcy's law. The gradient of the water table slopes from
site 2 to sites 4 and 8 at the rate of 47 feet in 960 feet (4.9 per-
cent) and 47 feet in 1170 feet (4.0 percent), respectively. Per-
meability of the material, determined by maximum water depth

Figure 3.   DEPTH TO WATER MAP —
MONROE COUNTY LANDFILL

+ DRILL SITE
⇀ FLOWLINE

MONROE COUNTY LANDFILL
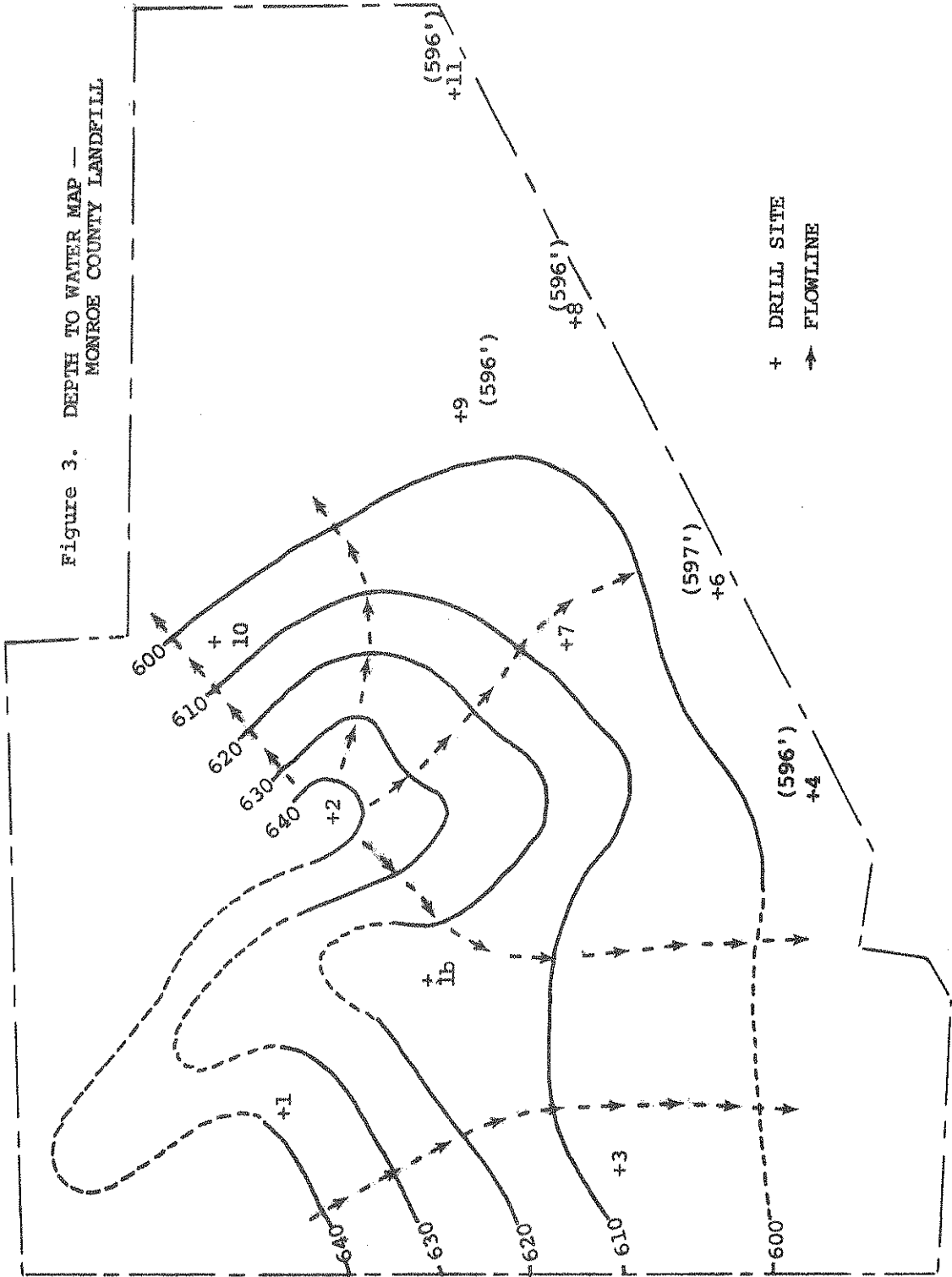
fluctuation is 0.024 in/hr or 0.047 ft/day (Table 3). Corresponding
flow velocities along the two traverses are 1.2 x 10^{-3} in/hr
(2.4 x 10^{-3} ft/day) and 1.0 x 10^{-3} in/hr (1.9 x 10^{-3} ft/day).

TABLE 3. Water Level Fluctuation, Monroe County Landfill

Date	Water Elevation	Fluctuation (ft)	Time (days)	Rate ft/day
Well 3B -- Ground Water:				
6/1/72	607.1			
		0.7	24	0.029
6/25	606.4			
		0.9	19	.047
7/14	605.5			
		0.5	15	.033
7/29	605.0			
		0.8	19	.042
8/17	604.2			
		2.0	43	.047
9/29	602.2			
		0.8	27	.030
10/26	601.4			
		0.2	23	.009
11/18	601.2			
		0.1	13	.008
12/1	601.3			
		1.8	63	.029
2/2/73	603.1			
Well 3A -- Perched Water:				
6/1/72	627.1			
		3.5	24	0.146
6/25	623.6			
		2.5	19	.132
7/14	621.1			
		0.4	15	.027
7/29	620.7			
		0.9	19	.247
8/17	619.8			
8/17 - 11/18	DRY	--	--	--
12/1	620.2			
		3.1	63	.049
1/26/73	623.3			

Water table elevations of the main zone of saturation in the valley
fluctuate as much as 6 feet annually and are highest in late spring and
lowest during late fall (Table 3). For example, at site 3 water was at
elevations 607, 604, 601, and 603 feet in June, August, November, 1972
and February, 1973. At site 8 water was at 599, 596, 593, and 595 feet
on corresponding dates.

The perched ground water in the valley at drilling sites 3, 9,
and 10 occurs in alluvial gravel above lacustrine silt and clay
(Table 1, Fig. 2). Precipitation infiltrates the thin and moderately
permeable surface silt, percolates downward through the quite per-
meable gravel, and accumulates on top of the very thick and relatively
impermeable silt and clay. The gravel thins downvalley from nearly
10 feet thick at site 10 to about 2 feet thick at site 9. Elevation
of the contact between the gravel and the underlying silt and clay
decreases downvalley from 593 feet at site 10 to 577 feet at site 9
along a gradient of 2.5 percent. As the permeability of gravel may
be as high as 0.81 in/hr (by laboratory permeability tests), the rate
of flow from Darcy's law is approximately 2.0 in/hr (4.1 ft/day).
Well 3A intercepts the perched-water zone and is about 350 feet from
the active fill areas in zones 1 and 2 (Fig. 4, cf. Fig. 1). At the
calculated rate of flow from the landfill to site 3, 85 days are re-
quired for contamination to reach well 3A. Elevation of the perched-
water table fluctuates as much as 7 feet annually (Table 3). During
the spring and summer of 1972 perched water in well 3A was at eleva-
tions of 627, 624, 621, and 620 feet from June 1 to August 17.

Although perched water was not present at upland sites during
drilling, it should occur. Precipitation should infiltrate the

surface soil and loess and percolate downward until the less per-
meable silt and clay are reached. Direction of flow should parallel
the hillslope along this less permeable subsurface contact, and this
water will eventually drain into the valley-fill sediments or into
the intermittent drainageways.

Prior to landfilling, surface water was supplied by runoff and
lateral flow of subsurface water down hillslopes into the tributary
drainageways. One tributary heads west of the landfill but crosses
the southwest corner of the landfill site. All water from the tri-
butaries collects in an artificial drainage ditch on the north side
of and parallel to Anderson Road (Fig. 1). The ditch then passes
under the road through a culvert near site 8 and flows south into
Beanblossom Creek.

Landfill Design. Landfilling utilizes the trench and area
methods (Beam et al., 1971) and began in January 1972 and is con-
tinuing at the present time. The trench method is used mainly on
the upland interfluves (Fig. 4). Trenches 25 feet wide and about
15 feet deep are excavated the entire length of a zone. Solid waste
is dumped, spread, and compacted in thin layers. The next adjoin-
ing trench is excavated simultaneously and spoil from the second
trench is spread and compacted over solid waste in the first trench
and so on.

The area method of operation is used to fill the tributary
valleys and consists of spreading solid waste in thin compacted
layers 50 feet wide in designated zones (Fig. 4). Cover material
is provided from excess earth of the trenches and from the area zones.
Top soil is stripped from the uplands and stockpiled for later use as
the final cover.

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MONROE COUNTY LANDFILL

Figure 4. ZONE MAP OF MONROE COUNTY LANDFILL
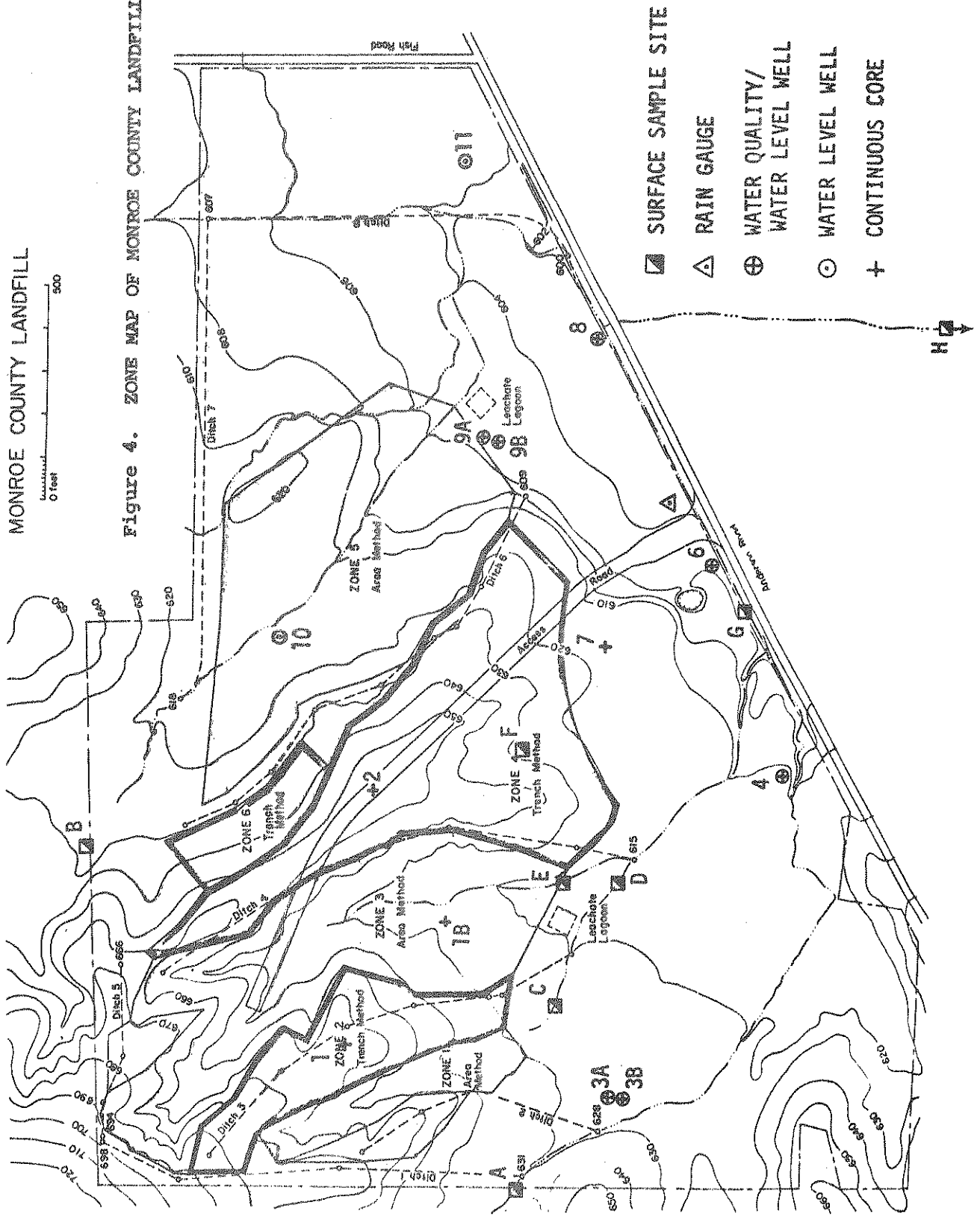
During trenching and filling the base of the garbage is brought
nearer the higher water table under the upland summits. During area
filling the waste is dumped on the original valley slopes and bottom
which become a buried land surface. Hence, the uplands are progres-
sively lowered at places and the valleys are built up through time.
After an estimated 10-year life, the original landscape will be re-
modeled to essentially one continuous hillslope that descends from
an elevation of 714 feet at the northwest corner of the area south-
eastward to 610 feet on the former valley bottom (Fig. 5, cf. Fig. 1).
The base of the interbedded waste and earth will be undulatory--higher
under former uplands and lower on former valley bottoms.

Changes in the landscape caused by landfilling may cause some
pronounced changes in the hydrologic regime of the area. After the
uplands are trenched and lowered, the base of the trenches will be
in the less permeable silt and clay. During and after filling of the
trenches by waste, precipitation will infiltrate and be trapped in
the fill which has greater porosity and permeability than the silt
and clay. As more than 70 percent of solid waste is paper and re-
lated organic products (Steiner et al., 1971), and as the field
moisture of such materials averages about 4 percent while the field
capacity is about 29 percent by volume (Remson et al., 1968), a
large volume of moisture may be contained in the solid waste. After
field capacity is reached, free water will accumulate in the filled
trenches and will become available for movement. After sufficient
head is reached, leachate should seep downward through the bottom
and laterally through the walls of the trenches.

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MONROE COUNTY LANDFILL

0 feet 500

Figure 5. FINAL CONTOUR MAP OF MONROE COUNTY LANDFILL
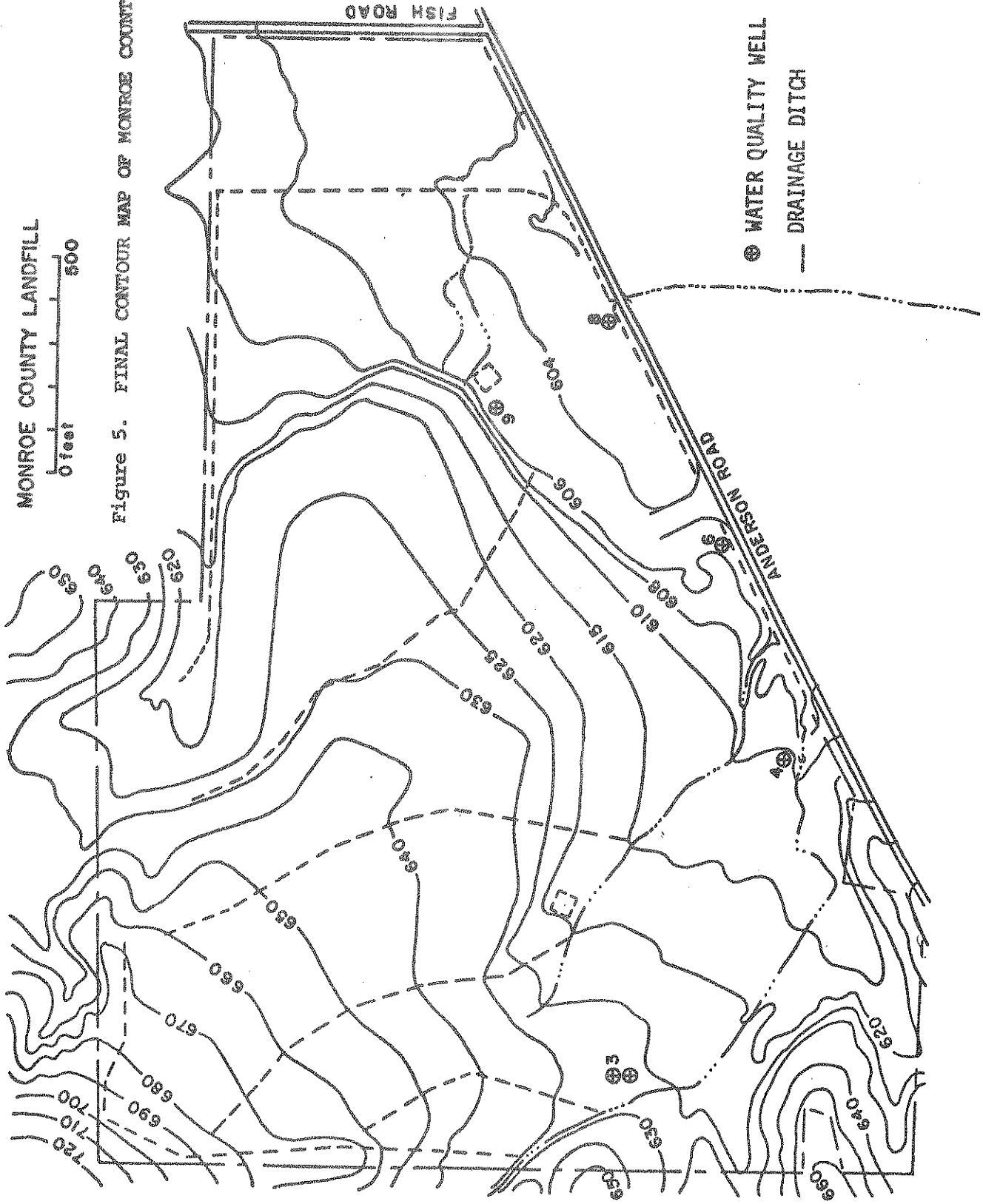
⊕ WATER QUALITY WELL

---- DRAINAGE DITCH

FISH ROAD

ANDERSON ROAD

In the original valleys filled with waste by the area method, water will infiltrate the final soil cover and accumulate in the waste above the buried land surface. Ultimately, leachate will seep downward to the original water table and laterally down the waste-buried valley.

The amount of time required for the waste to become saturated and for seepage to begin may be estimated. During a year the amount of water infiltrating the completed fill may be approximated by:

$$I = P - ET - R$$

where I is infiltration, P is precipitation, ET is evapotranspiration, and R is runoff--all in inches/year.

Precipitation is 44 inches annually (Table 2). ET estimated by the Thornthwaite method (Taylor and Ashcroft, 1972) is:

$$ET = -1.6 \frac{(10 \ T)^a}{I}$$

where ET is evapotranspiration in inches/month; T is the mean monthly temperature in $^{\circ}C$; I is the annual heat index calculated from the mean temperature for each month; and a is a constant based on I. Both I and a vary little for a given location. Potential ET is calculated for each month using the 1973 temperature data for Bloomington (Climatological Data, vol. 78, no. 13). The individual monthly values are summed to obtain the yearly estimate of 20.4 inches.

Runoff may be estimated from soil-erosion studies. Surface runoff and soil erosion are related since there can be no erosion without runoff. According to the universal soil-loss equation (Wischmeier and Smith, 1965) soil erosion per unit area (A) is dependent on several factors including erosive power of a given

rainfall (R), erodibility of the material (K), length (L) and gradient (G) of the slope, a cropping-management factor (C), and a factor (P) relating to erosion-control practices such as terracing, contouring, etc. If two areas are similar with respect to the different components of the soil-loss equation, then similar runoff and erosion should result.

In 1965 a loess-covered watershed in Iowa planted in brome grass had 44.3 inches of precipitation that produced 4.6 inches of surface runoff (Saxton et al., 1971). The Monroe County landfill receives about 44 inches of precipitation annually (Table 2). So, in comparing the two areas, the R factor in the soil-loss equation should be about the same. The vegetation (grass) and surface materials (loess) are similar, so the C and K terms should be similar. Neither area is contoured or terraced so the P factor should be about the same. The slopes in the Iowa study were 2 to 4 percent on the ridges and in the valleys and about 10 percent on the valley sides. The final hillslope at the landfill will have a gradient of about 6 percent. Although slight differences occur between the Iowa model and the landfill, the estimate of 4.6 inches of runoff is reasonable for the landfill.

Consequently, the amount of infiltration in the solid waste of the landfill is about 19 inches annually, that is, 44 (P) - 20.4 (ET) - 4.6 (R). A one-foot layer of waste contains about 0.46 inches of water and may hold as much as 3.44 inches at field capacity (Remson et al., 1968). Thus, 2.98 inches of precipitation are needed to bring each foot of garbage to field capacity. In a trench 15 feet deep, 44.7 inches of water are required to wet all solid waste in the

trenches to field capacity. At an infiltration rate of 19 in/yr, field capacity should be reached in slightly less than $2\frac{1}{2}$ years.

The wetting of waste has been observed in the Columbus landfill, described later in this report. Water level was observed in a well bottoming in the fine-textured sediments at the base of the waste fill. Free water appeared in the observation well about 16 months after filling began. Almost 48 inches of precipitation had accumulated and about 15 feet of uncovered waste had been filled. If field capacity had been reached at the time when free water appeared in the well, each foot of waste had absorbed 3.2 inches of water. The 16-month wetting time may be somewhat rapid, because the waste at this landfill was uncovered.

After field capacity is reached at the Monroe County landfill in $2\frac{1}{2}$ to 3 years, additional infiltration should cause saturation of the filled trenches. The time required for saturation also may be estimated by calculating the amount of water to fill all voids in the waste. Porosity (P) may be calculated as (Wisler and Brater, 1959):

$$ P = \frac{SW - D}{SW} $$

where $\underline{SW}$ is specific weight in lb/ft^3, and $\underline{D}$ is density in lb/ft^3. The density of dry waste is about 17.6 lb/ft^3 (Fungaroli and Steiner, 1968). The specific weight is about 79.3 lb/ft^3 for ordinary waste that contains about 78 percent paper and organic products, 6 percent metals, and 8 percent each of glass and dirt, rags, rubber, leather, etc. (Calculated from Fungaroli and Steiner, 1968). Thus, each cubic foot of waste should have about 0.78 ft^3 of pore space.

As the trenches contain 15 feet of waste, saturation of all voids will require about 140 inches of water. As effective infiltration is about 19 inches per year, the waste should be saturated in about 8 years after field capacity is reached in $2\frac{1}{2}$ years.

At any time that sufficient head is created by partial or complete saturation of a trench above its base, seepage may occur downward to the main ground-water zone. In the completed fill at site 1 (cf. Figs. 1, 3, and 4), the base of the trench will be about 7 feet above the original water table. Seepage from the trench should be transmitted through the intervening silt and clay, whose permeability is 0.047 ft/day, to the ground water in about 5 months. Thus, the minimum time for trench effluent to enter the ground water is about 3 years, that is, $2\frac{1}{2}$ years to reach field capacity in the waste plus the 5-month travel time through silt and clay.

In the areas filled by the area method in the original valleys (cf. Figs. 3 and 4), similar hydrologic conditions should develop. In addition, the original land surface that is converted to a buried surface by landfilling, probably will cause lateral seepage down the axis of the buried valley. Seepage of landfill effluent was observed and measured near the periphery of landfill zone 3 (Fig. 4) during the spring and early summer of 1974.

After completion of the landfill, the water table of the main ground-water zone ultimately will rise above its original position (Fig. 3) and subparallel the contours of the completed hill (Fig. 4). Flow paths and hydraulic gradients will be altered considerably and be directed toward the valley of Beanblossom Creek and the surface drainageways on the valley flat.

Water Quality. To evaluate the impact of landfill on water quality of the surrounding area, observation wells were installed around the perimeter of the designed completed landfill (Fig. 5). In addition, surface-water observation sites were selected along drainageways within and beyond the limits of the designed landfill.

Ground-water wells 3B, 4, 6, and 8 are located around the periphery of the planned final contour of the site (Figs. 1 and 5). These wells are bottomed on bedrock and are perforated to intercept the entire water column of the aquifer. If ground-water contamination is taking place at a site, it should be possible to detect it at these wells.

A shallow well (3A) intercepts the perched-water zone in the valley fill and is near the active landfill areas in zones 2 and 5 (Fig. 4). This well collects water that accumulates in the gravel layers and is bottomed in the silt and clay.

Surface-water sampling stations are in drainageways at points where leachate seeps from the landfill so that the chemical nature of the effluent may be determined. Also, samples are taken at different distances downstream to observe the dilution of the leachate with relatively uncontaminated water. Raw leachate seeping directly from the fill is sampled at site C (Fig. 1). It is sampled at site D after leachate has been channelled through a series of settling ponds. Site E is in a drainageway near the toe of the fill that has buried one of the previous tributary valleys. Water is sampled in the drainage ditch along Anderson Road (site G). Samples are also taken 700 yards downstream from site G at site H which is south of the culvert where the stream passes under the road.

Control wells are at site 9A for perched water in the valley and site 9B for the main ground-water body. These two control wells are in landfill zone 5 (Fig. 4) which has not been filled. Surface water control site A is in a drainageway west of the landfill (Fig. 1). Control site F is in a small first-order watershed that has had no landfilling.

Water samples are analyzed for K, Ca, Mg, Na, Fe, HCO_3, Cl, PO_4, SO_4, NO_3, electrical conductivity, (total dissolved solids, TDS), and hardness. Analyses from observation sites are compared to control sites and any water quality differences are then evaluated.

Means and standard deviations are calculated for all chemical constituents of samples from the control stations. Significant differences are tested by using the "student's t" distribution at confidence limits of 95 and 99 percent (Hoel, 1966). If the concentration of any constituent in a sample from an observation well or surface-water site on a given date exceeded the mean plus the confidence deviation of a sample from the control site, then that value was considered significantly different from the control value.

Variability of the chemistry of the natural ground water is represented by samples from control well 9B (Table 4). Samples of water possibly contaminated by leachate from the landfill (wells 3B, 4, 6, and 8) compared to the control gives the number of times the individual constituents differed significantly from control values (Table 5).

For 12 parameters among water samples from the 4 perimeter wells collected on 6 dates between February 1 and August 9, 1974, only 8 of 288 items differed significantly at the 95 percent confidence level from control-well water. None differed significantly at the 99 percent

TABLE 4. Variability of Water Chemistry for Control Well 9B, Monroe County
         Landfill Ground Water*

Parameter	Mean (ppm)	Std. Dev. (ppm)	Range (ppm)	Confidence Limit 95% (ppm)	99% (ppm)
Potassium	2.6	0.7	1.6 - 3.6	4.1	5.0
Calcium	51.7	6.4	45.7 - 61.0	65.3	74.1
Magnesium	20.2	8.5	4.6 - 27.5	38.3	49.7
Sodium	15.0	4.4	10.0 - 22.4	24.4	30.4
Iron	0.6	0.3	0.0 - 1.0	1.2	1.6
B.A.**	221.6	45.2	132.1 - 249.7	318.0	379.6
Chloride	25.8	2.3	24.5 - 27.9	30.7	33.8
Phosphate	0.0	0.0	0.0 - 0.1	0.0	0.0
Sulfate	62.6	33.4	21.3 - 115.5	133.8	179.3
Nitrate	1.6	1.1	0.6 - 3.0	3.9	5.4
TDS	26.5	14.1	250 - 285	295.1	314.3
Hardness	221.1	29.0	205.7 - 250.6	282.9	322.5

 *Six samples were analyzed.
**Bicarbonate Alkalinity.

level. There is no uniform pattern that shows greater concentrations
of certain chemical constituents in perimeter wells than in control
wells. Calcium, potassium, and phosphate were each greater on 2 occa-
sions, and nitrate and total dissolved solids were greater on one
date each. But in 280 of the 288 chemical analyses, water from the
perimeter wells did not differ significantly from control-well water.
Leachate from the landfill has not infiltrated the perimeter main
ground-water body.

TABLE 5. Frequency of Significant Differences in Ground Water Compared
to Control Well 9B.*

Parameter	Well 3B		Well 4		Well 6		Well 8	
	95%	99%	95%	99%	95%	99%	95%	99%
Potassium	O	O	O	O	O	O	2	O
Calcium	O	O	1	O	1	O	O	O
Magnesium	O	O	O	O	O	O	O	O
Sodium	O	O	O	O	O	O	O	O
Iron	O	O	O	O	O	O	O	O
B.A.**	O	O	O	O	O	O	O	O
Chloride	O	O	O	O	O	O	O	O
Phosphate	1	O	1	O	O	O	O	O
Sulfate	O	O	O	O	O	O	O	O
Nitrate	O	O	O	O	1	O	O	O
TDS	O	O	1	O	O	O	O	O
Hardness	O	O	O	O	O	O	O	O

*Six samples were analyzed.
**Bicarbonate Alkalinity.

The same procedure was used to evaluate water-quality differences
in the perched water. Water from well 3A was compared to water from
control well 9A (Tables 6 and 7). Among 48 analyses only 2 differed
significantly from control values. Nitrate and phosphate had higher
concentrations on one date each. As in deeper ground water, perched
water near the perimeter of the Monroe County landfill is not infil-
trated by leachate at the present time.

TABLE 6. Variability of Water Chemistry for Control Well 9A, Monroe County Landfill Perched Water*

Parameter	Mean (ppm)	Std. Dev. (ppm)	Range (ppm)	Confidence Limit 95% (ppm)	99% (ppm)
Potassium	8.0	1.1	7.1 - 10.0	10.3	11.8
Calcium	2.4	0.9	1.0 - 3.7	4.3	5.5
Magnesium	6.2	1.6	3.9 - 8.2	9.6	11.8
Sodium	15.7	3.8	11.4 - 22.4	23.8	29.0
Iron	0.8	0.3	0.0 - 1.0	1.4	1.8
B.A.**	30.5	10.7	15.6 - 41.6	53.3	67.9
Chloride	14.1	2.4	11.6 - 17.5	19.2	22.5
Phosphate	0.1	0.1	0.0 - 0.5	0.3	0.4
Sulfate	53.8	42.5	28.1 - 138.9	144.4	202.4
Nitrate	0.4	0.4	0.0 - 1.0	1.3	1.8
TDS	79.5	6.6	75 - 88	93.6	102.6
Hardness	33.5	8.7	20.2 - 42.3	52.0	63.9

*Six samples were analyzed.
**Bicarbonate Alkalinity.

TABLE 7. Frequency of Significant Differences in Perched Water Compared to Control Well 9A*

Parameters	Confidence Limits 95%	99%
Well 3A		
Potassium	0	0
Calcium	0	0
Magnesium	0	0
Sodium	0	0
Iron	0	0
Bicarbonate Alkalinity	0	0
Chloride	0	0
Phosphate	0	1
Sulfate	0	0
Nitrate	0	1
TDS	0	0
Hardness	0	0

*Four samples were analyzed.

To evaluate the effect of landfilling on the quality of water in the streams, water samples taken from within and downflow from the landfill were compared to water from control site A (Table 8). The number of times the observed values in the surface sites exceeded the confidence limits set by control site A are listed in Table 9.

TABLE 8. Variability of Water Chemistry for Control Site A, Monroe County Landfill Surface Water*

Parameter	Mean (ppm)	Std. Dev. (ppm)	Range (ppm)	Confidence Limit 95% (ppm)	99% (ppm)
Potassium	2.4	0.5	2.0 - 3.4	3.4	4.0
Calcium	10.2	4.3	2.1 - 14.1	18.9	23.8
Magnesium	6.5	0.9	5.4 - 7.4	8.3	9.3
Sodium	5.8	1.5	4.0 - 8.1	8.8	10.5
Iron	0.2	0.2	0.0 - 0.6	0.6	0.8
B.A.**	31.1	12.1	13.5 - 49.4	55.5	69.4
Chloride	13.3	3.7	11.1 - 16.4	20.8	25.0
Phosphate	0.1	0.2	0.0 - 0.3	0.5	0.7
Sulfate	29.9	5.5	0.0 - 48.2	41.0	47.3
Nitrate	2.1	1.5	0.2 - 7.6	5.1	6.8
TDS	65.9	10.1	48 - 79	86.3	97.9
Hardness	48.0	13.3	34.3 - 60.4	74.8	90.1

*Seven samples were analyzed.
**Bicarbonate Alkalinity.

Among 7 samples from site C (the raw leachate) collected between March 5 and June 7, 1974, results of 82 of 84 chemical analyses differed significantly at the 99 percent confidence level from control values.

Phosphate content was within control limits on 2 dates. Some concentrations of chloride, sodium, and TDS were 50 times greater than control values. At site D the effluent from the settling ponds differed

significantly at the 99 percent level from control values in 81 of 84 analyses. Concentrations of iron in 2 cases and phosphate in one case, were within the control limits. On many occasions, chloride, sodium, magnesium, alkalinity, and TDS were at least 30 times greater than control values. Analyses of water from site E, seepage from solid waste buried in a drainageway, differed significantly at the 99 percent confidence level from control values on 62 of 84 occasions. Concentrations were considerably lower than at sites C and D, and generally were between 2 and 10 times greater than control values for all constituents except iron and phosphate.

TABLE 9. Frequency of Significant Differences in Surface Water Compared to Control Site A*

Parameter	Site C		Site D		Site E		Site F		Site G		Site H**	
	95%	99%	95%	99%	95%	99%	95%	99%	95%	99%	95%	99%
Potassium	0	7	0	7	1	4	0	0	1	2	0	1
Calcium	0	7	0	7	0	7	0	0	1	3	0	0
Magnesium	0	7	0	7	0	7	0	0	2	3	0	1
Sodium	0	7	0	7	0	6	1	1	1	5	0	1
Iron	0	7	0	5	1	2	0	1	0	5	0	1
B.A.#	0	7	0	7	0	7	0	0	4	2	1	1
Chloride	0	7	0	7	0	7	0	0	2	1	0	1
Phosphate	0	5	0	6	0	3	0	0	1	0	0	0
Sulfate	0	7	0	7	1	4	1	0	1	4	0	1
Nitrate	0	7	0	7	0	1	0	0	0	0	0	0
TDS	0	7	0	7	0	7	0	0	2	5	0	0
Hardness	0	7	0	7	0	7	0	0	0	6	0	1

*Seven samples were analyzed.
**Two samples were analyzed.
#Bicarbonate Alkalinity.

Samples taken from site F, currently beyond landfill operations, differed significantly only 2 times at the 99 percent level and 4 times

at the 95 percent level. The 4 instances that were beyond confidence limits included sodium on 2 occasions and iron and sulfate on one date each. Contents at site G differed significantly from control values 36 times at the 99 percent level and 51 times at the 95 percent level, but in most cases, were either very closely approaching the upper confidence limit or were within control limits. The dilution of the landfill effluent with a much larger volume of relatively clean water is evident from the 2 analyses at site H which is 700 yards downstream from site G. Among 24 items 8 differed significantly at the 99 percent confidence level and 9 at the 95 percent level.

In the surface waters concentration of chemical constituents decrease with increasing distance from the source of contamination. For example, amounts of chloride, total dissolved solids, and sodium which are sensitive indicators of landfill effluent (Andersen and Dornbush, 1967), decrease with increasing distance from the landfill source at site C (Fig. 6). In this case, ion concentrations return to near natural values about 1200 feet from the source. Contamination is greatest near the landfill (site C), then decreases (sites D and E), but where the water leaves the landfill area (site G) it is still contaminated. Downstream at site H, however, the water chemistry is about normal for natural conditions. Therefore, water entering Beanblossom Creek is not seriously affected by landfill leachate at the present time.

Figure 6. DECREASE IN SURFACE WATER CONCENTRATION WITH DISTANCE
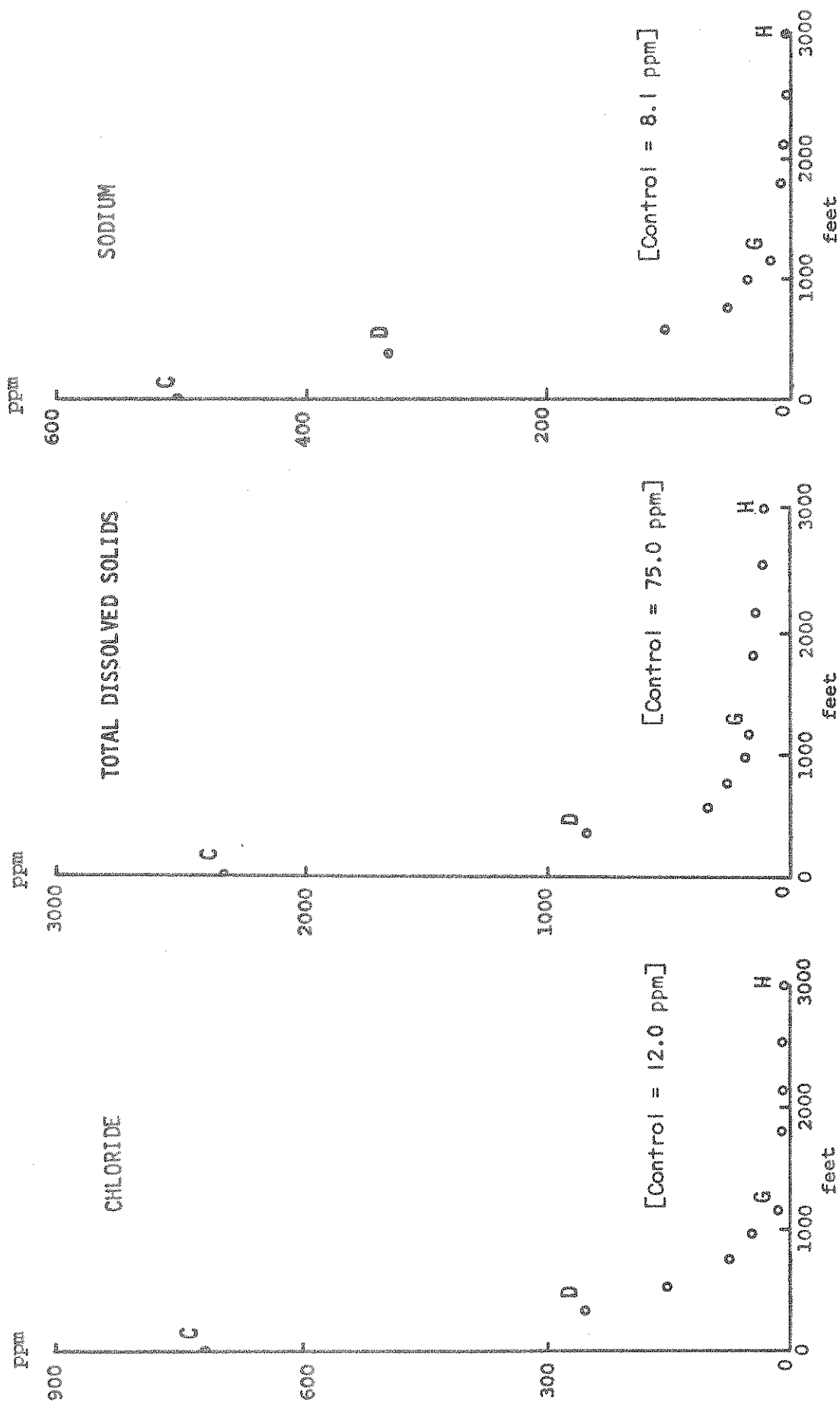
Summary of Analysis. (1) The hydrologic system at the Monroe County landfill has three components: (a) a main saturated zone that occurs in and just above bedrock under the uplands and in silt and clay in the valley-fill sediments, (b) a perched saturation zone in gravel that is interbedded in silt and clay above the main saturation zone in the valley, and (c) surface water in intermittent streams that drain the landfill area.

(2) At the present time, no significant change has occurred in the chemistry of ground water in the main saturated zone or perched-water zone peripheral to the landfill. Significant contamination of the main ground water in the near future is not anticipated, because permeability of the containing sediments is 0.047 ft/day. The perched water may become contaminated as the permeability of the containing gravels is many magnitudes greater, being 4.1 ft/day.

(3) Surface water in the streams that drain the landfill is currently contaminated by effluent many orders of magnitude greater than the natural chemical composition. Contamination decreases with distance in the drainage system from the perimeter of the landfill. At present, the surface water is diluted to normal composition prior to entering Beanblossom Creek.

(4) During construction of the landfill through its estimated 10-year life, the original landscape will be remodeled to essentially one continuous hillslope. The geohydrologic system will be altered with an ultimate rise of the water table above its original position under the hill and with flow paths directed toward the valley of Beanblossom Creek along steeper hydraulic gradients. Increased contamination can be expected in the subsurface, particularly in the perched ground water, and increased seepage and contamination can be expected in the surface water.

Brown County Landfill

Topography. The Brown County landfill is located about one
mile northeast of Helmsburg in the western half, $SE\frac{1}{4}$, $SW\frac{1}{4}$, section 23,
T. 10N., R.2E. The area is situated on a local topographic high
70 feet above and on the eastern valley slope of Dunnaway Creek
which is a major tributary of Beanblossom Creek (Fig. 7). The land-
fill site occupies 20 acres, but less than 2 acres have been used
since filling began in 1972. The site has two interfluves with
nose-slope gradients of about 2 percent and side-slope gradients
of 8 percent descending to first- and second-order drainageways.
These drainageways have incised Borden siltstone and descend to
the main valley along gradients of 5 to 6 percent.

Stratigraphy and Surficial Deposits. Six continuous cores
were taken from one interfluve where all trench filling has been
done (Fig. 7). The Borden siltstone bedrock (Wier and Gray, 1961)
is 9 to 12 feet below the ground surface under the interfluves.
Bedrock outcrops on valley slopes that descend to the bounding drain-
ageways. At drill sites 1 to 5 a zone of weathered siltstone con-
taining a fairly well-developed paleosol is immediately above bedrock.
The weathered bedrock is 6 to 9 feet thick, and the solum of the
paleosol is about 4 feet thick. Excluding the paleosol, weathered
bedrock contains 60 to 72 percent silt (Table 10). The paleosol
A and B horizons have, 58 and 45 percent silt, respectively; the
corresponding clay contents are 33 and 47 percent.

At site 6 near the eastern boundary of the area, a weakly de-
veloped paleosol is formed in Illinoian glacial till (Wayne, 1958).

Figure 7.   BROWN COUNTY LANDFILL

SURFACE SAMPLE SITE

DRILL SITE
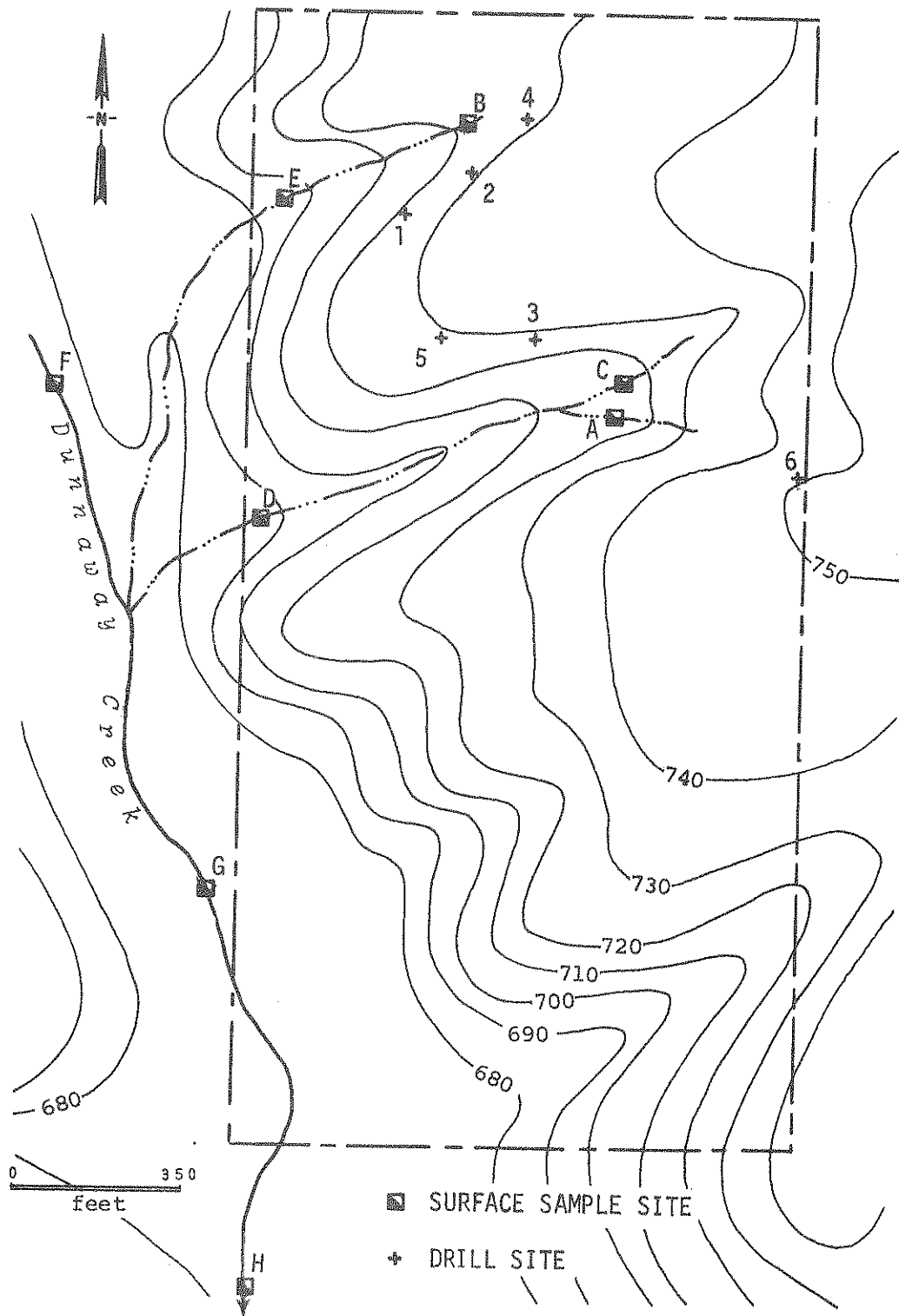
The C horizon of the paleosol contains 40 percent silt and less than 20 percent clay. The B horizon of the buried soil contains about 38 percent silt and 33 percent clay.
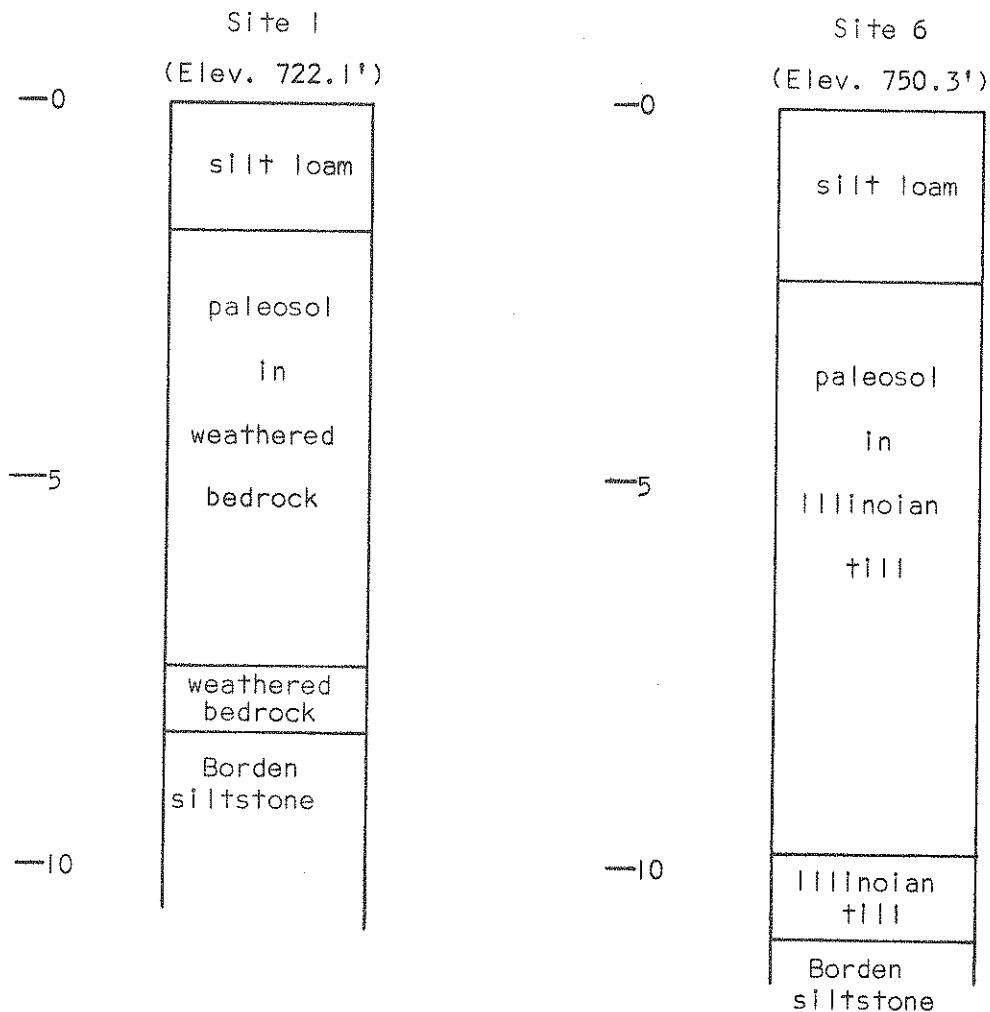
TABLE 10. Particle Size of Sediment at Sites 1 and 6

Depth (ft)	Material	Sand (%)	Silt (%)	Clay (%)
Site #1				
0 - 2	Loess	4.8	71.9	23.3
2 - 3.5	Ab horizon, paleosol	9.1	57.9	33.0
3.5 - 6	Bb horizon, paleosol	7.9	44.9	47.2
6 - 8	Weathered bedrock	5.2	71.1	24.6
Site #6				
0 - 2	Loess	5.0	68.5	26.5
2 - 4	Ab horizon, paleosol	27.4	55.3	17.3
4 - 7	Bb horizon, paleosol	28.1	38.7	33.2
7 - 10	Till	41.4	39.8	18.7

The surface deposit on the upland is a silt loam Wisconsinan loess, 2 feet thick, that contains 70 percent silt and 25 percent clay (Table 10). Generalized stratigraphic sections at sites 1 and 6 are given in Fig. 8. Cincinnati soils are dominant on the gently sloping uplands at the landfill site and are formed in the loess and the underlying substrata (Agric. Exper. Sta./Coop. Exten. Serv., Purdue Univ., 1971). In similar soils formed in thin loess over siltstone substratum, cation exchange capacities are 10 to 25 me/100g (Soil Survey Invest. Rept. 18, 1967).

Figure 8.   GENERALIZED SECTION DESCRIPTIONS

Site 1
(Elev. 722.1')

—0

| silt loam |
| paleosol
in
weathered
bedrock |
| weathered
bedrock |
| Borden
siltstone |

—5

—10

Site 6
(Elev. 750.3')

—0

| silt loam |
| paleosol
in
Illinoian
till |
| Illinoian
till |
| Borden
siltstone |

—5

—10

Hydrology.   Prior to construction of the landfill, runoff from
the interfluves flowed down side slopes and collected in the tributary
drainageways which, in turn, descend to Dunnaway Creek.   No ground
water was encountered during drilling in the late summer months of
1972 when rainfall was sparse.   During the wet spring months of 1974,
however, the intermittent drainageways continued to flow for several
days after all surface runoff had stopped.   This base flow is apparently

from infiltrated water that flows laterally in the subsurface.
This lateral hillslope seepage is the sole supply of water to the
drainageways during periods of no surface runoff.

Annual rainfall at the Brown County landfill is 42.7 inches.
Potential evapotranspiration, determined by the Thornthwaite method
(Taylor and Ashcroft, 1972) is 20.8 inches annually. If estimated
surface runoff is 4.6 in/yr as at Monroe landfill, then 17.3 inches
of water are available for infiltration each year.

From the previous calculations that about 3 inches of water are
required to bring a one-foot layer of waste to field capacity, about
30 inches of water are needed to wet the waste of a trench 10 feet
deep. At the infiltration rate of 17.3 in/yr, about 20 months are
needed to bring the waste of the filled trench to field capacity.

The time required for effluent to seep laterally to the drain-
ageways may be estimated from the permeability of the weathered bed-
rock (0.27 in/hr or 0.54 ft/day by laboratory permeability test).
The southern end of the dry-weather trenches are about 50 feet from
the tributary drainageways (Fig. 7), so at least 92 days are required
for travel time. Seepage from the trenches may appear in the drain-
ageways in about 2 years.

The drainageways on both sides of the interfluve have been dammed
at sites D and E. During periods of excessive runoff from the landfill,
the settling ponds become full and overflow. Effluent from these ponds
flows west and into Dunnaway Creek. The discharge from the landfill
area is relatively small compared to total flow of Dunnaway Creek
and is usually less than 10 percent (Table 11).

TABLE 11.   Discharge at Sites D, E, and G.(cfs)

Date	Site D*	Site E*	Site G**
3/15/74	0.2	0.1	12.3
4/8	1.1	0.5	166.2
5/15	0.2	0.1	11.4
5/23	0.5	0.1	12.3
5/30	0.3	0.1	20.7
6/7	0.1	0.1	7.0
6/24	0.01	0.04	8.3
7/11	0.02	0.02	2.0

*Settling pond.
**Dunnaway Creek.

Landfill Design.   Operation of the Brown County landfill began in June, 1972, and currently utilizes both the trench and area methods.   In dry weather trenches 20 feet wide, 10 feet deep, and 150 feet long are excavated on the interfluve paralleling the northern boundary of the landfill.   Since operations began, 7 trenches have been filled during the summers on the western end of this interfluve (Fig. 7).   During the winter of 1972-1973 trenches were also filled at the eastern end of the same interfluve.   There, 6 trenches were 20 feet wide, 8 feet deep, and 100 feet long.

The original operation plan called for only trench filling between the ravines (Regional Services Corporation, 1971), but procedures now include area-method filling during winter months (R. J. Wigh, personal communication, 1973).   Wet-weather filling during the winter of 1973-1974 was also done in an excavated cavity 20 feet deep and about 110 feet wide at the head of one drainageway (Fig. 7).   Waste was dumped

from a ramp, spread, and compacted, and was covered by earth daily.
This area-fill contains about 25 feet of interbedded solid waste
and earth. The final cover has a 5 percent slope down the drain-
ageway to the west.

During dry-weather operation, the entire interfluve will be
filled in a series of parallel trenches. At the present rate of
filling, the northern interfluve will be used for at least 5 more
years. Wet-weather filling will continue downvalley in the drain-
ageway for 2 or 3 years (T. V. McGuire, operator, personal communi-
cation, 1974).

Water Quality. As current seepage is mainly from the area fills
at the head of drainageways, water sampling for possible contamination
was focused on surface-water sites downslope. In the drainageways
sites A and B are control stations on the respective southern and
northern sides of the interfluve (Fig. 7). Leachate seeping from
the winter-fill area is sampled at site C but is diluted by control
water from site A which then collects in a settling pond. Outflow
from the settling pond is sampled at site D. Site B is a control
station, and overflow from the settling pond is sampled at site E.
Effluent from the two settling ponds flows west and drains into
Dunnaway Creek.

In Dunnaway Creek the control station is site F which is up-
stream from the landfill. Site G is downstream and should measure
any landfill effluent. Site H half a mile downstream from site G
may show the effects of an intervening large-scale cattle-feeding
operation along Dunnaway Creek.

As at Monroe landfill, comparisons were made of the water
chemistry between possibly contaminated samples and control samples.
Nine sets of samples were collected and analyzed for the period
March 5 through July 11, 1974. A total of 108 analyses were per-
formed on samples from each of sites C, D, E, and G and were com-
pared to control samples (Tables 12, 13, 14, and 15).

TABLE 12. Variability of Water Chemistry for Control Site A, Brown County
Landfill Surface Water *

Parameter	Mean (ppm)	Std. Dev. (ppm)	Range (ppm)	Confidence Limits 95%(ppm)	99%(ppm)
Potassium	3.1	1.6	1.6 - 6.3	6.1	7.6
Calcium	14.2	4.9	7.8 - 19.2	23.5	28.1
Magnesium	7.1	2.6	3.4 - 10.9	12.0	14.5
Sodium	6.5	3.1	1.5 - 10.6	12.4	15.3
Iron	0.8	0.4	0.3 - 1.5	1.6	1.9
B.A.**	45.0	7.9	37.4 - 58.8	60.0	67.4
Chloride	15.5	4.4	9.1 - 23.6	23.8	28.0
Phosphate	0.1	0.2	0.0 - 0.5	0.5	0.7
Sulfate	20.4	5.6	7.8 - 27.3	31.0	36.3
Nitrate	0.8	0.6	0.4 - 1.2	1.9	2.5
TDS	75.1	4.9	68.0 - 82.0	84.4	89.0
Hardness	64.8	12.7	47.6 - 89.3	88.9	100.9

*Nine samples were analyzed.
**Bicarbonate Alkalinity.

Samples from site C differed significantly from control values in
72 and 76 analyses at 99 and 95 percent confidence levels, respectively.
Chloride, TDS, alkalinity, and hardness differed significantly from
control values at the 99 percent level on each of the 9 sampling dates.
The parameters that differed significantly at the 95 percent level on
the fewest dates were: iron (4), nitrate (3), and phosphate (1).

TABLE 13. Variability of Water Chemistry for Control Site B, Brown County
Landfill Surface Water*

Parameter	Mean (ppm)	Std. Dev. (ppm)	Range (ppm)		Confidence Limits 95%(ppm)	99%(ppm)
Potassium	5.1	1.9	2.2 -	8.0	8.7	10.5
Calcium	11.0	4.2	4.2 -	16.0	19.0	22.9
Magnesium	7.0	2.8	1.9 -	10.3	8.9	15.0
Sodium	7.0	3.6	2.2 -	14.0	14.0	17.4
Iron	1.1	0.5	0.8 -	11.9	2.1	2.6
B.A.**	44.8	8.8	35.0 -	62.4	61.5	69.8
Chloride	16.2	4.3	7.1 -	21.1	24.3	28.4
Phosphate	0.0	0.1	0.0 -	0.2	0.2	0.3
Sulfate	25.3	7.5	15.5 -	37.8	39.5	46.6
Nitrate	1.5	2.0	0.1 -	6.6	5.3	7.2
TDS	80.0	10.5	68 -	100	99.9	109.8
Hardness	64.2	10.5	50.2 -	80.2	84.1	94.0

*Nine samples were analyzed.
**Bicarbonate Alkalinity.

TABLE 14. Variability of Water Chemistry for Control Site F, Brown County
Landfill Surface Water*

Parameter	Mean (ppm)	Std. Dev. (ppm)	Range (ppm)		Confidence Limits 95%(ppm)	99%(ppm)
Potassium	2.3	0.3	1.8 -	2.8	2.9	3.2
Calcium	8.6	3.3	5.1 -	14.4	14.9	18.0
Magnesium	6.2	1.5	3.6 -	9.4	9.0	10.5
Sodium	6.0	2.2	3.9 -	10.3	10.2	12.3
Iron	0.4	0.4	0.0 -	0.8	1.2	1.5
B.A.**	29.5	14.1	20.1 -	57.2	56.2	69.6
Chloride	10.1	3.1	4.6 -	15.6	16.0	18.9
Phosphate	0.0	0.0	0.0 -	0.1	0.0	0.0
Sulfate	32.9	5.0	27.3 -	38.2	42.4	47.1
Nitrate	0.6	0.8	0.5 -	2.5	2.3	2.9
TDS	67.1	11.7	52 -	88	89.3	100.3
Hardness	49.8	11.4	30.1 -	64.1	171.4	82.2

*Nine samples were analyzed.
**Bicarbonate Alkalinity.

TABLE 15. Frequency of Significant Differences, Brown County Surface Water*

Parameter	Mean (ppm) Site C[#]		Std. Dev. (ppm) Site D[#]		Range (ppm) Site E[+]		Confidence Limits 95%(ppm) Site G[&]		99%(ppm) Site H[&]	
	95%	99%	95%	99%	95%	99%	95%	99%	95%	99%
Potassium	1	5	0	1	0	0	0	0	0	0
Calcium	0	6	0	4	1	0	0	0	0	0
Magnesium	0	7	0	2	2	0	0	0	0	0
Sodium	0	7	1	2	1	0	0	0	0	0
Iron	3	1	0	5	0	6	0	0	0	0
B.A.**	0	9	2	4	0	3	0	0	0	0
Chloride	0	9	2	1	0	0	0	0	0	0
Phosphate	0	1	0	1	1	0	0	0	0	0
Sulfate	0	6	2	6	1	0	0	0	0	0
Nitrate	0	3	0	0	0	0	0	0	0	0
TDS	0	9	0	9	0	6	0	0	2	0
Hardness	0	9	2	3	0	1	0	0	0	0

*Nine samples analyzed -- only 4 at site H.
**Bicarbonate Alkalinity.
#Control was site A.
+Control was site B.
&Control was site F.

Analytical values of settling-pond effluent at site D differed significantly from control values 38 times at the 99 percent confidence level and 47 times at the 95 percent level. Total dissolved solids differed significantly from control values 9 times and sulfate differed significantly 6 times at the 99 percent level. Chloride differed significantly only one time at the 99 percent level and 3 times at the 95 percent level, and sodium differed significantly 2 times at the 99 percent level and 3 times at the 95 percent level.

At site E values differed significantly from control values 16 times at the 99 percent level and 22 times at the 95 percent level. At the 99

percent level, TDS differed significantly from control values 6 times, and bicarbonate alkalinity differed significantly 3 times. All other values, except those for iron, differed significantly 2 times or less at both confidence levels.

Samples were taken upstream and downstream from the junction where drainage from the landfill enters Dunnaway Creek (Fig. 7). Samples from downstream sites G and H were compared to control values from site F, and little if any differences exist (Tables 14 and 15).

Summary of Analysis. (1) The original hydrologic system in the landfill area consisted of two components. (a) Surface runoff from interfluves flowed down side slopes to drainageways and in turn, to Dunnaway Creek. (b) The water table of the ground water was well below the bottom of landfill trenches during the dry season, but lateral subsurface seepage contributed base flow to the drainageways during the wet season.

(2) Currently, effluent is mainly from the wet-weather landfill waste at the heads of the drainageways. This seepage contaminates surface water downslope and in the leachate ponds farther down valley. Dunnaway Creek, which receives overflow down valley from the ponds, is not being contaminated by leachate at the present time. The trench-fills on the interfluves are not contributing significant amounts of contaminants either in runoff or in sub-surface seepage.

(3) During future operations, buildup of the wet-weather landfills in the drainageways may increase discharge from the waste down slope to the leachate ponds and beyond to Dunnaway Creek. Although

travel time of subsurface seepage from saturated trenches to the drainageways will be about 2 years at the rate of about $\frac{1}{2}$ ft/day, adsorption of cations by the soils (CEC of 10-25 me/100g) should significantly decrease transmission of contaminants through the subsurface.

Columbus Municipal Landfill

Topography. The Columbus Municipal Landfill is located at the southeastern city limits in the $S\frac{1}{2}$, $SW\frac{1}{4}$, $SE\frac{1}{4}$, section 30, T.9N., R.6E. The site covers about 22 acres, but less than 3 acres were used since landfilling began in June, 1971.

The area is located in the White River valley and abuts the scarp of a terrace that is 20 feet above the level of the floodplain (Fig. 9). The scarp descends on a 16 percent slope from elevation 627 feet on the terrace to 608 feet on the floodplain. An abandoned stream channel, about 150 feet wide and 2 to 3 feet lower than floodplain level, is at the base of the scarp and crosses the study area from northwest to southeast.

The floodplain portion of the landfill area is within the parts of the White River valley that are subject to "intermediate regional flood" and "standard project flood" (Corps of Engrs., 1968). The intermediate regional flood corresponds to the 100-year flood, or a flood that may occur once in 100 years but in any year. The standard project flood is a predicted flood that is more severe and may be disastrous.

Stratigraphy and Surficial Deposits. The sediments in the landfill area are mainly sand and gravel which are overlain by finer-textured surficial materials. Depth to shale bedrock in the vicinity of the landfill site is about 60 to 80 feet (Davis et al., 1969). The site was extensively drilled, and continuous cores were collected of the sediment overlying the sand and gravel. These sediments contain large amounts of sand--43 to 59 percent on the terrace and

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Figure 9.   COLUMBUS MUNICIPAL LANDFILL SITE

WATER QUALITY/WATER LEVEL WELL

WATER LEVEL WELL

CONTINUOUS CORE

WELL NEST

DIRECTIONAL FLOW TRAVERSE

and 41 to 71 percent on the floodplain. The coarser-textured materials
underlying the surficial deposits contain more than 75 percent sand
(Table 16).

TABLE 16. Textural Data for Sites 2, 10, and 19

Depth (in)	Material	Sand (%)	Silt (%)	Clay (%)
Site 2 (abandoned stream channel)				
0 - 8	Loam	41.5	44.7	13.8
8 - 26	Sandy loam	70.6	7.6	21.8
26 -	Sand and gravel	76.7	19.5	3.7
Site 10 (terrace)				
0 - 17	Loam	43.2	44.4	12.4
17 - 40	Sandy loam	58.9	31.4	9.6
40 -	Sand and gravel	84.7	13.6	2.7
Site 19 (floodplain)				
0 - 71	Loam	40.7	40.3	18.9
71 - 79	Sandy loam	60.0	26.1	14.0
79 -	Sand and gravel	75.0	16.1	8.9

A typical profile of sediments on the terrace at site 10 contains
17 inches of loam and 23 inches of sandy loam with sand and gravel
which continue to more than 80 feet in depth. On the floodplain at
site 19, 71 inches of loam and 8 inches of sandy loam overlie sand
and gravel which continue downward for more than 60 feet. At site 2
in the abandoned stream channel, 8 inches of loam and 18 inches of
sandy loam are above sand and gravel which also descend to more than
60 feet. The sand and gravel, although below different features and
topographic levels, form a continuous aquifer under the landfill area.

The soils at the landfill are the Fox-Ninevah-Ockley association which is typical of level, well-drained, loamy soils on outwash sand and gravel (Agric. Exper. Sta./Coop. Exten. Serv., Purdue Univ., 1971).

Hydrology. The ground water of the area was observed in 15 wells (Fig. 9). Elevation of the water table fluctuates in the sand and gravel as much as 3 feet annually and is about $1\frac{1}{2}$ feet higher under the terrace than under the floodplain (Table 17).

TABLE 17. Ground-Water Elevation Changes

Site	Elevation (ft)	Water Table Elevation			
		7/28/71	10/2/71	1/25/72	4/26/72
	Terrace Sites				
10		603.0	601.4	603.7	604.0
11		603.1	601.5	603.8	604.1
12		603.2	601.6	603.9	604.2
32		603.2	601.6	603.9	604.2
	Mean + Std. Dev.	603.1+.1	601.5+.1	603.8+.1	604.1+.1
	Floodplain Sites				
18		601.8	600.2	602.3	602.7
19		601.7	600.1	602.2	602.6
20		601.8	600.2	602.3	602.7
21		601.7	600.3	602.4	602.8
	Mean + Std. Dev.	601.8+.1	600.2+.1	602.3+.1	602.7+.1

A depth-to-water map shows that the general direction of ground-water movement at the landfill site is to the southwest (Fig. 10). Direction of flow was also confirmed by injecting a brine solution in a center well and measuring electrical conductivity (TDS) changes

in 4 peripheral wells, each located about 3 feet from the center well (Fig. 9). The injected solution was $1\frac{1}{2}$ gallons of 12,500 ppm NaCl. Samples were taken from the 4 peripheral wells for 7 hours after injection. The dilution of the brine in the center well was also measured (Table 18).

Figure 10.   DEPTH TO WATER MAP--COLUMBUS MUNICIPAL LANDFILL
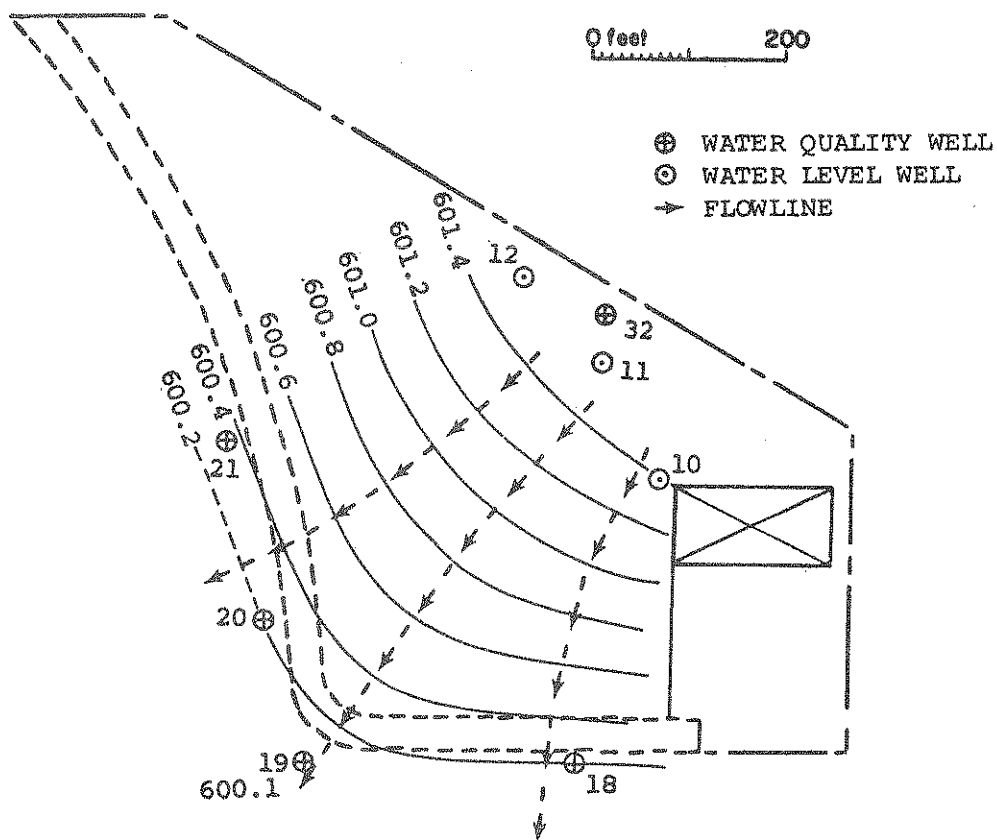
The largest increases in electrical conductivity occurred in the south and west wells and were still greater than original values 4 days later. The initial increases in the north and east wells were probably caused by dispersion of the brine solution, but values returned to normal after about 5 hours (Table 18).

TABLE 18. Electrical Conductivity Data and Flow Direction

Time (hrs)	Concentration (ppm NaCl)				
	Center	North	East	South	West
Before	435	435	435	435	435
1	2200x	450x	475x	520x	450x
2	1750x	465x	470x	545x	460x
3	1600x	440	475x	550x	470x
4	1450x	435	450x	570x	470x
5	1350x	440	440	585x	475x
6	1250x	440	445	580x	475x
7	1200x	435	440	580x	480x
98	635x	435	435	575x	480x

x Significant change.

The direction of ground-water movement determined here agrees with a previous study (Davis et al., 1969). Trend-surface analysis based on data from 38 observations showed that flow direction was toward the southwest and that monthly variations were minimal.

Rate of ground-water flow was determined from pumping tests. Transmissivity was calculated from the relation (Johnson, 1966):

$$T = \frac{264 \ (Q)}{\Delta h}$$

where T is the rate of flow of water, in gal/day, through a vertical

strip of the aquifer one foot wide under unit hydraulic gradient and where height equals the thickness of the aquifer; Q is the rate of pump discharge in gal/min; and Δh is the drawdown in feet. From the pumping tests, T was 88,000 gal/ft day or 11,750 ft^3/ft day. As the aquifer thickness is 70 feet (Davis et al., 1969), permeability is 167.9 ft/day. Using an effective porosity in the sand and gravel of 25 percent (Davis and DeWiest, 1966), Darcy's law is:

$$V = \frac{1}{n}\ k\ \frac{dh}{dl}$$

where V is velocity of flow in feet per day; k is the permeability in feet per day; dh/dl is the hydraulic gradient in feet per feet; and n is effective porosity in percent. From this relation, the rate of ground-water flow under the landfill is about 2 ft/day.

The Columbus area receives 41.2 inches of precipitation annually. Evapotranspiration, estimated by the Thornthwaite method (Taylor and Ashcroft, 1972) is 20.2 inches yearly. About 21 inches of water are available for runoff or infiltration each year, and runoff is minimal behind the dike (Fig. 9).

The effect of infiltration in the landfill waste was determined in a well emplaced in the waste. Well 13 was placed in the center of the filled area and was bottomed at elevation 603.7 in the fine-texture sediment underlying the landfill. Free water accumulated in the well after about 16 months (October, 1972) and continued to rise until about June, 1973, when water was standing at elevation 605.9 feet (Table 19). Water level has stabilized since that date.

After water samples are taken from the well, the water level rises
to its original position in a few hours.

TABLE 19.   Water Level Changes in Well 13, Columbus Landfill*

Date	Water Elevation (ft)
8/11/72	DRY**
10/18	603.9
11/8	604.3
12/18	604.5
2/2/73	604.6
5/30	605.2
6/5	605.8
6/26	605.9
7/12	605.8
7/25	605.9
10/3	605.9
12/1	605.8
2/11/74	605.9
3/11	606.0
6/26/74	605.9

* Well bottom elevation is 603.7.
** Well was dry prior to 10/18/72.

About 82.4 inches of precipitation were received on the land-
fill before the final cover was placed over the waste in May, 1973.
About 48 inches of precipitation were absorbed as field-capacity
moisture during the first 16 months. After field capacity was reached,
another 34.4 inches of precipitation accumulated in the waste as free
water which would saturate a thickness of 44.1 inches of waste when
corrected to a porosity of 78 percent.

Permeability of the sediments underlying the waste, as determined
by falling-head permeameter (Klute, 1965), is 0.002 in/hr (0.004 ft/day).

During the 8 months after field capacity was reached in the waste (October, 1972 - June, 1973) 0.96 feet (11.5 inches) of water should have seeped into the sediments. This seepage loss, corrected for 78 percent porosity, would decrease the amount of water ponded in the refuse by 14.8 inches. From these calculations, the saturated zone in the lower part of the waste should be about 2.6 feet (30 inches) thick. By actual measurement the zone is 2.2 feet (26.4 inches) thick. The 3.6 inches difference is probably due to evaporation from the uncovered surface of the waste.

After the final soil cover was placed on the landfill, the amount of water that could infiltrate the surface decreased. Of 42.1 inches annual precipitation, 20.2 inches could be lost to evapotranspiration, and runoff from the fill area could occur, because the dike was removed, and the fill area was no longer a closed system. Runoff is estimated as 4.6 in/yr (Saxton et al., 1971), but runoff may be slightly higher because a good grass cover was not established. From the calculations, 42.1 - (20.2 + 4.6), approximately 17.3 inches of water infiltrate the landfill each year. Under typical landfill conditions in the humid Midwest, approximately half of the yearly precipitation should be infiltration (Hughes et al., 1971). As the permeability of the underlying fine-texture sediments is 17.5 in/yr, water should percolate through the sediments underlying the landfill at about the same rate as water infiltrates the final cover of the landfill. Hence, the budget is balanced, and water level in the monitoring well is stabilized (Table 19).
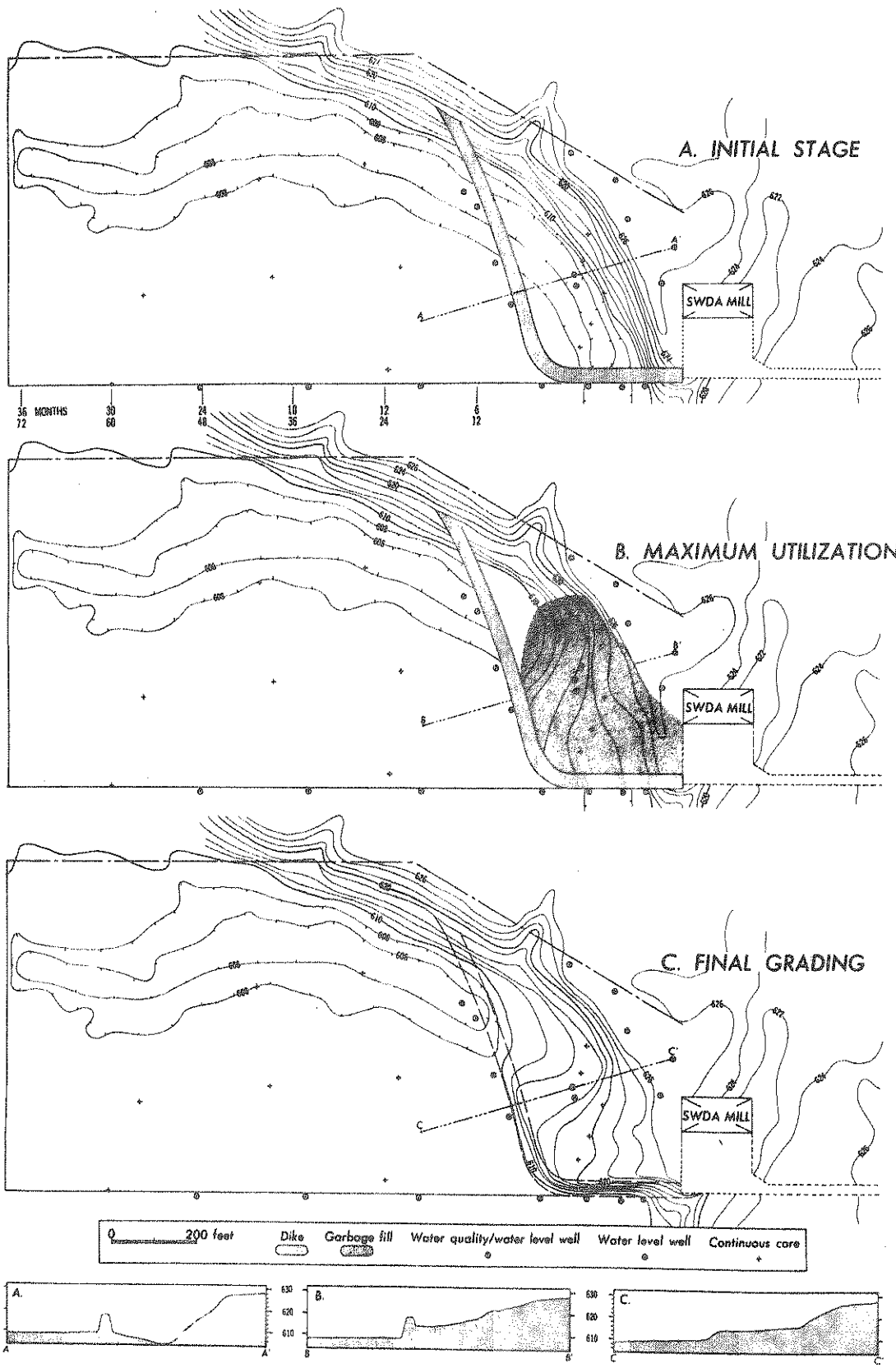
Landfill Design. Operation at the Columbus landfill began in
June, 1971. Solid waste was unloaded on a conveyor belt which carried
the refuse to a ball mill where glass, light metals, and other waste
including organic materials were pulverized. The waste was then
carried by another conveyor belt to a shredder where the organic
materials were reduced in size. The milled product was loaded into
a truck which dumped the processed waste in the fill area. The waste
was thinly spread and compacted in the landfill area on the floodplain.
No daily soil cover was used (Amer. Test. and Engr. Corp., 1970).

At the beginning of landfill operation a 7-foot high dike with
each end abutting the terrace was constructed on the floodplain at
the eastern end of the site (Fig. 9). As originally designed, the
area behind the dike was to be filled with waste to the level of
the top of the dike. Then, the procedure was to be repeated behind
another dike that would be constructed west of the filled area, etc..
It was expected that 3 years would be required for the entire landfill
site to be covered with 7 feet of waste. Then another 7-foot layer of
compacted waste was to be piled on top of the existing fill by another
series of dikes. After 6 years the area was to contain 14 feet of
shredded and compacted waste with a 2-foot cover of soil.

The landfill operation was stopped in June, 1972, due to ex-
cessive operating costs. By that time, the area enclosed by the
first dike had not been completely filled. Less than 12 percent
of the landfill surface area had been covered with waste.

The area was surveyed after operations ceased (Fig. 11).
About 900,000 cubic feet (33,000 cubic yards) of compacted waste
had been dumped in the landfill. If the entire area had been
filled with 14 feet of waste, the resulting volume would have

Figure 11.   STAGES OF THE COLUMBUS MUNICIPAL LANDFILL

A. INITIAL STAGE

B. MAXIMUM UTILIZATION

C. FINAL GRADING

SWDA MILL

0 _____ 200 feet   Dike   Garbage fill   Water quality/water level well   Water level well   Continuous core

been 1.7×10^7 cubic feet (625,000 cubic yards). Only about 5 percent of the planned volume was actually placed in the landfill. The waste was not covered until May, 1973. The dike was removed and used for cover material, and additional earth was hauled in to complete the job. No attempts were made to establish a vegetative cover on the finished surface. The completed landfill resting directly on the original floodplain is now an unvegetated hill sloping gently to the northwest.

Water Quality. As water infiltrates the landfill and percolates through the fine-texture material into the ground water, contaminants are leached from the waste. The degree of contamination in the subjacent aquifer depends on the ability of the sediment beneath the landfill to filter contaminants from the leachate. Cation exchange capacities (CEC) of the fine-texture sediments underlying the waste are 21.9 to 33.1 me/100g (Table 20). The filtering ability of these sediments was studied experimentally.

In a falling-head permeameter, leachate from the landfill waste was permitted to percolate through a core of sediment contained in the extraction Shelby tube (Fig. 12). Chemical compositions of the original leachate in the tube above the core and seepage waters from the bottom of the core were analyzed (Table 21). The latter samples were analyzed through a time sequence. During the experiment, 17.8 inches of fluid passed through the core.

Total dissolved solids in the outflow increased systematically through time (Fig. 13, Table 21). None of the ion concentrations was as great as in original leachate at the end of the 20 weeks, but they were still increasing at that time.

TABLE 20. Soil Chemistry Data--Columbus Landfill (me/100g)*

Depth (in)	Ca	Mg	K	Na	Cations (Sum)	CEC	BS (%)	OC+ (%)
0-1	15.3	3.2	1.5	0.2	20.2	21.9	92.2	1.6
1-2	16.3	1.4	1.4	0.1	19.2	28.1	68.3	2.1
3-4	12.8	2.9	1.2	0.1	17.0	29.0	58.6	3.6
5-6	14.5	2.5	1.3	0.1	18.4	33.1	55.6	4.3
7-8	13.7	2.6	1.2	0.1	17.6	32.6	54.0	4.2
9-10	8.5	2.2	0.4	0.1	11.5	24.3	47.3	5.1
11-12	4.8	1.7	0.2	0.1	6.8	22.0	30.9	4.0
13-14**	4.0	1.0	0.1	0.1	5.1	8.0	59.3	2.8
15-16**	2.2	1.0	0.1	0.1	3.4	5.3	64.2	1.8

 * Core sample from abandoned stream channel, near site 34.
** Sample was mostly sand.
 + Organic Carbon.

Permeability of the soil decreased considerably as leachate passed through the soil (Fig. 14) and was probably caused by plugging of pore spaces with organic complexes from the leachate.

After completion of the permeameter experiment, the soil through which the leachate had passed was chemically analyzed. One-inch increments of the core were analyzed for extractable cations, CEC, and organic carbon (Table 22). The values were compared to those of the original soil (Table 20). Organic carbon and base saturation in the contaminated core showed very systematic increases (Figs. 15A and 15B). CEC also increased (Fig. 15C) and relates to the increase in organic carbon (Figs. 15A and 15C). Although the soil does filter some of the contaminants from the leachate, progressively greater quantities pass through the soil with time and enter the ground water (Table 21).
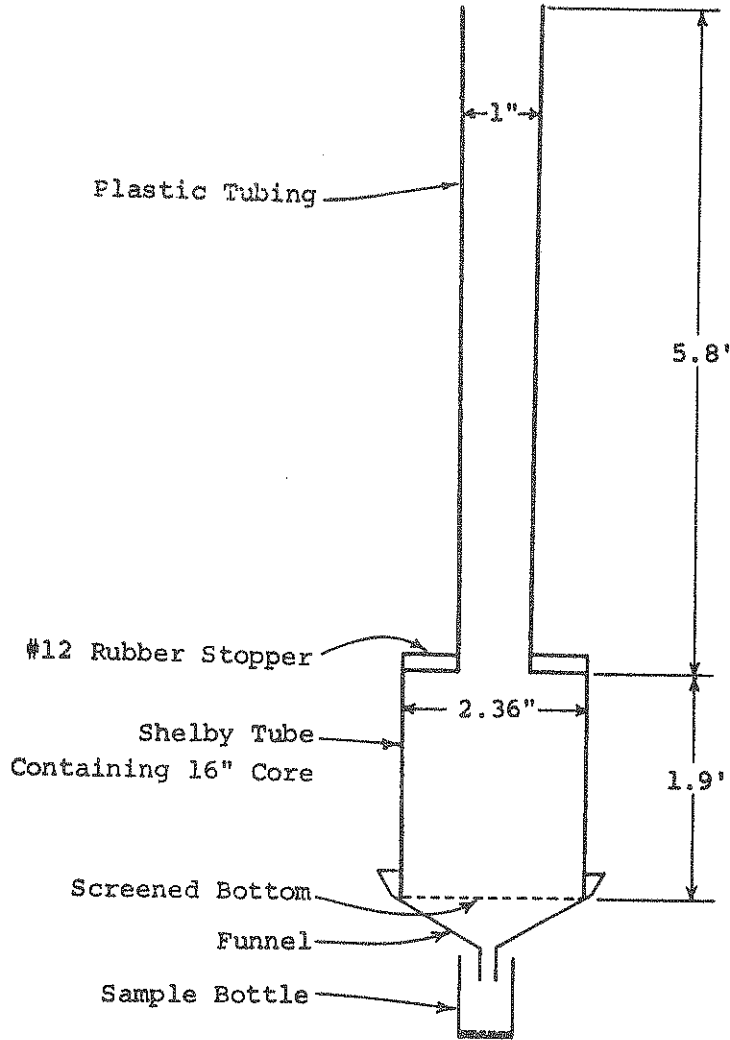
Figure 12.   LEACHATE PERMEAMETER

Plastic Tubing

←1"→

5.8'

#12 Rubber Stopper

2.36"

Shelby Tube
Containing 16" Core

1.9'

Screened Bottom

Funnel

Sample Bottle

TABLE 21. Water Chemistry of Outflow Samples--Leachate Permeameter

Sample	Time* (wks)	Ca (ppm)	Mg (ppm)	K (ppm)	Na (ppm)	TDS (ppm)
Effluent	-	160.0	59.0	5.1	114.0	910
1	0.6	49.1	9.0	1.5	11.1	300
2	1.1	52.3	10.1	1.3	10.6	315
3	1.6	50.2	11.8	1.0	12.3	320
4	2.1	37.4	14.0	1.0	13.1	335
5	2.9	28.9	17.0	1.5	17.4	355
6	3.5	24.7	21.6	2.0	20.8	410
7	4.0	20.0	22.3	1.9	21.5	430
8	5.0	17.1	25.5	2.2	31.0	495
9	6.0	15.4	28.6	2.2	41.2	540
10	7.0	14.1	27.7	2.3	49.5	555
11	7.5	19.1	31.2	2.5	60.5	565
12	8.4	19.4	33.1	2.5	63.0	590
13	9.3	23.5	40.4	2.5	66.7	615
14	11.0	33.0	45.9	2.5	68.8	650
15	13.5	43.2	47.2	2.6	79.7	720
16	18.5	62.9	50.8	3.0	85.9	750
17	20.0	66.9	52.2	3.5	93.7	760

* Time after beginning of experiment.

To evaluate the effect of landfilling on ground-water quality, observation wells were installed around the perimeter of the dike (Fig. 9). Wells 8, 2, and 1 intercept the 0 to 6, 3 to 16, and 12 to 30 feet depth intervals of the aquifer, respectively, on the south side of the dike (Fig. 16). Well 19 at the southwest corner of the dike and wells 20, 21, and 34 on the west side of the dike were installed to intercept water in the upper 5 feet of the aquifer.

Figure 13.   INCREASE IN TDS CONCENTRATION WITH TIME
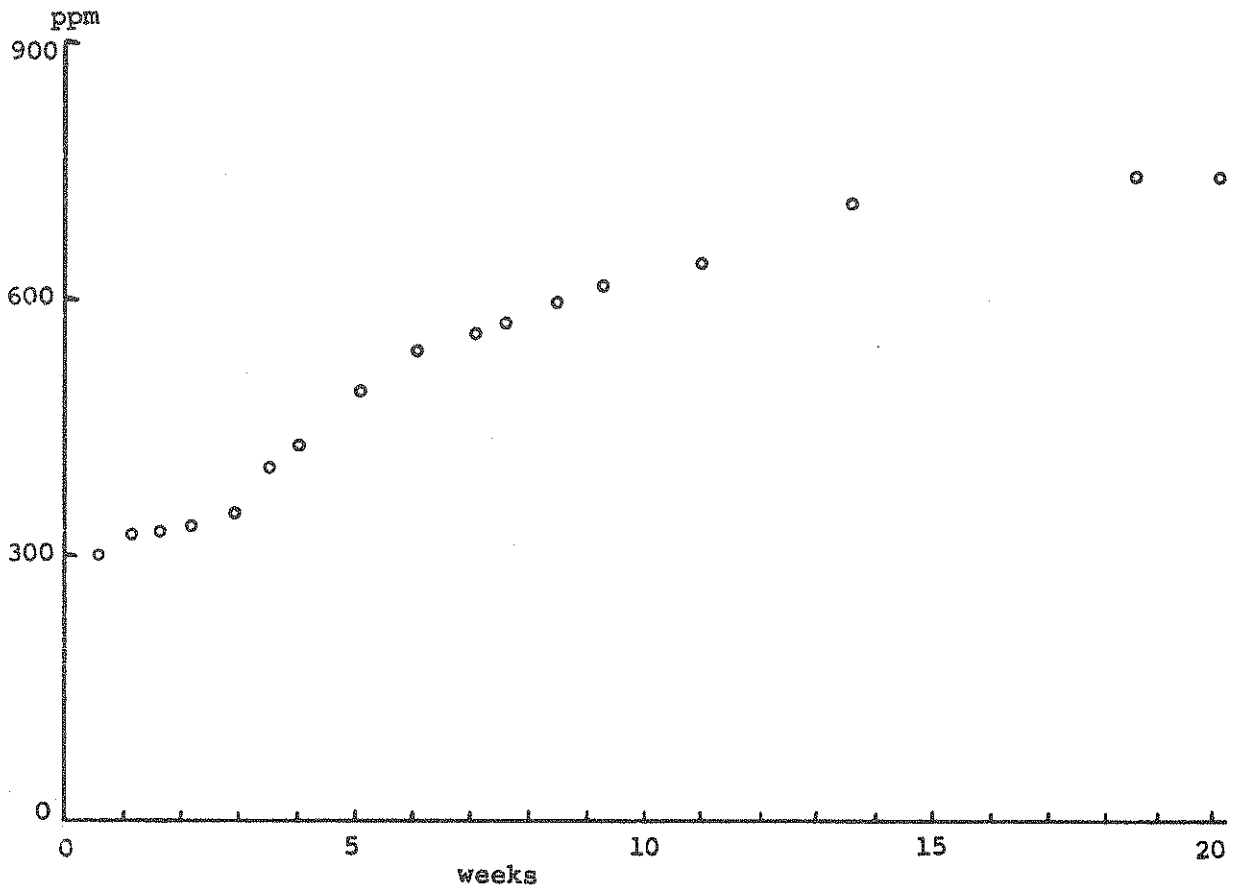
Wells 13 and 13X were installed within the filled area. Well 13 was

bottomed in the fine-texture sediment beneath the waste and collects

leachate that accumulates within the waste. Well 13X penetrates the

solid waste and underlying sediment and is used to sample the ground

water beneath the landfill (Fig. 9, cf. Fig. 16). Wells 5, 6, 7,

and 8 were placed in the direction of ground-water flow at distances

of 50, 100, 200, and 400 feet from the southwest corner of the dike
(Fig. 9). Analyses of water samples from these 4 wells should show
any contamination with distance from the landfill. Since the direc-
tion of flow is from the terrace toward the floodplain, background
values for water quality were determined in wells 32 and 32X on the
terrace (Fig. 9, cf. Figs. 10 and 16).

Figure 14. CHANGES IN PERMEABILITY WITH PASSAGE OF LEACHATE
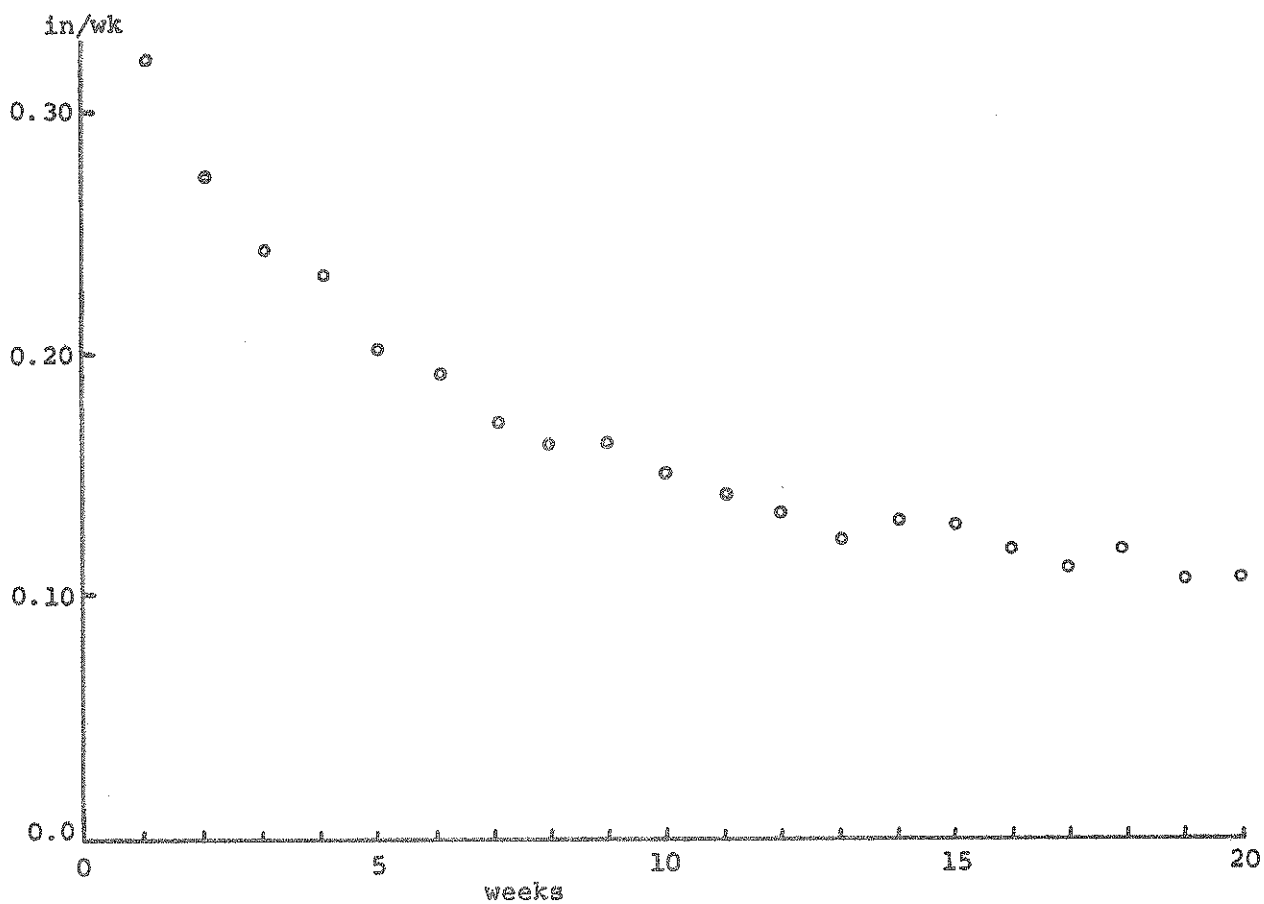
TABLE 22.   Soil Chemistry after Passage of Leachate--Permeameter
Experiment--Columbus Landfill (me/100g)

Depth (in)	Ca	Mg	K	Na	Cations (Sum)	CEC	BS (%)	OC* (%)
0-1	18.7	3.6	0.8	0.7	23.8	23.9	99.6	2.1
1-2	22.0	3.4	0.5	0.6	26.5	30.2	87.7	2.7
3-4	23.7	2.9	0.3	0.6	27.5	38.2	72.0	7.0
5-6	23.2	3.1	0.2	0.4	26.9	38.8	69.3	7.0
7-8	22.6	3.7	0.2	0.4	26.9	39.9	67.4	7.0
9-10	21.1	2.7	0.1	0.3	24.2	36.8	65.8	6.4
11-12	13.4	1.7	0.1	0.2	15.4	28.2	54.6	5.2
13-14**	10.1	1.2	0.1	0.2	11.6	18.1	64.1	3.8
15-16**	8.8	1.0	0.1	0.2	10.1	15.4	65.6	3.4

* Organic Carbon.
** Sample was mostly sand.

Means, standard deviations, ranges, and 95 and 99 percent con-
fidence limits were calculated for all chemical constituents of samples
from the two control wells (Table 23). No significant differences
occurred between the two controls.

Monthly water samples were collected from October, 1973 to July,
1974, from 9 observation wells--13 and 13X in the fill area, and wells
1, 2, 18, 19, 20, 21, and 34 around the perimeter of the dike. Nine
sets of samples were collected, and 12 chemical items were analyzed.
Concentrations in observation wells were compared to control values
for well 32X (Table 23), and the frequency of significant differences
was tabulated (Table 24).

TABLE 23. Variability of Water Chemistry for Control Well 32X, Columbus Landfill*

Parameter	Mean (ppm)	Std. Dev. (ppm)	Range (ppm)	Confidence Limit 95% (ppm)	99% (ppm)
Potassium	2.9	0.6	2.1 - 3.9	4.0	4.5
Calcium	96.4	16.3	65.3 - 111.8	125.9	139.3
Magnesium	30.0	4.6	24.7 - 43.2	38.3	42.1
Sodium	13.0	2.9	10.0 - 18.6	18.3	20.6
Iron	0.1	0.4	0.0 - 1.3	0.8	1.2
B.A.**	336.5	35.3	282.0 - 372.4	400.5	429.5
Chloride	23.3	1.0	22.2 - 25.2	25.2	25.9
Phosphate	0.0	0.1	0.0 - 0.2	0.2	0.3
Sulfate	48.6	5.5	41.7 - 57.8	58.6	63.1
Nitrate	52.3	41.5	19.3 - 148.4	127.5	161.6
TDS	400.4	42.5	340 - 440	477.5	512.3
Hardness	361.0	43.4	276.6 - 412.9	439.7	475.3

 * Twelve samples were analyzed.
** Bicarbonate Alkalinity.

Within 108 analyses chemical content of leachate from well 13 differed significantly from control values 90 times at the 99 percent level and 94 times at the 95 percent level (Table 24). Potassium, calcium, magnesium, sodium, alkalinity, chloride, TDS, and hardness differed at the 99 percent level from control values on 9 sampling dates. Calcium, magnesium, phosphate, sulfate, and nitrate were about 5 times greater than control levels, while TDS and hardness were about 10 to 20 times greater than background. Sodium, alkalinity, and chloride usually were 100 times greater than control levels.

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Figure 15. CHANGES IN SOIL CHEMISTRY WITH PASSAGE OF LEACHATE

A. ORGANIC CARBON
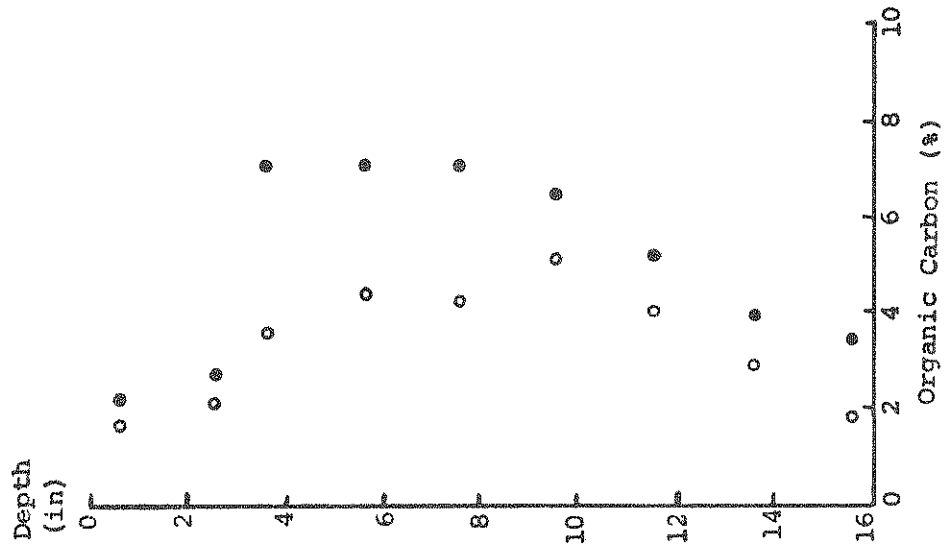
B. BASE SATURATION
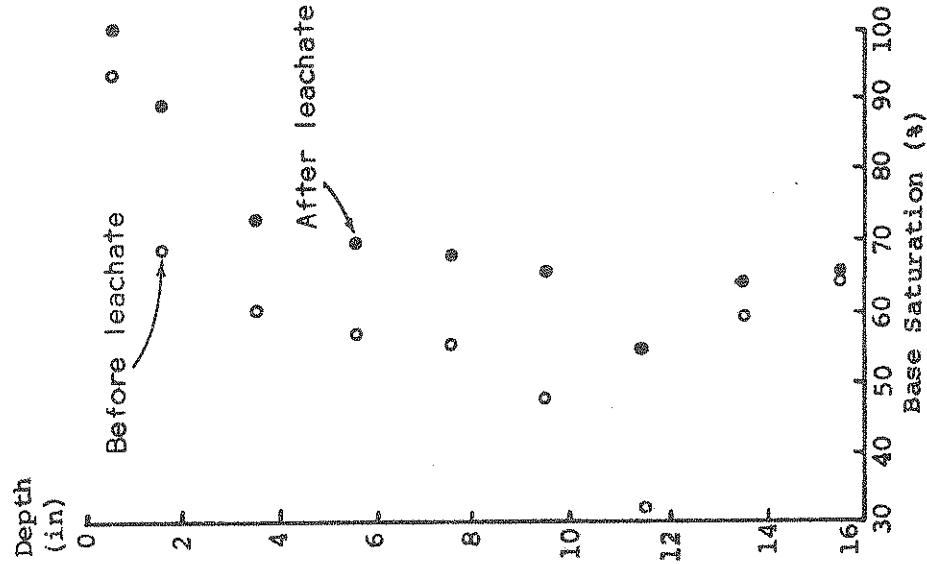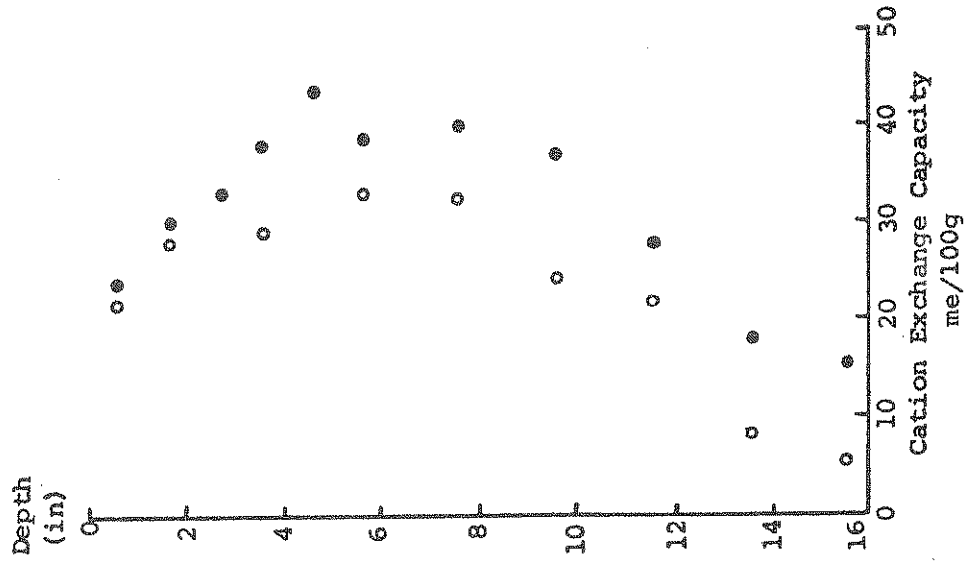
C. CATION EXCHANGE CAPACITY

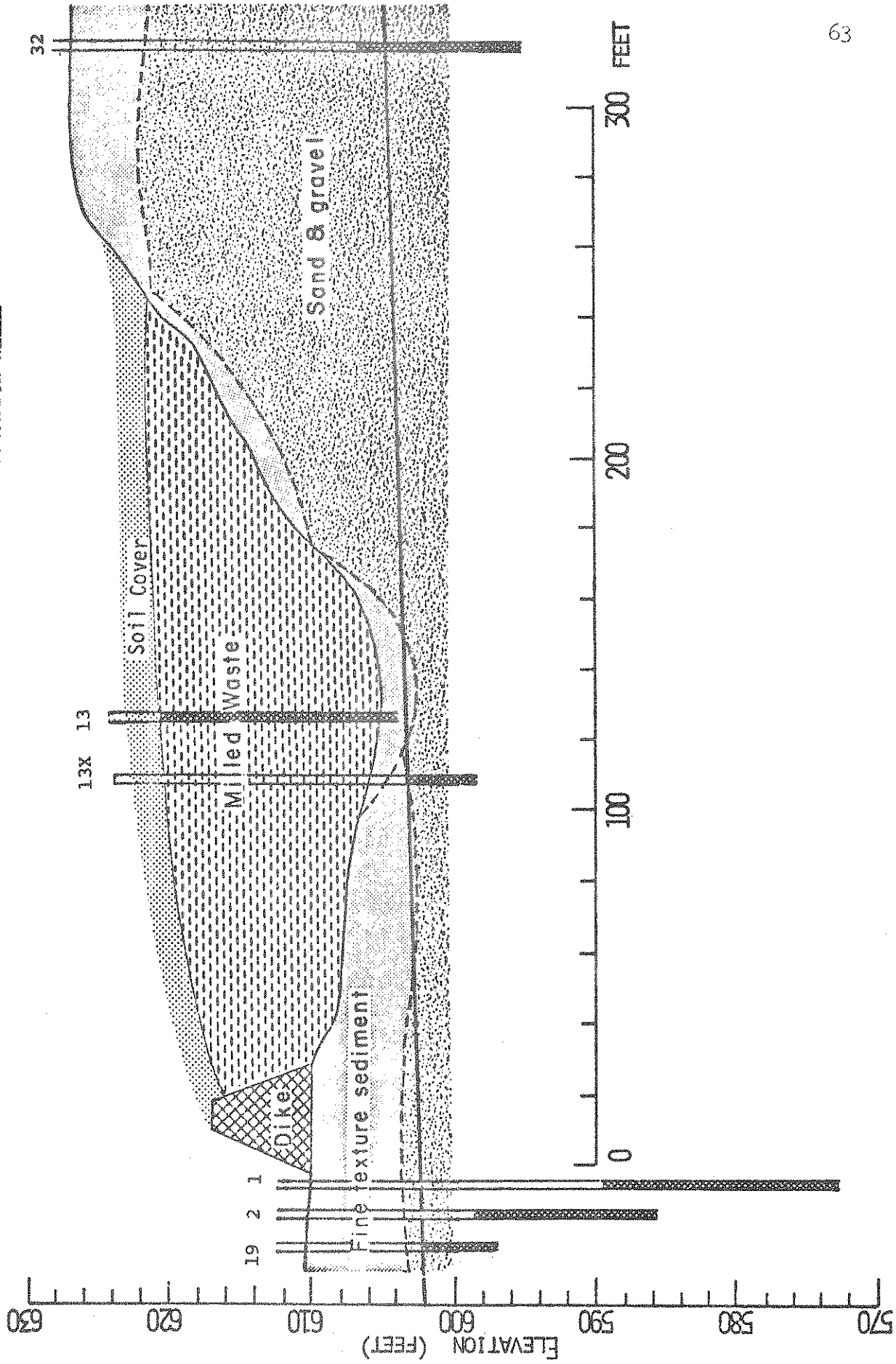Figure 16. CROSS-SECTION OF COMPLETED FILL WITH OBSERVATION WELLS

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TABLE 24. Frequency of Significant Differences in Ground Water Compared to Control Well 32X*

Parameters	Well 13 95%	99%	Well 13X 95%	99%	Well 18 95%	99%	Well 19 95%	99%	Well 2 95%	99%	Well 1 95%	99%	Well 20 95%	99%	Well 21 95%	99%	Well 34 95%	99%
Potassium	0	9	0	9	2	4	0	1	1	3	1	0	0	0	0	0	0	0
Calcium	0	9	0	1	0	8	3	0	1	1	0	0	0	0	0	0	0	0
Magnesium	0	9	0	9	0	8	2	3	0	3	0	0	0	0	0	0	1	0
Sodium	0	9	0	9	0	8	0	5	0	8	0	1	0	0	0	0	1	1
Iron	4	4	1	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0
B.A.**	0	9	0	9	0	9	2	6	0	6	0	0	0	0	0	0	0	0
Chloride	0	9	0	9	0	9	0	9	0	9	0	0	0	0	0	0	0	0
Phosphate	0	6	1	0	1	0	1	0	1	0	0	1	0	0	0	0	1	1
Sulfate	0	6	0	1	1	6	0	1	1	1	0	1	0	0	0	0	1	2
Nitrate	0	2	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
TDS	0	9	0	9	0	9	3	4	2	4	0	0	0	0	0	0	0	0
Hardness	0	9	3	6	0	6	2	2	0	1	0	0	0	0	0	0	0	0

*Nine samples were analyzed.
**Bicarbonate Alkalinity.

In the ground water beneath the landfill (well 13X) concentrations were significantly higher than background but were considerably lower than in leachate. Values were beyond control limits 63 times at the 99 percent level and 68 times at the 95 percent level (Table 24). Potassium, magnesium, sodium, alkalinity, chloride, and TDS were greater than control values at the 99 percent level on all 9 sampling dates. Iron (2), phosphate (2), and sulfate (1) were the fewest cases beyond confidence limits. With the exception of chloride which was 20 times greater than control levels in 2 cases, all other ions differed significantly from background but at less than 10 times greater than control.

Samples from well 18 on the south side of the dike differed significantly 68 times at the 99 percent level and 72 times at the 95 percent level. Alkalinity, chloride, and TDS differed at the 99 percent level on all 9 sampling dates while calcium, magnesium, and sodium differed from the 99 percent control limit on 8 sampling dates (Table 24).

Water chemistry in wells 18 and 13X were usually about the same, but well 18 had greater amounts of chloride on 4 dates. Sulfate was greater in well 18 on 9 of 12 dates (Table 25). Some other source may also be contaminating the water at well 18.

After the landfill operation ceased, the facilities were converted to a city maintenance garage. Vehicles are washed in the parking lot and salt, that is used on city streets, is stockpiled in the parking lot. Runoff from the parking lot flows along the south side of the dike, and part of the water ponds near site 18.

TABLE 25. Chloride and Sulfate Concentrations in Wells 13X and 18 (ppm)

Date	Chloride		Sulfate	
	Well 13X	Well 18	Well 13X	Well 18
10/10/73	157.9	226.2	12.7	87.9
12/1/73	570.5	311.2	15.6	21.3
1/25/74	239.9	595.8	12.5	36.4
2/25/74	484.9	505.0	4.9	68.7
3/27/74	204.4	57.8	16.6	105.6
4/26/74	217.8	231.1	6.2	67.9
5/29/74	284.5	206.5	12.7	61.6
6/20/74	260.0	164.5	112.8	74.7
6/26/74	248.8	219.0	11.1	15.0
7/2/74	297.6	162.7	15.3	73.1
7/26/74	666.7	217.8	126.9	85.0
8/15/74	277.6	185.6	283.3	83.0

Well 19 penetrates the upper 5 feet of the aquifer near the south-west corner of the site (Fig. 9, cf. Fig. 16) and is directly in line of ground-water flow from the terrace and landfill (Fig. 10). Samples from well 19 differed significantly from control values at the 99 percent level 31 times and 44 times at the 95 percent level. At the 99 percent level, chloride differed 9 times, alkalinity 6 times, sodium 5 times, and TDS 4 times (Table 24). All concentrations were less than 2 times above control levels. Some contamination is present in well 19.

To determine if leachate were in the ground water at greater depths than the upper 5 feet of the aquifer, well 2 and well 1 were installed to penetrate depths of 3 to 16 feet and 12 to 30 feet, respectively (Fig. 9, cf. Fig. 16). These wells are also in the direction of ground-water flow from the landfill (Fig. 10).

Samples from well 2 differed significantly from control values 36 times at the 99 percent level and 43 times at the 95 percent level (Table 24). At the 99 percent level, chloride differed 9 times, sodium 8 times, alkalinity 6 times, and TDS 4 times. Concentrations were generally greater than at shallower depth (well 19) and were usually about 3 or 4 times greater than control values.

Samples from well 1, the deepest zone of the aquifer, were usually within background limits. Among 108 analyses only 3 differed significantly from control values at the 99 percent level and 4 differed at the 95 percent level. Leachate from the landfill is contained in the upper 12 feet of the aquifer.

Samples from wells 20 and 21 on the west side of the dike (Fig. 9) did not differ from control values in 108 cases (Table 24). At well 34 in the abandoned stream channel near the northwest corner of the site, only 3 analyses differed at the 99 percent level and 5 analyses differed at the 95 percent level (Table 24). These differences were sulfate in 3 cases and sodium and phosphate in one case each. Leachate is not contaminating the ground water on the west side of the landfill.

Wells 5, 6, 7, and 8 were installed in the upper 12 feet of the aquifer on the south side of the landfill in the direction of flow to determine the distance of contamination from the landfill. These wells are 50, 100, 200, and 400 feet southwest of well 19, respectively, (Fig. 9). These wells align with wells 32X, 13X and 19. Five samples were collected from each of the 7 wells.

Analyses of control samples are listed with other samples from well 32X (Table 23). Analyses of samples taken from the 6 observation wells were compared to control values (Table 26).

TABLE 26. Frequency of Significant Differences--Directional Flow Traverse--Columbus Landfill*

Parameter	Well 13X 95%	99%	Well 19 95%	99%	Well 5 95%	99%	Well 6 95%	99%	Well 7 95%	99%	Well 8 95%	99%
Potassium	0	5	0	2	0	0	0	0	0	0	0	0
Calcium	0	0	1	0	0	0	0	0	0	0	0	0
Magnesium	0	5	1	3	0	0	0	0	0	0	0	0
Sodium	0	5	0	5	0	2	0	0	0	0	0	0
Iron	1	1	0	0	0	0	0	0	0	0	0	0
B.A.**	0	5	0	5	1	0	0	0	0	0	0	0
Chloride	0	5	0	5	1	4	0	0	0	0	0	0
Phosphate	0	1	0	0	0	0	0	0	0	0	0	0
Sulfate	0	3	0	0	0	0	0	0	0	0	0	0
Nitrate	0	0	0	0	0	0	0	0	0	0	0	0
TDS	0	5	0	5	0	0	0	0	0	0	0	0
Hardness	1	4	2	1	0	0	0	0	0	0	0	0

*Five samples were analyzed.
**Bicarbonate Alkalinity.

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Samples from well 13X, the water beneath the landfill, differed significantly from control values 41 times at the 95 percent level and 39 times at the 99 percent level (Table 26). Potassium, magnesium, sodium, bicarbonate alkalinity, chloride, TDS, and hardness differed on each of the 5 sampling dates. Most concentrations were 10 times greater than control levels.

Samples from shallow well 19 differed significantly from control values 30 times at the 95 percent level and 26 times at the 99 percent level. Chloride, sodium, bicarbonate alkalinity, and TDS differed on all 5 sampling dates. Concentrations were about 2 times greater than control values, so the level of contamination dropped considerably between wells 13X and 19.

Samples from well 5, 50 feet southwest of well 19, differed from control levels only 8 times at the 95 percent level and 6 times at the 99 percent level (Table 26). Chloride differed 5 times, sodium 2 times, and bicarbonate alkalinity one time. Concentrations were either or almost within control limits.

Wells 6, 7, and 8 at respective distances of 100, 200, and 400 feet from well 19, had no concentrations beyond confidence limits. Chloride concentration in well 6 was only slightly higher than control values in a few cases. There is little if any contamination in the ground water 100 feet from the landfill.

Summary of Analysis. (1) Ground water at the Columbus landfill is about 8 feet under the floodplain and less than 4 feet under the floor of the abandoned stream channel. Direction of flow is to the southwest and at a rate of about 2 ft/day. Annual fluctuation of the ground-water table is about 3 feet.

(2)   About 17.5 inches of precipitation infiltrate the landfill and percolate to the ground water each year. The sediments underneath the waste adsorb contaminants from the leachate, but because the sediments are thin, contamination appeared in the aquifer within one year after field capacity was reached in the waste.

(3)   Free water is standing in the bottom zone of the waste and has accumulated to a head of about $2\frac{1}{2}$ feet. Permeability rates of the fine-texture sediments that separate the waste from the ground water are about 0.004 ft/day (17.5 in/yr), but permeability is decreasing as organic complexes from the leachate are being adsorbed by the soil and are plugging the pore spaces in the soil. Consequently, elevation of the perched zone within the waste may be expected to rise in the future as percolation is partially blocked. However, contaminants continue to pass through the sediment to the ground water.

(4)   At present ground-water contamination is being detected south and southwest of the landfill. Contamination is confined to the upper 12 feet of the aquifer and decreases to approximate background levels within 100 feet from the landfill.

(5)   Landfill operations ceased after only 5 percent of the planned volume of waste had been buried. The contamination of ground water induced by this small volume is not severe. However, if the original planned volume, twenty times the present volume, had been buried, contamination would have been increased manyfold. Concentrations of contaminants would have increased in the ground water, and the areal distribution of contaminated aquifer would have been greater.

(6)   Landfill operations were halted not because of the possible contamination of ground water, but because of solely economic pressures. The operation was too expensive.

Critique and Recommendations

In the practice of solid waste disposal by land burial in southern Indiana, leachate from the waste can be expected to be introduced into ground and/or surface water. The level of contamination is dependent on an interrelation of (1) the original lay of the land, (2) the initial soil and sediment stratigraphy and the nature of the materials, (3) the original hydrologic system including ground and surface water, (4) the modification of the topography, stratigraphy including nature of materials, and hydrology induced by land burial, and (5) the nature and volume of waste and the method of land burial. Site selection for landfill must be based on these five criteria.

Topography is changed by land burial. Hill summits may be lowered during trenching and filling, and the land surface may be raised by area filling in adjoining valleys. At the Monroe County landfill, hill and valley terrain will be reshaped to essentially one hill. Similar but not as severe remodelling is underway at the Brown County landfill. At the Columbus landfill a sloping hill has been constructed on a previously nearly level floodplain. Reshaping the landscape changes the surface-water hydrologic system, which in turn, affects the original surface drainage at the down-slope periphery of the landfill.

The stratigraphy of the soils and sediments and the nature of materials are radically changed by land burial. In the trench method the soils and substrata sediments are excavated and replaced by waste and interlayered mixtures of soil and sediment materials. Yet, the natural soils and sediments remain in the walls and bottom

of the trench. The hydrologic nature of materials is changed during this reconstruction. Porosity and permeabilty are increased in the fill, and a great change in permeability is created at the sharp contact of the fill and the trench walls and bottom. These conditions now exist on the tops and nose slopes of hills at the Monroe County and Brown County landfills.

In the area-filling practice at these two landfills, small valleys between hills are filled with interlayered waste and soil up to the hilltops. Prior to burial these valleys contained drainageways that conducted surface runoff from the area. After filling by waste, the buried valleys are converted to subsurface collectors and drains for transport of leachate from the landfill. The effluent then joins the surface-drainage system at the margin of the landfill and contaminates surface water. This induced concentration of drainage is the current major problem at the Monroe County and Brown County landfills.

The foregoing example emphasizes the fact that the original hydrologic system of an area is altered by land-burial practices. In southern Indiana where average annual precipitation is more than 40 inches, almost half of the water is infiltration which will collect in the more porous waste. A "bathtub" effect is created on the hilltops and hillslopes as the waste ultimately becomes saturated in landfill trenches. In high rainfall areas, it is not uncommon for burial trenches to behave like tilted bathtubs--the trench fills with ground water, which then spills over the lower end creating surface seepage (Duguid, 1974).

Saturated trenches form a perched ground-water zone high on the remodelled landscape. This water must ultimately seep downward to the main water table causing it to rise, or the perched water may seep laterally to the filled valleys creating greater down-valley seepage through the area-practice filled waste. These effects should be anticipated at the Monroe County and Brown County landfills. As noted previously, seepage from the waste-buried valleys is occurring at these landfills.

Accumulation of free water in waste can also create a ground-water mound. At the Columbus landfill, the lower zone of the waste is saturated and the "created water table" is about 5 feet above the mean level of the natural water table. At times the waste water is perched, but with seasonal rise of the natural water, the two merge. Greater hydraulic head and gradients are created by the landfill. Any design of landfill practice and in particular, the containment of leachate on site, must take in consideration the altered hydrologic system.

As emphasized in this report and as a matter of common sense, the most important environmental aspect of disposal of buried waste by land burial is the possible deleterious effects of effluent on the quality of water. Any possible changes can only be determined by installation of ground-water wells and by surface-water stations where water samples can be collected and analyzed in the laboratory through time. Any changes in water chemistry will only be detected in a monitoring program.

If significant changes do occur in the quality of water, then the landfill practices should be modified to correct any cause for

concern. For example, at the Monroe County landfill effluent from the area-fill in waste-buried valleys is entering the surface-water drainage system, and leachate concentrations are detectable in those waters as they leave the landfill area. Will the concentrations increase beyond acceptable limits? Only continued monitoring will provide an answer to this question. Improvement of the system of leachate lagoons on site may be in order to correct the current problem and prevent any future compounding of the problem. A similar situation occurs at the Brown County landfill but to a lesser degree.

Based on our research during the period July 1, 1971 to November 30, 1974, recommendations for location design, and operation of sanitary landfills in southern Indiana are: (1) Upland sites should be used to avoid runon of water from higher areas. (2) All areas subject to flooding should be denied use as landfill sites. The Columbus landfill violates this rule as it is within the zone of the 100-year flood. (3) Sites should be used only where the highest elevation of the water table is many feet below the base of the solid waste. (4) The intervening material should preferably have high exchange capacity to permit maximum adsorption of contaminants from percolating leachate. These last two guidelines must be balanced. Although the sediments beneath the Columbus landfill have moderate exchange capacities and adsorption does occur, the depth to ground water is too shallow, so contaminants pass through the sediment into the water. (5) Adequate use should be made of leachate lagoons to control contamination of surface water. At the Monroe County landfill, chloride contents of raw-leachate seepage are 60 times greater than control levels, but

are reduced to 20 times at the overflow from a lagoon. A series of these ponds would further reduce contaminant concentrations that are discharged to the surface-water system. (6) A daily soil cover should be required at any landfill. At Columbus the refuse remained uncovered for many months. Direct infiltration from rainfall and snowmelt resulted in an open, decaying mass of waste, that among other things, was offensive to the eye and nose. (7) Rilling of the soil cover should be prevented. As rills cut into the buried waste, discharge of leachate is increased directly to the surface-water system. Rilling is common at the Monroe County and Columbus landfills. (8) Installation of ground-water wells and surface-water sampling stations should be required, and systematic water sampling and analysis should also be required. There is no other way to determine whether ground and surface waters will become contaminated above acceptable limits.

Although the Refuse Disposal Act of 1969, Indiana General Assembly and its implementation (Indiana State Board of Health, 1969) required only rudimentary inspection and monitoring of landfills, regulations have been changed as our research studies progressed. (Refuse Disposal Act, IC 1971, 19-2-1, as amended by Public Law 148, Acts of 1972; and the Environmental Management Act, IC 1971, 13-7.) Among other items, the recent Indiana Stream Pollution Control Board Regulation SPC 18, Solid Waste Management Permit Regulation of 1974 requires specific data on soils and geology, including physical, hydrologic, and ion-exchange properties of materials. It requires studies of ground water including monitoring wells and tests for water quality. Surface water must be controlled including runon,

runoff, and erosion. Restrictions are imposed on use of areas subject to the 100-year flood. Leachate control measures are required. Responsibility for implementation of the regulations is assigned to the owner and the person responsible for the quality of operation.

This report closes our study of landfills. At this time contamination of surface and ground water has occurred at the three landfills described in this report. Pollution, in the sense of exceeding limits of U. S. Public Health Service Drinking Water Standards, has not occurred at this time beyond the boundaries of the landfill areas. But, it must be emphasized that burial of waste has not been completed at the Monroe County and Brown County landfills. With increased volumes of waste planned for burial in future years, what will the effects be on the quality of surface and ground waters in those areas?

The Columbus landfill should not be a problem. Termination of that operation for economic reasons resulted in burial of only a small volume of waste. The large transmissivity of water through the aquifer should dilute any leachate from that waste within acceptable limits. If the original planned volume for disposal in that landfill had been done, continued study would be required.

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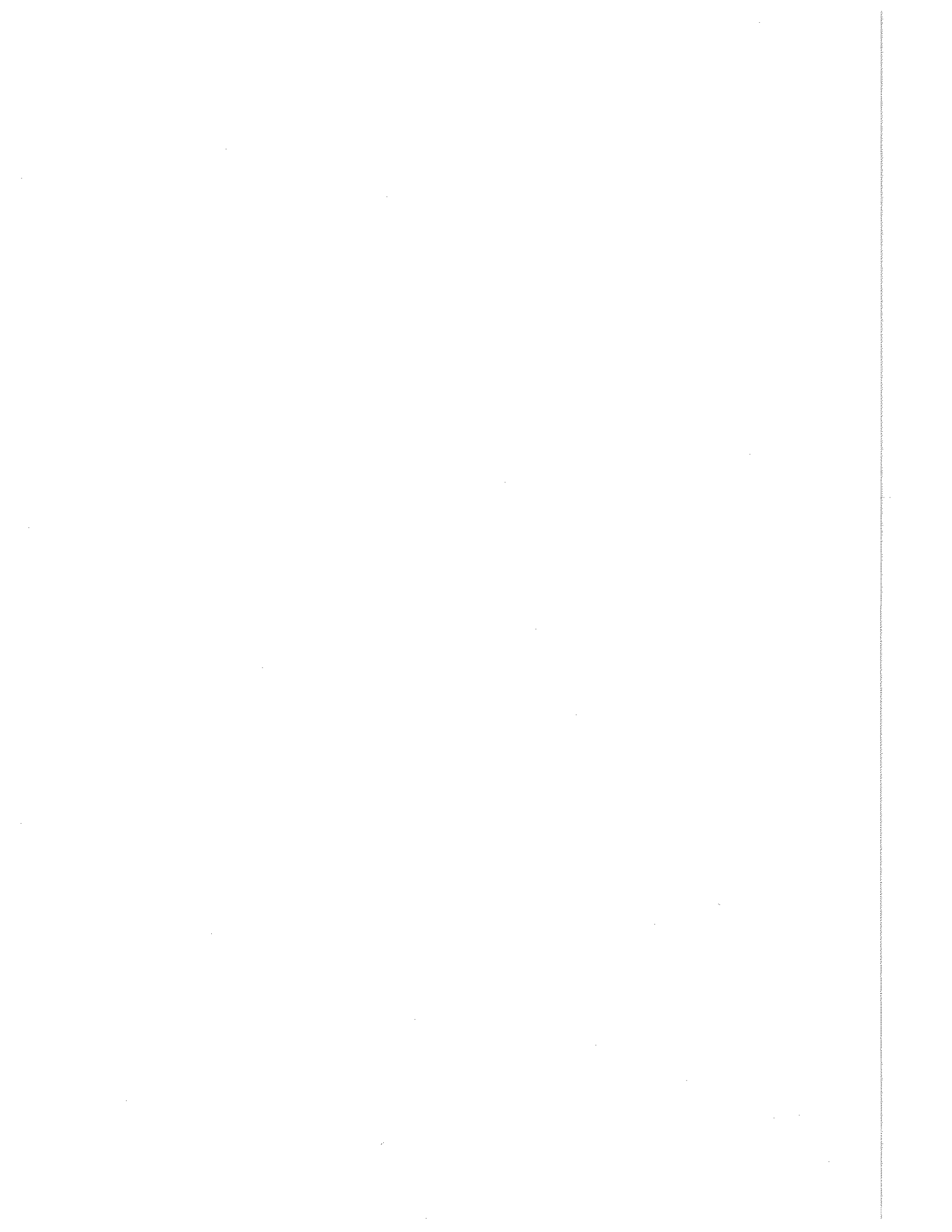
CARBON IN GROUND WATER AT THE COLUMBUS, INDIANA LANDFILL[*]

L. M. Games and J. M. Hayes

Introduction

Carbon is present in ground water in various forms and amounts, and sanitary landfills represent an important source of unnatural inputs of carbon compounds that can have an effect on ground-water quality. Little is known about the movement of carbon in normal ground water or the effects of a sanitary landfill on the presence of different molecular forms of the carbon, especially the inter- actions of a biological or physical nature that occur over a short time period. A study of a sanitary landfill and its effect on the carbon in ground water is a unique opportunity to gain information of this kind, since the presence of carbon in various compounds can be investigated before and after the influence of the landfill leachate.

Carbon appears in ground water in a large number of differing compounds so that the use of a single technique for the analysis of the total carbon is impractical. At the pH of most ground water, inorganic carbon is present as free CO_2 or as bicarbonate (Garrels and Christ, 1965). Inorganic carbon normally arises in ground water from essentially three sources, namely CO_2 from the atmosphere dis- solved in rainfall (a minor component), CO_2 from the microbial metabolism of organic compounds, and CO_2 from the dissolution of carbonate rock (Pearson and Hanshaw, 1970). The extent to which each of these contributes to the total in an individual ground- water sample is variable, depending on pH, P_{CO_2} in the soil, and

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[*] preliminary report

the concentration of inorganic ions which can interact with carbonate ions (Deines et al., 1974). Organic carbon is present in natural waters in many types of well-defined compounds, polar and non-polar, volatile and non-volatile (Croll, 1972). Less well-defined species, called humic and fulvic acids, are also present in varying amounts with a range of molecular weights (Gjessing, 1967). The few previous studies of the amount of dissolved organic carbon present in natural ground water indicate that the range is wide, the amount varying from 0.1 to 15 mg C/1 (Leenheer et al., 1974; Grigoropoulos and Smith, 1968; Robinson et al., 1967). Inorganic carbon has been analyzed in greater detail, both in measurement of ^{14}C isotope content for dating purposes (Hanshaw et al., 1965) and in studies of the hydrological cycle (Mook, 1967), and has an even greater range than the organic carbon.

The techniques described here have been developed to analyze the carbon present in discrete fractions (inorganic carbon, volatile organic carbon, non-volatile organic carbon, and CO and CH_4) and to use $^{13}C/^{12}C$ isotope ratio variations to detect changes in ground-water carbon resulting from the addition of landfill leachate. Measurements of the amount and isotope ratio of (a) ground-water carbon entering the landfill area, (b) leachate carbon existing within the landfill and presumably leaking into the ground-water system, and (c) carbon in ground water leaving the area provide the information needed to assess the extent to which simple mixing of the two reservoirs, "normal" and "leachate" carbon, takes place and dynamic interaction between the two occurs.

The focus of this study, which examines the Columbus, Indiana landfill, is to determine the effects of the landfill leachate carbon on the normal ground-water carbon by measuring the amount and isotope ratio of the individual carbon reservoirs and the mixture of the two. The study has been extended over a five-month period to measure the range of natural variation that occurs in these parameters in both pools. Not all of the possible biological and physical processes that mediate the relationships and exchanges of carbon between the various carbon compounds in ground water have been fully character- ized. For this reason, the procedure is designed to measure the carbon in four discrete fractions that account for the total carbon present and that are, to a certain extent, unique in the processes leading to their formation.

Use of Isotope Ratio Results

To measure the extent to which simple mixing or dynamic inter- action occurs, the origin of the carbon that is found in the "down- stream" ground water must be determined. Carbon isotope ratio variations have been used for some time by geochemists to distinguish various pools of carbon (Degens, 1969) and have been previously used to measure the proportion of normal to pollutional carbon present in the dissolved organic carbon of a marine-bay system (Calder and Parker, 1968). The technique is used here for the same reason with the additional purpose of measuring the extent of interaction between the various carbon pools.

Carbon isotope ratios are normally reported by geochemists in terms of $\delta^{13}C$:

$$\delta^{13}C = \frac{^{13}C/^{12}C \text{ sample} - ^{13}C/^{12}C \text{ std.}}{^{13}C/^{12}C \text{ std.}} \times 1000^*$$

Figure 1 shows the natural range of variations of $\delta^{13}C$ for various carbon pools. The differences are caused by kinetic and equilibrium isotope effects in both physical and biological processes involving carbon compounds.

The use of isotope ratio measurements as a tracing technique in this study is dependent upon the existence of a difference between the $\delta^{13}C$ values of the landfill leachate carbon and the normal ground-water carbon. If mixing of the two carbon pools occurs, ground water in the "downstream" wells will contain carbon with an intermediate $\delta^{13}C$ value. The simple mixing model can be formalized as shown in the equation in Figure 2 (Calder and Parker, 1968).

If the two pools of carbon, "normal" and "leachate," are mixed, two possibilities for the resultant $\delta^{13}C$ exist. If simple mixing occurs, the final $\delta^{13}C$ can be calculated for each of the four fractions of carbon by the equation in Figure 2. If interactions occur between the different fractions of the two pools or between the

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* The isotope ratio results are thus reported in terms of parts per thousand (o/oo) differences between the sample and a selected standard, in this report the PDB limestone(Craig, 1953). Samples enriched in ^{13}C relative to this standard have a positive $\delta^{13}C$ and samples depleted in ^{13}C have a negative $\delta^{13}C$.
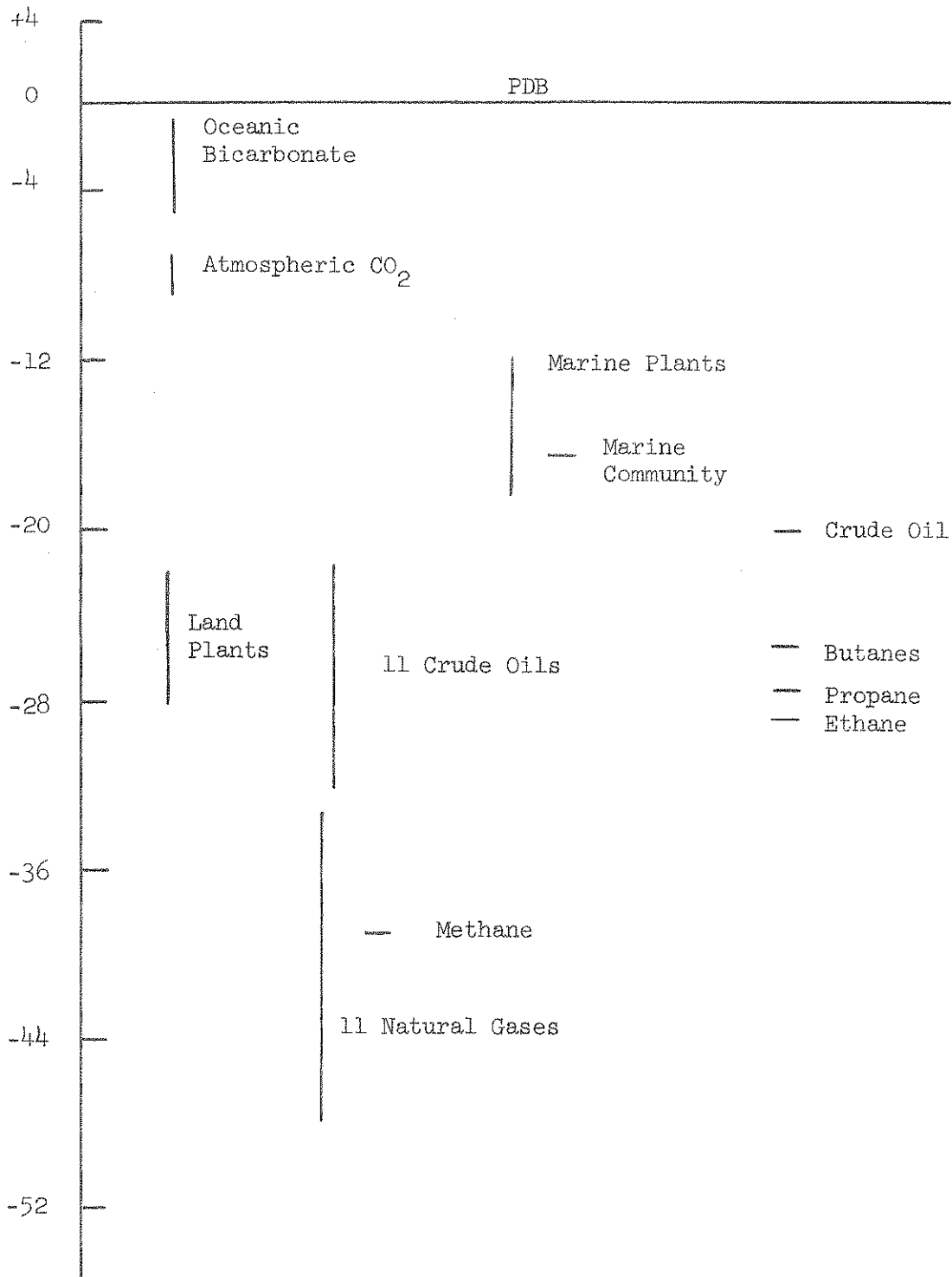
Figure 1. Data and figure taken from Calder and Parker (12).

Groundwater
In

Landfill
Leachate

Groundwater
Out

$$\delta^{13}C_I(C_I) \qquad + \qquad \delta^{13}C_L(C_L) \qquad = \qquad \delta^{13}C_O(C_O)$$
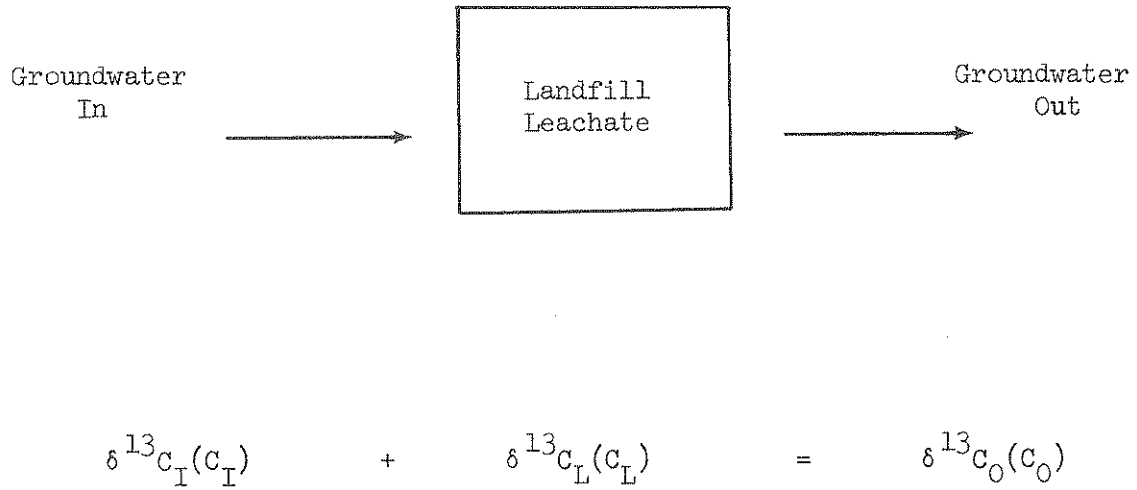
Figure 2. The $\delta^{13}C$ values refer to the isotope ratios of the carbon in the various pools and the C_X terms refer to the carbon concentrations.

mixture of the two and other species in the soil, the $\delta^{13}C$ of the resultant mixture will not be given by equation 2, and further study of the factors that might cause this effect will be necessary. By utilizing the equation, an estimate can be made of the extent to which the two processes (simple mixing and dynamic interaction) occur.

Experimental

Sampling. The ground-water samples for the analysis of non-volatile organic carbon are retrieved from the wells with glass bailers, placed in glass-stoppered containers, returned to the laboratory, filtered through previously cleaned Whatman GF/C glass fiber filter pads and 0.45μ Millipore filters, and stored at $4^\circ C$ until analysis. Since the other fractions include volatile compounds that might escape if suitable precautions are not taken, they are collected in sample vessels like the one shown in Figure 3. These vessels are evacuated and sealed in the laboratory, lowered into the wells, and opened by snapping the breakseal with a string. They are then raised from the wells and quickly closed. This method prevents loss of any volatile compounds and avoids mixing of the CO_2 present in the ground water with atmospheric CO_2. Filtration of these samples is avoided for the same reason, and they are stored as collected at $4^\circ C$. No samples are stored for more than one week before analysis.

Separation of Carbon into Specific Fractions. The carbon in the water is separated into four fractions on the basis of volatility and molecular type, and these fractions are:

Breakseal

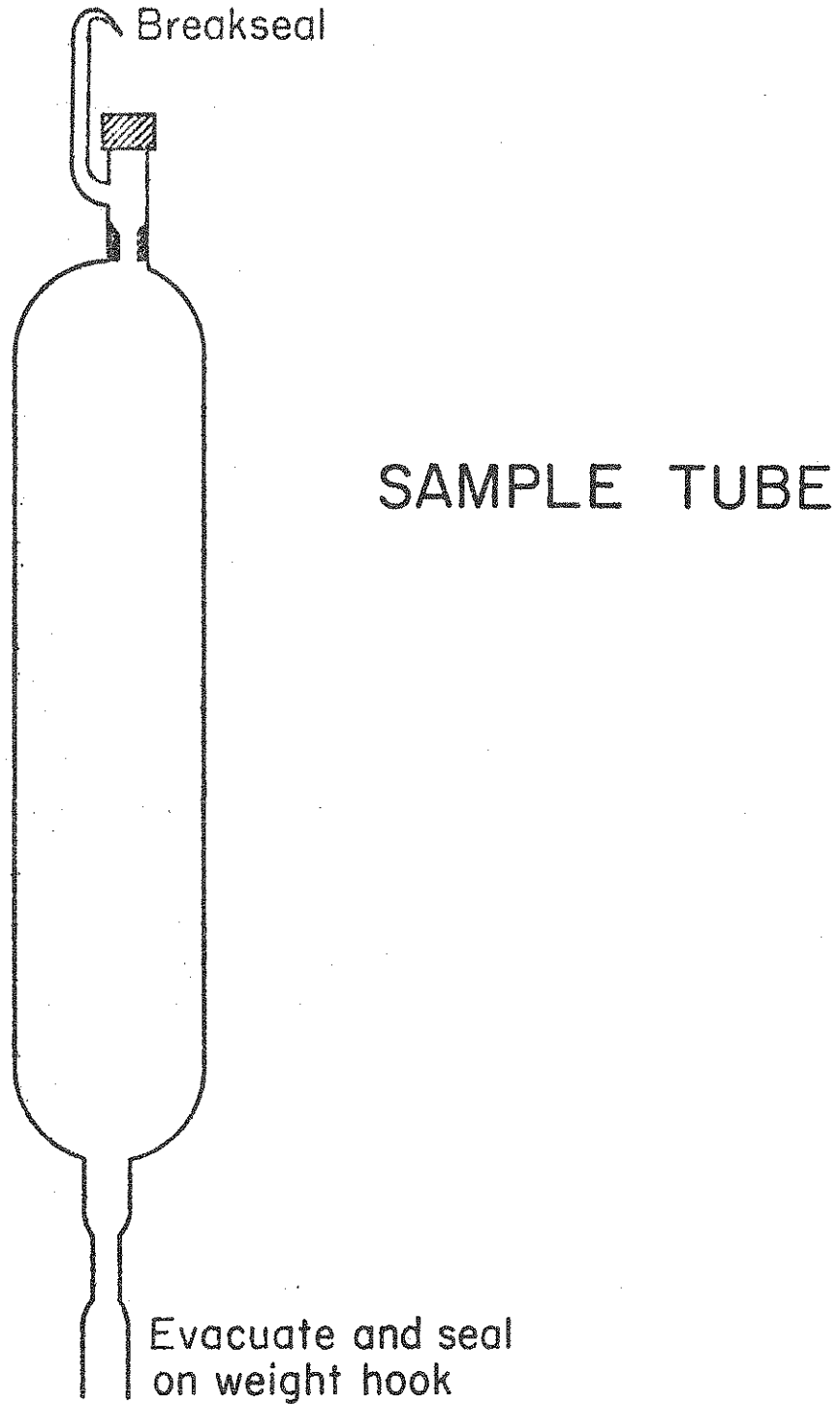SAMPLE TUBE

Evacuate and seal
on weight hook

Figure 3. Collection Vessel for samples containing
volatile carbon. After evacuation the tube is sealed
and a hook attached to hold a weight which pulls the
vessel under water.

(1) Inorganic Carbon--This fraction contains all free CO_2, bicarbonate, and carbonate. The ground-water samples are acidified to shift the equilibrium between these species to free CO_2, and cryogenic pumping is utilized to separate the CO_2 from the water. The method resembles that of Mook (1967).

(2) Volatile Organic Carbon--This fraction contains all organic material that can be stripped from the water at room temperature with a 10-15 ml/min stream of oxygen during a three-hour period. It includes volatile hydrocarbons and other carbon compounds with sufficient vapor pressure to escape the water at atmospheric pressure. The carbon dioxide is removed from the gas stream by a bed of soda asbestos prior to combustion of the volatile organic carbon and collection as CO_2.

(3) Non-volatile Organic Carbon--All organic carbon present except that removed as volatile organic carbon is obtained by photo-oxidation of oxygen-saturated water with a 1200 W mercury arc UV lamp. This converts the organic carbon to CO_2 so that it can be stripped from the water and measured. The technique is essentially that of Williams (Williams and Gordon, 1970) as developed by Armstrong et al. (1966) with a few modifications.

(4) Carbon Monoxide and Methane--In this analysis for specific components of the volatile organic fraction, the CO and CH_4 are separated from each other and the remainder of the volatile carbon by selective combustion and trapping techniques. The CO is oxidized to CO_2 by Schutzes reagent (Stevens and Krout, 1972; Smiley, 1949) and the methane combusted to CO_2 following the removal of the CO-derived CO_2 from the gas stream.

All of the various analyses are performed on one all-purpose vacuum line, and the amounts of material in each fraction are reported in ppm C (mg C/l) as CO_2. After conversion to CO_2 and measurement of the amounts present, the samples are analyzed by isotope ratio mass spectrometry.

Further details of the analytical procedure, including measurements of standards, will be published elsewhere (Games and Hayes, in preparation), but Table 1 shows the reproducibility of measurements on natural ground-water samples for the first 3 fractions and also shows the average blanks for the system.

TABLE 1. Analysis System Blanks and Reproducibility

Fraction	Average Blank[*]	Reproducibility of Measurement of Natural Samples
	ppm C	ppm C
Non-volatile Organic Carbon	0.1	$\pm$ 0.17
Volatile Organic Carbon	.017	$\pm$ .02
Inorganic Carbon	.005	$\pm$ 2.2

* System blank obtained by using water with all carbon removed.

Results

This section deals with the carbon content of ground water in the area of the Columbus, Indiana landfill. The well-number designations in this section of the report are different from Waldrip and Ruhe (this report), and the correlation is given in Table 2.

TABLE 2. Key to Well Designation

Well Number Used Here	Well Designation[*]
1	32
2	18
4	19
8	13
12	1
13	2

[*] Waldrip and Ruhe, this report.

The Columbus, Indiana landfill is located above a thin fine-texture sediment overlying a sand and gravel aquifer (Fig. 4). If leachate moves from the landfill into the sand and gravel layer, wells located down gradient in ground-water flow may reveal this movement by a rise in carbon content. In addition, the landfill is located only a few feet above the normal water table, and during times of high rainfall mixing of the leachate and ground water might be expected to occur. The landfill was filled with wastes between June, 1972 and June, 1973, then covered with 3 feet of topsoil; and the results reported here cover the period from January to June, 1974.

The water wells monitored around the landfill were chosen based on hydrologic information referred to earlier in this report. A well about 30 feet upstream in ground-water flow from the landfill was used as a control. A well drilled in the waste itself was used to sample the leachate, and 4 wells below the landfill were then sampled to monitor leachate transport. These 4 wells are all located approximately 10 feet away from the landfill at different depths as shown in Figure 4, and according to hydrologic studies are
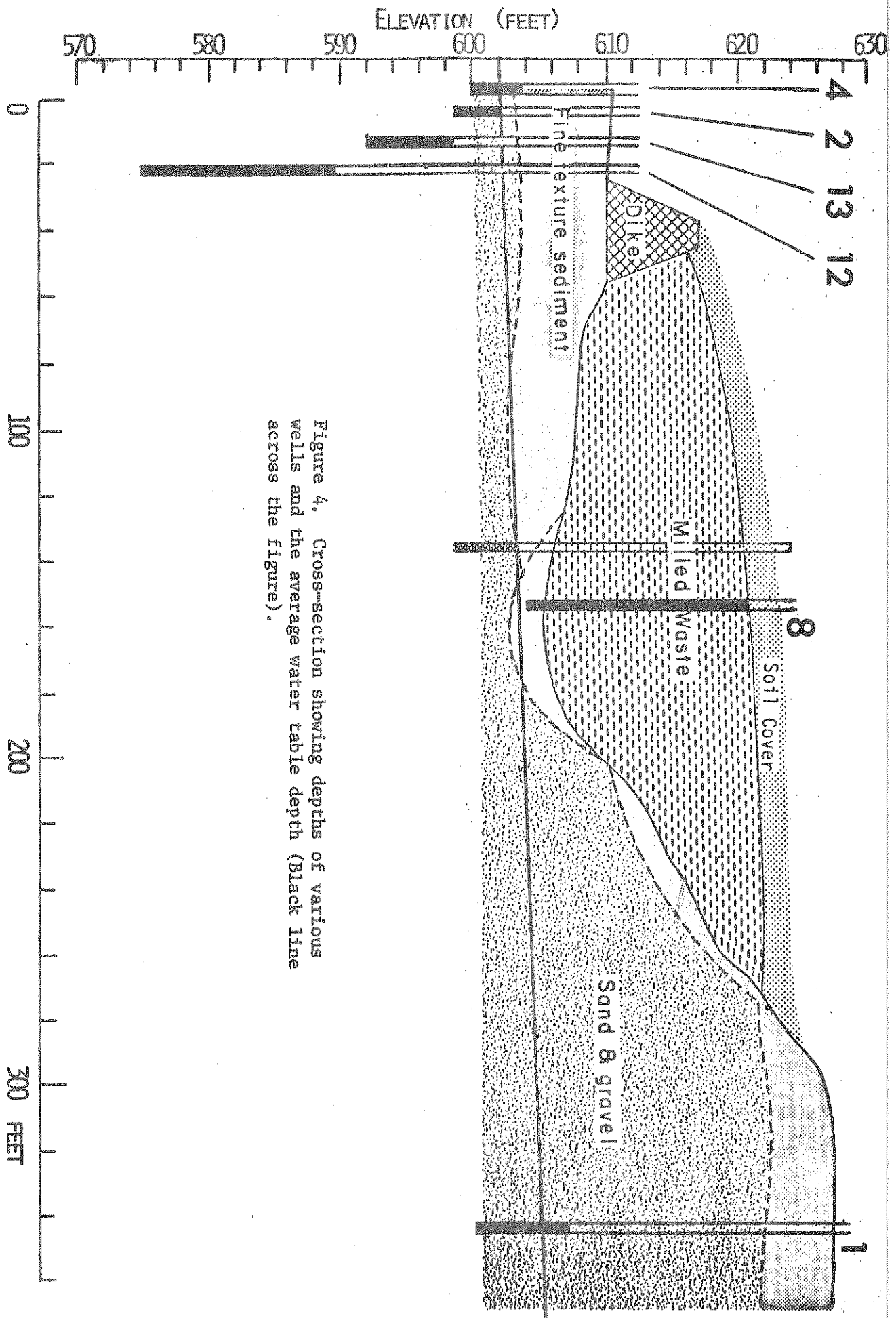
Figure 4. Cross-section showing depths of various wells and the average water table depth (Black line across the figure).

in the path of ground-water movement from the landfill. The 6 wells
were sampled once every 2 weeks in the 5-month period for volatile
organic and non-volatile organic carbon, but the inorganic carbon
was only sampled once every 6 weeks. The carbon monoxide and methane
analyses were started in June, 1974, and only limited data are avail-
able. However, the results obtained from the analysis of these frac-
tions, especially the methane fraction, have been extremely useful
in understanding the long-term results.

Table 3 shows the average amounts of carbon present in the
leachate, well 8, and the normal ground water, well 1, for the volatile
organic, non-volatile organic, and inorganic fractions. The dif-
ferences are quite large, as expected, and the detection of additions
of leachate to ground water appears to be relatively straightforward.
Any significant addition of leachate to the aquifer would result in
levels of carbon much higher than those present in the normal ground
water, even considering any variation in amounts that occurs over the
5-month time period.

TABLE 3. Average Amount of Carbon in Normal Ground Water and Landfill
Leachate During the Period Studied

Fraction	Normal Ground Water (ppm C/l)	Leachate (ppm C/l)
Non-volatile Organic Carbon	3.0	800
Volatile Organic Carbon	0.3	40
Inorganic Carbon	100	1600

Figure 5 contains the results of the isotope ratio analyses for the leachate and the control-well carbon. In order to make use of the isotope ratios as a means of tracing the leachate, a significant difference must exist between the isotope ratios of the carbon in the 2 wells (Fig. 2). The $\delta^{13}C$ values for the volatile and non-volatile organic carbon in the control well overlap and average about -27 o/oo, which is in the range expected for land plant-derived organic material (Fig. 1). The leachate volatile and non-volatile organic fractions have a much greater $\delta^{13}C$ variation during the time studied. Although there are large differences on specific dates, the average values do not differ significantly from the isotope ratios of the carbon in the natural ground water. The only difference appears to be that the volatile organic carbon in the leachate is consistently depleted in ^{13}C relative to the non-volatile organic carbon, while the two fractions overlap in the control-well water. This is probably due to the large percentage of isotopically light methane present in the leachate volatile organic fraction.

The isotope ratios of the inorganic carbon are extremely different in the leachate and control-well water. Although the inorganic carbon $\delta^{13}C$ of the control-well water is shown as non-variant over the 21-week period, the value used (-16 o/oo) is the average of a 10-month period, and the individual values are also given. The inorganic carbon of the leachate averages about +22 o/oo over the 10-month period, which is well beyond the range expected. Possible reasons for this result will be discussed later, but this fraction may be useful in determining what fraction of the carbon in downstream wells is a result of leachate transport out of the landfill.
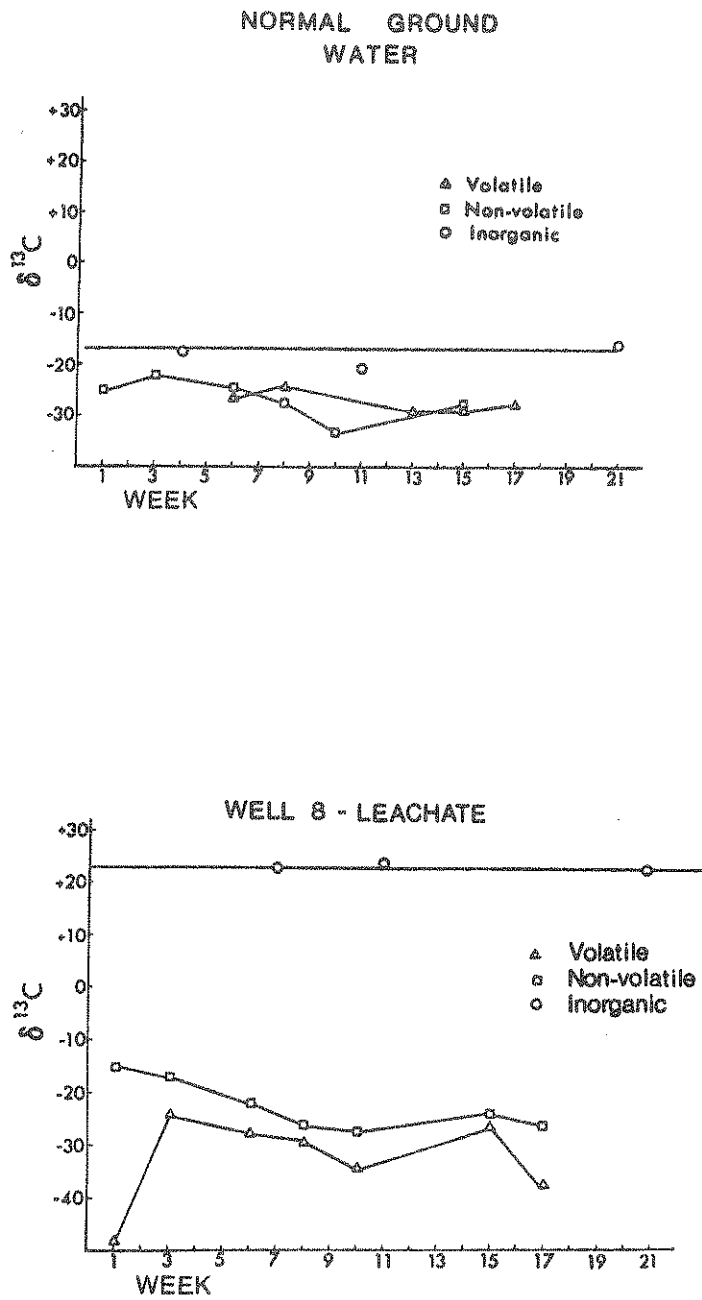
NORMAL GROUND
WATER

WELL 8 - LEACHATE

Figure 5. The time scale begins Jan. 30, 1974 and
extends to June 17, 1974.

96

Figure 6 shows the amounts of carbon present in the volatile and non-volatile organic fractions of the 4 downstream wells (2, 4, 12, and 13). The leachate and the control-well results are included for comparison. Figure 6 indicates that wells 2 and 13 consistently contain volatile organic material in excess of that present in well 1. The amount of volatile carbon in the control well is not a minimum amount but only a general guideline for the amount of carbon present in the normal ground water in the area. Figure 6 shows that the control well also functions in the same capacity for the non-volatile carbon, since some of the wells occasionally have carbon present in even lower amounts than the control well. The non-volatile organic fraction is less definitive than the volatile organic fraction, and none of the downstream wells contain significantly more non-volatile carbon compounds. Some factor is causing increased amounts of volatile organic material to appear in wells 2 and 13, but is not affecting the non-volatile organic content to any significant extent. Table 4 contains the results of inorganic carbon analyses prior to and during the same 5-month period that the other fractions were monitored. Again, well 2 contains carbon in excess of that in the control well, but in this case well 13 is definitely not affected.

The results for the 3 main fractions of carbon indicate that the landfill causes a change in the carbon content of the ground water in a selective manner. Well 2 has about twice as much volatile organic and inorganic carbon as the control well, but the non-volatile organic fraction is not increased. In well 13, only the volatile organic carbon content is increased. These facts indicate that some
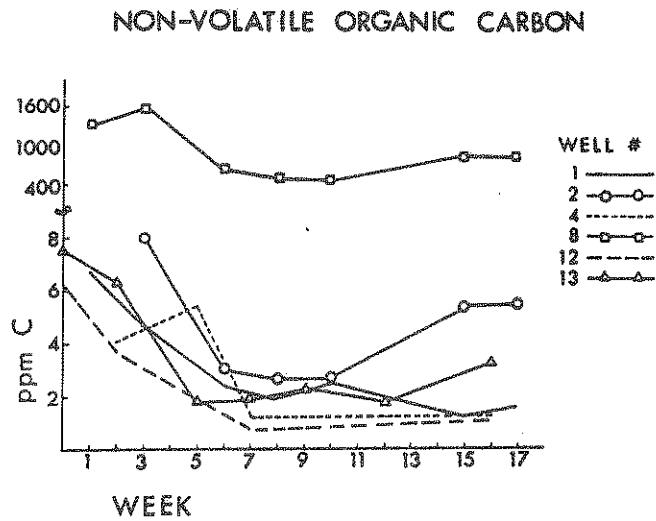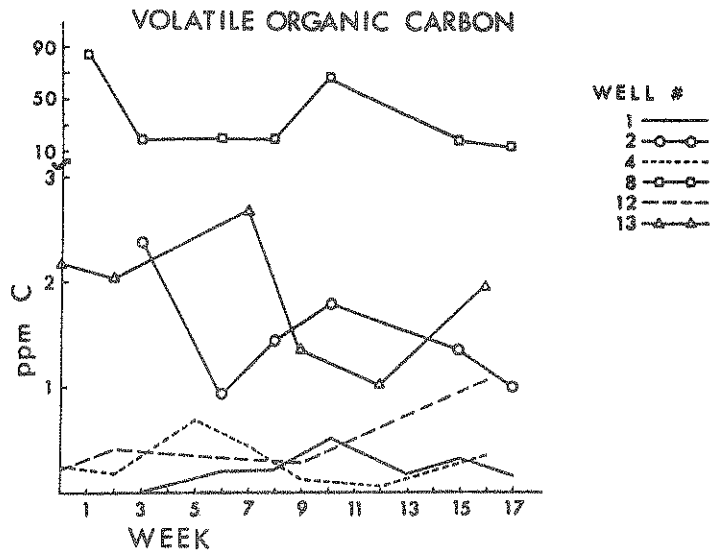
Figure 6.   The time scale begins Jan. 30, 1974.
The results for all six wells are included.

other factor may cause increases of carbon in these wells in a selective manner, and that leachate may not be transported from the landfill in the straightforward manner expected. The non-volatile organic carbon, being highly ionic, may be adsorbed in the soil before reaching the downstream wells.

TABLE 4. Inorganic Carbon (ppm C/l)

Date	Well 1	Well 2	Well 4	Well 8	Well 12	Well 13
8/13/74	104.2		106.6	1640		
8/28/73	91.1	229.2				
10/23/73					77.7	74.0
2/25/74	113.3	201.2			89.4	94.2
3/25/74				1615		
4/15/74	116.9	171.9	94.1	1528	85.4	107.9
6/17/74	104.6			1742	81.2	109.2

To ascertain whether the various carbon fractions reflect movement of the leachate from the landfill, the chloride contents of each well during the same 5-month period were compared to the carbon contents. Figure 7 contains the results of the chloride analyses reported earlier. Chloride concentrations have been considered in the past to be a fairly good indicator of the movement of leachate in ground-water systems, because chloride is relatively unaffected by exchange reactions with soil (Apgar and Langmuir, 1971; Zanoni, 1972). Well 2 is definitely increased in chloride content, and well 13 is slightly affected. Only in well 2 is there a significant increase in
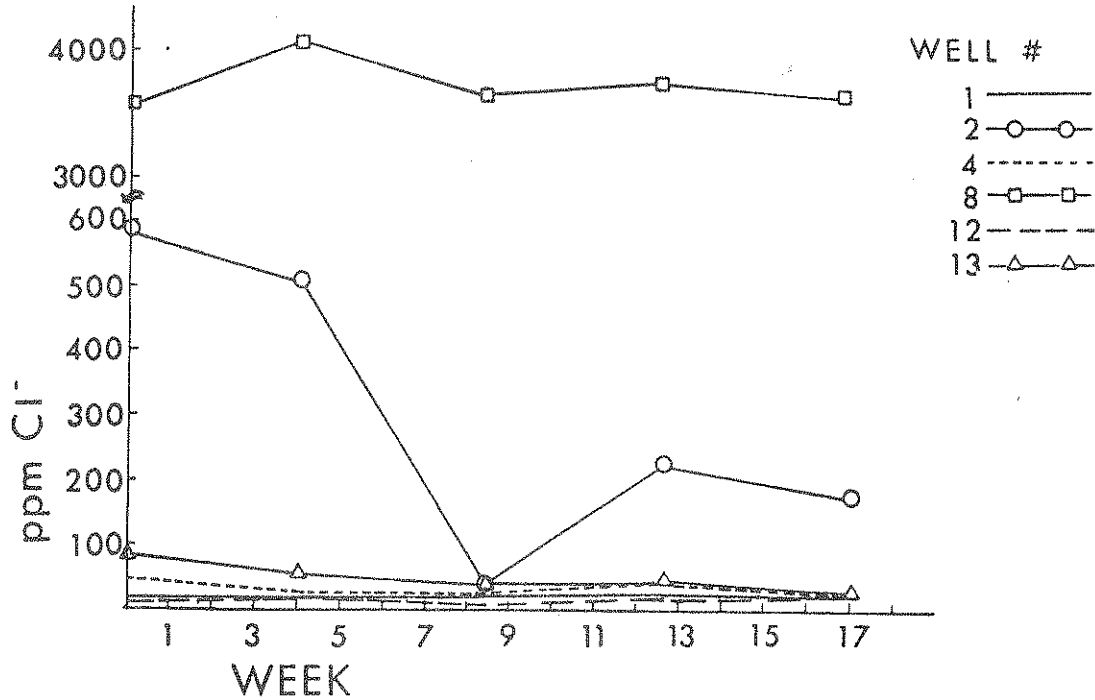
Figure 7. These are chloride contents, from the previous
section of this report, of the six wells monitored for
carbon content. The same time period, beginning Jan. 30,
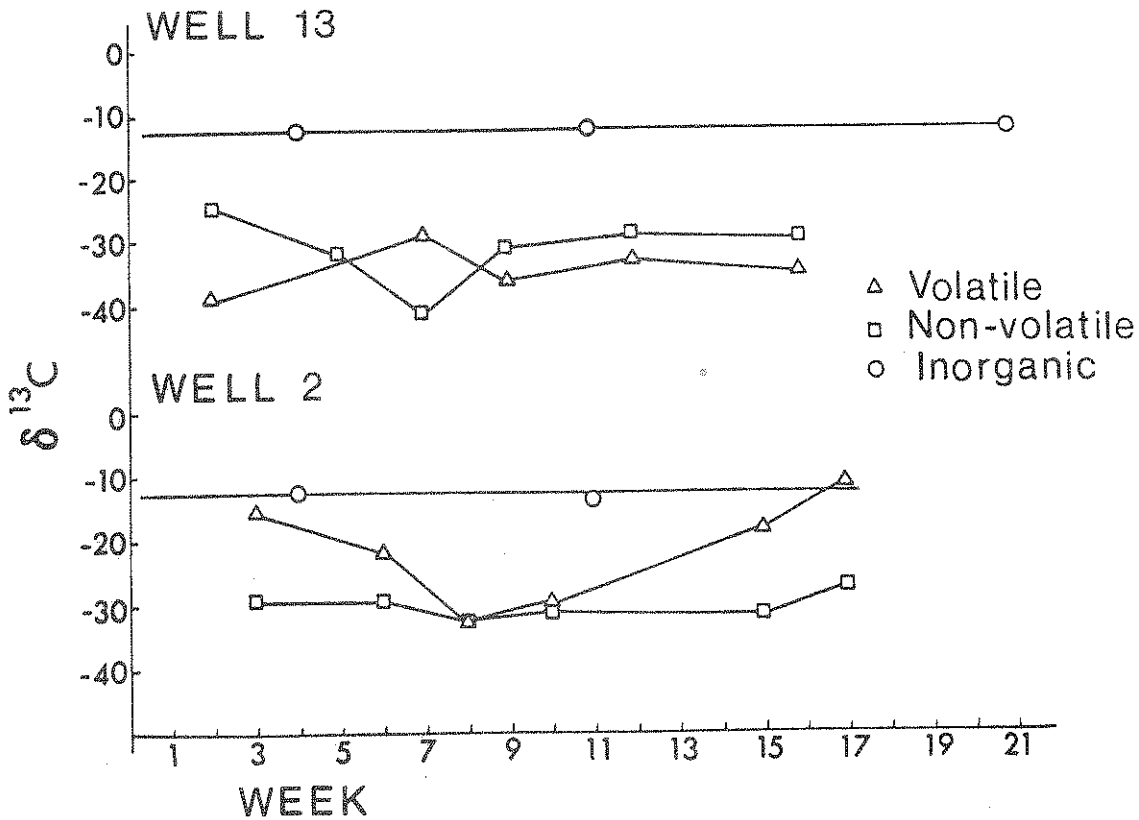1974, is shown here.

Figure 8. The isotope ratio results for the samples from
wells 2 and 13 for the period beginning Jan. 30, 1974.

inorganic carbon and an unambiguous increase in chloride content. Sampling has not been frequent enough to establish any definite relationship between the carbon and the chloride contents. More frequent and lengthy sampling would probably show even more variation with time than the results presented here.

The $\delta^{13}C$ results for the volatile organic and non-volatile organic fractions of the two wells (2 and 13), possibly contaminated on the basis of the amount of carbon present, are shown in Figure 8. The non-volatile carbon in both wells is more depleted in ^{13}C than the control or leachate water. The volatile organic and non-volatile organic carbon in well 13 overlap in the same manner as the control well, but the volatile organic carbon in well 2 is always isotopically heavier than both the non-volatile carbon in that well and all the organic material in the control and leachate wells. The relationship of this isotopically heavy carbon to the landfill is difficult to understand, since the leachate volatile carbon is isotopically light. Further confirmation of this result is found in the methane fraction, as will be discussed later.

Table 5 lists the inorganic carbon $\delta^{13}C$ results for all of the wells. All 4 downstream wells have $\delta^{13}C$ values intermediate between the control and leachate water. Well 2, expected to be isotopically heavy because of its higher inorganic content which presumably comes from the leachate, is not significantly different from the other downstream wells.

TABLE 5. Inorganic Carbon - $\delta^{13}C_{PDB}$ (o/oo)

Date	Well 1	Well 2	Well 4	Well 8	Well 12	Well 13
8/13/73	-14.7		-11.2	+14.0		
8/28/73	-14.9	-12.2				
10/23/73						-11.3
2/25/74	-17.5	-12.2			-10.0	-12.6
3/25/74				+22.8		
4/15/74	-20.8	-13.5	-15.8	+23.1	-10.8	-12.5
6/17/74	-15.8			+22.3	-10.9	-12.9

Careful study of Table 5 indicates that well 1, the control well, has an average $\delta^{13}C$ that is about 4 o/oo lower than other wells which also appear to be unaffected by the inorganic carbon from the landfill leachate (wells 4, 12, and 13). Since all wells are in the same aquifer, this lower value suggests that some secondary contamination of the control well may be occurring. The possibility also exists that all of the downstream wells have some leachate contamination, but a measurement of the ^{13}C of the inorganic carbon in another control well in the same area gave -11.7 o/oo, suggesting that the original control well is unique in some respects as yet undetermined. The amount of inorganic carbon in this second control well, however, is essentially the same as that of the primary control well.

The equation in Figure 2 can be used to calculate the expected isotope ratio of the inorganic carbon in well 2. Well 2 water

contains 200 mg/l inorganic carbon, which is approximately half leachate
and half ground water derived if we assume a simple mixing of control-
well water with the leachate. Assuming a mixture of 100 mg/l C with
a $\delta^{13}C$ of -11.7 o/oo and 100 mg C/l with a $\delta^{13}C$ of +20 o/oo, the $\delta^{13}C$
of well 2 would be +4.2 o/oo. Apparently the increase in amount of
inorganic carbon is not a direct result of leachate contamination,
since the expected positive shift in $\delta^{13}C$ does not occur. The actual
result for well 2, from Table 5, is -10 o/oo, and either some other
process more complicated than simple mixing is taking place or the
increase of inorganic carbon is caused by a secondary source of
contamination.

The analysis for CO and CH_4 was instituted to explain (a) the
isotopically heavy volatile organic carbon in well 2, (b) the iso-
topically heavy inorganic carbon in the leachate, (c) the lack of a
change in $\delta^{13}C$ of the inorganic carbon in well 2, and (d) the appar-
ently selective effect of the leachate in well 13. The isotopically
heavy inorganic carbon in the landfill leachate can only be explained
by requiring that some other fraction of carbon is becoming extremely
light. Previous research has shown that in anaerobic environments
the methane-producing bacteria present make methane depleted in ^{13}C,
while the CO_2 evolved in these biologically complicated environments
is enriched in ^{13}C compared to metabolic CO_2 resulting from aerobic
bacterial decomposition (Oana and Deevey, 1960). These bacteria
obviously function in landfills (Apgar and Langmuir, 1971) and flooded
soils (Bell, 1969). Carbon monoxide has also been observed being
emitted from decomposing plant material (Wilks, 1959; Loewus and
Delwiche, 1963; Stevens, 1972), and a search for it was also

instituted. The carbon monoxide from plant sources is usually assumed
to come from chlorophyll degradation (Loewus and Delwiche, 1963), and
its isotope ratio has been measured as about -25 o/oo (Stevens, 1972).
However, the presence of CO in ground water has never been reported.
The isotopically heavy volatile organic material in well 2 seems to
contradict expectations, and its methane $\delta^{13}C$ was measured to deter-
mine if the methane or other volatile carbon caused its high $\delta^{13}C$.

The results of the analyses for carbon monoxide and methane are
shown in Table 6. Wells 1, 4, and 12 contained no methane or carbon
monoxide and are not included in the table. The small amount of CO
found in the leachate has yet to be substantiated by other means, and
its $\delta^{13}C$ is not in the expected range (Stevens, 1972). The methane
of the leachate is distinctly depleted in ^{13}C, about -48 o/oo. There
is a 70 o/oo difference between the methane and carbon dioxide of
the landfill leachate (CO_2 = +20 o/oo and CH_4 = -48 o/oo), about
the same difference found by Oana and Deevey (1960) in the mud gases
from some New England lakes (CO_2 = 0 o/oo and CH_4 = -75 o/oo). They
found that the organic substrate in the lake mud had a $\delta^{13}C$ of
-30 o/oo, and a corresponding $\delta^{13}C$ for the organic matter leading
to the isotope composition reported here would be -5 o/oo, assuming
that the same type of bacterial ecology is present in both cases.
Combusted samples of the landfill waste have an average $\delta^{13}C$ of
-23.9 o/oo so that the food-chain system in the landfill environment
is apparently different in some way from the lake-mud environment.
Further work to elucidate the causes of the heavy carbon in the
leachate inorganic carbon is being conducted. This includes measurement

of the $\delta^{13}C$ of methane and CO_2 produced by other sanitary landfills and by a sewage digester.

TABLE 6. CO and CH_4 Fraction

Sample (date)	ppm C		$\delta^{13}C_{PDB}$ (o/oo)	
	CO	CH_4	CO	CH_4
Well 2				
6/17/74	<0.04	.21	--	- 3.3
7/22/74 A	< .04	.11	--	- 2.9
7/22/74 B	< .04	.15	--	- 3.2
Well 8				
6/17/74	0.18	5.71	-15.2	-49.4
6/24/74	.09	3.98	-11.3	-47.8
Well 13				
6/17/74	< .04	.80	--	-37.7
Average Blank	.002	.017	-33.6	-40.9

The $\delta^{13}C$ of the well 2 methane fraction is isotopically the heaviest natural methane ever reported. Volcanic and metamorphic methane with $\delta^{13}C$ values as high as -10 o/oo have been reported (Dubrova and Nesmelova, 1968), but these possibilities appear untenable in this case. This $\delta^{13}C$, -3 o/oo, accounts for the isotopically heavy volatile organic fraction discussed earlier and shown in Figure 8. Because of the unusual nature of this result, a further confirmation was sought. The method of analysis for methane used to obtain these data did not specifically exclude a very slight contamination by

ethane or ethylene. A somewhat more complicated system of measure-
ment, modeled after work by Swinnerton and Linnenbom (1967) was
instituted and the analyses repeated. Since the results are for
a later date, October 15, 1974, they do not strictly correlate
with the data in Table 6. However, the same amounts of methane
were found and although the methane was not quite as isotopically
heavy, $\delta^{13}C$ = -9.7 o/oo and -10.1 o/oo for duplicate determinations,
the results indicate that the methane is indeed outside the range
of any previously reported data.

Well 2 is thus unusual in two respects: (a) the high inorganic
carbon content without a change in isotope ratio of the carbon, and
(b) the isotopically heavy methane. These observations suggest that
well 2 is contaminated by a secondary source, but the isotope ratio
results indicate that if the secondary source exists, it must be
unusual in some respects.

Any explanation of this finding must await further study, but
various possibilities are currently being explored. First, the
possibility exists that the methane-producing bacteria in the area
of this well are biologically different than those observed in
natural situations previously (Oana and Deevey, 1960; Rosenfeld and
Silverman, 1959). Very little work has been done with these bac-
teria in pure culture, and only one measurement of the isotope
ratio of the methane produced by a pure culture has been made (pri-
vate communication from R. S. Wolfe). The second possibility
is that the organic substrate in this area is being depleted with
no replacement, and that the remaining organic material has already

been substantially depleted in ^{12}C by bacterial action. Silverman and Rosenfeld (1959) observed this effect in a culture of bacteria growing on a limiting amount of substrate. This second possibility, if true, suggests that the landfill and well 2 were once connected but are not at present interacting in any way. The particulate material in this well is currently being investigated, but no data are presently available. The last possibility is that a large fractionation of the methane is occurring between the landfill and the area of well 2, but this hypothesis is much more difficult to test.

The methane from well 13 is in the range expected for contamination from the landfill, but this well does not show significant increase in any of the other carbon pools. If the water flowing through well 13 does contain landfill leachate, some explanation of the lack of increase in inorganic carbon content and of the lack of a change in its δ^{13}C must be made. The well does have only a slight increase in chloride content, so that the possibility does exist that the leachate contribution is too small to be detected in a rise in the concentration of any fraction or change in the isotope ratio of any fraction that is already present in the unpolluted ground water. A second suggestion might be that the non-volatile organic carbon from the leachate, probably highly ionic, is being adsorbed by the soil, and that the inorganic carbon content and isotope ratios are controlled by factors outweighing the landfill effects where little transport of contaminants occurs.

Conclusions

Although this study is not yet completed, the findings indicate
the need to study more than one type of carbon compound when assess-
ing the chemical effects of unnatural inputs of carbon on a ground-
water system. The large variation of the amounts and isotope ratios
with time emphasizes that ground-water systems must be studied over
a long period to gain any insight into the interactions taking place
between normal and contaminant carbon.

The isotope ratio results are more complicated than expected,
and interpretation requires careful consideration and knowledge of
the hydrologic parameters and the bacterial population present.
The limitations of the data acquired to date are obvious in the
numerous possible explanations that can be offered, and an example
of the more extensive information that would be useful is found in
the 10-year study of Golwer et al. (1973) of a sanitary landfill
site in West Germany. The isotope ratio results are, however,
extremely useful in considering possible sources of any carbon
present, especially in this case for well 2. The data show that
the major contamination is not indicative of landfill leachate
transport and also limit the choice of alternative secondary
sources.

All four of the fractions are shown to be diagnostically useful,
although in this situation the non-volatile organic carbon appears
to have the least value. This is probably due mainly to the low
rate of leachate transport. However, it might be true in all cases
involving anaerobic decay, since the volatile organic carbon fraction

in such processes would be expected to show the greatest change

(Zanoni, 1972). In addition, the non-volatile fraction, being the

most likely to be adsorbed, is undoubtedly the most efficiently re-

moved from the ground water by the self-purification properties of

the aquifer, and methods of assessing this factor are being considered.

The landfill is having a minimal effect on the surrounding aquifer,

as evidenced by the only slight increases found in any of the carbon

fractions and the lack of any striking effect on the carbon isotope

ratios. However, the isotope ratio results in many cases are quite

unexpected, and the experiments required to clarify the causes of

these effects are currently being done.

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