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STATE OF CALIFORNIA
EDMUND G. BROWN
Governor

EFFECTS OF REFUSE DUMPS ON GROUND WATER QUALITY

1961

THE RESOURCES AGENCY OF CALIFORNIA
STATE WATER POLLUTION CONTROL BOARD

Publication No. 24

THE RESOURCES AGENCY OF CALIFORNIA
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Administrator

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FOREWORD

Early in 1960 a special meeting of the Los Angeles Regional Water Pollution Control Board was held to consider a request for modification of certain requirements regarding disposal of refuse in a gravel pit located near the center of a major ground water basin. Testimony offered during the meeting revealed a difference of opinion as to the possible effect of refuse disposal upon the ground water quality of the basin. There appeared to be a clear and pressing need for more factual information and data pertaining to refuse disposal in alluvial formations and protection that should be afforded to ground water basins.

Some years before, the State Water Pollution Control Board had sponsored research studies of the extent of pollution of ground water by alkalies and salts leached from refuse dumps. Projects undertaken by the University of Southern California under the direction of Professor Robert C. Merz resulted in State Board Publications No. 2, "Investigation of Leaching of Ash Dumps" (1952) and No. 10, "Investigation of Leaching of a Sanitary Landfill" (1954).

In August, 1958, a large rise in hardness was observed in water from a well adjacent to a refuse dump. The available data on the changes of the chemical quality of this water were assembled by the Los Angeles Regional Board. Later, the regional board contracted with the California Department of Water Resources for an investigation of ground water impairment in the basin. The results showed a build-up of carbon dioxide, apparently of refuse origin, with an associated increase of hardness.

In response to the need for further information, the State Board's research consulting board (Jack E. McKee, Sc.D., Erman A. Pearson, Sc.D., and Richard D. Pomeroy, Ph.D.) recommended that a project be undertaken to collate all available data bearing on pertinent aspects of the problem. In light of this undertaking it was suggested that the collation project include a recommended program of specific research projects needed to fill gaps found to exist in present knowledge of the subject.

On September 1, 1960, the State Board contracted with Engineering-Science, Inc., Arcadia, to undertake a study for the collation, evaluation and presentation of data on effects of refuse dumps on ground water quality. On the succeeding pages is printed the contractor's report submitted on August 29, 1961. Background information and objectives are shown in the report transmittal letter and Chapter I - Introduction. Findings are summarized in Chapter VII - Recommendations. The latter chapter includes a list of specific research objectives or needs, with relative priorities for future projects. Printing and distribution of the report as Publication No. 24 was authorized by the board on November 1, 1961.

Although the investigation reported herein was conducted under the sponsorship and direction of the State Water Pollution Control Board, the conclusions and recommendations given in the report are those of the research contractor and do not necessarily reflect opinions or policies of the board.

EFFECTS OF REFUSE DUMPS
ON GROUND WATER QUALITY

Prepared for

State of California
State Water Pollution Control Board

by

Engineering-Science, Inc.
150 East Foothill Boulevard
Arcadia, California

August 1961

ENGINEERING-SCIENCE, INC.

HARVEY F. LUDWIG, PRESIDENT
RUSSELL G. LUDWIG
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150 EAST FOOTHILL BL
ARCADIA, CALIFORNI
ELLIOT 9-9381

29 August 1961

State Water Pollution Control Board
1227 "O" Street
Sacramento 14, California

Gentlemen:

In accordance with Standard Agreement 12-19, dated September 1, 1960, we are herewith submitting our final report, entitled "Effects of Refuse Dumps on Ground Water Quality". This work was performed under my direction, with the consulting assistance of Mr. Francis R. Bowerman. Dr. Andrew L. Gram of our staff was project engineer.

In the report, information considered pertinent to the evaluation of potential ground water pollution by refuse dumps is developed and reviewed. Recommendations are given for a series of research projects which we believe should be undertaken without delay in order to furnish data needed to establish appropriate requirements for refuse dumping in gravelly areas.

Respectfully yours,

Harvey F. Ludwig

Harvey F. Ludwig, President
Engineering-Science, Inc.

HFL:jed

San Francisco Area Office and Research Center: 4144 Telegraph Avenue, Oakland 9, California

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I. INTRODUCTION

This report has been prepared by Engineering-Science, Inc., in accordance with the terms of Standard Agreement 12-19 of the California State Water Pollution Control Board.

As a result of the rapid population growth of metropolitan Los Angeles and the increasing urbanization of outlying areas, it has become increasingly difficult to find suitable convenient locations for land refuse disposal. The problem has further been aggravated by the abolition of all household incineration as a measure to control air pollution. Consequently the possibility of using spent sand and gravel pits, of which there are a considerable number in the San Cabriel and San Fernando valleys, has received more and more serious consideration. Because of the permeable nature of the ground in the gravel pit regions, and because many of the pits have been excavated below the ground water table, there has been just apprehension concerning the possible pollution of the ground water should any uncontrolled dumping of refuse be permitted.

The availability of close-in refuse disposal sites is evidently in the public interest, just as is a ground water supply unimpaired from its natural condition. Further, the empty gravel pits are worthless eyesores, whereas filled with refuse or any other material, their surfaces would become available for recreational, industrial, parking, and other useful purposes. There is a clear need for technical information which could be used to determine under what circumstances the disposal of refuse into gravelly areas will not constitute a menace to the ground water quality.

Objectives

The principal objective of the report was to organize and review the available information concerning the effects of refuse fills on the quality of the adjacent ground water. The technical information lacking for a satisfactory understanding of how to make use of gravel pits for refuse disposal without ground water degradation was thus made apparent, and a research program designed to furnish the missing facts was recommended.

Specific objectives of the report, as expressed by the State Water Pollution Control Board in Standard Agreement 12-19, were the following:

1. Assemble and collect all available information from publications and from such public and private reports that may be obtained.

2. Analyze any available data relative to changes in composition of gases in the soil atmosphere and their movement through the ground and their absorption into ground water.

3. List the chemical, biological, hydrologic, geologic, and other factors that may affect the extent of ground water alteration by buried rubbish, clarifying the state of knowledge in each case.

4. Analyze data relating to the changes in the composition of chemical and biological constituents of the ground water, both for situations where the ground water table is within the land fill and where the fill is above the water table.

5. Evaluate the effect of movement of water and gases through the soil on the constituents of the air and water, respectively, with regard to adsorption, desorption, ion-exchange and similar phenomena as they relate to water quality.

6. Analyze data relative to the rates of decomposition of organic matter in sanitary land fills under various conditions of compaction, moisture, soil content, and composition of the trash.

7. Prepare a comprehensive report of the findings and conclusions of the investigations.

Scope of Study

The search for information covered published reports on the subject of refuse disposal, papers published in the technical journals, and such unpublished data as could be obtained from public and private sources through local inquiry and by correspondence. Some fifty individuals throughout the United States in sanitary engineering and allied fields were contacted for references and information. Especially valuable sources of direct information on refuse pollution of ground water were Merz's report of studies made at the Riverside, California, sanitary fill (Reference 18), and the recent report by the British Ministry of Housing and Local Government on experimental studies of pollution by refuse dumped into wet and dry pits (Reference 33). In developing pertinent facts to the problem at hand, it was necessary to extend the literature investigation to the fields of soil science, hydrology, physical chemistry, and biochemistry.

In presenting the acquired data and information in this report, an attempt has been made to establish all of the factors which might participate in any effects of refuse fills on the ground water, and to evaluate them individually. The data apparently lacking for a thorough appraisal of the problem have been summarized at the end of each section.

Acknowledgements

At the beginning of the project, an Advisory Committee was formed of persons interested by reasons of their own work in the results of the investigation. The committee met to discuss and constructively criticize each of two

progress reports, and prepared detailed comments on the initial draft of this final report. Engineering-Science, Inc. wishes to express its sincere gratitude to the members of the advisory committee for the time and effort they donated. Following is the composition of the committee:

Mr. Linne Larson, Executive Officer, Regional Water Pollution Control Board No. 4, Chairman of the Advisory Committee.

Mr. F. R. Bowerman, Assistant Chief Engineer, County Sanitation Districts of Los Angeles County.

Mr. Dewey Dye, Engineer, Department of Water and Power, City of Los Angeles.

Mr. Carl Fossette, Executive Secretary, Central Basin Municipal Water District.

Mr. Judson A. Harmon, Supervising Sanitary Engineer, California Department of Public Health.

Mr. Howard H. Hawkins, Director, Upper San Gabriel Valley Municipal Water District.

Mr. Raymond Hertel, Senior Water Pollution Control Engineer, Regional Water Pollution Control Board No. 4.

Mr. Norman B. Hume, Director, Bureau of Sanitation, City of Los Angeles.

Mr. Finley Laverty, Assistant Chief Engineer, Los Angeles County Flood Control District.

Mr. Ben R. Paris, Director, Bureau of Street Maintenance, City of Los Angeles.

Mr. Arthur Pickett, Division Engineer, Los Angeles County Engineer's Office.

Mr. Brennan S. Thomas, Chief Engineer and General Manager, Long Beach Water Department.

Mr. A. E. Thompson, Refinery Manager, Mobil Oil Company.

Mr. Ralph Thorsen, President, Owl Park Corporation.

Mr. David B. Willets, Supervising Hydraulic Engineer, California Department of Water Resources.

Many individuals and agencies have contributed time and invaluable information for use in the study. Among the public agencies supplying data were the California State Water Pollution Control Board, Regional Pollution Control Board No. 4, the California Department of Water Resources, the Bureau of Sanitation of the City of Los Angeles, the Los Angeles County Engineer's Office, and the City of Pasadena Water Department.

In the preparation of the report, Engineering-Science, Inc., employed the consulting services of Mr. Francis R. Bowerman, who contributed valuable information on specialized phases of the study.

II. VERTICAL WATER MOVEMENT

Water applied to the surface of a refuse fill may have two important effects: One is the effect on the refuse decomposition rate, which is profoundly dependent on moisture content; the other is the leaching and transport of soluble materials in the refuse downward to reach ground water. In assessing both of these effects, a primary consideration is the amount of water reaching the refuse and its rate of arrival. These factors are determined by the rate at which water is applied to the fill surface--through precipitation, irrigation, or surface runoff--and by the retention and transmission characteristics of the refuse and its cover.

Theoretical Considerations

The movement of water in a porous medium occurs under the action of gravity and pressure differences and is influenced by surface tension and other forces associated with the tendency for moisture to adhere to the solid surfaces. In saturated portions of the medium the moisture content per unit volume is constant and consequently there are no net forces of the surface type. In that case, the flow rate is proportional to the combined gradient of pressure and gravity potentials, or to the gradient in static head. In symbols,

$$v = -k \text{ grad}\left(\frac{p}{w} + h\right) \qquad (1)$$

where v is flow per unit area of medium, $\frac{p}{w}$ is pressure head, h is potential head, and k is a constant known as the hydraulic conductivity of the porous medium. This expression is Darcy's Law and it holds very well for flow velocities sufficiently low that the flow is essentially laminar through the interstices of the medium.

In regions where the pores are not completely filled with water, a spacial variation in moisture content is accompanied by forces tending to transfer moisture from damper to drier areas. These forces can be considered as the gradient of a potential known as the "capillary potential" or "suction", which is the work required to remove a unit weight of water from the medium to a continuous body of water. The term "suction" is derived from the fact that the capillary potential corresponding to a given moisture content is equal to the vacuum which must be applied to a body of water in contact with the medium in order to maintain equilibrium. The total force acting on an element of moisture in an unsaturated region is the gradient of the gravitational and capillary potentials. Darcy's Law still holds, but with the conductivity varying as a function of the moisture content:

$$v = -k(u) \text{ grad } (S+h) \qquad (2)$$

where u is the local moisture content per unit bulk volume of medium, and -S is the suction.

The variation in suction and hydraulic conductivity with moisture content is shown in Figure 1 for several media. The suction increases from zero, at a moisture content equal to the porosity of the medium (saturation), to a very high value as the moisture content approaches zero. With a media composed of relatively uniform size particles, the suction curves tend to exhibit a range of moisture content over which the change in suction is small, whereas the suction in well-graded media increases without inflection. Conductivity has its maximum value at saturation and decreases rapidly as the medium becomes drier.

In order to analytically calculate the progress of moisture movement in a medium of known and uniform properties, the appropriate differential equation may be obtained by combining Darcy's Law with the equation of continuity, which expresses the fact that any local rate of change in moisture content must correspond to a difference in the rate of moisture flow into and out of the vicinity. The equation of continuity is:

$$\frac{\partial u}{\partial t} + \operatorname{div} v = 0 \qquad (3)$$

Substituting for v the expression given by Darcy's Law (Equation (2)),

$$\begin{aligned} \frac{\partial u}{\partial t} &= \operatorname{div} (k \operatorname{grad} (S+h)) \qquad (4) \\ &= \operatorname{div} (k \operatorname{grad} S) + \frac{\partial k}{\partial z} \end{aligned}$$

where z is the vertical space coordinate. Defining a "diffusivity" D as

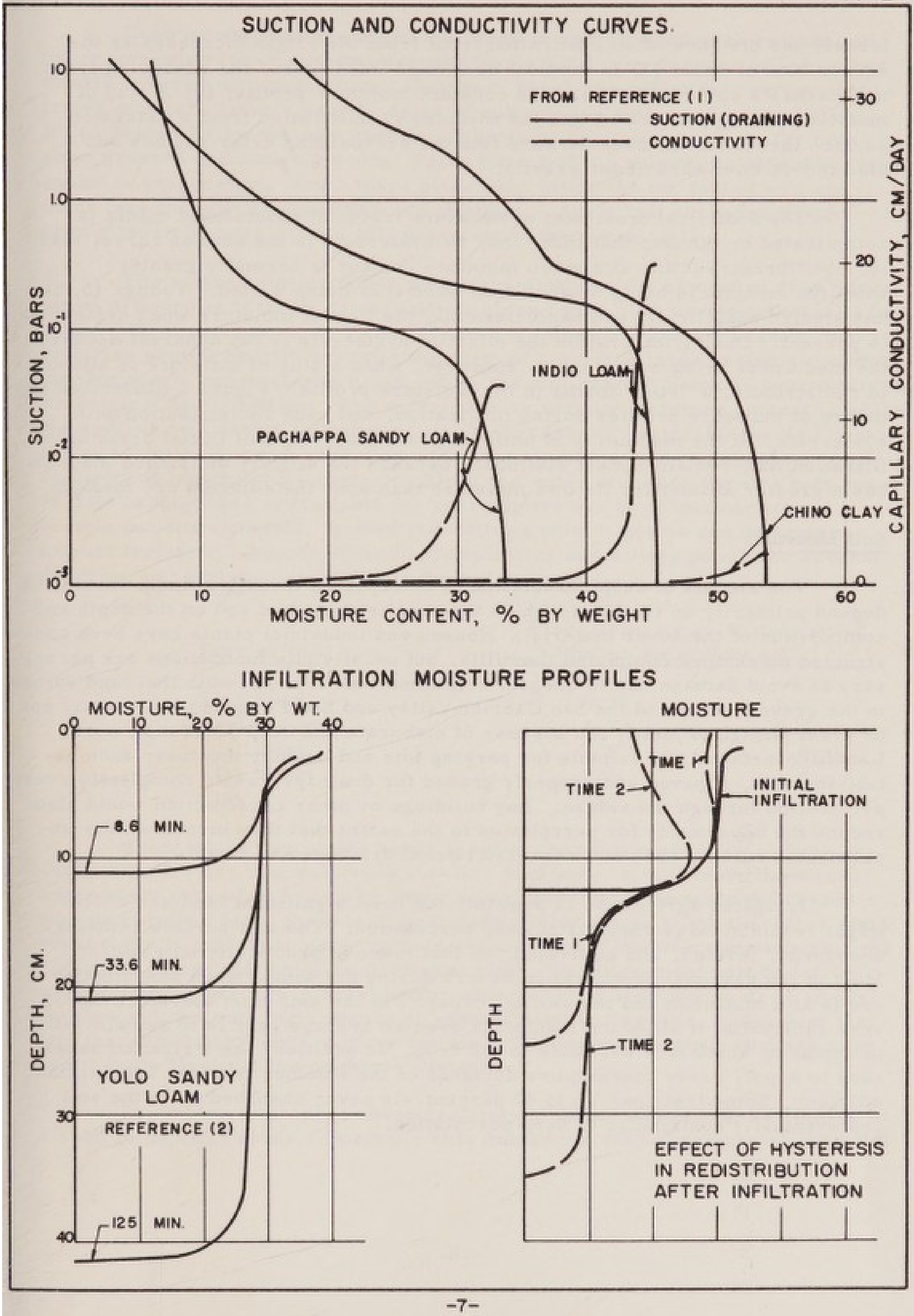
$$D(u) = k \frac{dS}{du}$$

Equation (4) becomes

$$\frac{\partial u}{\partial t} = \operatorname{div} (D \operatorname{grad} u) + \frac{\partial k}{\partial z} \qquad (5)$$

Equation (5) gives the time rate of change of moisture content at any point as a function of the moisture content and its space derivatives and can be solved for particular sets of boundary conditions if the functions D(u) and k(u) are known. Normally the integration will have to be performed by some approximation or iterative procedure. Gardner and Mayhugh (1) have established the solution for horizontal movement for the case in which the diffusivity D can be approximated by an exponential function of moisture content. Philip (3, 4) has developed a series solution applicable to several cases of horizontal and vertical flow. In general terms the results, which have been substantiated by experiment, are the following: (1) Moisture infiltrating into a dry medium tends to move as a fairly well defined front--the drier the medium, the sharper the front; (2) In horizontal one-dimensional movement, and in vertical movement through media so fine-grained that gravity is small compared to capillary

FIGURE I



SUCTION AND CONDUCTIVITY CURVES.

INFILTRATION MOISTURE PROFILES

forces, the distance of an infiltration front from the origin increases as the square root of time; (3) In vertical downward infiltration, the advancing front approaches a constant velocity and constant moisture profile; (4) A slug of moisture having penetrated into the medium by infiltration from a saturated surface thereafter diffuses outward into the surrounding drier regions and also moves downward under gravity.

The analytical treatment of moisture travel in unsaturated media is complicated by the fact that there may be hysteresis in the suction curve; that is, equilibrium suction at a given moisture content is normally greater when the medium is being drained than when it is being wetted. Youngs (5, 6) has studied analytically and experimentally the flow of moisture when hysteresis is present. During infiltration the effect of hysteresis is not apparent because the medium is being wetted only. However, when a slug of moisture is allowed to redistribute, a "step" forms in the moisture profile. Figure 1 illustrates the nature of moisture profiles during infiltration, and upon redistribution with hysteresis. If the medium is of uniform grain size, a small initial depth of infiltration may remain almost stationary because the slightly moistened medium has a greater affinity for its own moisture than does the adjacent dry medium.

Soil Moisture

The amount of deep percolation to be expected through a dump cover will depend primarily on the use to which the area is subjected and on the depth and composition of the cover material. Houses and industrial plants have been constructed on shallow completed landfills, but usually pile foundations are necessary to avoid damage due to unequal settlement and it is possible that land values in the gravelly areas of the San Gabriel Valley and San Fernando Valley may not be great enough to justify the expense of elaborate and deep foundation work. Landfill surfaces are suitable for parking lots and outdoor theatres; such establishments, if paved and properly graded for drainage, would completely avert percolation through the refuse. Any buildings or other construction would also reduce the opportunity for percolation to the extent that they increased the impermeable surface area and enhanced lateral drainage and runoff.

Irrigated agriculture is probably the most significant land use which might result in large amounts of deep percolation. The soil surface remains uncovered, leveled, and cultivated, so that there is practically no runoff. Most of the seasonal precipitation occurs during the winter when consumptive use is at a minimum and the storage capacity of the soil must therefore be very substantial if all of the rainfall in even an average year is to remain within depths at which it is available to the crop. In addition, the irrigation necessary to supply heavy consumptive demands of the summer is never 100 percent efficient. Some fraction--up to 40 percent--is never absorbed from the soil and eventually contributes to deep percolation.

Land on which native brush and weed growth is allowed to remain will exert somewhat lower consumptive use than a field crop because of being less thickly covered with vegetation. However, brush can effectively remove infiltrated moisture from a greater depth of soil and the amount of precipitation which passes below the root zone is of the same general magnitude for both sown grasses and native growth. From bare soil the only water loss other than runoff is evaporation, which takes place only within the top foot of soil and is much less than the consumptive use of plants. About half of the infiltrated precipitation on bare ground will percolate to the water table.

The surface soil or cover material influences the amount of deep percolation principally through its capacity to store water. As precipitation falls on a dry soil, it penetrates as a sharp front and, when infiltration ceases, the moisture profile shortly assumes the form of a surface layer having a moisture content nearly constant with depth, while the soil beneath remains practically in its original dry condition. The more-or-less definite moisture content to which the surface soil drains is known as its "field capacity"; actually the soil continues to drain, but at a very slow rate because of the reduced hydraulic conductivity at low moisture content. All of the water stored in the soil root zone at field capacity is not available for consumptive use by plants because at a certain moisture content, termed the "wilting point", plants can no longer extract moisture. Approximate field capacities and wilting points for several different soil textures are given in the following table:

Type of Soil	Field Capacity	Wilting Point
	inches water/foot soil	
Fine Sand	0.5	0.2
Sandy Loam	1.7	0.5
Silt Loam	2.5	1.0
Clay Loam	3.3	1.4
Clay	4.5	1.7

Water which percolates below the root zone can move upward when the surface layers dry, but only very slowly. Similarly, it will move downward appreciably only when driven by more water descending from above. Years may be required for a particle of percolating moisture to reach the water table, but if there is sufficient water applied annually to force some moisture past the root zone, there will be some downward flow. The movement will be fairly rapid following periods of heavy infiltration and very slow during dry periods; the flow will become more steady at greater depths, however, because of the fact that capillarity tends to move moisture from wetter regions to drier ones and thereby level out the moisture content.

The moisture storage characteristics of dump covers may be limited by a shallow depth of cover. Normally only one or two feet of compacted material would be used and a considerable part of this has been observed to sift down into the refuse. However, refuse itself has significant moisture holding capacity and the top layer, especially, being fairly accessible to atmospheric oxygen, will tend to compost into an absorbent humus. It is therefore quite possible that the moisture holding capacity of the surface refuse may be greater than that of the cover material, particularly if the latter is native sand.

Water Balance

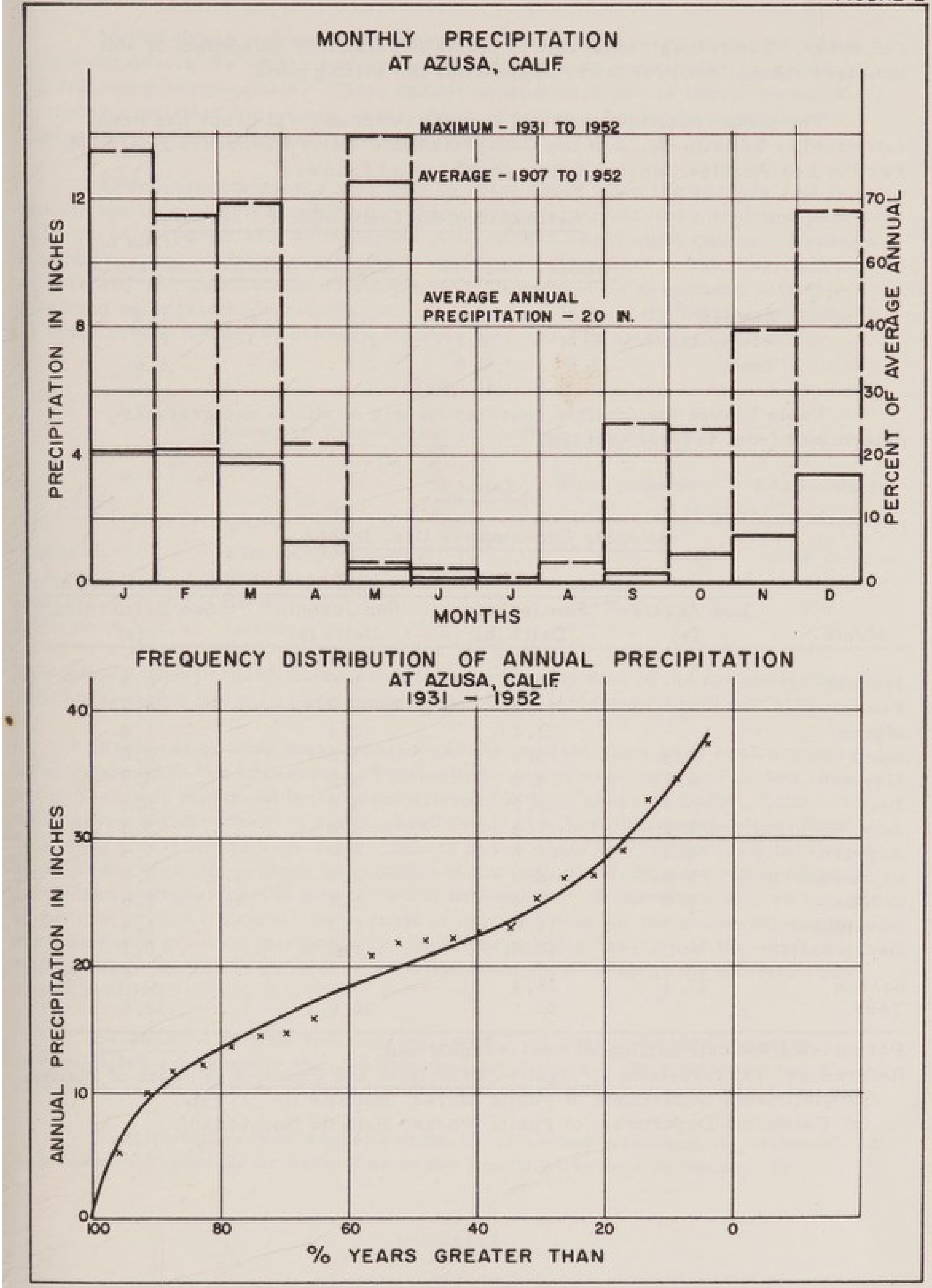
It is possible by means of a water budget to estimate the amount of water which will enter a refuse fill from the surface during the filling operation and after completion. The only significant sources of water are precipitation and irrigation; it is assumed that all present and future fills can and will be graded so that surface runoff from adjacent areas does not flow across the refuse-filled area. Water may be removed from the fill to the atmosphere through evaporation and transpiration. The excess of the water applied during any period over evapo-transpiration may be stored to some extent in the surface layers, but any water which passes below the plant root zone is no longer susceptible to appreciable surface losses, and will eventually reach the ground water table. On reasonably fine-textured soil, it is possible, through judicious irrigation practice, to avoid large amounts of deep percolation, but such procedure will result in the gradual accumulation of salts in the soil as water is lost by evapo-transpiration. In order to avoid adverse effects on the irrigated crops it is therefore necessary that some leaching below the root zone occur.

Precipitation: The normal annual rainfall in Los Angeles County ranges from about 10 inches to more than 30 inches, the higher values being registered in the mountainous regions. Figure 2 shows average and maximum monthly precipitation recorded at Azusa, and also the frequency distribution for annual precipitation. The average annual rainfall at Azusa is about 20 inches; because of the proximity of the mountains, this value is somewhat greater than that for most level regions in the San Gabriel Valley in which refuse disposal sites might conceivably be located.

The monthly distribution of rainfall follows the typical Southern California pattern; nearly all of the year's rain comes during the winter months of December through March. Even the maxima recorded for the months of May through August are very small. In even the wettest years there is a four-month summer period during which the soil has adequate opportunity to lose its accumulated moisture by evapo-transpiration.

Evapo-transpiration: The amount of water lost to the atmosphere from a given land area depends on the type of soil and vegetation and is closely correlated with climatic factors such as temperature, humidity, and wind velocity. The consumptive use also depends on the amount of water available. Once the soil has dried throughout the depth of the root zone, no more appreciable loss

FIGURE 2



MONTHLY PRECIPITATION AT AZUSA, CALIF.

FREQUENCY DISTRIBUTION OF ANNUAL PRECIPITATION AT AZUSA, CALIF. 1931 – 1952 can occur. The transpiration rate of plants is relatively independent of soil moisture content until the latter approaches the wilting point.

The mean consumptive use of several common local crops has been estimated in Bulletin No. 2 of the California State Water Resources Board (7). For the Los Angeles interior region these are as follows:

	Estimated Mean Consumptive Use, ft/yr				
	Alfalfa	Pasture	Orchard	Walnuts	Citrus & Subtropical
Applied	2.3	2.5	1.5	1.6	1.4
Precipitation	1.3	1.1	1.3	1.3	1.2
Total	3.6	3.6	2.8	2.9	2.6

Table I gives the monthly consumptive use of alfalfa and grass as determined from several sources.

Table I

Monthly Consumptive Use, Inches

	Alfalfa		Pasture	Bermuda Grass
Month	Los Angeles (a)	San Joaquin Delta (b)	San Joaquin Delta (b)	San Bernardino (c)
January	-	(0.7)	1.0	0.8
February	-	(1.0)	1.2	0.7
March	-	1.2	2.4	1.0
April	3.3	3.6	3.0	2.0
May	6.7	4.8	3.0	4.3
June	5.4	6.0	3.0	5.2
July	7.8	7.8	3.0	5.5
August	4.2	6.6	3.0	5.3
September	5.6	6.0	2.4	2.9
October	4.4	2.4	1.8	2.8
November	-	(1.2)	1.2	1.4
December	-	(0.8)	1.0	0.7
Season	37.4	38.4	-	-
Year	-	42.1	26.0	32.6

Parentheses indicate estimated non-seasonal use.

References: (a) Israelsen: Irrigation Principles and Practice (8), Data of Blaney.

(b) California Department of Public Works, Bulletin No. 27 (9).

(c) California Department of Public Works, Bulletin No. 44 (10).

Native brush and weeds in Southern California use soil moisture while it is available in the winter and spring and then die or remain dormant until the following rainy season. Their annual consumptive use is therefore equal to the precipitation minus any water which percolates below the root zone during periods of exceptionally wet weather (11).

Evaporation from the soil may be quite rapid when the surface soil is moist, but is greatly retarded after the top inch or so becomes dry, and practically no evaporation occurs at depths greater than about eight inches. Because the surface soil moisture condition is so heavily dependent on the distribution of rainfall, any estimate of monthly evaporation must be associated with the assumed or predicted precipitation. Reference (11) lists the following depths of evaporation measured between October and April in Southern California.

Table II

Measured Evaporation in Southern California

Place	Soil	Precipitation	Evaporation
		(inches)	
Anaheim	Fine Sandy Loam	12.4	6.0
Ontario	Sand over Silt Loam	11.5	4.7
Ontario	Sand	12.8	6.8
Glen Avon Hts.	Loam	12.2	8.1

The evaporation in these measurements was roughly half of the precipitation; this observation has also been made in the ASCE Hydrology Handbook (12).

Evaporation rate measurements made on field plots have occasionally been reported in the literature. Veihmeyer (13) obtained only 3.5 inches loss from bare soil sheltered from precipitation for four years at Davis, Calif. Veihmeyer and Brooks (14) and Richards et al (15) have observed evaporation following irrigation of bare soils; in both bases about two inches loss developed in 60 days, with the greater part of this occurring within 20 days. Evaporation from land surfaces during any period of dry weather is therefore limited to about two or three inches. In periods of frequent rainfall the evaporation rate may be much greater, particularly where the holding capacity of the surface soil is large enough to provide a continuous supply of evaporable moisture between storms.

Balance: Table III and Figures 4 and 5 indicate the water budget of refuse-filled areas under various assumed conditions. The quantities and distributions of the various additions and withdrawals were estimated as follows:

Precipitation--For average rainfall, 20 inches was used, distributed as shown in Figure 2, or according to the monthly average at Azusa. In estimating effects during a wet year, a total precipitation of 170 percent of average was used, or 34 inches. This would be exceeded at Azusa in only 10 percent of all years. Its monthly distribution was taken as proportional to the average monthly distribution.

Evapo-transpiration--The annual consumptive use of irrigated grass is estimated at 36 inches, which is intermediate between the 32 inches found for Bermuda grass (Table I) and the 43 inches estimated by the California Water Resources Board for irrigated pasture. Its monthly distribution has been taken as that shown in Figure 3, which has ordinates approximately proportional to the measured values for Bermuda grass listed in Table I, but smoothed out slightly.

Native brush and weeds were assumed to use water at the same rate as irrigated grass during that portion of the year in which there is soil moisture from precipitation at some level above the wilting point. Consumptive use throughout the remainder of the year was taken as zero.

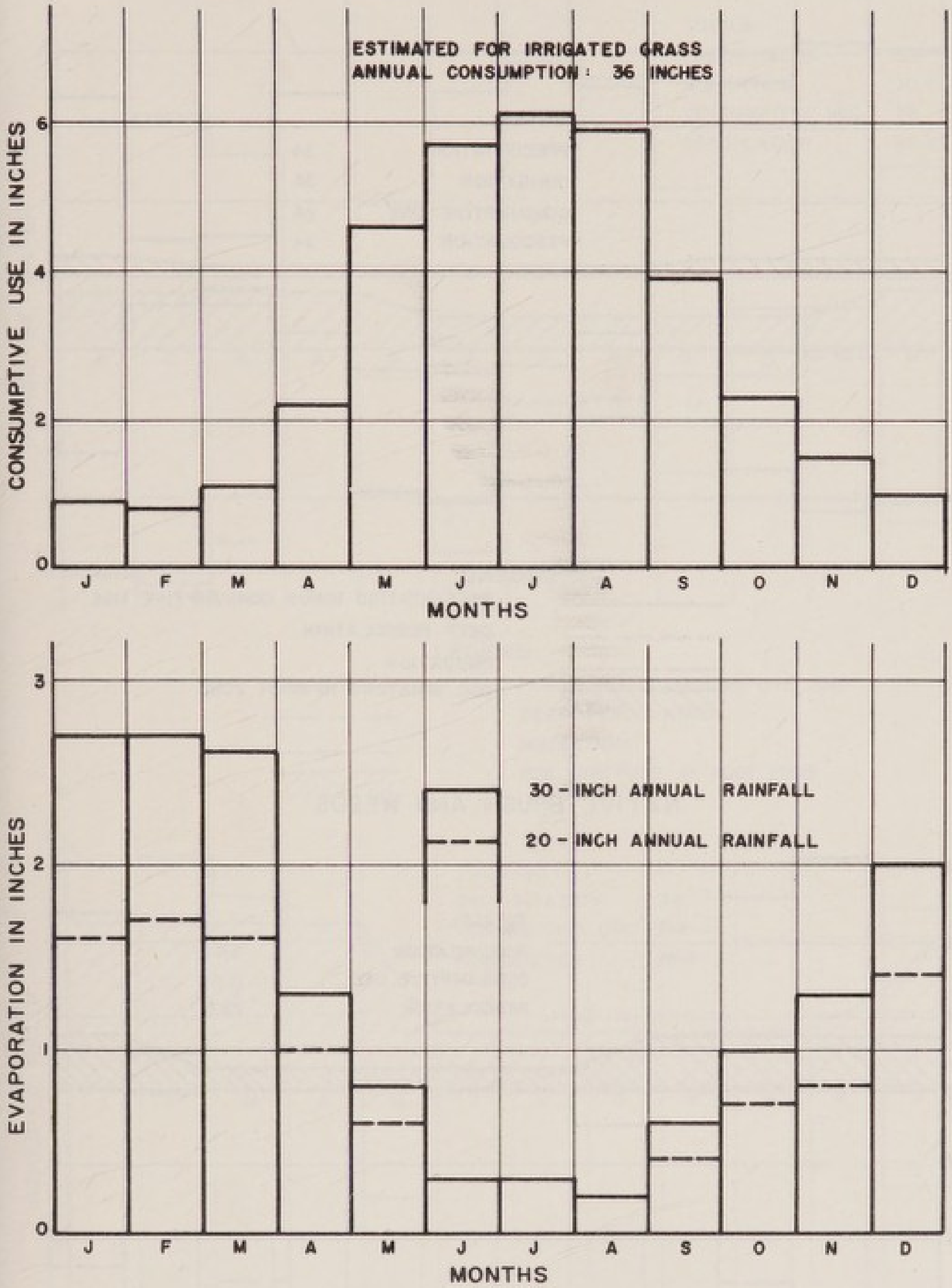
Evaporation from bare soil was assumed to proceed as indicated in Figure 3. The distribution was prepared from a chart prepared by Meyer (12, p. 132) which gives monthly evaporation as a function of precipitation and mean temperature. Temperatures used were mean values tabulated by the U. S. Weather Bureau for Pasadena, California. The total year's evaporation is indicated to be 10. 4 inches for a 20-inch yearly precipitation, and 15. 8 inches for 34 inches of precipitation; the ratio of evaporation to precipitation of about 50 percent is in agreement with the values of Table I. To allow for reduced evaporation opportunity from the permeable cover materials under consideration, coefficients of 0. 6 and 0. 8 for fine sand and sandy loam, respectively, were applied to the monthly values shown on Figure 3.

In the early years of a dump's existence the evaporation may be quite high because of the high temperatures occurring in decomposing refuse. The data of Merz shown in Figure 6 indicate evaporation of over 40 inches per year under conditions where moisture was continuously being applied. However, decomposition must eventually cease, and at that time the evaporation from a fill surface having a cover thicker than about one foot will be the same as that from equivalent soil not underlain by refuse.

Irrigation--It has been assumed that irrigation water would be applied to the grass surface as needed to satisfy the consumptive requirement. Some loss of the water to deep percolation is unavoidable because of nonuniformities in depth of application, depth of penetration, distribution of roots, etc., and losses will clearly be the greatest in coarser soils which have low moisture retention capacity and permit more rapid movement. Experiments mentioned in Davis (16, p.804) indicate percolation loss of 12 percent in silt soils up to 25 percent for coarse sandy soils. Herein a loss of 20 percent has been assumed for sandy loam, and 30 percent for fine sand. It is expected that the grass would

FIGURE 3

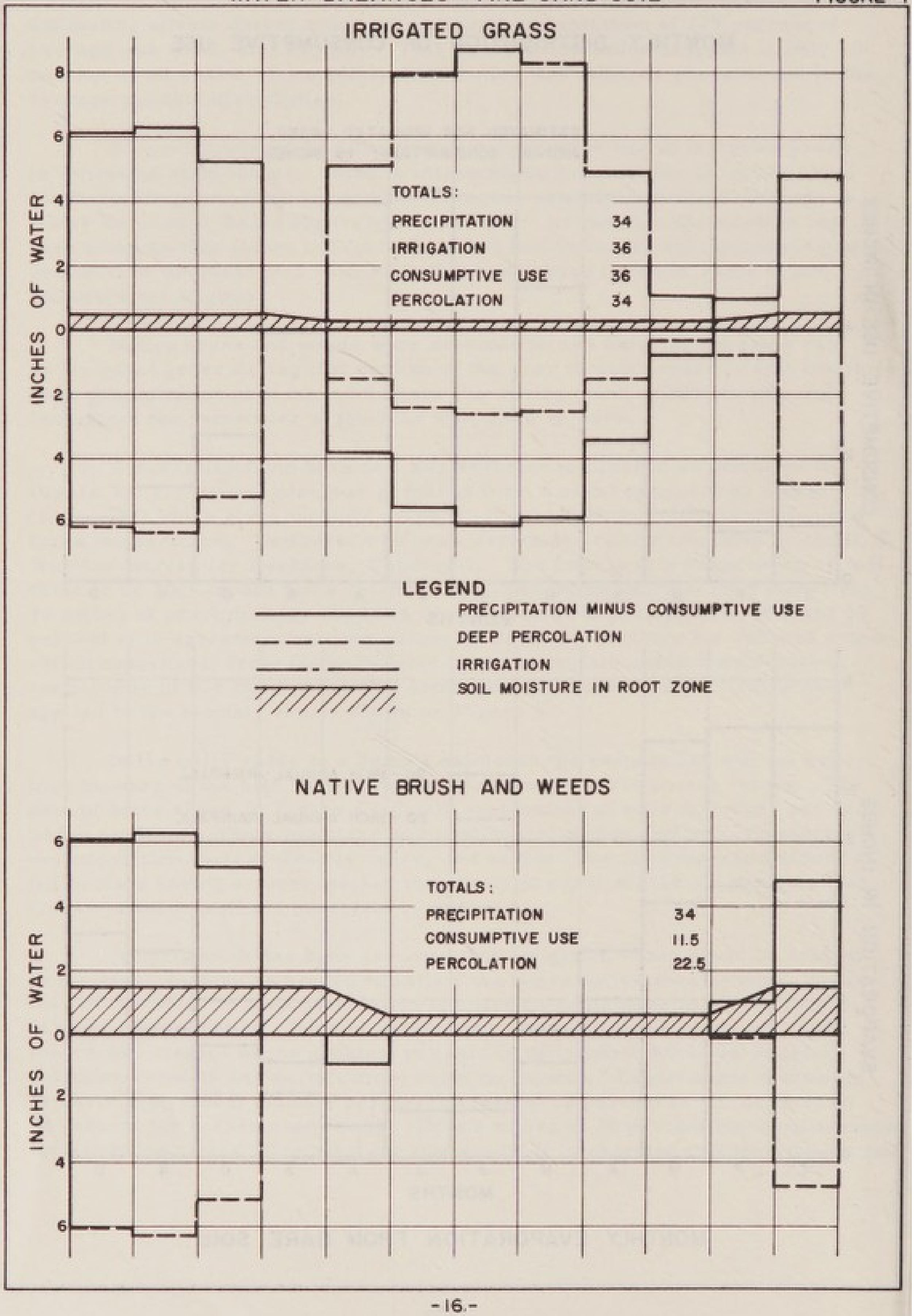
MONTHLY DISTRIBUTION OF CONSUMPTIVE USE



MONTHLY EVAPORATION FROM BARE SOIL

WATER BALANCES - FINE SAND SOIL

FIGURE 4



WATER BALANCES - SANDY LOAM SOIL FIGURE 5

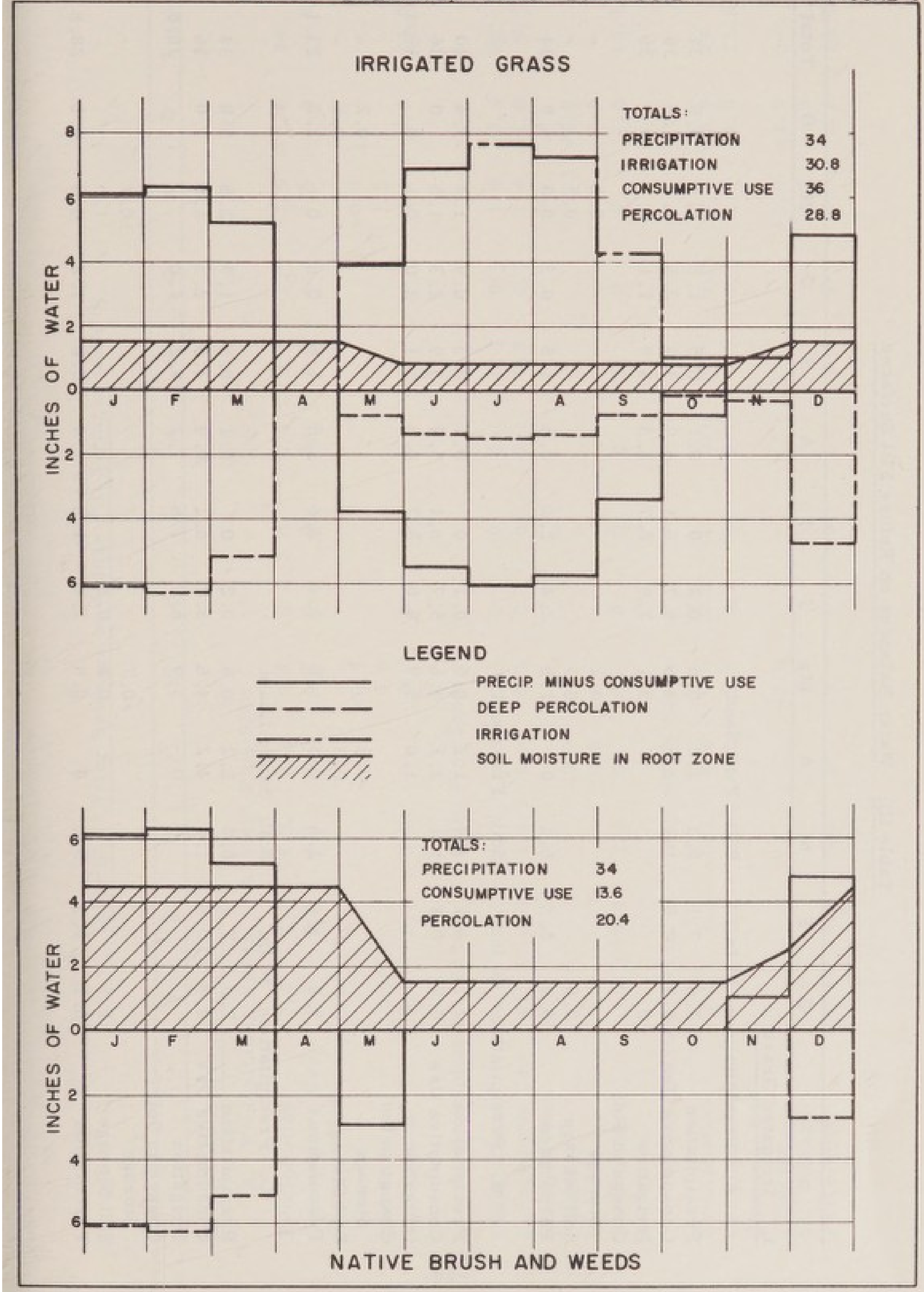


Table III: Water Balances on Refuse Fill Surfaces

	J	F	M	A	M	J	J	A	S	O	N	D	Total
I. Irrigated Grass													
A. Precipitation: 34 In.			Soil: Fine Sand										
Precipitation	7.0	7.1	6.3	2.2	0.8	0.2	0	0.1	0.5	1.5	2.5	5.8	34
Consumptive Use	0.9	0.8	1.1	2.2	4.6	5.7	6.1	5.9	3.9	2.3	1.5	1.0	36
Irrigation	0				5.1	7.9	8.7	8.3	4.9	1.1	0	0	36
Change in Soil Storage					-0.2						+0.2		-
Soil Storage	0.5				0.3						0.5		-
Percolation	6.1	6.3	5.2	0	1.5	2.4	2.6	2.5	1.5	0.3	0.8	4.8	34
B. Precipitation: 20 In.			Soil: Fine Sand										
Precipitation	4.1	4.2	3.7	1.3	0.5	0.1	0	0.1	0.3	0.9	1.4	3.4	20
Consumptive Use	0.9	0.8	1.1	2.2	4.6	5.7	6.1	5.9	3.9	2.3	1.5	1.0	36
Irrigation				1.0	5.9	8.0	8.7	8.3	5.1	2.0	0.1	0	39.1
Change in Soil Storage				-0.2								+0.2	-
Soil Storage	0.5			0.3								0.5	-
Percolation	3.2	3.4	2.6	0.3	1.8	2.4	2.6	2.5	1.5	0.6	0	2.2	23.1
C. Precipitation: 34 In.			Soil: Sandy Loam										
Precipitation	7.0	7.1	6.3	2.2	0.8	0.2	0	0.1	0.5	1.5	2.5	5.8	34
Consumptive Use	0.9	0.8	1.1	2.2	4.6	5.7	6.1	5.9	3.9	2.3	1.5	1.0	36
Irrigation				0	3.9	6.9	7.6	7.2	4.2	1.0			30.8
Change in Soil Storage					-0.7						+0.7		-
Soil Storage	1.5				0.8	0.8					1.5		-
Percolation	6.1	6.3	5.2	0	0.8	1.4	1.5	1.4	0.8	1.2	0.3	4.8	28.8

Table III: Water Balances on Refuse Fill Surfaces (cont'd)

	J	F	M	A	M	J	J	A	S	O	N	D	Total
II. Native Brush and Weeds													
A. Precipitation: 34 In. Soil: Fine Sand													
Precipitation	7.0	7.1	6.3	2.2	0.8	0.2	0	0.1	0.5	1.5	2.5	5.8	34
Consumptive Use	0.9	0.8	1.1	2.2	1.7	0.2	0	0.1	0.5	1.5	1.5	1.0	11.5
Change in Soil Storage					-0.9						+0.9		-
Soil Storage	1.5				0.6						1.5		-
Percolation	6.1	6.3	5.2	0	0	0	0	0	0	0	0.1	4.8	22.5
B. Precipitation: 20 In. Soil: Fine Sand													
Precipitation	4.1	4.2	3.7	1.3	0.5	0.1	0	0.1	0.3	0.9	1.4	3.4	20
Consumptive Use	0.9	0.8	1.1	2.2	0.5	0.1	0	0.1	0.3	0.9	1.4	1.0	9.3
Change in Soil Storage				-0.9								+0.9	-
Soil Storage	1.5			0.6								1.5	-
Percolation	3.2	3.4	2.6	0	0	0	0	0	0	0	0	1.5	10.7
C. Precipitation: 34 In. Soil: Sandy Loam													
Precipitation	7.0	7.1	6.3	2.2	0.8	0.2	0	0.1	0.5	1.5	2.5	5.8	34
Consumptive Use	0.9	0.8	1.1	2.2	3.8	0.2	0	0.1	0.5	1.5	1.5	1.0	13.6
Change in Soil Storage					-3.0						+1.0	+2.0	-
Soil Storage	4.5				1.5		1.5				2.5	4.5	-
Percolation	6.1	6.3	5.2	0	0	0	0	0	0	0	0	2.8	20.4

Table III: Water Balances on Refuse Fill Surfaces (cont'd)

	J	F	M	A	M	J	J	A	S	O	N	D	Total
III. Bare Earth													
A. Precipitation: 34 In.			Soil: Fine Sand										
Precipitation	7.0	7.1	6.3	2.2	0.8	0.2	0	0.1	0.5	1.5	2.5	5.8	34
Consumptive Use	1.6	1.6	1.5	0.8	0.5	0.2	0.2	0.1	0.3	0.6	0.8	1.2	9.4
Change in Soil Storage							-0.2		+0.2				-
Soil Storage	0.4						0.2		0.4				-
Percolation	5.4	5.5	4.8	1.4	0.3	0	0	0	0	0.9	1.7	4.6	24.6
B. Precipitation: 20 In.			Soil: Fine Sand										
Precipitation	4.1	4.2	3.7	1.3	0.5	0.1	0	0.1	0.3	0.9	1.4	3.4	20
Consumptive Use	1.0	1.0	1.0	0.5	0.4	0.2	0.1	0.1	0.2	0.4	0.5	0.8	6.2
Change in Soil Storage						-0.1	-0.1		+0.1	+0.1			-
Soil Storage	0.4					0.3	0.2		0.3	0.4			-
Percolation	3.1	3.2	2.7	0.8	0.1	0	0	0	0	0.4	0.9	2.6	13.8
C. Precipitation: 34 In.			Soil: Sandy Loam										
Precipitation	7.0	7.1	6.3	2.2	0.8	0.2	0	0.1	0.5	1.5	2.5	5.8	34
Consumptive Use	2.2	2.2	2.1	1.0	0.6	0.2	0.2	0.1	0.4	0.8	1.0	1.6	12.4
Change in Soil Storage							-0.2		+0.1	+0.1			-
Soil Storage	1.1						0.9		1.0	1.1			-
Percolation	4.8	4.9	4.2	1.2	0.2	0	0	0	0	0.6	1.5	4.2	21.6

be irrigated when the root zone soil moisture content has been reduced by transpiration to 30 percent of the holding capacity between wilting point and field capacity.

Soil Characteristics--In order to estimate percolation loss which would be conservatively high, the water balance was carried out on soils of relatively low moisture holding capacity. These were fine sand and sandy loam having the following field moisture limits:

	Field Capacity (inches/foot)	Wilting Point (inches/foot)
Fine Sand	0.5	0.2
Sandy Loam	1.5	0.5

The effective depth of the root zone was taken as one foot for the irrigated grass and three feet for native brush and weeds. On bare earth it was assumed that moisture percolating below a depth of nine inches would not be susceptible to evaporation.

The estimated values for annual deep percolation range from about 20 inches for wild brush on sandy loam soil to about 36 inches for irrigated turf in a year of exceptionally high rainfall. Percolation is considerably lower during average rainfall years and would be lower still if the root zone were deeper and composed of more finely textured soil. For the conditions assumed, storage of water in the surface soil has an almost negligible effect toward reducing percolation losses. Blaney (11) has observed that at least 19 inches of seasonal rain must fall on native brush before there is any deep percolation and that in coarse soil some percolation is obtained through grass and weeds after 10 to 12 inches of rain has fallen. However, the soils on which those experiments were made were all much finer textured than the sand and sandy loam considered herein as representing probable types of readily available dump cover material; also the root zones for brush and weeds extended much deeper than the one and three feet suggested herein for grass and sparse brush respectively.

The amounts of percolation estimated by the water balance are conservatively high. They would be greatly reduced by an increase in assumed soil moisture storage capacity and, in particular, either by increasing the effective depth of the root zone or by increasing the moisture retention characteristics of the material composing the root zone. While the root zone of turf grass could hardly be expected to extend itself appreciably below the one-foot depth assumed, plants such as brush and trees produce roots down to 15 feet or more, and the suitability of decomposed refuse as a growing medium therefore becomes an important question.

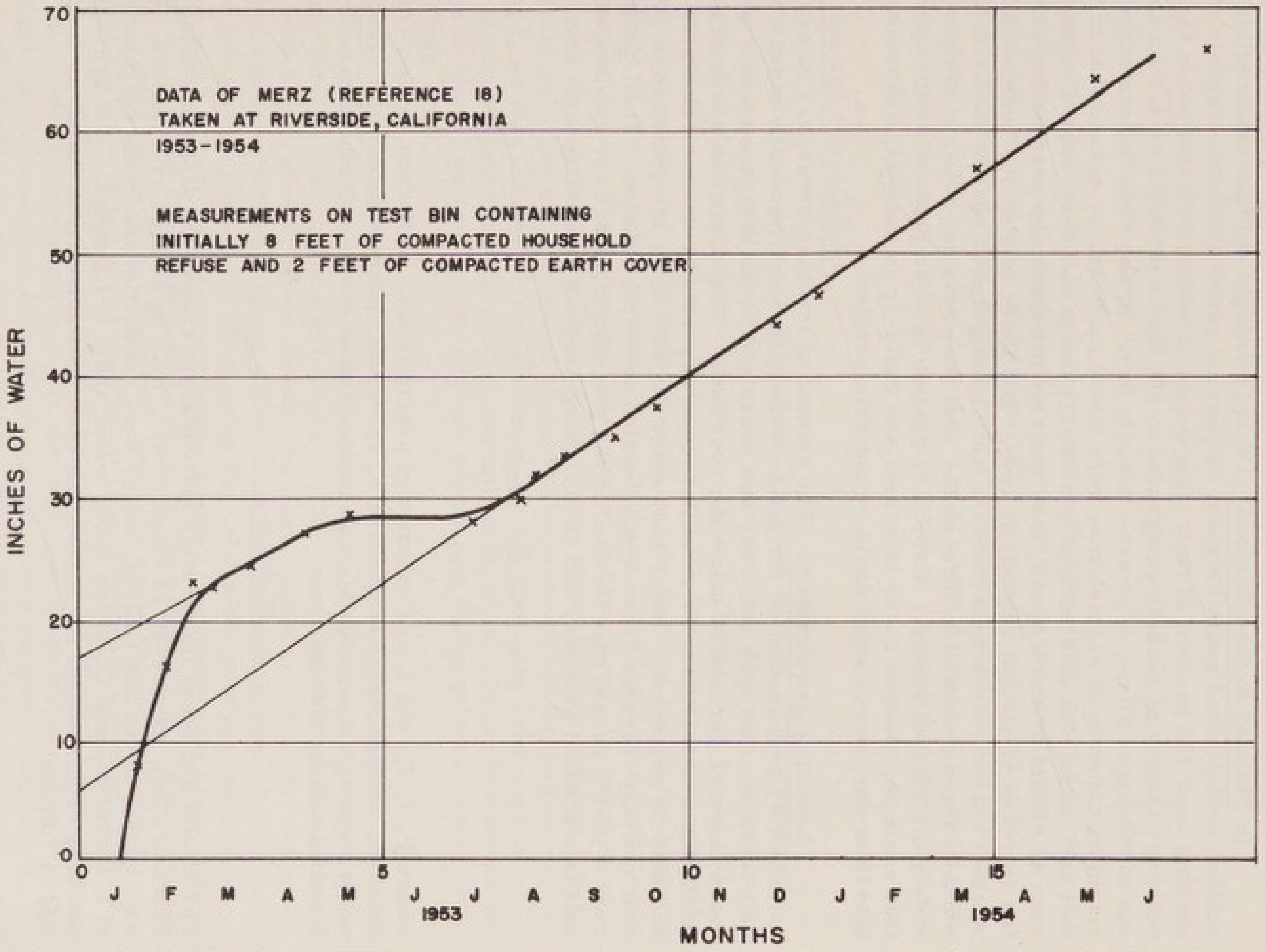
There are literature references concerning the suitability of completed fills for the growing of grass and forage crops (17); however, little experience in the growing of trees or other deep-rooted plants is evidenced. In Los Angeles County, the Sanitation Districts make it a practice to construct into the top layer of refuse an earthen bowl of some ten-foot radius in which trees are planted for park purposes; the added depth of soil is deemed better suited to root growth than the refuse. The most convincing argument as to the suitability of completed fills for growing cover crops is the rapidity with which native grasses and weeds re-establish themselves on covering layers of earth, even without planned seeding or irrigation. A lush growth of grass can be grown within weeks after the depositing of refuse if a nominal amount of seed is sown and watered.

Moisture Retention by Refuse: Assuming that most plants can derive moisture and nutrients from buried refuse, at least in a partially decomposed condition and with a certain amount of soil material present from the surface cover and intermediate layers, the moisture retention capacity then becomes a controlling factor in determining the magnitude of deep percolation. In the Riverside studies of Merz (18), water was applied to the surface of a test bin full of refuse with a soil cover at the rate of one inch per week. Figure 6 shows the cumulative excess of applied water over that removed from the bottom as percolate. The initial amount of water required to raise the column to field capacity was about 15 inches. From about the seventh month to the end of the experiment--a period of about nine months--the rate of moisture disappearance was constant; the rate of evaporation was therefore relatively unaffected by any variations in seasonal environmental factors. However, between May and July of 1953 there is a step in the consumption curve; actually slightly more water was released as percolate during this period than was applied. From the corresponding temperature rise it might be supposed that the period was one of accelerated decomposition and, if the evaporation rate was nearly equal to the value obtained during the latter two-thirds of the experiment, there was an apparent decrease in the field capacity from 15 inches to about 6 inches over the moisture initially present in the refuse. If the moisture required to raise the 2-foot soil cover to field capacity was one inch per foot, the absorption capacity of the refuse (which had settled initially from its original 8-foot depth to about 5.5 feet) was about 2.5 inches/foot before the moisture release occurred and about 0.75 inch/foot afterward. However, some fraction of the released water could have been produced biochemically from decomposition reactions. As indicated in Section III, this would probably be somewhat less than one inch per foot.

A study performed by the British government on ground water pollution due to refuse fills (33) has yielded good information on the moisture retention capacity of the refuse listed for Watford, England, in Table VII. The material packed into one-foot diameter pipe columns would absorb about 1.5 inches of water per foot of refuse depth. In a pilot plant fill a five-foot depth of the same refuse, compacted to 780 lb/cu yd and covered with 1.5 feet of earth, absorbed 7.3 inches of rainfall before producing appreciable percolate. However the

FIGURE 6

EVAPORATION FROM EXPERIMENTAL LANDFILL SURFACE
CUMULATIVE DIFFERENCE BETWEEN APPLIED WATER AND REMOVED PERCOLATION



saturating rainfall arrived over a period of about four months, so that there was considerable opportunity for evaporation. The actual retention was therefore of the order of one inch per foot, or possibly somewhat less.

In Planned Refuse Disposal (19) it is brought out that combustible rubbish as collected contains about 40 percent paper products, capable of absorbing over 250 percent of their own weight in water. On this basis the absorbing capacity of rubbish compacted to 600 lb/cu yd would be 4. 5 inches/foot due to the paper alone, if it were initially dry. In any case the moisture-retaining capacity of rubbish may very likely approach or exceed that of loam soil and, if so, the existence of reasonably dense and deep-rooted vegetation, such as alfalfa or thick brush, may entirely prevent deep percolation of rainfall in years of normal rainfall or less. But there is still the question as to whether buried refuse behaves as a truly homogeneous porous medium, or whether rapid percolation through the root zone may not occur at isolated, but widely distributed points. Also, if irrigation is to be practiced, it is necessary that some percolation occur to arrest salt build-up in the soil, supposing that losses to deep percolation could be entirely avoided.

Leaching Requirement: The amount of leaching necessary to remove salts left by evaporation of irrigation water can be estimated by means of a salt balance. Salt may enter the soil dissolved in the irrigation water and leave in water percolated below the root zone. The quantities which are added in the form of fertilizer or removed in the harvested crop are relatively minor and may be neglected. Water may enter the soil from precipitation or irrigation and may leave as evapo-transpiration or by percolation. Total quantities over a year's period may be used in the calculation, since the accumulation of salt is a gradual process, and the leaching need not be continuous. In symbols,

$$P + I = \frac{E}{L} \quad \text{and}$$

$$I c_i \doteqdot L c_o$$

where P, I, E, and L are volumes of precipitation, irrigation, evapo-transpriation, and leaching, respectively; c_i is salt concentration in the irrigation water; and c_o is permissible salt concentration in the soil moisture. Solving for the leaching volume,

$$L = \frac{(E-P)\,c_i}{c_o - c_i} \qquad (6)$$

For alfalfa and other moderately salt-tolerant crops, the allowable total salt concentration, expressed as conductivity, may be as high as 8 mmhos. Local well waters in the San Gabriel Valley have conductivities in the neighborhood of 0. 6 mmhos, and Colorado River water about 1. 1 mmhos. The average annual leaching requirements for alfalfa and turf, estimated from the above formula, are as follows:

Annual Leaching Requirement, inches

	Alfalfa	Grass
Precipitation	20	20
Evapo-transpiration	43	36
Leaching Requirement:		
Well Water Irrigation	1.9	1.3
Colorado River Water	3.7	2.6

The amount of required leaching is very nominal, therefore, provided irrigation water of reasonably good quality is available. Occasional excessive rainfall and irrigation losses will normally be more than sufficient to take care of the required leaching, unless special measures are taken to prevent deep percolation. The minimum leaching requirement represents a long-term average value for the deep percolation which must be permitted if the surface soil is to remain fertile.

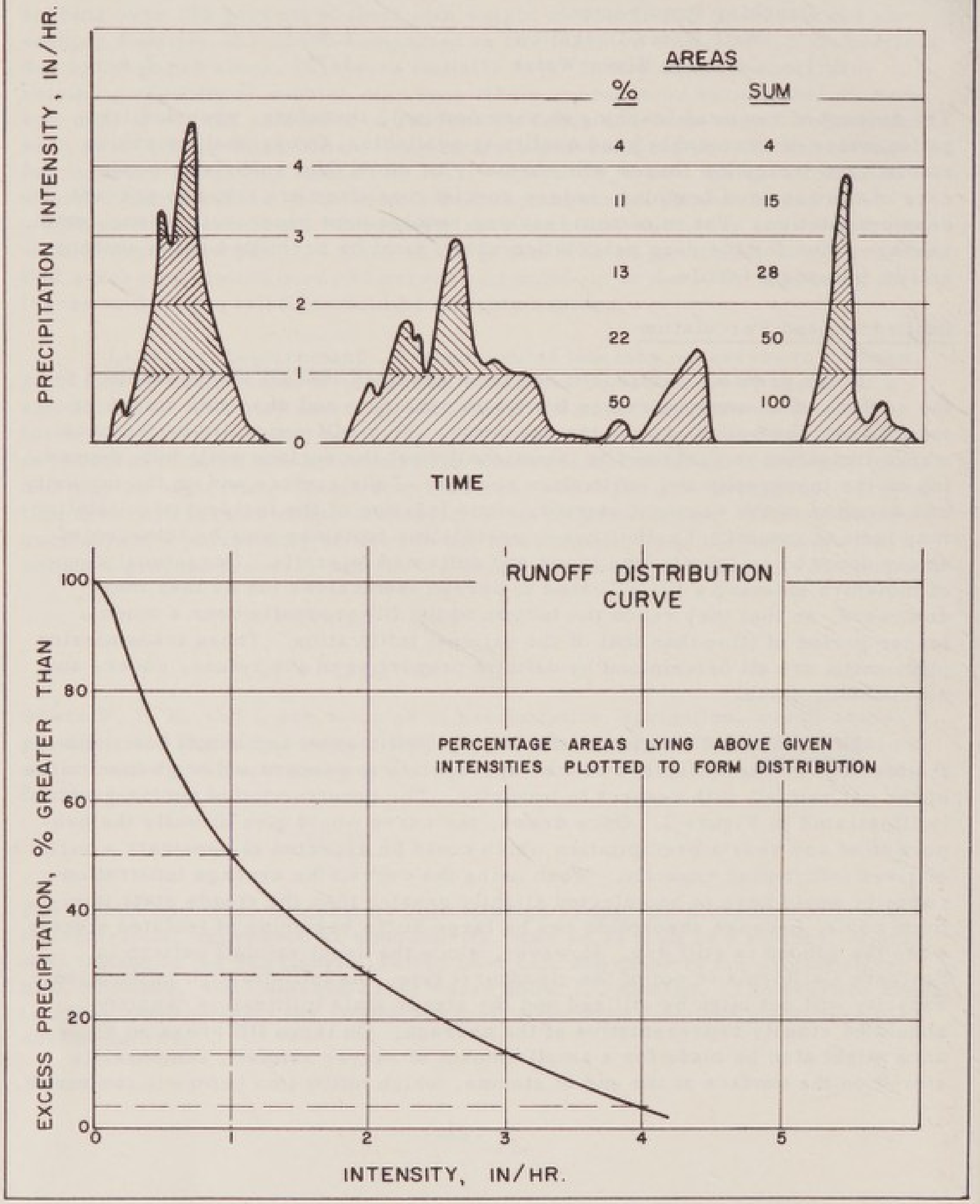
Infiltration and Percolation

In the previous sections it has been assumed that all water supplied to the surface of an unpaved refuse fill would penetrate and descend, unless removed from the soil by evapo-transpiration. This will generally be the case where irrigation is practiced to intentionally wet the surface soil, but, depending on the topography and infiltration capacity of the surface and on the intensity and duration of the seasonal storms, some fraction of the incident precipitation may instead run off. Furthermore, percolating moisture may be intercepted at any depth by an impervious layer and deflected laterally. Occasional slugs of moisture entering a fill will tend to spread themselves out as they move downward, so that they reach the bottom of the fill gradually over a much longer period of time than that of the original infiltration. These transmission phenomena are all determined by definite properties of the refuse, cover, and surrounding earth.

Infiltration: To estimate the average infiltration and runoff fractions of the incident precipitation it would be appropriate to prepare a distribution curve of the net rainfall with respect to intensity. The construction of such a graph is illustrated in Figure 7. Once drawn, the curve would give directly the proportion of any year's precipitation which could be expected to penetrate a surface of given infiltration capacity. When using the curve, the average infiltration capacity would have to be selected slightly greater than the steady state minimum value, because absorption can be large at the beginning of isolated storms when the ground is still dry. However, since the usual rainfall pattern in Southern California is not of the cloudburst type, the initially high infiltration capacity will not often be utilized and the steady state infiltration capacity should be closely representative of the average. On large fill areas an allowance might also be made for a small amount of excess rainfall, temporarily stored on the surface at the end of storms, which infiltrates before it can runoff.

FIGURE 7

CONSTRUCTION OF RUNOFF DISTRIBUTION CURVE



Another approach toward estimating average infiltration is illustrated by Figure 8. The upper graph shows rainfall intensity distributions which were observed at Edwardsville, Illinois, over a one-year period (20). They are different from the curve suggested in Figure 7, since they represent the distribution of total rainfall with respect to intensity, not that of the net rainfall in excess of infiltration at the indicated intensities. The lower curves were empirically prepared from City of Los Angeles data, and give infiltration rate versus precipitation intensity for surfaces of various permeabilities (21). By dividing the frequency curve into intensity groups, determining the percent infiltration for each group from the lower graph, multiplying the percent infiltration by the fraction of precipitation falling in each group, and summing, the total infiltrated fraction is obtained. Proceeding in this way with the total year's rainfall curve at Edwardsville, the infiltration would be 90, 75, and 50 percent of the rainfall, respectively, for sand, loam, and clay. However, the corresponding results when the low-intensity type rainfall curve of January-April is used are 97, 90, and 60 percent infiltration for sand, silt, and clay. It is clear that where an appreciable fraction of the rainfall comes at high intensity, the infiltration through a refuse fill can be greatly reduced by using a fairly impermeable cover, and providing adequate surface drainage. Unfortunately the steepness of the rainfall intensity distribution curves in the intensity range corresponding to usual infiltration rates makes it necessary that the distributions be determined with considerable accuracy and from data taken in the locality being investigated.

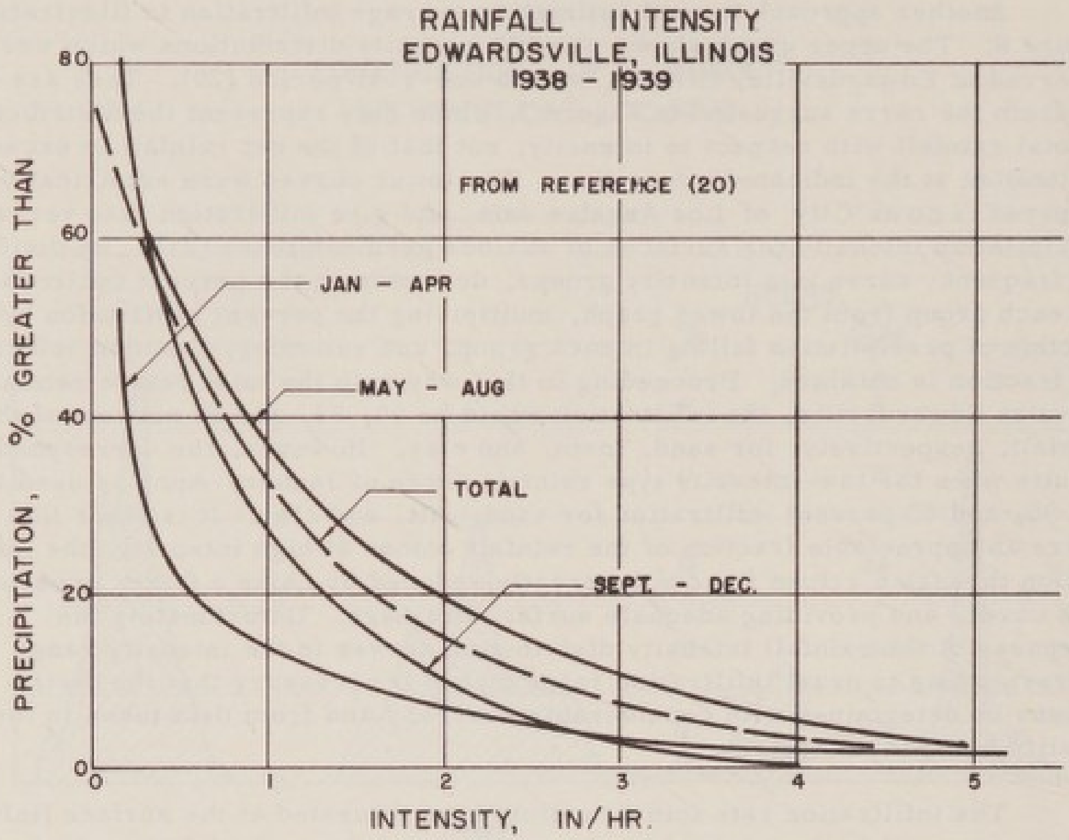
The infiltration rate into a medium kept saturated at the surface (infiltration capacity) is determined by the structure of the surface layer which actually contains percolating water. Free et al (22) measured infiltration rates on a large number of different field soils and concluded that the most important factors tending to promote infiltration were the non-capillary porosity and organic content of both surface soil and subsoil (non-capillary porosity is the fraction of air-filled voids in a porous medium at field capacity). Density and clay content of the subsoil were the most important characteristics tending to retard the infiltration rate. The rate values measured during the third hour of saturation of previously dampened soil ranged from nearly zero to five inches per hour. Musgrave and Free (23) had earlier noted that infiltration rates decrease with increasing soil moisture content and that a cover of close vegetation (alfalfa and bluegrass) permitted nearly double the rate on otherwise equivalent bare ground.

It has already been noted that the infiltration rate from saturation into a relatively dry soil decreases with time. Various formulas have been proposed to describe the infiltration rate curve. Horton (24) has suggested the expression

$$i = i_m + (i_o - i_m) e^{-k_f t} \qquad (7)$$

where i is the infiltration rate at any time t, i_o and i_m are the initial and ultimate rates, respectively, and   is a constant. Swartzendruber and Huberty (25)

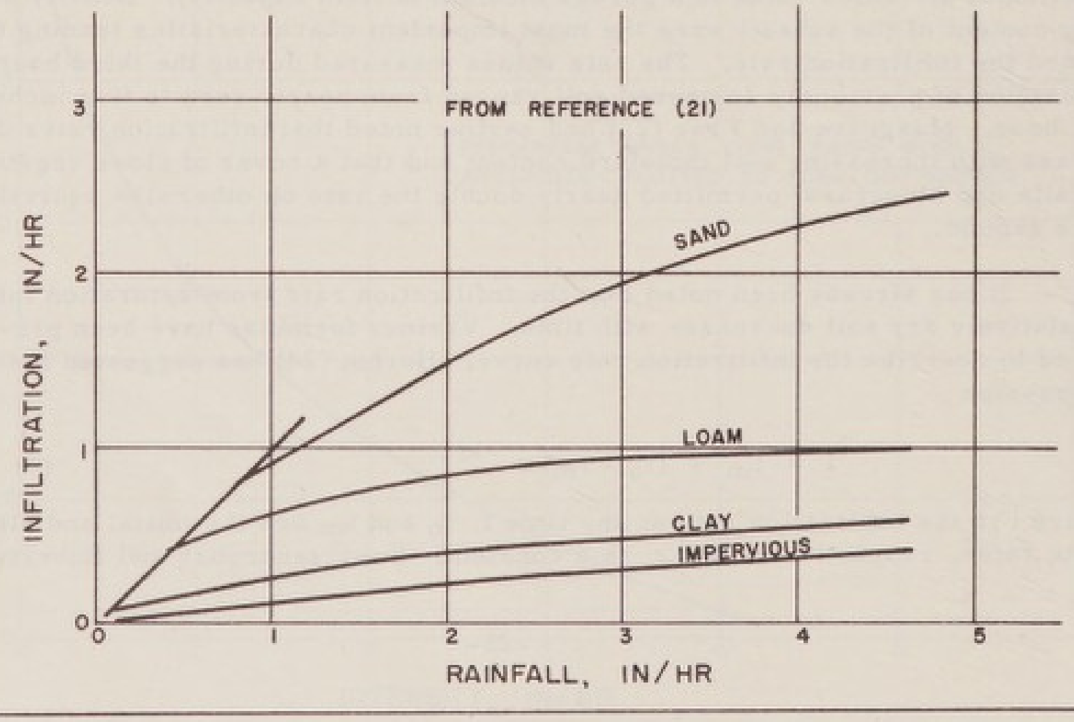
FIGURE 8



RAINFALL INTENSITY
EDWARDSVILLE, ILLINOIS
1938 - 1939

FROM REFERENCE (20)

RAINFALL-LOSS RATE CURVES



FROM REFERENCE (21)

have compared several proposed infiltration formulas, and have found that for a short-term measurement, the expression

$$i = At^B \tag{8}$$

is as simple and accurate as any. Philip (26) obtained the following formulas as an approximation to his solution of the differential equation of unsaturated flow for infiltration conditions:

$$i = i_m + \frac{B}{\sqrt{t}} \tag{9}$$

Both Philip's and Horton's formulas represent the infiltration rate as decreasing asymptotically to a final value greater than zero. However, Philip concluded from a comparison of the various expressions fitted to actual data that Horton's formula was not very satisfactory in following the initial sharply curved portion of the curve. In any case, the quantity of principal interest in estimating average infiltration is the final minimum rate, i_m. If the time required to substantially attain the minimum rate were large, the average infiltration rate might be appreciably higher than the minimum. This period may vary from almost zero where an antecedent storm has soaked the ground, to over an hour for certain soils in an initially dry state.

While the grain size and composition of a cover material have a predominant effect on infiltration rate, several other factors also influence the soil structure to a marked degree; for this reason it is not possible to accurately predict the rates which will be realized on a given area. Raindrops themselves falling on bare soil tend to compact the surface layer and wash fines into the larger pores, thereby effecting a rapid reduction in permeability. Most important is the effect of vegetation; the growth of roots creates large pores and promotes the aggregation of soil particles. Foliage protects the surface against the impact of drops and encourages the formation of a highly permeable surface layer of humus. It also retards surface flow velocity, so that the washing of turbidity is minimized and the time of water retention on the surface is increased. The one-hour infiltration rates for three general groups of soils are listed in Reference 12 as follows:

Table IV

Soil Infiltration Rates

Group	One-Hour Infiltration Rate Range, inches/hour
High (Sandy; friable silt loam)	0.5 to 1.0
Intermediate (Silt loam and light clay loams)	0.1 to 0.5
Low (Clays; heavy soils)	0.01 to 0.1

In the spreading of sewage effluent on very permeable sand at Azusa, it has been possible to sustain 0.6 inch/hour infiltration for 30 days or more despite clogging by suspended solids (27).

The range of "cover factors", to be applied to the above rate values for bare soil, is also given in Reference 12:

Table V

Infiltration Cover Factors

Cover		Cover Factor
Type	Condition*	Range
Grass	Good	3.0 - 7.5
	Medium	2.0 - 3.0
	Poor	1.2 - 1.4
Close Growing Crops	Good	2.5 - 3.0
	Medium	1.6 - 2.0
	Poor	1.1 - 1.3
Row Crop	Good	1.3 - 1.5
	Medium	1.1 - 1.3
	Poor	1.0 - 1.1

*Definitions: Good: Dense cover; area planted for several years or more.
Medium: Cover density and yield 30 to 80% that of "good" area.
Poor: Cover density and yield less than 30% that of "good" area.

Percolation: As water percolates downward through a dump cover, it may encounter layers of different structure and properties; in any event there will be a change between the cover and the refuse itself. Darcy's Law for unsaturated flow (Equation 2) indicates that any moisture movement must occur in the direction of decreasing gravity and capillary potential. The gravity potential gradient is constant, while capillary potential gradients may be very large and usually control the rate of flow throughout most of the moisture content range. From the suction curves of Figure 1, it may be seen that at a given suction, the moisture content in fine-grained media is much greater than in coarse media. Therefore, in order for water to pass from a fine medium into a coarse one, since flow normally can occur only in the direction of increasing suction (decreasing capillary potential), the fine medium must first become nearly saturated. This phenomenon has been demonstrated in soil columns by Colman and Bodman (28) and in beds of graded fine glass beads by Palmquist and Johnson (29). In the latter experiments, the movement into the coarse layer, when it finally occurred, was through isolated columns of wetted medium; most of the coarse layer never became damp. In the case of flow from a coarse medium into an underlying finer stratum, the rate of infiltration is controlled entirely by the fine particles (provided there is a fairly large difference in the two gradings). If water is applied more rapidly than the fine-grain infiltration capacity, either the upper layer will eventually become completely saturated, or water will flow off laterally.

A question preliminary to that of defining the behavior of moisture movement through refuse is to what extent the refuse resembles a normal porous medium. As deposited in a fill, it is extremely non-homogeneous, since large loads of predominately one material or another are constantly being deposited and do not receive appreciable mixing. Aside from the variations in composition within a dump, it is not certain that moisture movement obeys the same laws therein as those known to be suitable for granular media. It seems probable that refuse in general, and in particular combustible rubbish as it is normally collected, will have pore sizes distributed about two vastly different orders of magnitude. These will be (1) the interstices of the individual objects which compose the rubbish, namely paper, plant tissue, textiles, and other porous ingredients; and (2) the much larger voids scattered among the conglomerate of the component objects. The finer pores will exhibit the phenomena associated with capillarity--retention of water, transmission toward decreasing moisture content in all directions, and low hydraulic conductivity. The large voids will behave either as channels or air pockets, depending on whether they are or are not interconnected. If the foregoing hypothesis is correct, refuse should behave as a fine-grained porous medium until the small-scale pores in any portion of it become saturated. At this point there will be a large apparent increase in hydraulic conductivity as water begins to fill the large voids and flow through them.

Prevention of Percolation

The most obvious means of reducing the volume of water passing vertically through a fill is that of providing an impermeable layer at some level to intercept any incident moisture and divert it outside of the filled areas, or otherwise waterproofing. Paving the surface would probably involve the least risk of failure, since any flaws which might develop would be visible. However, the cost of paving would be significant, difficulty with maintenance due to continued uneven fill settlement might be experienced, and paving might conflict with the intended use of the site. Another possibility is that of using a relatively impermeable material for the final refuse cover. The percolation rate could be greatly reduced over that of sand, for example, but careful field studies would be necessary in order to accurately estimate the actual infiltration. There would be uncertainties regarding the perforating effect of roots and animal burrows. It is conceivable that with a sufficiently thick layer and effective surface drainage, rainwater could be entirely excluded. However, any irrigated plant growth would require up to four inches of leaching annually to prevent eventual retardation due to salinity. Mr. J. F. Furness, River Inspector of the Surrey County Council, England, has informed that recently permission was granted in that county for use of a dry sand pit as a disposal site, with the provision that the surface be waterproofed with clay and sloped for drainage.

Finally, the impervious layer might be placed at the bottom of the refuse fill. In this location there would be the advantages of having the impervious layer bedded on firm material which would not be likely to subsequently settle and roots and animals could not easily reach it. If clay were used, it could be kept continuously moist and therefore relatively impermeable at all times. Once placed, however, a bottom coating could not be inspected or repaired except at great expense.

Lee (30) has described the successful waterproofing of the lagoon at the San Francisco Exposition in 1937 by means of a clay lining. In that case the 10-inch clay layer, after being placed and compacted, became dry and developed large cracks. When the lagoon was filled the seepage amounted to 0.9 inches/day. To reduce the seepage, the lagoon was drained and refilled with sea water, which was allowed to remain until 40 inches had infiltrated. This treatment increased the proportion of sodium ions adsorbed on the clay with respect to the calcium and magnesium. The latter divalent ions had been effective in keeping the clay in a flocculated and hence permeable condition. When fresh water was put back in, the seepage rate dropped slowly as the excess salt water was leached out and the clay swelled, until it finally reached 0.1 inch/day in about four months.

If a bottom lining is to function effectively in intercepting percolation, the water which falls upon it must remain ponded or else be removed. Coating the sides of the pit as well as the bottom would retain the water, but if the lining were at all permeable the infiltration through it might increase with the rising head to the point where seepage was finally occurring at the rate of percolation. To minimize this possibility and also to avoid lining a great depth of pit wall, the impervious bottom could be sloped and covered with a layer of sand and gravel, so that all drainage would arrive at a sump structure, and could be pumped to a sewer whenever it accumulated. The walls would still have to be lined to a height of a few feet in order to retain the drainage within the pit.

Summary of Information Needed

Information presently available indicates that the possibility of obtaining deep percolation through a refuse dump located in Los Angeles County depends on the thickness and nature of the cover material, on the use of the covered site, and on characteristics of the precipitation cycle and of the refuse itself. Under continued adverse conditions (permeable cover, careless irrigation, poor drainage) percolation amounting to one or two feet a year is very probable. However, measures designed to limit infiltration or to limit the amount of water reaching the refuse surface, appear capable of reducing percolation to a negligible rate.

The principal types of data lacking for a confident appraisal of effects associated with the vertical movement of water through refuse are the moisture transmission characteristics of the refuse and the infiltration of precipitation in the vicinity of proposed dump sites. The specific evaluations to be obtained are the following:

A. Capillary Moisture Properties

1. Suction-moisture content curve of combustible or mixed rubbish, and

2. Capillary conductivity and diffusivity of rubbish. These data should be obtained in the laboratory and replicated in order to compensate for non-homogeneity.

3. Moisture holding capacity (field capacity) of rubbish in its original and decomposed states.

4. Moisture profile within a bed of rubbish during infiltration, including observation of its uniformity. The data of Items 3 and 4 would be most meaningful if obtained from actual fills or portions of fills, with controlled moisture application.

B. Infiltration

1. Determination of rainfall intensity distribution from gauges located as close as possible to likely sites.

2. Infiltration capacity of types of cover which might conceivably be used or proposed, and quantitative evaluation of infiltration actually realized through various cover types.

C. Miscellaneous

1. Suitability of rubbish as a growing medium for different plants, and effective depth of root zones.

2. Trial use of impermeable layers and drainage systems to intercept percolating water. Studies should be performed in pilot plants or in sections of full-scale dumps where water is available.

III. DECOMPOSITION PROCESSES

Freshly deposited refuse contains organic and inorganic substances which can be leached out immediately by vertically or horizontally percolating water. However, as decomposition proceeds, more of such materials will be produced or liberated. If the refuse is subjected to continuous or frequent leaching, the rate of decomposition will control the rate at which polluting substances are extracted from the fill. The rate and extent of carbon dioxide production are of considerable interest and are directly related to the decomposition process.

Basic Considerations

The natural decomposition of organic refuse is carried out through the agency of bacteria and other micro-organisms. The organisms use the organic matter as food, converting it to their own cell substances. The process of conversion requires chemical energy; this is supplied through biochemical reactions collectively known as respiration. If oxygen is present in the environment, the organisms which come to predominate will be those whose respiration consists of oxygen reacting with organic substances to produce carbon dioxide and water; organic nitrogen is released as ammonia:

Aerobic Respiration

$$CH_a\, O_b\, N_c + 1/4\,(4+a-2b-3c)\, O_2 \rightarrow CO_2 + 1/2\,(a-3c)\, H_2O + cNH_3 \qquad (10)$$

In the absence of molecular oxygen, respiration occurs as other energy-yielding reactions. The products of such anaerobic respiration are numerous, but many can serve as foodstuffs for other organisms and the final products, when a wide variety of organisms are participating in the fermentation, are principally methane and carbon dioxide gases:

Anaerobic Respiration

$$\begin{aligned} CH_a\, O_b\, N_c + 1/4\,(4-a-2b+3c)\, H_2O \rightarrow\ & 1/8\,(4-a+2b+3c)\, CO_2 \qquad (11)\\ & + 1/8\,(4+a-2b-3c)\, CH_4 + cNH_3 \end{aligned}$$

Anaerobic respirations are considerably less efficient than the aerobic in terms of the energy produced from a given amount of consumed organic matter; usually more than 90 percent of the material metabolized by an anaerobic organism is converted to respiration products and only ten percent or less to cell substances. However, the cell material of the organisms themselves is subject to decomposition and, after sufficient time under suitable environmental conditions, nearly all of the readily decomposable organic matter originally present will have been converted to the aerobic or anaerobic respiration products mentioned.

The organic matter of refuse contains a small percentage of nitrogen, which enters in the reactions of decomposition and in fact is necessary for the life processes of the organisms. As fermentation proceeds the nitrogen is released to the environment, where it is available for reuse by other organisms. Under anaerobic conditions, the released nitrogen is in the form of ammonium salts, but if free oxygen is available some or all of the ammonia nitrogen may be oxidized to nitrate and its predecessor, nitrite.

Initially refuse contains mineral substances in quantities usually small compared to the organic content. These substances may be of varying degrees of solubility, ranging from virtually zero for earth and rubble to readily soluble salts, such as chloride and sulfate. The salts may be deposited on the surface of refuse particles or entrapped, for example, within the cells of vegetable matter. The superficial soluble substances can be leached out of the refuse at any time, but the internally held salts can only escape by slow diffusion or be released when their organic containers decompose away. Certain substances, such as calcium carbonate and iron metal or oxide, are insoluble at high pH, but are easily dissolved in acid. Thus, if the carbon dioxide from aerobic or anaerobic respiration accumulates to any extent, percolating waters in which it dissolves will become acid and begin to dissolve any acid-soluble substances present in or beneath the refuse.

Some of the organic respiration products are soluble and may be leached out of the refuse, along with very fine, suspended organic particles. After leaving the refuse, such materials will normally be surrounded by sufficient oxygen to permit their aerobic breakdown. Decomposition processes therefore need not be confined to the fill itself, but may continue in the adjacent ground and water, taking up oxygen and producing carbon dioxide.

Composition of Refuse

The material being deposited at present in Los Angeles County land fills is mostly combustible rubbish; that is, relatively dry substances which can be burned. In 1956, the Los Angeles County Sanitation Districts (19) estimated the per capita production of combustible rubbish at 2.1 lb/day, with garbage and non-combustible rubbish each at 0.5 lb/day, all subject to routine collection. An additional 0.5 lb/day was allowed for materials, combustible and non-combustible, not subject to routine collection. Garbage collected separately is normally fed to hogs, and a large part of the non-combustibles collected separately is salvaged so that only some fraction of the collected amount of these substances ever reaches the large refuse fills. However, recent trends have been toward the combined collection and burial of all of these several categories of refuse in one single effort.

Physical Composition: Probably the most practical means of characterizing a given refuse to distinguish it from that being collected or deposited somewhere else is by defining its physical composition. In this sense there are only a relatively limited number of major refuse components; they in turn have a limited range of characteristics. In addition to classifying refuse as garbage, combustible rubbish, and non-combustible rubbish, these categories can be broken down into the materials they comprise.

Table VI lists the results of two reasonably large samplings of collected rubbish. The summer sample reported by the Sanitation Districts was residential combustible rubbish, whereas the ASCE Committee data was on material of whatever nature delivered to a dump. Some of the differences in volume percentage composition can be attributed to this difference in source, but the major variance is due to the difference in season. In summer the plant materials amounted to 60 percent by volume, compared to about 10 percent in winter. Paper products were 64 percent in winter, but only 34 percent in summer. Of all the substances listed, the only ones susceptible to rapid decomposition are the vegetation and garbage. For the summer sample of combustible rubbish, these amounted to 64 percent by weight.

Table VI

Physical Composition of Rubbish

	July 1954 (19)			Dec 1957 (31)
	% by Weight	Density lb/cu yd	% by Volume	% by Weight
Paper and Cardboard	30.2	250	34	64
Rags	1.0	185	1	-
Vegetation - Small	21.7	565	11	10
- Large	42.4	240	49	
Wood	1.9	185	3	14
Garbage	0.5	-	1	12
Miscellaneous Combustible	1.3	-		
Non-Combustible	1.0	390	1	
	100.0		100	100

Reference 19: County Sanitation Districts of Los Angeles County, Planned Refuse Disposal (1955).

Reference 31: Am. Soc. Civ. Engrs., Comm. on Sanitary Engineering Research, "Refuse Volume Reduction in a Sanitary Landfill". Proc. Am. Soc. Civ. Engrs., 85, No.SA6, 37 (1939).

The composition of refuse collected in urban areas in more northern latitudes is very different from that of Los Angeles. Table VII gives the physical composition of house refuse from New York in 1940 (32), from Watford, England about 1955 (33), and from Manchester, England in 1932 (34). The samples exhibit similar trends, and differ mainly in the relative amounts of fine dust and cinders they contain. Compared to Los Angeles County refuse, the refuse from colder climates contains a much larger proportion of garbage and much less paper and garden vegetation. Ashes and cinders form a very small fraction of the Los Angeles rubbish since heating is done almost exclusively with natural gas. The moisture content of the New York and Manchester refuse was about 20 percent, as compared to 45 percent for Los Angeles rubbish.

Table VII

Physical Composition of Urban Household Refuse

	New York(32)	Watford England(33)	Manchester England (34)			
	Percent Collected Weight	Percent Collected Weight	Percent Collected Weight	Percent Dry Weight	Volatiles, % Initial Dry Weight	
					Initial	After 12 mo
Inorganic						
Fine Dust		18.1	34.6	43.1	9.1	9.1
Ashes	7.8					
Cinders		18.2	14.5	18.1	12.4	12.4
Metals	6.6	6.5	4.2	5.5	0	0
Glass	6.3	7.7	3.0	3.9	0	0
Miscellaneous		1.8	1.7	2.2	0	0
	20.7	52.3	58.0	72.8	21.5	21.5
Organic						
Fine Dust		(incl above)	0.8	0.5	0.5	0.3
Vegetable and Putrescible	43.8*	22.2	24.3	5.7	4.8	0.5
Bones		0.6	0.5	0.5	0.1	0.1
Paper	25.5	20.8	12.5	15.6	14.7	7.4
Wood	5.9					
Rags	1.8	2.1	1.2	1.5	1.4	1.0
Miscellaneous	2.3	2.0	2.7	3.4	3.2	3.0
	79.3	47.7	42.0	27.2	24.7	12.3
Total	100.0	100.0	100.0	100	46.2	33.8
			Original Moisture Content: 23.8% Collected Weight			

* Food Refuse

Table VIII gives the percentages of constituents in the municipal refuse of Chandler, Arizona, as determined by year-long sampling. Shown for comparison is composition of Berkeley, California, refuse as found in eleven truckloads from a cross-section of residential and light commercial areas. The percentage of combustible materials is greater in the Chandler refuse, probably because of greater production of garden vegetation in the warmer climate.

Table VIII

Physical Composition of Mixed Municipal Refuse

	% Collected Weight	
	Berkeley, Calif.[1]	Chandler, Ariz.[2]
Compostable Material		
Paper		30.0
Vegetation		35.7
Wood		1.8
Garbage		12.0
Fines		7.0
	67.7	86.5
Cans	9.8	5.7
Other Metals	0.9	0.2
Glass	11.7	4.5
Rags	1.6	1.0
Miscellaneous	7.9	2.1
	100.0	100.0

[1] From Reference 37.

[2] From data furnished by P. P. Maier, U.S. Public Health Service.

The physical composition of residential garbage at East Lansing, Michigan has been reported by Mallison and Hohloch (35). As collected separately and wrapped, the average analysis includes paper at 33 percent of dry weight, and water 143 percent of dry weight.

The average composition of non-combustible rubbish in Los Angeles County as given in Reference 19 is the following:

	Percent by Weight
Cans	35
Other Metals	10
Glass	15
Rubber	2
Moisture	3
Other	35
	100
Density	350-400 lb/cu yd

Chemical Composition: As far as effects on ground water are concerned, the principal interest of refuse composition is in determining the quantities of polluting substances which may become available, and the rate at which such substances are actually liberated. A detailed knowledge of organic compounds present would be of value only if the products and rates of their breakdown were also known; this is not the case. The activities of the responsible microorganisms are affected by numerous uncontrollable factors and a given compound may give rise to many products in various proportions. It is therefore more practical to study decomposition processes directly in the heterogeneous mixtures which actually occur. In the absence of good data, however, the composition with respect to chemical elements yields estimates of the maximum quantities of various substances which could reach the ground water over a period of time.

Table IX gives partial analyses which have been made on several types of refuse. Because of the scantiness of the determinations, it is not possible to make good comparisons of the different materials. Their sources and physical composition are reflected in their volatile contents; the refuse of New York and Manchester, which contained large quantities of ashes, cinders, and dust, was less than 50 percent volatile, while materials composed chiefly of combustible rubbish were about 70 percent volatile. The volatile percentage of garden vegetation was about 80.

The carbon analyses for combustible refuse materials are all of the same order of magnitude and from these figures it may be concluded that in Los Angeles rubbish, carbon amounts to 50-55 percent of the volatile fraction. Total nitrogen in vegetation is about two percent, and nearly zero in paper; Table IX indicates that the nitrogen in rubbish volatiles is about 1.5 percent.

Reliable data which would indicate the quantities of minerals and minor elements to be found in refuse are not widely available. The figures in Table IX for potassium suggest that dry combustible rubbish probably contains about 0.5 percent. Pomeroy (36) obtained about twice as much sodium as potassium from a sample of Long Beach rubbish. Probable phosphorus concentrations appear to be less than 0.5 percent.

Some indication of the magnitude of various components in mixed household refuse can be obtained from Merz's results on leaching through a test bin (18). The total weight of the components leached can be calculated from the concentrations and volumes of percolate extracted during the 18-month period of the experiment. While the concentrations of most substances had dropped to small fractions of their initial values by the end of the experiment, they were still being extracted at an appreciable rate. Also, some of the volume figures are missing and the most that can be said about the total weights of the various substances in the initial refuse is that they were somewhat greater than the amounts extracted. The refuse weight is not known either, but assuming that its initial moisture content was 50 percent and that after it had compacted

Table IX

Chemical Composition of Refuse

Percent of Dry Weight

	Volatile	BOD	COD	C	Total N	P	Na	K	Chloride Cl
N.Y. Fines & Organics (32)	12	0.6	3.6		0.16				0.08
Manchester (England) Refuse (34)	46				1.0				
Watford (England) Organics* (33)				35	0.62				0.17
Long Beach Refuse (36)							0.52[1]	0.27[1]	
Berkeley Compost Mat'l (37)	71.5			35.7	1.07	0.51		0.69	
Chandler, Arizona Refuse (Wiley)	68			39	0.56				
Garden Trimmings (31)	81.3								
Garden Trimmings (37)	79.0			39.2	2.03	0.22		0.84	

* Includes combustible items and fine dust, but not cinders.

[1] Assuming moisture content 100% of dry weight.

initially to a depth of 5.5 feet, its density was 550 lb/cu yd, an approximate dry weight value can be obtained upon which to base percentages of extracted substances. The results are as follows:

Table X

Material Leached From Riverside Landfill*

Substance Extracted	Percent of Dry Refuse Weight
Alkalinity (as $CaCO_3$)	0.78
Calcium (Ca)	0.16
Magnesium (Mg)	0.029
Sodium (Na)	0.15
Potassium (K)	0.18
Total Iron (Fe)	0.019
Chloride (Cl)	0.22
Sulfate (SO_4)	0.022
Inorganic Phosphate (PO_4)	0.0014
Organic Nitrogen (N)	0.032
Ammonia Nitrogen (N)	0.072
BOD	2.54

Leached Volume: {0.21 gal/lb
3.5 in/ft

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* Based on data of Merz (18), assuming dry density of 275 lb/cu yd.

Some of the substances listed, notably calcium, iron, and phosphate, were probably present largely as combinations of low solubility; for them the extracted quantities represent only a small fraction of the total.

Aerobic Processes

Most sorts of organic matter are capable of being utilized as food materials by micro-organisms of some description. The particular organisms which prevail in any small particle of decomposing organic matter will be those most adaptable to the existing environmental conditions. In a medium such as refuse, the major selective factors, other than the composition of the refuse, are the presence or absence of oxygen, the temperature, and the moisture content. While the simpler organic compounds can be directly assimilated by the decomposing organisms, more complex materials must first be broken down by externally secreted enzymes to simpler units. The respiration products of a particular organism may be organic compounds, but these in turn may serve as raw material for some other organism. In aerobic decomposition, the ultimate fate of all organic carbon is its conversion to carbon dioxide; water is also produced. Organic nitrogen appears first as ammonia, which subsequently may be oxidized to nitrite and nitrate in the respiration of certain organisms. It is always possible for intermediate products to be removed from a fill (by leaching, for example), but these will continue to decompose, provided they remain in contact with organisms capable of operating on them in the new environment.

Organic matter leaving a fill with moving water can be conveniently measured as biochemical oxygen demand (BOD), which is the amount of oxygen utilized by micro-organisms in decomposing the substances. However, BOD analysis of the solid matter composing the body of a fill would give a smaller value than the amount of oxygen which the same material would ultimately consume; BOD is actually the oxygen taken up under specified conditions in five days, and hence measures only organic compounds which are readily decomposable.

Since the different organic substances present in refuse decompose at various rates, it is to be expected that in freshly deposited refuse there will be rapid initial decomposition activity which will decrease as the readily attacked components become depleted. An investigation of the specific breakdown reactions occurring in refuse would not be practical, but qualitatively it seems apparent that the easily metabolized materials will include sugars, starch, fats, and food proteins. The slowly decomposable fraction will chiefly comprise fibrous cellulosic materials. Actually cellulose is subject to the action of certain bacteria and fungi, but rapid decomposition of any substance requires that physically it be finely divided in order to offer a large surface accessible to the organisms. Organic structural materials such as wood fiber are therefore very slow to decay.

In order to establish the maximum rate at which refuse may be expected to decompose aerobically, it is instructive to consider the results of experiments on refuse composting, wherein the objective is rapid stabilization. Following is the description of an experimental composting procedure (37): The wood, rags, and non-combustible items were removed from Berkeley, California, municipal refuse, and the remaining fraction was mechanically ground and spread in windrows to compost. Every few days the material was turned over; temperatures were taken frequently. In from four to eight days, the temperature within the heap rose to a maximum of about 165°F, at which value it remained for several days and then began to drop. The compost was considered finished and the experiments were terminated when the temperature had fallen to about 130°F. The composting period ranged from two to three weeks and apparently depended on the ratio of carbon to nitrogen in the initial mixture; the richer the refuse was in nitrogen, the more quickly it composted. Reductions in carbon and nitrogen during the process varied considerably, but the carbon loss was generally 50 to 55 percent, and that of nitrogen, 30 to 40 percent. Moisture contents between 40 percent and 70 percent maintained a maximum composting rate. An excessively damp mixture tended to become anaerobic, presumably because of reduced accessibility of oxygen to the interior of the pile.

Aerobic respiration is equivalent to combustion and the same amount of heat is liberated during the decomposition of an organic substance as would result from its being burned. Heat effects in a refuse fill are of some interest because they influence the density of the refuse atmosphere and thereby affect gas movement. As an illustration of aerobic heat phenomena, a heat balance can be made using data from the Berkeley composting experiments described above. In one particular test, the mixture had the following composition at the beginning and end of a six-day period:

	Percent of Dry Weight	
	Beginning	End
Ash	24	38
Carbon	35	31
Water	98	65
Temperature	14°C	68°C
Average Temperature	58°C	

Assuming that the ash fraction was unchanged during the process, the reductions in the original carbon and moisture were 43 percent and 58 percent, respectively. It is further assumed that the compounds undergoing decomposition were carbohydrates such as starch or cellulose; the reaction was then:

$$C_6 H_{10} O_5 + 6O_2 \qquad 6CO_2 + 5H_20$$

The water produced in the reaction amounted to 0.19 gm/gm dry solids (which in a refuse fill with density 550 lb/cu yd and 50 percent moisture, would be 0.4 in/ft of fill depth). During the composting, oxygen was consumed and was replaced by carbon dioxide. Heat was liberated in the reaction; it was disposed of in warming the refuse, in evaporating some of the water present, and in heating the exhaust gases, which were nitrogen and carbon dioxide. The details of the heat balance are given in Appendix A. Of the heat produced, only 65 percent can be accounted for in the ways just mentioned. In addition, some heat was absorbed chemically in the growth of the micro-organisms, but the heat efficiency of biochemical synthesis is very low and would not have amounted to more than 30 percent. The remaining 35 percent or more had to be dissipated to the atmosphere by ventilation. Since the same maximum temperature (165°F) was reached and sustained repeatedly in different trials, it is very probable that this temperature represents the highest under which the organisms could continue to survive actively and that the rate of decomposition during the period of maximum temperature was being limited by the rate of heat removal.

In the Berkeley composting experiments, conditions for rapid decomposition had been optimized; the refuse was crushed and homogenized and small enough piles were used that air could enter by draft or diffusion to supply oxygen. The refuse in a covered fill is in a coarse, non-uniform state, and adequate oxygen is by no means assured, so that aerobic decomposition will normally proceed at a much lower rate than in composting. Eliassen (38) has reported laboratory results taken from the 1940 New York landfill study indicating that the rate of breakdown of food refuse decreases markedly as it is mixed with increasing amounts of paper, and attributed the effect to separation of the putrescible particles. Also demonstrated was the dependence of breakdown rate on moisture; maximum activity, as measured by the rate of carbon dioxide evolution, was obtained with moisture contents between 40 and 80 percent (37, 38).

The temperature of buried refuse appears to follow a typical pattern of rising to a maximum value within a comparatively short time after being placed and then gradually falling to that of the surrounding earth over a period of several years. Apparently the initial use corresponds to rapid aerobic breakdown of easily decomposable materials. Nedry (39) has suggested that the temperature is in fact a good measure of the extent of decomposition. His experience with a six-foot-deep sanitary fill at Coeur d'Alene, Idaho, is that the refuse temperature rises 20°F above the ground temperature in three months, and is back down in two to three years. Dunn (40) has found maximum temperatures of 101°F in the Seattle Union Bay fill, occurring two weeks after completion of 20-foot refuse depth. In the New York Landfill Survey (32) temperatures were measured at various depths for about two months after filling. A maximum of about 160°F was obtained at the three-foot depth after about a week; the temperature then dropped slowly and irregularly to about 140°F after two months. The temperature at seven feet required two weeks to reach a maximum, but then remained close to the three-foot temperature. At eleven feet the temperature remained about 55° lower than the temperature at higher levels.

The ASCE Committee on Sanitary Engineering Research has reported temperature measurements at six-foot depth in a fill at Monterey Park, California, which was receiving mostly combustible rubbish (31). The temperatures ranged from 85° to 122°F, and averaged 98°. The greatest value was found in the oldest fill area, which had been in place for 1-1/2 years. Temperature readings were included in the Riverside experiments of Merz, and were summarized in the report (18) as follows:

	Bin No. 1	Bin No. 2	Landfill
Max. Temp. reached, °F	138	157	124
Days to reach Max. Temp.	8	8	2
Temperature after 30 days, °F	114	153	96
after 60 days	100		92
after 90 days	97		90
after 120 days	88		88
after 1 year	76		94
after 17 months	90		84

It appears that fill temperatures occasionally do approach the 165°F maximum obtained in composting, but that weeks rather than days may be required to reach it, and usually somewhat lower maxima are observed. In composting, the same aerobic decomposition process which occurs in a landfill is executed in a much shorter time. Widely varying temperatures within the same fill, such as reported by the ASCE Committee, are suggestive of variable decomposition rates due to the heterogeneous distribution of easily decomposable materials.

The state of decomposition of refuse as judged by its physical appearance has been noted occasionally in the literature. Schneider (41, 42) has described the stabilization of household refuse in a New Orleans fill, in which the material was sufficiently innocuous after three years that it could be re-excavated. The rapid breakdown was attributed to a high annual rainfall (60 in/yr) and to the warm humid climate. On the other hand, test holes dug in a 12-year old San Francisco fill indicated very little progress in the decomposition (76). Vegetables were still intact and cans were still shiny. With respect to the description of fill material which has undergone appreciable decomposition, observations made in the Riverside bin studies (18), the New York Survey (32), and the British experiments at Bushey (33) are rather uniform. After a year or so, garbage and fine vegetation was no longer identifiable, but wood and paper were comparatively unaltered. The paper was soaked and disintegrated readily, but the print was still legible.

Included in Table VII is the volatile content of the Manchester, England, refuse at the beginning and end of a one-year period in the fill. The reduction was about 27 percent; excluding the volatile matter contained in cinders, the percentage destruction was 37. Data supplied by J. S. Wiley from experiments at the Phoenix Field Station of the U. S. Public Health Service indicate changes in the household refuse buried in a small fill for about one year. Reductions in volatile matter and carbon were about 25 percent.

Reduction in organic nitrogen of refuse apparently lags slightly behind carbon destruction. Probably carbohydrates are more rapidly handled than the nitrogenous substances, and some released nitrogen will be retained by the organisms as a necessary growth element. Also, the ammonia released upon breakdown is volatile only at high pH; as ammonium salts it will remain in the refuse until leached. In the British Manchester study (34) analyses of the refuse for various forms of nitrogen were made during the decomposition process. The data are not sufficient to permit calculation of removals, but the changes in the distribution of the nitrogen among its various forms is evidenced. Most of the nitrogen, which amounted to about one percent of the dry refuse, was present throughout as organic nitrogen. Initially ammonia nitrogen was about 20 percent of the total; but it decreased to less than half that value in one year, while nitrate, initially zero, increased to as much as 0.1 percent. Other data demonstrating nitrification are given in the recent British government report (33). In leached aerobic samples of refuse, nitrate and nitrite began to appear at the expense of ammonia and organic nitrogen as soon as the concentration of organic matter in the percolate had fallen off to a low value.

It may be concluded that although a composting operation can accomplish 50 percent reductions of volatile matter and carbon in two or three weeks, the same destruction in a landfill takes at least a year, and normally will take a much longer time. The probable retarding factors are the coarse unmixed condition of the refuse particles, poor oxygen supply, and excessively low or high moisture content.

Anaerobic Processes

If oxygen is excluded from any portion of a refuse fill, aerobic decomposition is impossible, and only anaerobic or facultative organisms will be able to grow. In this case, carbonaceous matter is converted to carbon dioxide and methane, the relative quantities of each depending on the compounds undergoing breakdown. Equation 10 indicates this relation generally for any compound. Carbohydrates produce methane and carbon dioxide in equal amounts; proteins result in slightly more methane than carbon dioxide, and fats give more than twice as much methane as CO_2.

Both gases are regularly found in refuse fills, but in widely variable proportions. The percentages are determined not only by the compounds being anaerobically decomposed, but also by the rate of aerobic carbon dioxide production and by the rates at which the gases are being removed. The source and movement of the gases therefore cannot be confidently ascertained from gas analysis alone. However, the presence of methane is a sure indicator of anaerobic action, just as carbon dioxide in the absence of methane demonstrates the functioning of aerobic processes.

In contrast to the large heat effects of aerobic action, anaerobiosis is accompanied by comparatively small heat releases, and in fact some of the regularly occurring reactions absorb heat. However, rates of anaerobic breakdown are increased at higher temperatures, at least up to about 130°F.

The rates of anaerobiosis which prevail in actual refuse fill have not been quantitatively observed; since both aerobic and anaerobic action may occur simultaneously in different parts of the same fill, and succeed each other as the decomposition continues, it is not a simple matter to distinguish them. Qualitatively it would appear that the normal series of events is for the fill to undergo rapid aerobic decay when the refuse is first put in place and then gradually become anaerobic as easily digested substances become depleted and the slower heat production no longer encourages oxygen entry by convection. After a period of anaerobiosis which may last several years, decay has slowed to the point where the aerobic processes can again be supported on the oxygen diffusing into the fill at a nominal rate. In an experimental fill at Bushey, England, anaerobic conditions set in shortly after placement and persisted for the entire observation period of 33 months (33). Older fills excavated in the same area showed increasing decay of the refuse with age, but no return of aerobic conditions until between five and ten years. In the 10-year-old fill even the paper had disappeared and iron was rusty. These fills had been subjected to heavy rainfall and percolation.

Rogus (43) has indicated that the New York City landfills are aerobic in the top two to four feet, but remain anaerobic below that depth. Gas analyses made during the Riverside study (18) showed that methane was generally present. Percolate from Test Bin No. 1 was noxious for over a year, and anaerobic decay was apparently more active in it than in the second test bin, which received only natural precipitation and had remained fairly dry.

In addition to the anaerobic respirations of organic compounds, there are energy-yielding reactions which involve only inorganic substances. One of these is dentrification, or the reduction of nitrate to nitrogen gas. This will not normally be a part of refuse decay, since nitrate is not usually present. However, the production of sulfide from sulfate appears to be very common. In England, several instances have been reported (44, 45) in which refuse dumped into water-filled gravel pits caused anaerobic conditions; hydrogen sulfide production was encouraged by high sulfate concentrations leached from the ash fraction of the refuse and appreciable atmospheric nuisance resulted. Data from the experimental fill at Bushley, England (33) include some interesting effects in connection with sulfate reduction. Percolating rainwater leached continuous low concentrations of sulfide; they were as high as 30 ppm near the beginning of the experiment, but had dropped to about two ppm after 2-1/2 years. Sulfate, however, appeared in high concentrations only after winter periods of wet weather and was very low during dry seasons. Sulfate which had been reduced within the fill had remained as insoluble sulfides, probably of iron, judging from analysis of the material excavated at the end of the study.

Summary of Needed Information

The physical and chemical composition of Los Angeles County refuse is not too precisely known, but could be determined at any time by appropriate sampling procedures if it were desirable. Composition data would be of some value in characterizing and comparing refuse from different sources, but the important question bearing on potential pollution is not the total content of the refuse in undesirable substances, but rather the rate at which they will leave the fill. The polluting substances to be considered are either dissolved solids or gases; the former may leave a fill only when leached, but the gases may move under the influence of gravity and pressure difference. Some of the solids become leachable only upon decomposition, and gases are produced only as the result of decomposition. The following points should be investigated:

1. Establishing of the factors which determine the pattern of aerobic and anaerobic decomposition. Since the results desired are mainly

qualitative, the investigation could be most profitably carried out on an existing full-scale landfill.

2. Determination of aerobic and anaerobic decomposition rates under the range of conditions which would normally prevail in a refuse fill. The measurement of rates should be based on gas production.

3. Measurement of release of leachable material during the course of decomposition. Items 2 and 3 would most conveniently be performed in the laboratory where sampling and control would present no unusual problems.

IV. GAS PRODUCTION AND MOVEMENT

Unlike moisture, gases are not attracted appreciably to the particles of granular media, and are therefore free to circulate wherever they are impelled by unequal pressures. The most important gaseous product of refuse decomposition is carbon dioxide; it is quite soluble in water, and upon being dissolved forms a weak acid capable of dissolving calcium carbonate and other minerals quite insoluble in pure water. For the most part such materials susceptible to the dissolving action of carbonic acid are undesirable in water supplies. Pollution of the ground water by this means, while possibly not as severe as that associated with substances actually leached out of a refuse fill, may be potentially difficult to prevent.

Basic Considerations

The composition of the gas produced upon decomposition of refuse depends upon the original composition of the refuse and on whether or not oxygen is admitted. As indicated in Section III, the over-all reactions for the complete decomposition of refuse having the empirical formula $CH_aO_bN_c$ are the following:

Aerobic Process

$$CH_aO_bN_c + 1/4\,(4+a-2b-3c)\,O_2 \longrightarrow CO_2 + 1/2\,(a-3c)\,H_2O + cNH_3 \qquad (10)$$

Anaerobic Process

$$\begin{aligned} CH_aO_bN_c + 1/4\,(4-a-2b+3c)\,H_2O \longrightarrow\ & 1/8\,(4-a+2b+3c)\,CO_2 \\ & + 1/8\,(4+a-2b-3c)\,CH_4 + cNH_3 \end{aligned} \qquad (11)$$

It may be seen that anaerobically one mole of gas is produced from each gram atom of carbon consumed, while in aerobic breakdown the volume of carbon dioxide produced may be greater or less than the volume of oxygen utilized. In cellulose and other carbohydrates there are exactly half as many oxygen atoms as hydrogen atoms, but most other common natural organic substances contain smaller amounts of oxygen with respect to hydrogen. Thus in the aerobic stabilization of refuse, slightly more free oxygen will be consumed than carbon dioxide produced, and there will be a slight net inflow of air into the fill. Since oxygen is available within the refuse when it is first placed, the initial decomposition will be aerobic, but unless diffusion and convection provide adequate mixing of the internal and external atmospheres, the oxygen in the interior of the fill will be soon depleted, and thereafter the decomposition will proceed anaerobically. Once anaerobiosis sets in there is a net production of gas, which, in the absence of compensating factors, will tend to drive away outside air and thereby maintain the anaerobic state.

The movement of gases through porous media is governed by Darcy's Law when the flow is laminar. However, the conductivity of a medium varies inversely as the viscosity of the fluid, and since carbon dioxide is less than 1/50 as viscous as water, its flow rate would be 50 times as great for the same pressure drop. Turbulent interstitial flow is therefore quite possible for gas in natural underground situations, and in this case the flow rate is no longer proportional to the pressure drop. But also, because of the relatively high conductivity of porous media to gases, the latter will readily escape from a source into the surrounding medium without the creation of large pressure differences.

Whether the rate of gas flow is proportional to the pressure gradient or not, it is convenient to modify the equation to account for compressibility whenever large pressure differences are involved. Darcy's Law is still

$$v = -\frac{k_g}{w_g} \text{ grad } p \tag{12}$$

where v is the volume flow per unit area of medium, grad p is the pressure gradient, w_g is the unit weight of the gas, and k_g is a constant characteristic of the medium and gas. For dry medium, and gas at pressures somewhere near atmospheric, k_g is related to the hydraulic conductivity as follows:

$$k_g = k \frac{\nu}{\nu_g} \tag{13}$$

where ν and ν_g are the kinematic viscosities of water and gas. As the gas flows through a given region its volume will increase downstream because of the falling pressure, but its mass rate of flow will remain constant. If the gas remains at constant temperature, which will be nearly the case except at very high velocities or when there are natural temperature variations in the medium, it will obey Boyle's Law,

$$pV = p_o V_o \tag{14}$$

where V is the specific volume of gas at pressure p, and V_o its specific volume at pressure p_o. The mass rate of flow, expressed as the volume flow of gas at some standard pressure (p_o) is then

$$\begin{aligned} v_o &= v\,(V_o/V) = v\,(p/p_o) \\ &= -\,(k_g/w_g p_o)\,(p \text{ grad } p) \\ &= -(k_g/2w_g p_o) \text{ grad } p^2 \end{aligned} \tag{15}$$

Thus it may be seen that Darcy's Law for isothermal gas flow is that the flow rate is proportional to the gradient of the square of absolute pressure. For flow rates sufficiently great that Darcy's Law ceases to be accurate, the equation can be modified by the addition of an exponent to the pressure gradient (47):

$$v_o = - \frac{k_g}{2w_g p_o} (\text{grad } p^2)^{1/n} \qquad (16)$$

In addition to the movement of gases brought about by pressure differences, transfer can also occur at significant rates by means of molecular diffusion. Fick's Law applies, and can be written

$$q = - D_p \text{ grad } c \qquad (17)$$

where q is the transfer rate per unit area of medium, D_p the molecular diffusivity, and grad c the concentration gradient. The diffusivity in the porous medium will be less than that in free space because of the volume occupied by solid particles which is not available for transfer.

Gas Quantities

As an indication of the amounts and proportions of gases which might be produced from decomposing refuse, Table XI gives computed quantities based on the reactions represented by Equations 10 and 11. By "exhaust gas" is meant the gas composed of respiration products and the nitrogen remaining after the removal of oxygen from air to form the product gases.

The composition and production rate of gas will vary between the values indicated, according to the particular compounds undergoing decomposition at any time. Proteins and fats occur mainly in garbage, and since they are highly susceptible to bacterial action, their respiration products should appear soon after placement in a dump. The anaerobic breakdown of fats can continue for a period of years, judging from the high methane content of the gas produced. In combustible rubbish the bulk of the organic material will be carbohydrates, chief among which is the cellulose of paper and wood. Cellulose does not usually decompose rapidly, but the sugars and starch to be found in food wastes and plant tissue are exceptionally digestible substances. In a fill containing mostly combustible rubbish, the decomposition products should always be predominately those of carbohydrate reduction, with protein and fat decomposition products present to some extent during the initial stages.

Comparing aerobic and anaerobic gas products, Table XI shows that one pound of carbohydrates upon decomposing yields 1.65 lb of carbon dioxide aerobically, but only 0.85 lb anaerobically. In protein and fat breakdown the difference in the two sorts of respiration with respect to CO_2 formation is still greater. Other things being equal, it would therefore appear desirable to maintain anaerobic conditions within buried refuse deposits in order to minimize CO_2 production.

In the aerobic respiration of fats and proteins a smaller volume of carbon dioxide is produced than that of oxygen taken up. With carbohydrates the oxygen consumed is replaced one-for-one by carbon dioxide. A small amount of oxygen may be absorbed and used in the syntheses of cell material by the organisms during periods in which their population is increasing, but minor volumes of

Table XI

Gas Production from Organic Substances

Type of Compound	Carbohydrate		Protein		Fat	
Composition, % by Weight						
Carbon		45		53		77
Hydrogen		6		7		12
Oxygen		49		23		11
Nitrogen		0		17		0
Composition, atoms/atom C						
Carbon		1		1		1
Hydrogen		1.67		1.58		1.89
Oxygen		0.83		0.33		0.11
Nitrogen		0		0.27		0
Aerobic Respiration						
Production & Utilization/lb Compound						
	lb	cu ft	lb	cu ft	lb	cu ft
O_2 Uptake	1.20	14.5	1.46	17.6	2.92	35.2
CO_2 Produced	1.65	14.5	1.95	17.1	2.82	24.8
Net Uptake		0		0.5		10.4
Production & Utilization/lb Carbon						
O_2 Uptake	2.67	32.2	2.75	33.2	3.79	45.8
CO_2 Produced	3.67	32.2	3.67	32.2	3.67	32.2
Composition Exhaust Gas, % by Volume						
N_2		79		80		84
CO_2		21		20		16
Av. Molecular Weight		31.3		31.2		30.5
Anaerobic Respiration						
Gas Production/lb Compound						
CO_2	0.82	7.3	0.95	8.3	0.82	7.2
CH_4	0.30	7.3	0.36	8.8	0.72	17.6
Total Produced		14.6		17.1		24.8
Gas Production/lb Carbon						
CO_2	1.83	16.1	1.78	15.6	1.07	9.4
CH_4	0.67	16.1	0.69	16.6	0.94	22.8
Composition Exhaust Gas, % by Volume						
CO_2		50		49		29
CH_4		50		51		71
Av. Molecular Weight		30.0		29.7		24.2

Gas volumes at 1 atmosphere, 70°F

carbon dioxide may also disappear by dissolving in the refuse moisture. Over a long period of aerobic action of refuse containing mostly combustible rubbish the carbohydrate breakdown process will predominate, and the volume of carbon dioxide given off will be only slightly less than the oxygen consumption.

Because of the existence of at least three common classes of organic substances which may simultaneously decompose, and the fact that each can decompose in two different ways, it is not usually practical to definitely ascertain the mechanisms occurring from gas analyses alone. From Table XI it may be verified that large percentages of methane relative to carbon dioxide are indicative of anaerobic fat fermentation, and as Merz (18) has pointed out, a higher concentration of nitrogen than that found in the atmosphere (79 percent) suggests aerobic destruction of fats.

Several reported refuse gas analyses are listed in Table XII. In all large operating fills there is a considerable excess of methane over carbon dioxide, and the methane concentrations reach 60 percent or more. Simultaneous aerobic and anaerobic decomposition is reflected in the widely variable nitrogen percentages at low oxygen concentration; extensive anaerobic gas production will drive out the nitrogen while nearly all oxygen will be consumed aerobically and independently of the anaerobiosis. The Long Beach fills are illustrative of a decreasing rate of aerobic breakdown with age, the oxygen being higher, and the carbon dioxide lower, in the older fills.

The Riverside gas analyses suggest the progression of initially aerobic breakdown with high nitrogen and CO_2 concentrations, followed by the increasingly anaerobic decomposition associated with low nitrogen and high methane percentages. In the older samples the methane has decreased and the nitrogen increased, indicating the return of aerobic conditions. In Merz's studies (18), Test Bin No. 1, which was being subjected to percolation, became increasingly anaerobic during the four months of sampling while Test Bin No. 2, having remained relatively dry, never developed significant anaerobic breakdown. The lack of moisture apparently either maintained the refuse and its soil cover relatively pervious to the outside air, or discouraged decomposition to an extent that inward diffusion of oxygen could keep pace with the process.

Very little data is available upon which to estimate rates of gas production, and as pointed out in Section III, the quantitative measurement of decomposition rates would be very desirable. Pomeroy (36) concluded, after examination of three trash dumps in the Long Beach area, that only a minor part of the combustible organic matter is susceptible to rapid biological oxidation; from measurements of carbon dioxide diffusion from the surface of Paramount Dump, then several years old, he estimated the aerobic decomposition rate at 0.3 percent per year.

Table XII

Refuse Gas Analyses

Source	Percent by Volume CO_2	CH_4	O_2	N_2	H_2
Seattle (48)	40	60			
San Francisco (49)*	25	60	0	6	8
New York (50)	30	45	3	22	0.1
Arlington, Mass. (51)	32	43		17	
Los Angeles**					
Branford	40	46			
Holly - Age 2-1/2 yr	32.0	43.4		17.2	
Long Beach, Calif. (36)					
Spring St. - Age 20 yr	2.1		19.4		
Dominguez - Age 8 yr	11.0		7.2		
Paramount - Age 5 yr	26.8		3.2		
Riverside, Calif. (18)					
Fill Age 2-4 mo	36	5	1.3	57	
Fill Age 3	18	60	1.0	20	
Fill Age 4	31	65	0.8	3	
Fill Age 4-5	29	33	1.3	38	
Fill Age 5-6 yr	18	11	0.5	71	
Test Bin No. 1 (Damp)					
Age 1 mo	4	0	13	83	
Age 2 mo	35	25	2	38	
Age 4 mo	38	46	0.4	15	
Test Bin No. 2 (Dry)					
Age 1 mo	14	0	4	82	
Age 2 mo	10	9	8	73	
Age 4 mo	5	3	15	77	

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* Original analysis corrected for dilution with air by assuming oxygen was zero in refuse gas.

** Data of Bureau of Sanitation, City of Los Angeles.

Gas Properties

There are four gases which commonly occur at appreciable concentrations in the atmospheres of refuse fills: nitrogen, oxygen, carbon dioxide, and methane. Small amounts of hydrogen formed in certain anaerobic fermentations are occasionally measured, but for practical purposes hydrogen and all other rare gases may be neglected. At all temperatures and pressures which would conceivably be obtained in and around a fill, the gases closely follow the perfect gas law

$$pV = \frac{RT}{M} \qquad (18)$$

where p is absolute pressure, V the specific volume (cu ft/lb), T the absolute temperature, M the molecular weight, and R the universal gas content, which is equal to 1,544 ft-lb/mol-°F. For a mixture of several gases, the molecular weight to be used in Equation 18 is an average obtained by weighting the molecular weights of the components by their volume percentages.

Of particular interest to the question of refuse gas movement is the fact, implicit in Equation 18, that at a given pressure the unit weight (1/V) of any gas mixture is directly proportional to its average molecular weight and inversely proportional to its absolute temperature. All of the exhaust gases from aerobic respiration listed in Table XI have average molecular weights greater than that of air, which is 28.9. Surrounded by an atmosphere of air at 70°F, the aerobic exhaust gases would tend to sink unless their temperatures were sufficiently high--115°F for the carbohydrate gas, and 100°F for the exhaust from fat decomposition. On the other hand, the anaerobically produced gases from fats and proteins are lighter than air at the same temperature. The anaerobic carbohydrate gas is heavier and would have to be at 90°F in order to have the same density as air at 70°F. The unit weight of gases at one atmosphere may be obtained from Figure 9 as a function of temperature and average molecular weight.

Gas Movement

Pressure Gradient: When differences in gas pressure are created in a porous medium a flow will be established in accordance with Darcy's Law. Because of the fact that soil is quite permeable to gases and the area of earth through which gas may flow into and out of a refuse fill is relatively great, large pressure differences are not likely to occur. It is therefore permissible in calculating flow rates to consider gases as incompressible fluids. The proper form of Darcy's Law is then Equation 12,

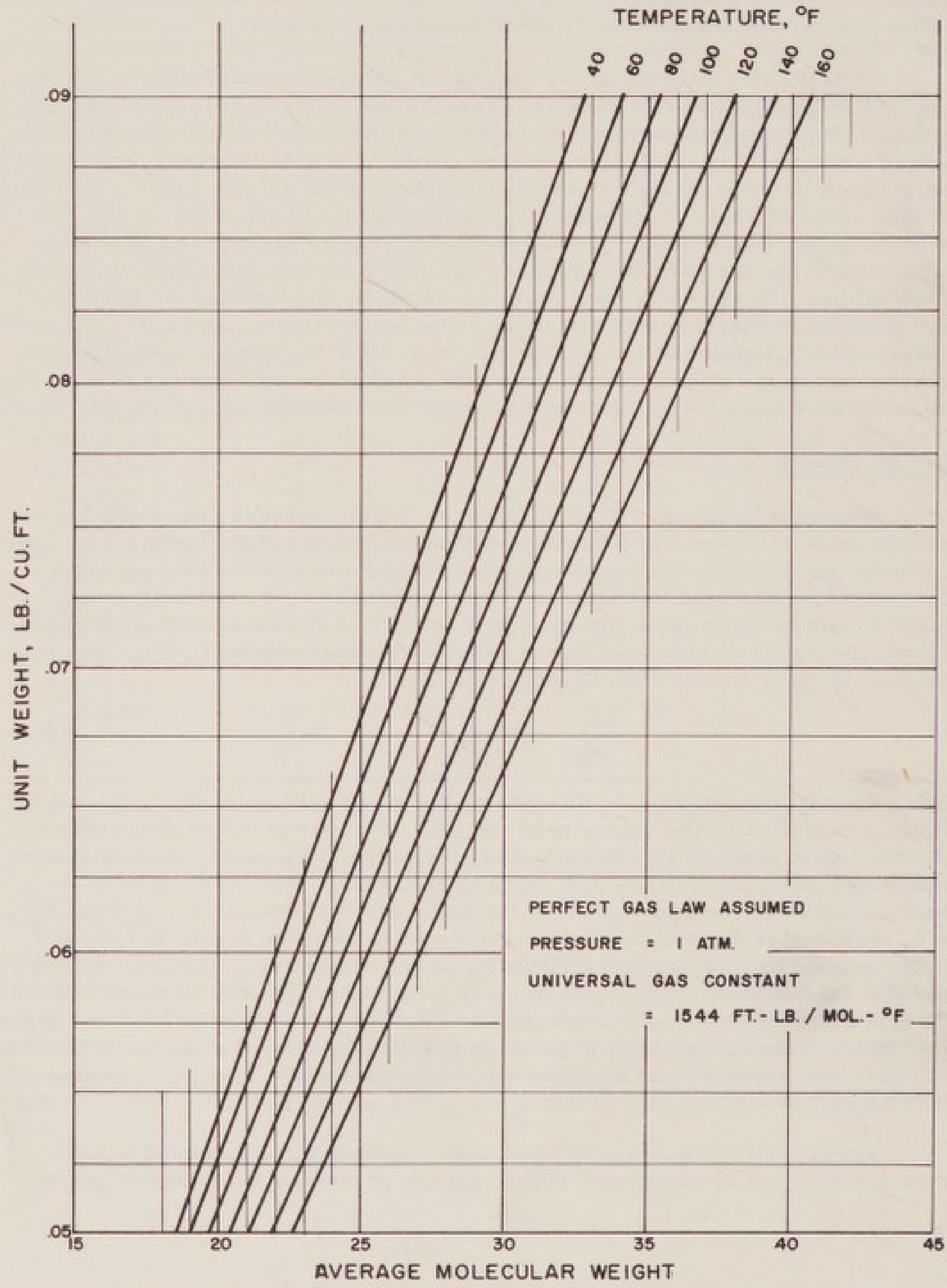
$$v = -\frac{k_g}{w_g} \text{ grad } p \qquad (12)$$

The hydraulic conductivity k_g depends on the medium through which the gas is flowing, and also on the viscosity of the gas. Because of the profound effect of the medium's particle size distribution, the range of hydraulic conductivity in commonly encountered soils and sands is enormous. The viscosities of the various gases in question are all of the same order of magnitude; at 70°F for air, carbon dioxide, and methane, respectively, they are 0.018, 0.015, and 0.011 centipoise. Water has a viscosity of 1.0 centipoise. In order to establish general magnitudes for the rates of gas flow which micht occur around refuse fills, Table XIII lists approximate values for the conductivities of natural granular media. The conductivity to gases is drastically decreased by the presence of moisture, especially in the finer textured media which have high moisture holding capacities.

Pressure differences tending to cause motion of gases around buried refuse may arise in two ways. First, any net production or uptake of gas by

FIGURE 9

UNIT WEIGHT OF GASES



PERFECT GAS LAW ASSUMED

PRESSURE = 1 ATM.

UNIVERSAL GAS CONSTANT

= 1544 FT.- LB. / MOL.- °F

Table XIII

Approximate Hydraulic Conductivities

Medium	k (Water) ft/day	k_g/w_g(Gas)* ft^4/lb-day
Alluvial Sand[1]	250	267
Silt Loam	1.0	1.07
Clay Loam	0.2	0.21

* Viscosity = .015 cp.

[1] Pomeroy (52) has estimated that at a slope of 5.5 ft/mile the ground water flow rate in the San Gabriel Basin alluvial sands is 0.2 to 0.4 cu ft/sq ft/day, which corresponds to a conductivity of 192 to 384 ft/day.

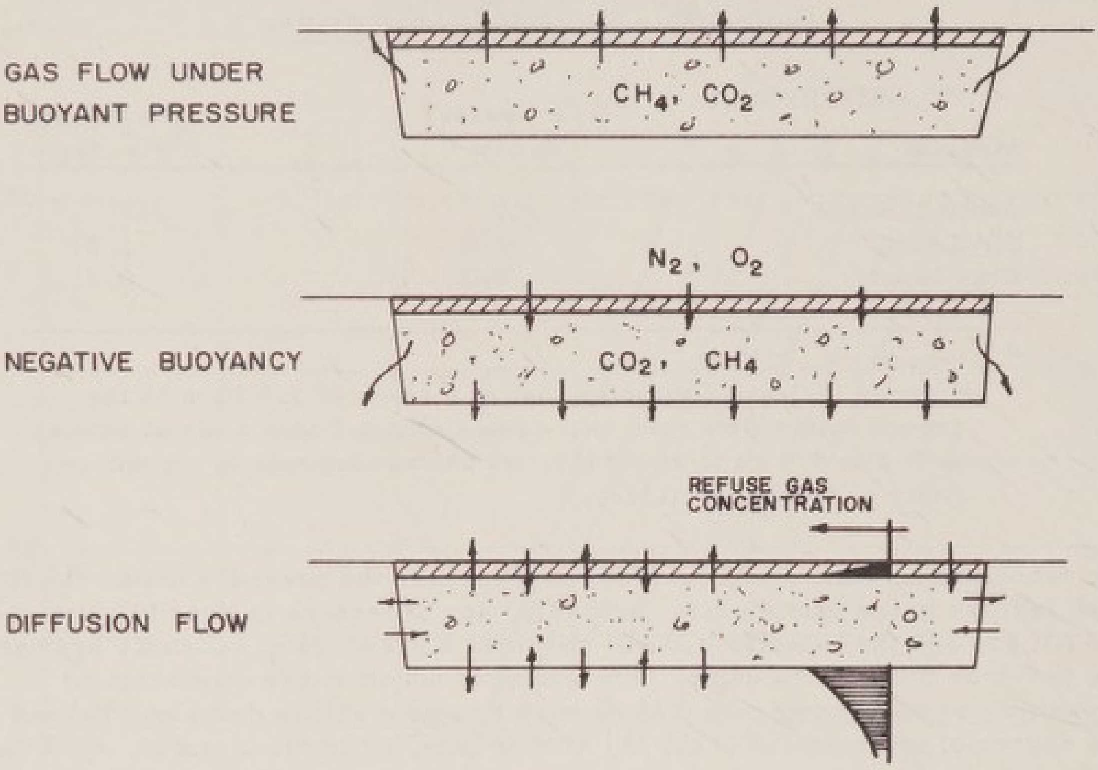
the decomposing refuse will increase or decrease the pressure inside the fill with respect to the outside air. Secondly, any difference in density between the fill gas and the exterior air will result in a pressure of buoyancy against the cover or floor of the dump. The extent to which either displacement pressures or buoyancy pressures develop around a refuse dump will depend on the degree of confinement of the fill atmosphere, or in other words, upon the hydraulic conductivity of the bottom, sides, and cover of the dump. It is to be presumed that the refuse itself will have a very high permeability compared to the pit boundaries, although this might be an oversimplification if the fill contained large amounts of fine material.

If there is a net production of gas lighter than the ambient air, as would be the case if anaerobic fat fermentation were the predominant process, the gas will rise to the fill cover and begin to flow out. When the cover is tight and gas evolution relatively rapid, the gas will displace the air within the refuse and, forming a layer underneath the cover, will flow out the sides as well. As the depth of light gas increases the buoyancy pressure against the cover and sides will increase, with a corresponding increase in flow rate. Finally, if the rate of gas production is greater than the capacity of the walls and cover to conduct it away, the gas will completely permeate the refuse atmosphere and, flowing out through all boundaries, will build up pressure within the fill cavity until the outflow rate becomes equal to the production rate. The case of net production of a gas heavier than air, such as anaerobic carbohydrate decomposition at fairly low temperatures, is similar to that just described except that the gas first descends to the floor of the dump and fills it from the bottom up. These flow situations are illustrated in Figure 10.

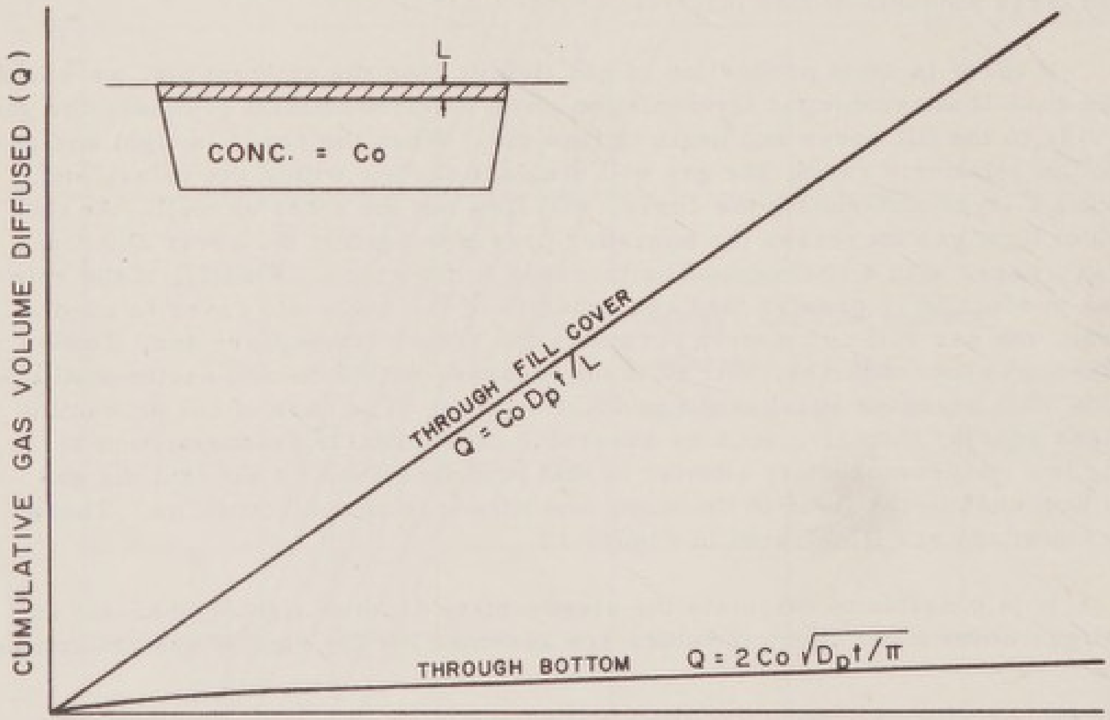
It is possible to calculate the steady-state depth of lighter-than-air exhaust gas under a fill cover if values are assumed for the rate of gas production,

FIGURE 10

GAS FLOW PROCESSES IN REFUSE FILLS



DIFFUSION OF GAS FROM REFUSE FILL



the gas composition and temperature, and the conductivity of the cover. As an example, a 30-foot-deep fill containing 4.5 lb of carbon per cu ft which is decomposing at the rate of one percent per year will produce the anaerobic fat breakdown gas of Table XI at the rate of 0.12 cu ft/day (70°F) per sq ft of fill surface. If the interior of the fill is at 90°F, the gas volume will be 0.125 cu ft/day/sq ft, and its unit weight, from Figure 9, 0.060 lb/cu ft. Air at 70°F has a unit weight of 0.075 lb/cu ft. If a three-foot depth of cover be considered, made of the materials listed in Table XIII, the gas will pile up to a depth of 25 ft under the silt loam, but to only 0.1 foot under the alluvium cover, while with the clay loam cover the fill will be completely under pressure (unless the sides are rather permeable). It is apparent that the relative conductivities of the various portions of the fill boundary must be very important in determining the course of gas flow. Also, the magnitude of the gas production rate may determine whether the majority of the gas is vented to the atmosphere or directed into underground strata; a light gas produced at a nominal rate may be able to seep out through the cover, whereas rapid production would back it up into the ground.

Whenever a light gas is being produced at the bottom of a fill, or a heavy gas at the top, there will be a mixing action as the gases of initially different composition and temperature convect through each other. It is possible that such mixing processes may significantly influence the nature of the decomposition processes; light anaerobic exhaust gas may rise from the bottom of the fill and suffocate aerobic breakdown occurring at the top. Heavy, but warm, aerobic gases may transfer heat downward to promote the rate of anaerobic breakdown beneath. The net result of mixing, after heat and gas generation has settled down to low values and pressure gradients have disappeared, will be a stable atmosphere within and around the fill; the unit weight of the gas in a column of earth and refuse will then increase with depth.

Diffusion: Molecular diffusion may actually effect more significant transfer of refuse gases than mass flow under a pressure gradient. The two processes may occur simultaneously and independently, except that both are restricted by decreasing porosity of the medium. The rate of diffusive transfer is proportional to the concentration gradient of the diffusing substance, and the constant of proportionality is the diffusivity, D_p. Diffusivities of different gases moving through each other in free space are inversely proportional to the absolute pressure and vary approximately as the three-halves power of absolute temperature. Values of the diffusivity of oxygen and carbon dioxide into air given in Reference 53 and corrected to a temperature of 70°F, are the following:

	Diffusivity, sq ft/day
Oxygen	17.5
Carbon Dioxide	14.0

A number of measurements of diffusivity in porous media have been reported. Usually the results have been expressed as a formula giving the ratio of diffusivity in the medium to free space diffusivity as a function of porosity. Buckingham's original work in 1904 (54) was that the diffusivity varied as the square of porosity:

$$\frac{D_P}{D_o} = P^2 \tag{19}$$

where D_o is the diffusivity of any system of gases in free space, and P the fraction of the medium occupied by gas. More recently Penman (56, 55) has given his results as a linear expression:

$$\frac{D_P}{D_o} = 0.66P \tag{20}$$

Rust et al (51), using a non-steady state system for measuring carbon dioxide diffusion, obtained the relation

$$\frac{D_P}{D_o} = 0.04 + 0.60P \tag{21}$$

Actually all of these expressions were suggested as approximations to fit somewhat irregular data. In all three sets of experiments, and in those of Taylor (58), the relative diffusivity-porosity plots were slightly concave upward, and the results are not inherently conflicting. Any curve, in order to be strictly correct, would have to indicate zero diffusivity at zero porosity, and a diffusivity of one at 100 percent porosity. Porous medium diffusivity does not seem to be affected to any extent by the particle shape or size distribution. Moisture has the direct effect of reducing the porosity and accordingly may be responsible for considerable variability of the diffusivity in media of high moisture holding capacity.

To illustrate the order of magnitude of diffusion rates through dump cover layers, one may consider the example of the three-foot cover having a porosity of 0.20. The diffusivity for oxygen, according to Penman's formula, would be about 17.5 x .66 x .20, or 2.3 sq ft/day. With oxygen at a concentration of 21 percent in the outside air and at zero inside the cover, the transfer rate would be 2.3 x 0.21/3, or 0.16 cu ft/sq ft/day. With reference to the example previously cited of exhaust gas flow under a pressure gradient and to Table XI, the dump oxidizing itself aerobically at the rate of one percent per year requires only 0.12 cu ft of oxygen per sq ft per day, so that oxygen supplied only by diffusion from the surface would be adequate to sustain aerobic breakdown of the entire fill, if it could be distributed throughout. Concurrently with the diffusion of oxygen into a dump, the carbon dioxide produced aerobically from that oxygen, together with any carbon dioxide and methane produced anaerobically, will diffuse outward at a comparable rate.

The diffusion of gaseous mixtures is rather complicated analytically because the different diffusivities of the component gases can result in the generation of pressure gradients and associated mass flow. To a first approximation, however, and in view of the undertainties which will always exist regarding the rate of gas production and the characteristics of the porous media, the diffusion process around refuse fills may be pictured as outward movement of some interior gas and the corresponding inward movement of air. An average free space diffusivity should then be used, and would amount to about 16 sq ft/day.

For given diffusivity, the rates of gas flow out of a dump can be estimated by considering the interior top and bottom as extensive plane surfaces. Flow out of the relatively thin cover will quickly approach the steady state given by Equation 17, namely

$$q = c_o D_p / L \tag{22}$$

where c_o is the gas concentration within the fill and L the thickness of the cover. Flow out of the bottom of the dump will be rapid at first, but will decrease continuously as more of the adjacent earth is filled with gas, and the concentration gradient flattens out. For this case the solution of Equation 17 is

$$q = c_o \sqrt{D_p / \pi t}$$

where t is time after the diffusion commences. The value of the inside concentration c_o will assume a value at which the rate of diffusion from the fill top and bottom balances the rate of gas production. It will vary with time as the rate of diffusion from the bottom decreases and as the rate of gas production also falls off. Figure 10 gives the cumulative gas flow from the top and bottom of a fill, and illustrates how the proportion leaving through the cover will increase with time if the internal concentration remains constant. If the diffusivities of the cover and bottom are the same, and a reasonably thin cover is used, the vastly greater proportion of the gas will pass out through the cover.

The extent of penetration of gas through the bottom of a fill may be estimated if diffusivity in the earth and the gas concentration within the fill are known. Equation 17 solved for diffusion from an infinite plane surface kept at constant concentration is

$$c = c_o \operatorname{erfc} (x / 2 \sqrt{D_p t}) \tag{24}$$

where c is the gas concentration at any distance x below the fill bottom, and t is time after the onset of diffusion. Using a diffusivity D_p of 2.1 sq ft/day Equation 14 indicates that the gas concentration in the earth will reach 10 percent of its value in the fill at 11-foot depth in 10 days, and at 105-foot depth in 1,000 days. Actually the concentration would build up somewhat more slowly than this, since Equation 24 does not account for the gas diffusing laterally after it enters the earth.

Reducing the possibility of percolation through refuse fills by making the cover impermeable would have the adverse effect of restricting the flow of gases to the atmosphere. Although coarse soil layers are easily penetrated by gases, a cover of fine material may significantly restrict gas transfer, especially when damp. It may be desirable, therefore, to rely on a bottom lining to restrict percolation and downward gas movement and to use a permeable cover to encourage ventilation. If both top and bottom were sealed, gravel-filled vents might be used to provide for gas escape, but their effectiveness and design requirements would be determined by the permeability of the refuse, which would no longer be negligible compared to that of the exits. Eliassen et al (51) have described such a procedure to intercept refuse gas moving horizontally from a dump.

Underground Gas Movement

Any gas having penetrated into the ground through the bottom and sides of a fill will continue to move under the convection and diffusion processes just described. Heavy gases will fall, spreading laterally whenever they encounter a less permeable layer, until they reach ground water or become mixed with the ground atmosphere. Heterogeneous composition of the earth sediments will direct both convective and diffusive transfer preferentially in particular directions.

As it travels through the underground, the gas will come in contact with the soil moisture and eventually with the ground water table and capillary fringe. The gas components will tend to dissolve in the water to the limit of their solubility, but the process may be very slow. The solubilities of the four commonly occurring refuse gases at one atmosphere pressure and 70°F are as follows:

Table XIV

Solubilities of Gases

One atm, 70°F

Gas	Solubility, ppm
Carbon dioxide	1,720
Methane	24
Oxygen	44
Nitrogen	19

Carbon dioxide is the only appreciably soluble one; methane and nitrogen are of virtually no importance in connection with ground water pollution since they are inert and undetectable at their maximum conceivable concentrations.

Oxygen is necessary for the complete stabilization of organic matter and any decomposable substances which enter the ground water will immediately begin to deplete the dissolved oxygen supply. Since the solubility of oxygen is very low and its rate of replenishment slow in the absence of surface agitation, the dissolved oxygen will quickly disappear under any sizeable underground organic pollution load and the stabilization will then be limited by the rate of transfer of oxygen from gas to liquid and by the rate of dilution with adjacent waters containing dissolved oxygen. Rates of ground water reaeration cannot be estimated reliabily from available information, but are of some importance since large concentrations of BOD can be leached out of refuse. However, the significance of organic pollution will generally be overshadowed by the problem of inorganic salts; organic matter will be oxidized in the ground at some rate, however slow, while mineral matter must remain in the ground until it flows out. In the incident described by Roessler (59) of pollution of a well water supply by a garbage fill, only slight increases inthe permanganate oxygen demand (a chemical measure of organic matter) occurred in wells located about a mile from the pollution source, whereas the increase in hardness was over 500 ppm. However, the pollution had taken nine years to reach the wells. In Merz's measurements of leaching at Riverside (18), the BOD of the water under the fill was rarely more than a few parts per million and could easily be handled by the existing dissolved oxygen in a short time.

If leaching water must percolate downward an appreciable distance before reaching ground water and at a slow annual rate of infiltration, the organic matter it contains on leaving the dump may have adequate time for complete oxidation in the zone of aeration. Among the recent experiments by the British Ministry of Housing and Local Government (33) was that of applying the leachate from dumped refuse to six-foot-deep sand and gravel filters. An influent BOD of 250 ppm was reduced to about 30 ppm in a coarse gravel filter and to less than 15 ppm on fine gravel and sand.

Carbon dioxide not only is fairly soluble in water, but once dissolved it can react with solid calcium carbonate to form soluble calcium bicarbonate. The carbon dioxide thus removed leaves room for more to dissolve, and the net result of the process is that calcium carbonate is dissolved until the water is saturated with it at the equilibrium carbon dioxide concentration. This is in fact the principal undesirable effect associated with carbon dioxide in ground water; although the entire process will reverse itself when the ground water flows to a region where the soil atmosphere is low in carbon dioxide, it occurs very slowly, and the calcium hardness will remain for years.

The reaction of carbon dioxide, which after dissolution is in the form of carbonic acid, with calcium carbonate can be written as follows:

$$H_2CO_3 + CaCO_3 \rightarrow Ca^{++} + 2HCO_3^- \qquad (25)$$

At chemical equilibrium the law of mass action requires that

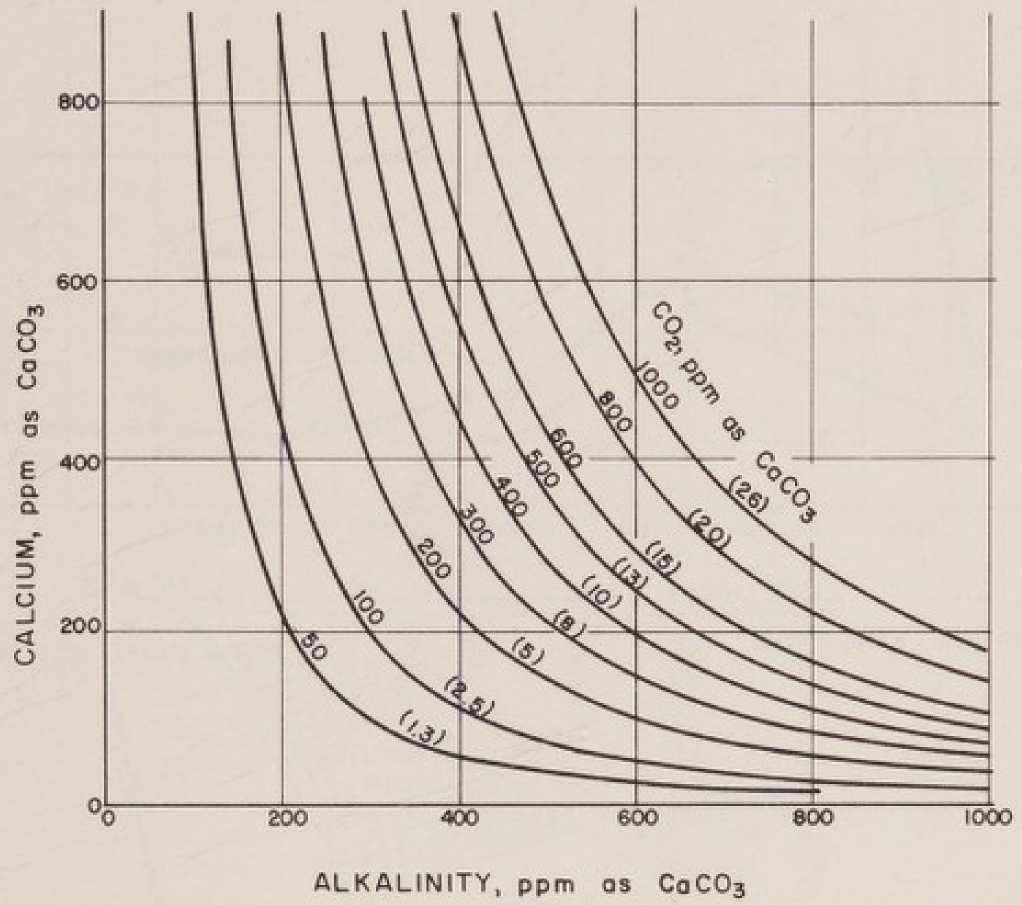
$$(Ca^{++})(HCO_3^-)^2 / (H_2CO_3) = K \qquad (26)$$

where the parentheses refer to the molar concentration of the different reactants, and K is the equilibrium constant. The bicarbonate concentration is substantially equivalent to the alkalinity at pH's less than about 8.0. The equilibrium concentration of carbon dioxide is proportional to its partial pressure in the atmosphere over the solution (Henry's Law). Figure 11 illustrates the relation expressed in Equation 26, and gives the equilibrium calcium concentration as a function of alkalinity and carbon dioxide concentrations. Since the appearance of calcium as the result of dissolving calcium carbonate is accompanied by an equivalent increase in alkalinity, the point on Figure 11 representing a water in which such process is occurring must move through equal horizontal and vertical distances. If the initial calcium and alkalinity are plotted on the chart, the concentration of calcium carbonate dissolved when the water reaches equilibrium with carbon dioxide at any given concentration may be found by drawing a 45° line from the initial point to intersection with the appropriate CO_2 curve. The horizontal or vertical distance between the initial and final points then represents the dissolved calcium carbonate.

An outstanding example of the build-up of carbon dioxide, probably of refuse origin, in the underground with associated hardness increase is offered by the experience with the Mayflower Well in Monrovia, California. Figure 12 shows the location of wells and test holes around a refuse dump. The normal ground water flow is in a southwesterly direction, but water spreading a short distance downstream from the dump has reversed the flow occasionally. In August 1958 a large rise in hardness was observed in water from Mayflower Well. The changes of the chemical quality of this water were assembled by the Los Angeles Regional Water Pollution Control Board and are shown in Figure 12, together with comparative analyses from the Eighth Avenue well located about one-half mile west, which was unaffected. In October 1959 the Los Angeles Regional Water Pollution Control Board entered into a contract with the California Department of Water Resources to make an investigation of ground water impairment in the Main San Gabriel Basin. Under this contract the Department of Water Resources, in April 1960, drilled Test Holes 1, 2, and 3; gas samples were taken at various depths and the ground water encountered was analyzed. At that time the ground water was at about 100-foot depth and the bottom of the refuse fill is about 27 feet below the surface. The results, taken from Reference 60, are given in Table XV.

The gas composition in each test hole was reasonably constant with depth. The nitrogen content was about 58, 32, and 78 percent in Test Holes 1, 2, and 3, respectively; it increased with distance from the dump, suggesting increasing dilution of anaerobic decomposition gases with air. The top sample in Hole 2 was in the refuse itself and contained equal amounts of CO_2 and methane, the composition of exhaust from anaerobic carbohydrate breakdown. Elsewhere the methane concentrations were lower, probably because methane has a greater diffusivity than that of carbon dioxide. The carbon dioxide concentrations in the water of the test holes, while much higher than values usually obtained from ground water, were still much lower than their

DISSOLVING OF CALCIUM CARBONATE BY CARBON DIOXIDE



THE POINT REPRESENTING WATER WITH GIVEN Ca AND ALKALINITY MOVES ON A 1:1 SLOPE TO EQUILIBRIUM AT THE GIVEN CO_2 CURVE. THE HORIZONTAL OR VERTICAL DISTANCE MOVED IS THE CONCENTRATION $CaCO_3$ DISSOLVED. FIGURES IN PARENTHESES ARE % CO_2 IN GAS AT EQUILIBRIUM WITH THE INDICATED CO_2 LIQUID CONCENTRATIONS.

FIGURE 12

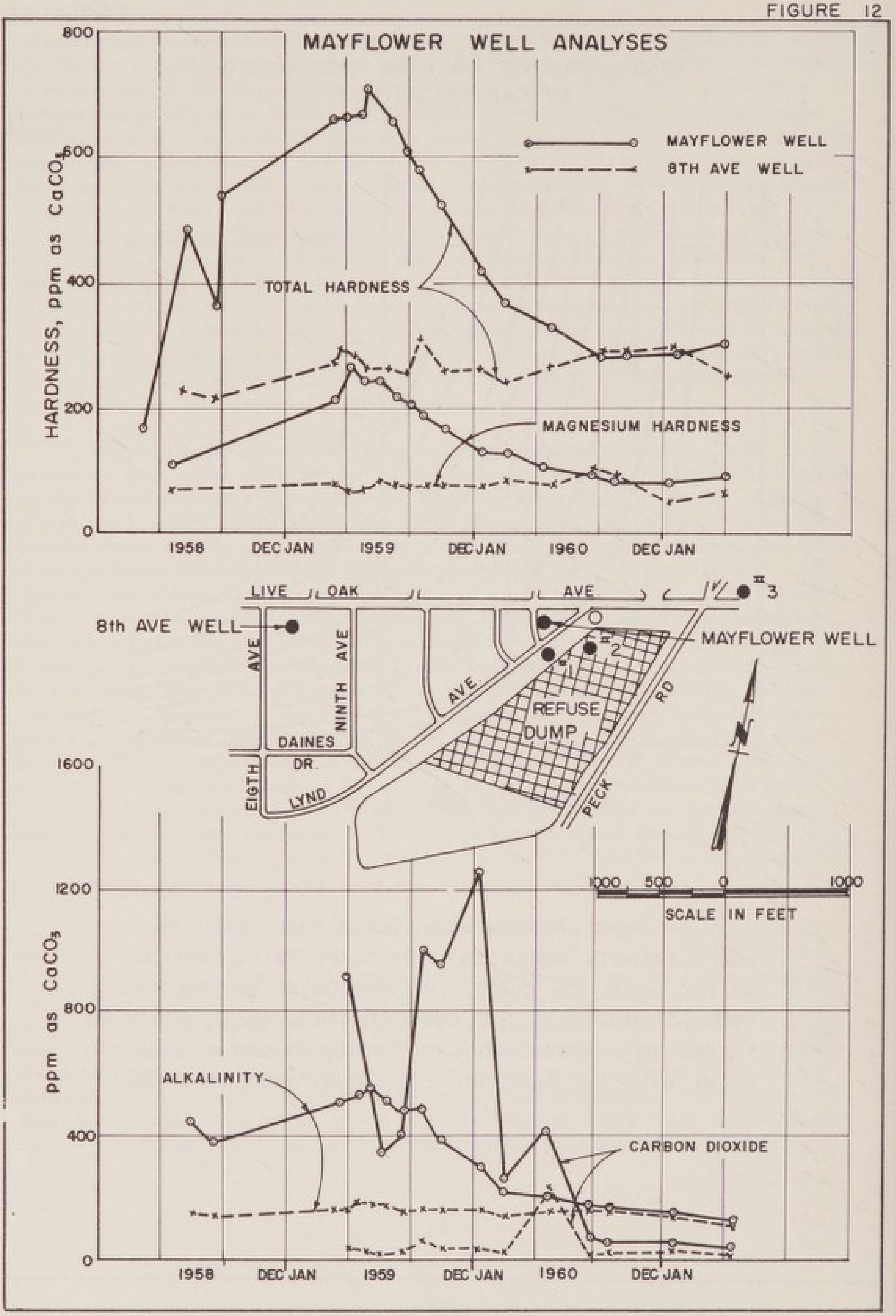

MAYFLOWER WELL ANALYSES
MAYFLOWER WELL
8TH AVE WELL
HARDNESS, ppm as CaCO$_3$
800
600
400
200
0
TOTAL HARDNESS
MAGNESIUM HARDNESS
1958
DEC JAN
1959
DEC JAN
1960
DEC JAN
LIVE
OAK
AVE
#3
8th AVE WELL
MAYFLOWER WELL
#1
#2
AVE
NINTH AVE
AVE
REFUSE
DUMP
RD
DAINES
DR.
EIGTH
LYND
PECK
N
1000
500
0
1000
SCALE IN FEET
ppm as CaCO$_3$
1600
1200
800
400
0
ALKALINITY
CARBON DIOXIDE
1958
DEC JAN
1959
DEC JAN
1960
DEC JAN

TABLE XV

Results of Mayflower Test Hole

Gas and Water Analyses

Gas

	Percent by Volume			
	CO_2	CH_4	O_2	N_2
Test Hole No. 1				
Dept, ft: 20	28.6	17.6	1.8	52.0
40	26.2	12.9	2.8	58.1
60	29.8	10.6	1.2	58.4
80	32.2	14.0	1.1	52.7
100	12.6	6.5	14.8	66.1
Test Hole No. 2				
Depth, ft: 20	43.5	44.9	1.7	9.9
60	31.4	14.7	3.8	50.2
80	45.3	21.9	1.9	30.9
96	40.8	22.2	2.7	34.3
Test Hole No. 3				
Depth, ft: 20	0.03	0.0	21.5	78.5
40	9.5	0.0	9.7	80.8
60	15.4	0.0	6.4	78.2
80	17.0	0.2	4.1	78.7
96	23.7	1.3	1.5	73.5

Water *

	Parts per Million				
	Alkalinity	CO_2 as $CaCO_3$	Total Hardness	Chloride (Cl)	pH
Test Hole No. 1	300	152	377	26	7.0
Test Hole No. 2	480	670	489	43	6.5
Test Hole No. 3	545	425	663	48	6.8

---

* Analyses at maximum hardness during pumping.

saturation values under CO_2 gas at the percentages found in the deepest gas samples. Carbon dioxide was still going into solution, but apparently at a very slow rate and it is even possible that most of the CO_2 in the ground water arrived there after being dissolved in soil water percolating down from the surface. On the other hand, the analyses indicate, with reference to Figure 11, that the water was fairly close to saturation with calcium carbonate. The test hole water was not analyzed for calcium, but comparison with the Mayflower Well water analyses suggests that the total hardness was about 65 percent calcium. On this basis the water from Test Holes 1 and 2 was undersaturated with calcium carbonate, while the Test Hole 3 water was somewhat supersaturated. The possibility of the hardness having been leached from the dump itself is not probable because of the relatively low chloride concentrations found throughout.

The carbon dioxide and hardness in the Mayflower Well water eventually disappeared completely, suggesting that only a portion of the ground water near the well had been affected--possibly the surface layer--and was finally pumped out. It is to be noted that a considerable fraction of the increased hardness was due to magnesium; probably the mineral being dissolved to release it was a complex calcium magnesium carbonate (dolomite).

Subsequent gas samples were taken by the Department of Water Resources along the south and east boundaries of the dump early in 1961. The samples were removed through well points driven to 40 feet. All were relatively low in carbon dioxide (less than eight percent) except one at the southwest corner of the dump, which contained 50 percent CO_2 and 4 percent methane.

Summary of Needed Information

Available information, including some actual experience, indicates that gases are produced in significant quantities from refuse and can travel through the underground to pollute the ground water. However, the circumstances under which such pollution does in fact happen are still obscure. Because of the difficulty of simulating the large-scale conditions which apparently control gas movement around dumps, the most fruitful approach in investigating gas motion appears to be observations made on operating fills. The following needs are suggested:

1. Quantitative measurement of gas composition and production rate in conjunction with decomposition experiments described in Section III.

2. Monitoring of gas composition within existing fills and in the surrounding ground atmosphere. Such information would be most valuable if monitoring were begun at the time a fill was put into operation, since the displacement of the soil air by the refuse gases would be easy to detect and follow. It is of particular importance to determine the length of time over which gas pollution will continue to be a hazard, since this will affect the durability and cost of remedial measures.

3. Determination of the suitability of certain techniques suggested for limiting or controlling the flow of gases, such as relatively impermeable clay or layers, sheets of plastic materials, and soil-cement or soil-asphalt layers. In addition, the practicality of using natural draft ventilation through gravel-packed tubes or trenches reaching the ground surface, or of forced-draft ventilation with blowers to cause controlled gas flow, should be tested under full-scale conditions. Such tests would not be inexpensive, but are deemed necessary prior to the acceptance of such corrective measures for any large-scale refuse dumping program which would contemplate placement of refuse into alluvial deposits comprising principal ground water storage basins.

V. LEACHING

The transfer of soluble non-volatile substances from refuse dumps to the ground water is accomplished by means of a leaching process. Either vertically or horizontally moving water must physically pass through the refuse, and by intimate contact, extract soluble matter from it. The extraction may proceed to near completion in small or large volumes of water, depending upon how accessible the soluble matter is, and on how rapidly it is created by decomposition processes. The dissolving of calcium carbonate from the aquifer minerals by water high in carbon dioxide may also be considered a leaching operation.

Theoretical Considerations

Leaching may be visualized as a special case of extraction of substances adsorbed onto solid particles. If an extracting liquid is allowed to remain in prolonged contact with a partially saturated adsorbent, the adsorbed substance will go into the liquid until an equilibrium finally prevails. At equilibrium there is a definite relation between the concentration of the soluble in the liquid and its adsorbed concentration on the solid particles. The equation which expresses this relationship is termed an adsorption isotherm. In the case of the inorganic salts contained in refuse there will be little or no true adsorption, but as water flows through the refuse, some of the soluble materials will be retained in thin liquid films surrounding the particles or possibly trapped within plant cells. It will therefore appear that adsorption is occurring with a linear isotherm; that is, the equilibrium between liquid and solid phases will be such that the amount of solute remaining with the solid will be directly proportional to the concentration in the liquid.

There are at least two theoretical models which might be used to describe the leaching process. One is that proposed by Glueckauf (61) in connection with his theory of chromatography, in which it is assumed that the leaching liquid is in equilibrium with the solid being leached at all points of the bed, but that longitudinal diffusive processes disperse the solute as it flows. The differential equation describing the process for the case of a linear isotherm is identical with that of hydraulic mixing of a tracer in flow through a porous medium; this equation and its solutions are given as Equations 38 to 42 of Section VI. For the case of elution of a bed saturated with a leachable substance, the appropriate solution is

$$c = c_1 + \frac{(c_o - c_1)}{2} \operatorname{erfc} \frac{(1 - \bar{v}t/x)}{(2\sqrt{(D_m/x)(\bar{v}t/x)})} \qquad (26)$$

where c, c_o, and c_1 are, respectively, the concentration of leaching substance in the effluent, in the liquid initially saturating the bed, and in the applied leach water. T is time, x the depth of the bed, $\bar{v}$ the percolating velocity, and D_m a constant of the medium which describes the degree of hydraulic dispersion.

The other promising model for leaching is implicit in the treatment of column ion-exchange and adsorption processes by Thomas (62), and Hiester and Vermeulen (63). In contrast to Glueckauf's model, equilibrium throughout the bed is not assumed, and the adsorption isotherm need not be linear, but the possibility of hydraulic dispersion is neglected. Instead of equilibrium, the adsorption is presumed to occur at a rate proportional to the concentration of the solute in the liquid phase, and also to the difference between the actual and maximum possible concentrations of the solute adsorbed on the solid particles. A reverse reaction or desorption proceeds at a rate proportional to the concentration of adsorbed solute. In symbols

$$\frac{\partial q}{\partial T} = kc\,(Q - q) - \frac{k}{K^{ad}}\,q \tag{27}$$

where q is the concentration of adsorbed solute per unit weight of adsorbent, T is time, Q is the exchange capacity, k is the adsorption rate constant, and K^{ad} is the adsorption equilibrium constant. The equation expressing conservation of leaching material within the bed can be written

$$\rho_b \frac{\partial q}{\partial V} + f_E \quad \frac{\partial c}{\partial V} + \frac{\partial c}{\partial v} = 0 \tag{28}$$

where ρ_b is the bulk density of the adsorbent, f_E the fraction of total volume filled with moving liquid, and v the total bed volume. The solution of Equations 27 and 28 can be expressed in terms of three dimensionless parameters which, for the case of elution of a saturated bed, are defined as follows:

$$\begin{aligned} r &= 1 + K^{ad}\,c_o \\ s &= \frac{kQ\ \rho_b}{1 + K^{ad}\,c_o}\left(\frac{v}{R}\right) \\ t &= \frac{k\,c_o}{K^{ad}}\left(\frac{V - vf_E}{R}\right) \end{aligned} \tag{29}$$

R is the volume rate of flow.

In case the rate of mass transfer of solute from adsorbent to liquid is being controlled by a slow rate of diffusion through stagnant liquid films surrouding the adsorbent particles, or through surface layers of the particles, the analysis must be modified. The parameters s and t are replaced by Σ and Θ, which, for the case of internal diffusion controlling, are defined as

$$\Sigma = \frac{k_p a_p kQ \rho_b}{1 + K^{ad} c_o} \left(\frac{v}{R} \right)$$

$$\theta = \frac{k_p a_p . k c_o}{K^{ad}} \frac{(V - vf_E)}{R} \qquad (30)$$

where k_p is the mass transfer coefficient and a_p the adsorbent surface area per unit bulk volume. The solution now depends upon the parameters r, Σ, and θ, instead of r, s, and t. The parameters r and Σ remain constant for any given saturated bed, while θ increases with the volume of eluting water put through the bed. Generalized breakthrough curves have been worked out for several values of r and Σ by Hiester and Vermeulen; they are plots of the quantity $(c_o - c)/(c_o - c_1)$ versus θ/Σ.

If the solid material composing the bed is a poor adsorbent, the value of K^{ad} will be small and r will be nearly equal to one. The adsorption isotherm in this case is linear, and it happens that the breakthrough curves for diffusion controlling become identical with those corresponding to control by the adsorption-desorption rate. Curves for r = 1 are given in Figure 13.

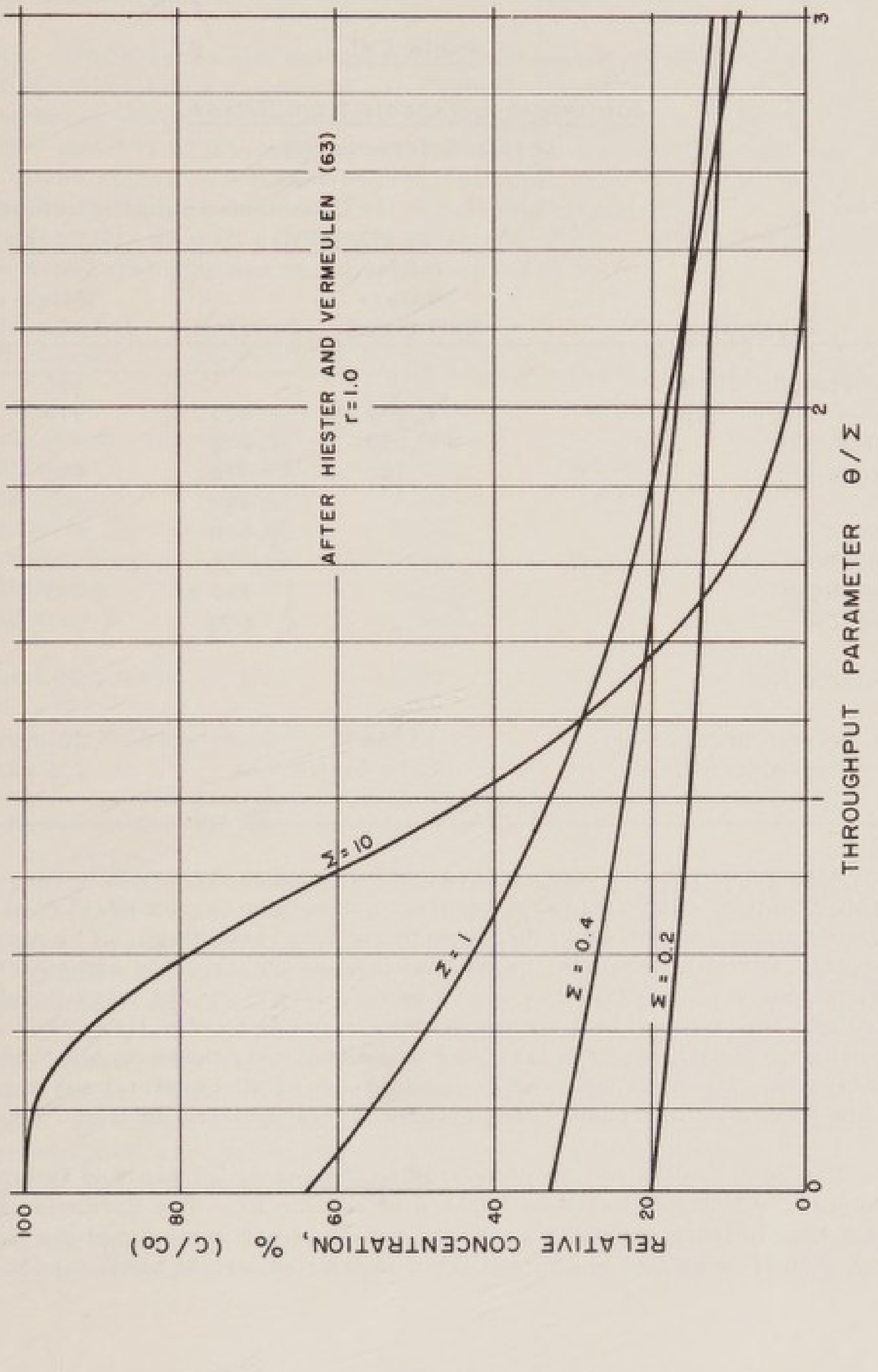
Quantities of Leachable Material

Much of the organic and inorganic matter contained in refuse can be extracted immediately by leaching. Additional material will be freed in time but normally will become available only after the worst part has been leached out. While the seriousness of any pollution will depend on the rate of leaching, the total amount of available leachable substances furnishes an idea of the possible over-all damage to a water supply and also reveals what substances are most likely to cause problems if minor percolation does occur.

Household Refuse: The problem of ground water pollution due to refuse dumped in spent sand and gravel pits has existed in England for several years and the British Ministry of Housing and Local Government has recently published the results of research designed to answer some of the basic questions (33). The basic investigation was to determine the quantity and composition of percolate from natural precipitation in dry refuse fills, and also the nature of the leachate from refuse deposited directly into the ground water. The results are quite complete, but unfortunately the British refuse is rather different from that produced in Southern California, as has been pointed out in Section III. Table XVI lists the quantities of various chemical substances extracted from refuse in the dry and wet fills. The 6-foot-deep dry fill received only natural precipitation (65 inches in 2-1/2 years), whereas in the wet fill the refuse was completely submerged and subjected to horizontal flow of water at the rate of about 3.4 gal/day/sq ft. Less chemically oxidizable organic matter (permanganate oxygen consumed) was leached out during the course of the experiments than that extractable from the fresh refuse because stabilization was occurring within the fills. However all other substances were leached in greater quantities

FIGURE 13

GENERALIZED DESORPTION BREAKTHROUGH CURVES



than were initially present in soluble form. Somewhat greater quantities were leached from the wet fill in 20 months than from the dry fill in 33 months, but a much greater volume of water passed through the wet refuse and the water was clearly in more intimate contact with the soluble substances.

Table XVI

Substances Leachable from Refuse

(From Reference 33)

	Quantities in mg/kg refuse			
	Dry Fill		Wet Fill	
Substance	Initial Water-Extractable	Leached	Initial Water-Extractable	Leached
O_2 Consumed: 30 min			685	385
4 hr	2,100	375	970	600
Chloride (Cl)	1,080	1,270		1,050
Ammonia (N)	340	370	195	550
BOD		2,490		5,150
Organic C		1,630		2,850
Organic N		72		75
Sulfate (SO_4)		840 as SO_4	545	1,300
Sulfide (S)				110
Albuminoid N				50
Period of leaching:	33 mo		20 mo	
Percolate volume:	0.14 gal/lb		2.6 gal/lb	

With regard to the completeness of leaching in the dry fill, the measures of organic matter--BOD, organic carbon, and oxygen consumed--had all fallen off to low concentrations when the experiment was terminated. The ammonia and organic nitrogen were also nearly exhausted, but chloride and sulfate were still appearing at concentrations of 800 ppm or more. In the wet fill refuse was dumped into the water over a period of time in order to simulate operating conditions in gravel pits, and the last load was added only three months before the final analysis. However, the end concentrations of all chemical substances were all below 80 ppm, the highest being chloride at 78 ppm and BOD at 74 ppm.

The British data can be compared to the quantities leached in the Riverside studies, which are listed in Table X of Section III. The Riverside figures are only approximate since the weight of refuse is not known, and the figures in Table X should be halved to convert them to undried weight basis (approximate).

The chloride and nitrogen values are of the same order of magnitude as the British data, but the leached BOD at Riverside was much greater than the British and probably reflects the greater amounts of garden vegetation in Southern California refuse.

Ash: While ashes are not a major component of Southern California rubbish fills, there are a number of ash deposits, mostly adjacent to municipal incinerators, which offer some threat to the ground water quality. The ash material itself is of interest because it exhibits a property which may also belong to mixed refuse to a lesser degree, namely, base exchange capacity. Merz's study of ash dump leaching (64) has demonstrated the nature of the leachable material, as well as the rate of its extraction. The average leachable quantities of mineral ions are summarized as follows:

Substance	Leached with Water lb/cu yd	Leached with Water % by weight*	Acid Extract lb/cu yd	Acid Extract % by weight*
Sodium	1.48	0.078	5.46	0.29
Potassium	0.92	0.049	7.24	0.38
Calcium	0.40	0.021	48.50	2.57
Magnesium	0.27	0.014	4.61	0.24
Chloride	1.65	0.087		
Sulfate	4.20	0.22	5.62	0.30
Alkalinity	0.79	0.042		
Nitrate - N	0.15	0.0025		
Total Measured	10.5	0.56	70.9	3.8
Leached Volume:	0.038 gal/lb* 4.3 in/ft			

* Based on ash weight of 70 lb/cu ft.

The total weight of measured leached substances amounted to only 0.6 percent, whereas the material soluble in acid, not including chloride and carbonates, was four percent. As Merz noted, only a small fraction of the mineral matter of the ash was easily extractable. One reason for this is the low solubility in water of a large part of the acid extractable material, much of which was apparently calcium carbonate. However, Merz demonstrated in another phase of the investigation that the ash has a base exchange capacity of about 1,260 grains Ca per cu ft, or at an ash unit weight of 70 lb/cu ft, about 13 meq/100 gm. Assuming the attached cations had an average equivalent weight of 30, they would account for another 0.4 percent of acid extractable material not susceptible to leaching by pure water.

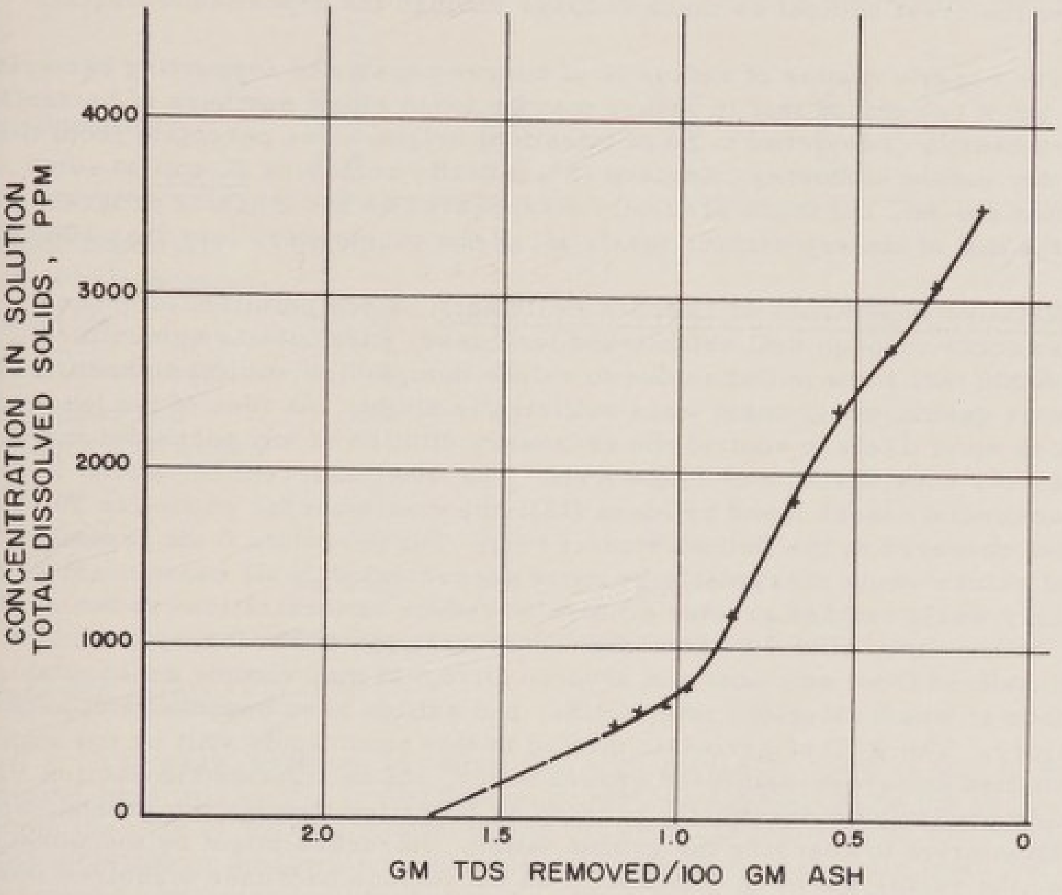
The extracted salts obtained by Merz may be compared to a sample of Alhambra incinerator ash analyzed by the Department of Water Resources. The material was leached with distilled water by the process of mixing the ash and water together, settling, decanting half of the liquid, and replacing with fresh water. Total dissolved solids were determined on the decanted liquor and the procedure was repeated until 14 liters of water had been passed over the 2,270-gm ash sample. The results may be plotted as the adsorption isotherm shown in Figure 14. There the concentration of solids in the solution at any stage is plotted against the total weight of the material desorbed from the ash. By extrapolating the isotherm to zero supernatant concentration, the weight of extractable solids initially present with the ash is made manifest. Because of the steepness of the low concentration portion of the curve the extrapolation cannot in this case be performed very confidently, but it would appear that the leachable solids amounted to at least 1.5 percent of the ash, and were probably closer to 2.0 percent. As determined by the breaking point of the curve, the easily extractable solids were about one percent of the ash, and probably included a large fraction of non-adsorbed salts. The lower value obtained by adding up the quantities of individual ions in Merz's experiments may be attributed to some prior leaching of the ash by rain while in the dumps, and possibly to the presence of substances other than those measured.

An incident has been described by Haupt (65) wherein a quarry pit used as an ash dump polluted a row of wells 1,000 feet away. The refuse had apparently been leached by rain water, and the concentration of total dissolved solids beneath the refuse was 5,000 ppm. Concentrations of other substances at that point were total hardness 2,600 ppm (of which only 300 ppm was carbonate hardness), sulfate 1,370 ppm, carbon dioxide 100 ppm, iron 28 ppm, and manganese 14 ppm.

Reported Experiences: When organic refuse is dumped directly into water the concentrations of salts obtained are usually not as great as in the case of leaching by percolation because of the relatively large volume of water immediately available for dilution. However the entire body of water soon becomes septic, and as sulfate reduction sets in the pit presents a large surface from which hydrogen sulfide can diffuse to the atmosphere. Longwell (46) has given the analysis of water in a gravel pit into which refuse had been dumped: BOD 975 ppm, chloride 128 ppm, total hardness 523 ppm, and sulfide 9.5 ppm. McDermott (66) has reported measurements on a small stream flowing through a gravel pit filled above the water table with refuse. The effluent BOD was about 480 ppm. In the New York landfill survey (32) sea water entering one of the fills with the tide returned on each cycle with up to 12 ppm sulfide, 100 ppm BOD, and 50 ppm ammonia nitrogen.

The ability of carbon dioxide to dissolve minerals in the ground has already been mentioned. The Pasadena, California, Water Department has suffered two examples of this phenomenon which resulted in serious impairment of ground water quality. In 1934 and 1935 heavy runoff following forest fires deposited a large amount of organic debris and fine sediment behind Devils Gate Dam in the Arroyo Seco (77). The water from galleries beneath the Arroyo soon

LEACHING OF INCINERATOR ASH – EQUILIBRIUM LINE



DATA OF CALIFORNIA DEPARTMENT OF WATER RESOURCES ON ASH FROM ALHAMBRA MUNICIPAL INCINERATOR acquired unpleasant tastes and odors, increased in hardness by 300 ppm, and in iron by 7 ppm. Heavy growths of crenothrix bacteria appeared in the galleries. Several of the municipal wells had to be abandoned. The organic matter apparently has persisted or has been replenished slightly by succeeding years of runoff, and an analysis of the gallery water in January 1961 gave 125 ppm carbon dioxide, 5.5 ppm iron, and 298 ppm hardness. In 1943 a similar series of events occurred behind a debris dam in Brown Canyon, farther up the Arroyo Seco. Here the high concentrations of iron, hardness, and carbon dioxide were largely relieved when weep holes in the dam were plugged, and the stream could pass over the crest without as much seepage through the deposited material.

The organic matter of refuse is of course capable of supporting bacterial growth, but it is curious that in refuse may be found small numbers of bacteria which are usually considered to be of intestinal origin. The percolate from the leached dry refuse at Bushey, England (33) initially contained E. coli at some one million per ml, but these gradually disappeared as the leaching progressed until at the end of the experiment nearly all of the counts were less than 100 per ml.

Relative Importance of Leached Pollutants: Since pollution of ground water can occur through well established land uses, particularly agriculture, it is probable that some pollution due to refuse dumps also would be permitted if the water quality impairment were sufficiently slight. An idea of the leached substances most likely to control the necessary dilution of any percolate may be obtained from consideration of Table XVII. The maximum concentrations listed are approximately those found by Merz (18); the maximum for sulfide is 30 ppm, which was observed in the British studies (33). The percolate from greater depths of refuse would presumably be more concentrated in all substances, and accordingly would require greater dilution to reduce concentrations to acceptable limits. In any case, it will be seen that concentrations of hardness, iron, and nitrate (oxidized from ammonia and organic nitrogen) may remain unacceptable at dilutions at which chloride, total solids, and sulfide have become relatively satisfactory. The BOD of percolate diluted to this point would still be too high to be satisfied in oxygen-saturated ground water, but as indicated in Section VI, BOD would probably be drastically reduced by filtration through the ground, and by oxidation prior to reaching the ground water. Hardness might be the worst problem; the values shown in Table XVII do not include hardness dissolved from the ground by carbon dioxide.

Deleterious substances other than those specifically listed in Table XVII or previously mentioned are undoubtedly present in refuse, and are capable of being leached into the ground water. Included in domestic refuse would be medicines, inflammable liquids, insecticides, weed killers, and other chemicals sold for specific household purposes; any refuse leachate would contain quantities of boron, copper, chromium, and other substances whose adverse effect on crops or humans is recognized. However the percentage of such materials in the refuse and in the percolate therefrom would be so minute compared to the amounts of undesirable materials herein considered, that they could never constitute an independent problem. However specific liquid or solid industrial wastes which might be dumped in truckload quantities could be a potential hazard to ground water.

Table XVII

Dilution of Percolate from Refuse

Substance	Concentration, ppm Percolate (maximum)	1:100 Dilution	1:1,000 Dilution	USPHS Drinking Water Standards
Total Dissolved Solids (inorganic)	20,000	200	20	500
Chloride	2,200	22	2.2	250
Total Hardness	8,000	80	8	-
Ammonia and Organic Nitrogen	1,300	13	1.3	NO_3 - N: 10
Iron	300	3	0.3	0.3
Sulfide	30	0.3	0.03	-
BOD	30,000	300	30	-

Rate of Leaching

The basic problem in connection with refuse leaching is that of determining what will be the concentration of various substances in the percolate, given the rate of water application and the physical make-up of the refuse bed. The situation is analogous to the elution of adsorbed materials from beds or columns of granular media, but there are at least three complicating factors that make any satisfactory theoretical treatment almost hopeless. These are: (1) the fact that the refuse is not initially soaked with water, and therefore whe the first percolate appears the bed has already been leached to some indeterm nate extent; (2) refuse is so coarse and nonuniform that the hydraulic irregularities of the flow may completely obscure any orderly desorption behavior; and (3) the refuse undergoes microbial attack during the period that it is being leached, thereby altering its composition and properties. However, the adsorption theory is of some value in permitting a qualitative visualization of the relationships among some of the variables, and possibly also in extrapolating available data.

Theoretical treatments applicable to column adsorption phenomena have been fairly successfully applied to the leaching of salts from soil. Van der Molen (67) has employed the equations of Glueckauf to the measured rate of removal of sea salt from reclaimed lands in Holland, and obtained reasonable agreement with theory for permeable soils. Gardner and Brooks (68) have applied Hiester and Vermeulen's theory to the leaching of salts from soils in columns. The breakthrough curves obtained experimentally could be fitted quite well with the theoretical curves, but it was noticed that the hydraulic dispersion theory would have worked just as well.

A family of generalized breakthrough curves as given by Hiester is shown in Figure 13. They all correspond to a value of 1.00 for the equilibrium parameter, r; this is the case for leachable materials which are weakly adsorbed by the material of the bed. It is presumed that the leachable substances of refuse are loosely held, so that the curves shown in Figure 13 should apply. In order to fit the general curves to a particular experimental breakthrough curve, values must be assigned to two parameters; these are the column capacity parameter Σ, and a factor B which will convert the actual throughput to the throughput parameter Θ/Σ. With reference to Equations 30, the constants A and B must be determined, where

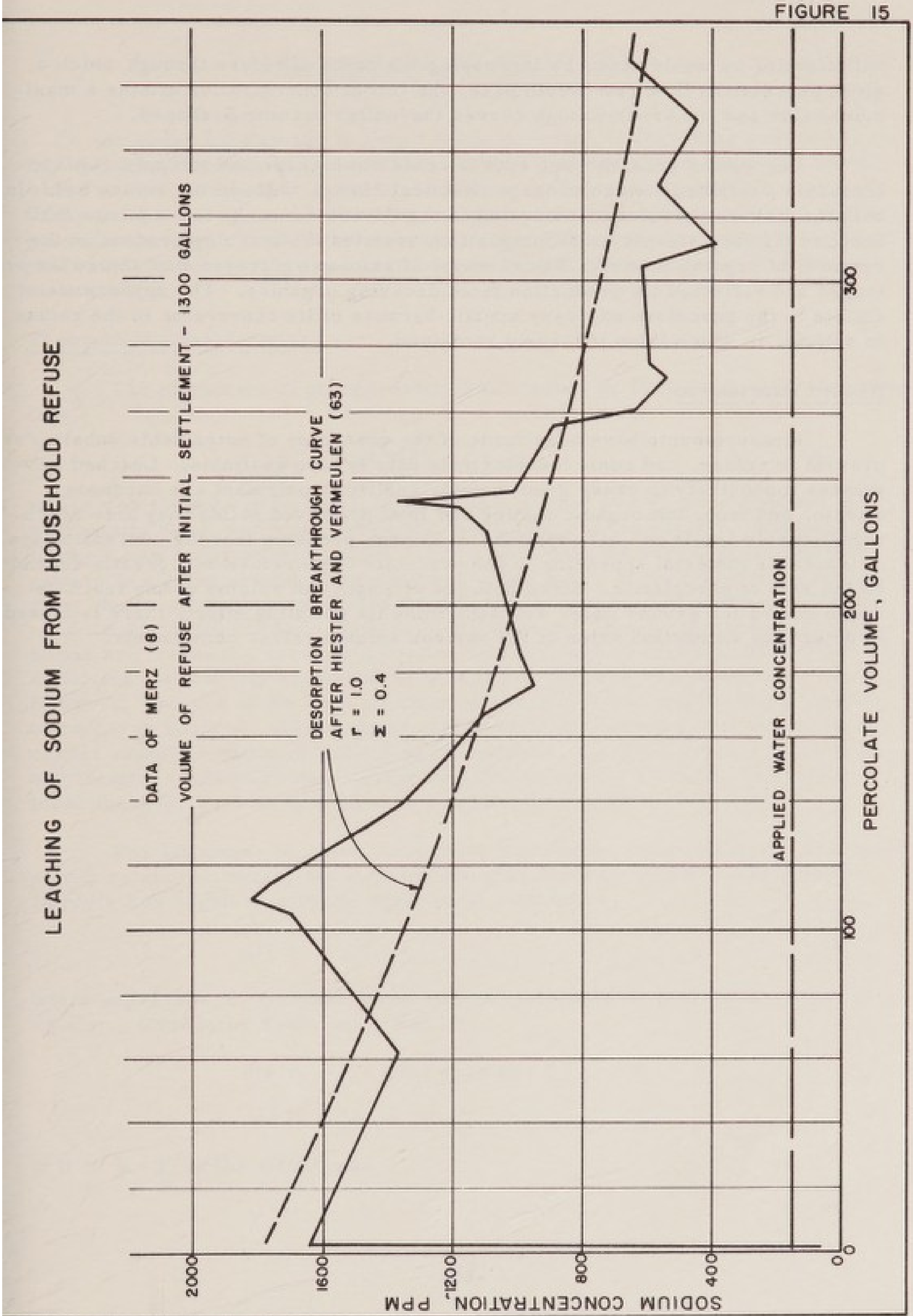
$$\Theta = Av/R \qquad (31)$$

$$\Theta/\Sigma = B\,(V - vf_E)/v \qquad (32)$$

The quantity vf_E is the volume of water saturating the bed before percolation begins; the origin of the breakthrough curves of Figure 13 is therefore the moment when fresh eluting water begins to flow out the bottom of the bed after having displaced the original liquid. In reasonably dry refuse the first water which appears will be essentially all water applied at the top, and $V\text{-}vf_E$ can therefore be taken as the volume of percolate.

As an Example, Figure 15 shows the concentration of sodium obtained by Merz (18) from the percolate of Test Bin No. 1 at Riverside, plotted against volume of percolate. The Hiester breakthrough curve for $\Sigma = 0.4$ has been fitted, using values for the constants of A = 0.00022/day and B = 0.076. Because of the large fluctuations in concentration, a longer period of data would be necessary to establish precise values, and the curve as it is could be fitted about as well for any value of Σ between 0.3 and 0.8. However, since Σ is inversely proportional to the flow rate, the curves of Figure 13 indicate the effect of changing the rate of percolation. If it should be doubled, giving Σ a value of 0.2 instead of 0.4, the curve would become flatter; the initial concentration of sodium would be reduced by possibly one-third, and the ion would be removed from the bed in a greater volume of water. Reducing the percolation rate would have the opposite effect of producing a more concentrated percolate at first, and leaching the salt out in a smaller volume of water. The theoretical curves also suggest that when the column capacity parameter Σ is increased

FIGURE 15



LEACHING OF SODIUM FROM HOUSEHOLD REFUSE sufficiently, as would occur by increasing the depth of refuse through which a given percolation flow rate would pass, the initial concentration attains a maximum value and the breakthrough curves thereafter become S-shaped.

Desorption breakthrough curves could not be expected to apply at all to leachable substances which undergo chemical change while in the refuse bed. In both the British percolation study and in the Riverside work, the leachate BOD dropped off very steeply as decomposition assisted the leaching process in the removal of organic matter. The removal of ammonia nitrogen was somewhat prolonged and reflected its production from decaying organics. The appearance of sulfate in the percolate was very erratic because of its conversion in the refuse to sulfide, in which from it largely remained.

Needed Information

Measurements have been made of the quantities of extractable substances present in refuse, and some leaching rate data is also available. Leached substances most likely to cause ground water quality impairment are hardness, nitrate, and iron, but organic matter and total dissolved solids may also reach objectionable levels in well supplies. It is very probable that the concentrations of leachable material appearing in the percolate from refuse will greatly depend on the rate of percolation. Since both the strength and volume of any leachate which enters the ground water will determine its polluting effect, there is a need to determine extraction rates of the various soluble refuse components.

VI. TRAVEL OF POLLUTION

In order for damage to result from the entry of degrading substances into the ground water, it is necessary that the affected water move to a point where it is withdrawn for some beneficial use. The ground water in most areas moves slowly, and will carry polluting materials with it. Mixing of the ground water and its load of dissolved matter also occurs during travel, and will result in decreasing concentrations as the pollutants move away from their source.

Theoretical Considerations

The movement of ground water is described by Darcy's Law for saturated flow,

$$v = - k \text{ grad } (h + \frac{P}{w}) \qquad (1)$$

where v is the rate of flow per unit of aquifer, grad $(h + \frac{P}{w})$ the gradient of hydrostatic head, and k the hydraulic conductivity of the aquifer. Although Darcy's Law holds at every point within a saturated porous medium, the flow pattern in a given system is determined by the shape of the impervious boundaries, the location, extent, and magnitude of the sources and sinks, and by the conductivity of the medium throughout its volume. Natural ground water situations are frequently very complicated, and it would be impractical to make sufficient conductivity determinations to characterize a region with any detail. However, because of the huge volumes of water involved and the fixed location of recharge sources, the direction and speed of ground water movement do not change rapidly. Although subject to fluctuations, the ground water velocity at any location tends to remain relatively constant unless there are important local changes, such as operation of a spreading ground or well field.

For situations in which the aquifer boundaries and conductivity are known or can be approximated, the flow pattern may be described by making use of Darcy's Law together with the equation of continuity,

$$\text{div } v = 0 \qquad (33)$$

which expresses the fact that water cannot be created or destroyed within the aquifer. Combining Equations 1 and 33,

$$\text{div } v = - k \text{ div grad } (h + \frac{P}{w}) = 0$$

$$v^2 (h + \frac{P}{w}) = 0$$

If $H = h + \frac{P}{w}$ is the static head,

$$v^2 H = 0 \qquad (34)$$

If the boundary conditions are sufficiently simple, Equation 34 can be solved analytically to give H (and by differentiating, v) at all points. There are graphical and numerical techniques for obtaining approximate results in more difficult cases. Equation 34 also describes the steady flow of electricity and heat in conducting media, as well as several other physical phenomena; solutions obtained in these problems therefore apply to corresponding cases of ground water flow, and electrical model analogs can be used to obtain experimental solutions.

One flow case of some interest is that of a well pumping at a constant rate from an aquifer carrying ground water at a uniform constant velocity. If it is assumed that the direction of flow is everywhere parallel to the water table, which is nearly correct except in the well's immediate vicinity, the surfaces of constant head are all vertical and the flow pattern is that shown in Figure 16. The well draws all of its water from a strip of aquifer whose maximum width is Q/vd, where Q is the well pumping rate, v the flow per unit area in the aquifer, and d the depth of the aquifer. If it were not for dispersion phenomena, no polluting substances which might enter the ground water outside of the zone of influence could ever reach the well. Moreover, if a well's zone of influence lies partially under a dump undergoing leaching, the refuse percolate destined to reach the well will arrive diluted by the entire ground water flow within the zone of influence.

Experiments with tracers, both in the field and in laboratory columns, have repeatedly demonstrated the fact that a given blob of water flowing in a porous medium does not remain intact as it moves, but instead comingles with the adjacent water and thereby becomes spread over an increasingly greater volume. It has been shown that the dispersion process may be described by a diffusion equation. Fick's Law of diffusion states that the rate of mass transfer of a diffusing substance is proportional to its concentration gradient, or, in symbols,

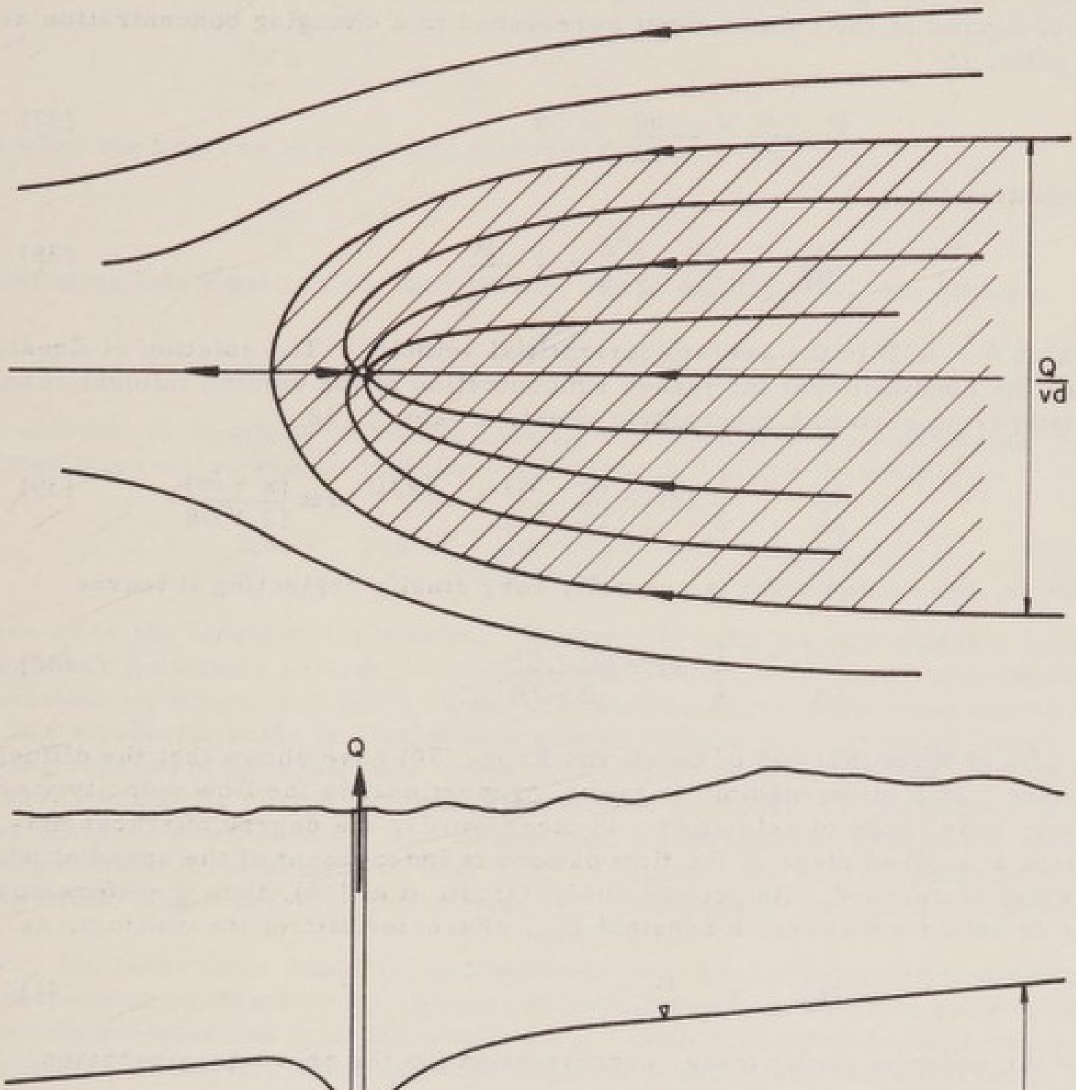
$$q = - PD \text{ grad } c \qquad (35)$$

where q is the rate of transfer across a unit area, c is concentration, P the porosity, and D the diffusion constant, which has the dimensions of length^2/time. However in flow through porous media the diffusion is apparently brought about by variations in velocity through the pores and by variable lengths of paths of different particles of water. It turns out that the diffusion constant D varies with direction, and in particular, is greater in the direction of flow than in a perpendicular direction.

Considerable experimental work has been performed to measure dispersion in laboratory columns packed with porous media. In this case, the boundary essentially limits the observable dispersion to only one direction, that of the column axis. The rate of transfer of a tracer is then equal to the sum of two effects, namely, dispersion superimposed upon carriage at the average flow rate. In symbols,

FIGURE 16

ZONE OF INFLUENCE OF WELL



$$q = vc - PD \frac{\partial c}{\partial x} \qquad (36)$$

where x is the length coordinate along the column. The continuity equation for tracer, expressing the fact that any difference in the transfer rate on either side of a point in the column must correspond to a changing concentration at that point, is

$$P \frac{\partial c}{\partial t} + \frac{\partial q}{\partial x} = 0 \qquad (37)$$

Substituting for q,

$$\frac{\partial c}{\partial t} = D \frac{\partial^2 c}{\partial x^2} - \bar{v} \frac{\partial c}{\partial x} \qquad (38)$$

in which $\bar{v} = v/P$ is the average interstitial velocity. The solution of Equation 38 for the condition that the tracer concentration of the column influent changes abruptly from zero to a thenceforth constant value c_o is

$$\frac{c}{c_o} = \frac{1}{2} \left(\text{erfc} \frac{(x - \bar{v}t)}{(2\sqrt{Dt})} + e^{\bar{v}x/D} \text{erfc} \frac{(x + \bar{v}t)}{(2\sqrt{Dt})}\right) \qquad (39)$$

However, the second term is normally very small; neglecting it leaves

$$\frac{c}{c_o} = \frac{1}{2} \text{erfc} \left(\frac{x - \bar{v}t}{2\sqrt{Dt}}\right) \qquad (40)$$

The data of Rifai (69) and of Orlob and Krone (70) have shown that the diffusion constant D in a given medium is nearly proportional to the flow velocity $\bar{v}$ over a fairly wide range of velocities. In other words, the degree of tracer dispersion at a given stage of the flow pattern is independent of the speed at which the stage is reached. As pointed out by Orcutt et al (78), it is therefore possible to define a dispersion constant D_m, characteristic of the medium, as

$$D_m = \frac{D}{\bar{v}} \qquad (41)$$

Then the equation giving tracer concentration for the above case becomes

$$\frac{c}{c_o} = \frac{1}{2} \text{erfc} \left(\frac{x/\bar{v}t - 1}{2\sqrt{D_m/\bar{v}t}}\right) \qquad (42)$$

The pattern of pollutant concentration downstream from an extensive source can be calculated by making use of several approximations and idealizations. Taking x as distance in the direction of flow and y as distance in a horizontally perpendicular direction, the equation of continuity can be written

$$P \frac{\partial c}{\partial t} + \frac{\partial q_x}{\partial x} + \frac{\partial q_y}{\partial y} = 0 \qquad (43)$$

where q_x and q_y are rates of pollutant transfer in the x and y directions. Here c is the concentration of pollutant over a vertical probe through the surface aquifer. Transfer by the diffusive process in the x direction is small compared to mass movement in the ground water, and may be neglected. Then

$$q_x = vc$$
$$q_y = - PD \frac{\partial c}{\partial y}$$

and when the process has reached a steady state,

$$\frac{\partial c}{\partial t} = 0$$

Substituting into Equation 43, and making use of Equation 41, one obtains

$$\frac{\partial c}{\partial x} = D_m \frac{\partial^2 c}{\partial y^2} \qquad (44)$$

The solution of Equation 44 for the case in which the pollutants are being emitted from a line source is shown in Figure 17. It can be written

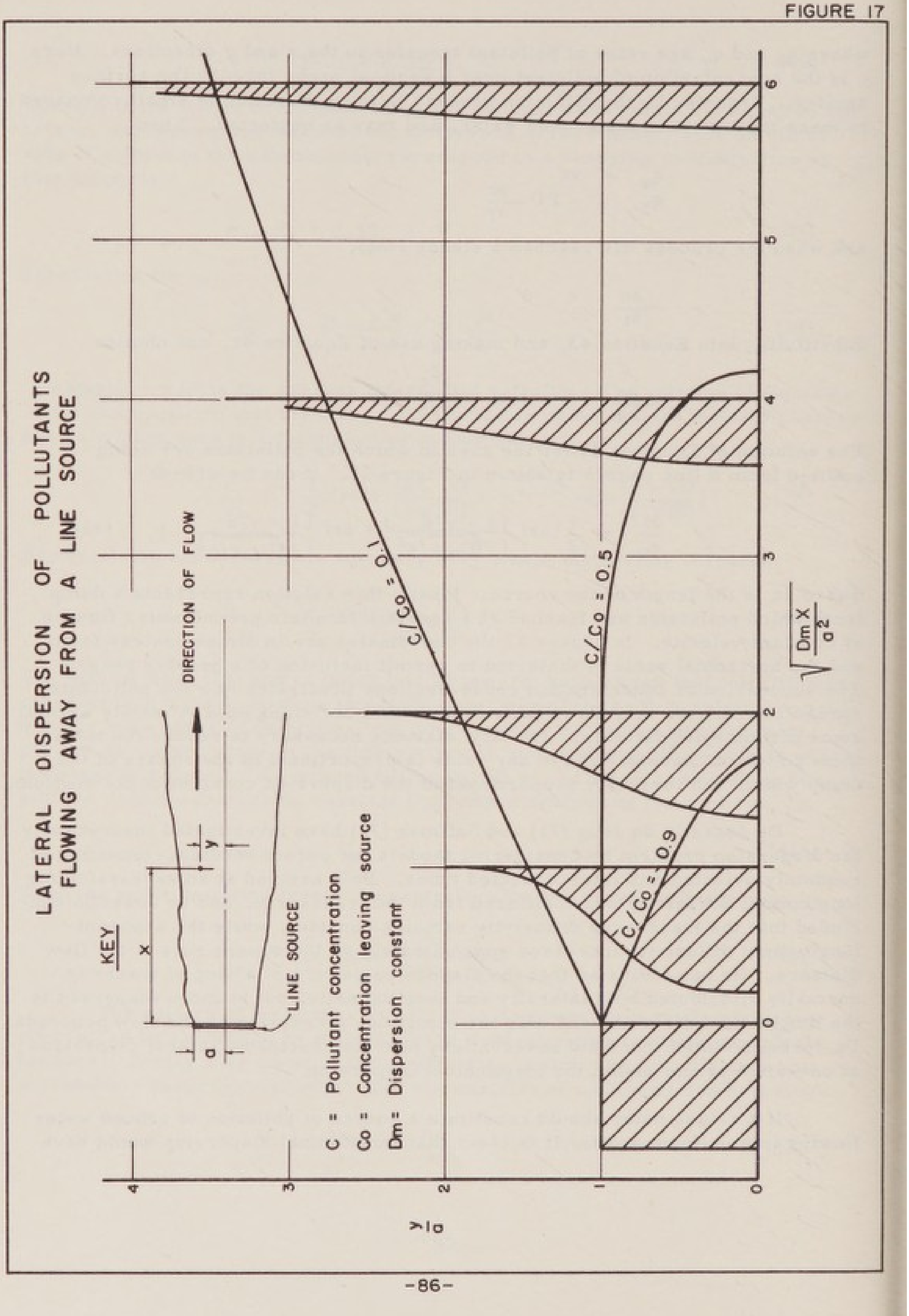
$$\frac{c}{c_o} = \frac{1}{2} \left(\text{erf} \left(\frac{1 - y/a}{2\sqrt{D_m x/a^2}} \right) + \text{erf} \left(\frac{1 + y/a}{2\sqrt{D_m x/a^2}} \right) \right) \qquad (45)$$

where 2a is the length of the source. Ideally this solution represents a dump from which pollutants are leached at a constant rate into ground water flowing at constant velocity. In Figure 17 the coordinates are in dimensionless form and the horizontal scale is distorted to permit inclusion of a greater range. The succession of concentration cross-sections illustrates how the pollutants spread laterally as they flow, their distribution becoming approximately normal some distance downstream. The flow distance necessary to reduce the maximum pollutant concentration to any value is proportional to the square of the dump width, and inversely proportional to the dispersion constant of the medium.

De Josselin de Jong (71) and Saffman (72) have investigated theoretically the dispersion problem by considering models for porous medium consisting of randomly oriented and interconnected tubes. Both arrived at an expression for longitudinal diffusivity which differed from the transverse, and de Josselin concluded that the transverse diffusivity remains constant, while the apparent longitudinal diffusivity increases approximately as the square root of the flow distance. He has indicated that the dispersion pattern of a slug of tracer is normally distributed both laterally and longitudinally, but is more dispersed in the longitudinal direction and with the elongation increasing as the flow proceeds. De Josselin stated that field observations have demonstrated lateral dispersion of one-sixth to one-eighth the longitudinal dispersion.

If a refuse dump should constitute a source of pollution to ground water flowing in a uniform aquifer it is clear that longitudinal dispersion would have

FIGURE 17



LATERAL DISPERSION OF POLLUTANTS FLOWING AWAY FROM A LINE SOURCE

KEY

C = Pollutant concentration

Co = Concentration leaving source

Dm = Dispersion constant

little effect on the concentration of pollutants appearing in downstream wells, except possibly at a very great distance from the dump. The longitudinal spreading would cause the pollutants to appear earlier than would be predicted from the average flow velocity, but the concentration would rise to the same level as would occur in the complete absence of dispersion. On the other hand transverse dispersion would dilute the pollutants by spreading them out laterally and downward in the aquifer, but would thereby increase the extent of the affected zone. If the aquifer is not uniform, but consists of layers of varying conductivity with large-scale irregularities, then both longitudinal and transverse dispersion will be effective in diluting the pollutants.

Dilution in Ground Water

The upper layer of ground water may become strongly polluted by leaching directly through a bed of refuse, or by receiving percolate from an overlying deposit. Thereafter dilution of the pollutants may proceed in at least two different ways, namely, (1) by mixing with adjacent ground water as it enters a well casing and is pumped out, and (2) by dispersion into the adjacent ground water during flow.

Effect of Wells: With reference to Figure 16, it may be seen that in the absence of horizontal dispersion, any refuse percolate entering a well's zone of influence will be diluted into the entire volume of ground water entering the well. The dilution may not take place until the moment when the pollutants enter the well casing to mix with clean water converging from throughout the zone of influence. As an example, according to the Riverside leaching studies of Merz (18) it might be possible to leach 0.8 lb of sodium per cu yd of fill in the first year. In a dump 30 feet deep and 500 feet long in the direction of ground water flow, sodium would be released at the rate of 1.2 lb/day per foot of dump width. Then if the aquifer were 100 feet deep and the ground water were moving at 0.5 cu ft/sq ft/day, the sodium concentration, averaged over the aquifer depth, would be about 380 ppm greater than that of the unaffected water. A well located downstream such that its zone of influence lay completely within the shadow of the dump would contain at least this concentration of sodium regardless of how much vertical mixing occurred between dump and well. If the well penetrated only a shallow depth of the surface aquifer, the full depth of flow in the aquifer might not be available for dilution of percolate, and the concentration of pollutants in the well discharge could be greater than the aquifer average. On the other hand, a well whose zone of influence included some area not overlain by the dump, or which also pumped water from lower aquifers, would produce smaller concentrations of pollutant. In any case hydraulic mixing or dispersion in the horizontal direction only will be effective in reducing the concentration of pollutants in wells below dump sites, and then only if the mixing is great enough to bring fresh water in from outside of the zone of influence.

It may be noted that in wells having their influence zones either completely within or completely without the dump's shadow, the pollutant concentration is independent of the ground water velocity. However, the width of the zone of influence is inversely proportional to the ground water velocity; therefore the influence zones of randomly located wells in the area downstream from a dump will be less likely to intersect the dump if the ground water is moving rapidly, but wells located directly in the path of the pollution will have less chance of collecting fresh diluting water from outside the dump shadow. Also it is clear that a dump of given area will produce lower concentrations of pollutants in downstream wells if it is spread out perpendicularly to the direction of ground water flow, but that more wells are likely to be affected by some pollution.

Dispersion: Pollutants having entered the ground water within a limited region may be diluted by mixing laterally with the surrounding clean water. However, the effectiveness of such dispersion is not very great in homogeneous formations consisting of sand and small gravel. The dispersion constant (D_m of Equation 41) as determined from Rifai's data (69) with sand of 1.1 mm effective size is about 0.23 cm for dispersion in the direction of flow. The theoretical analysis of de Josselin (71) has indicated that the dispersion constant perpendicular to the flow should be proportional to the average length of pore channels between junctions, i.e., to the average particle size. Also, the lateral dispersion constant, according to de Josselin's theory and observations, is but a small fraction of the longitudinal constant; this has been confirmed by Bear and Todd (79) who observed the dispersion of a blob of salt injected into water flowing through a bed of fine glass beads. The salt pattern was elongated in the direction of flow, with the long dimension about five times its width. The lateral dispersion constant for one-mm sand is therefore about one-fifth of 0.23 cm, or about 0.05 cm. With reference to Figure 17, the value of $\sqrt{D_m x/a^2}$ corresponding to $D_m = 0.05$ cm, a dump width of 200 feet ($a = 100$ ft), and a downstream distance x of 10 miles, is about 0.1. The lateral dispersion over this long distance would therefore be practically undetectable if the aquifer were uniform fine sand. Actually many surface aquifers consist of cobbles and boulders, so that the effective size might be 0.1 ft or more, with a correspondingly larger value for D_m. However, even with $D_m = 0.2$ ft, the 10-mile flow distance and 200-foot dump width make $\sqrt{D_m x/a^2}$ equal to only about 1.0; the dispersion would still be rather slight, and the central portion of the pollution would scarcely be diluted at all. Dilution of ground water pollutants by means of dispersion induced by random velocity variations is not to be relied upon for effective abatement, except where the flow irregularities are of large amplitude, such as might be induced by numerous boulders or lenses of fine sediments in the aquifer.

Reported Experience

There is considerable information available on the transmission in the ground water of contamination from refuse dumps, but the data are generally inadequate to permit determination of the dilution being afforded. The only well-defined example of pollution travel from a dump in Southern California is that of the Riverside sanitary fill, which was studied at length by Merz (18, 73). In that case the bottom of the fill was in partial and intermittent contact with the ground water, and therefore subject to some direct leaching. There were definite increases in mineral and organic constituents as the ground water passed under the fill, but at a distance of one-half mile the difference in quality from the native ground water was scarcely detectable. It was concluded that mineral matter in the ground water stream undergoes limited vertical diffusion, and that in aquifers of greater thickness than 100 feet the bottom water probably remains unimpaired.

A case of refuse dumping into water-filled gravel pits at Egham in Surrey County, England, has been described by Furness (45). Three neighboring pits were filled with household refuse between 1940 and 1960, the greater part being placed between 1954 and 1960 at the rate of 100,000 tons per year. The water depths averaged 12 feet, and the ground water was about 8 feet below the ground surface. The flow velocity was about two feet per day. Analyses of water from the filled pits and from downstream pits and wells have been made periodically. Results have been that the chloride concentration rises from about 50 ppm in the native water to over 800 ppm in the pits receiving refuse. In adjacent downstream gravel pits the maximum recorded chloride concentration was 290 ppm; chlorides generally decreased away from the source, and in a quarry 3,500 feet downstream the maximum was 70 ppm. The full effects of the refuse are not yet known because of the slow rate of travel, and the maximum salt concentrations in the pits next to those being filled were observed only in April 1961, so that they are not necessarily yet diminishing permanently. However, organic and bacterial pollution has disappeared within one-half mile from the refuse pits. Considerable dilution has clearly taken place during the flow, and has probably been promoted by the presence of a fairly large area of water-filled pits along the path of the flow.

Bodies of open water lying in the path of ground water flow will increase mixing of the stream in two ways. One results simply from the freedom of motion afforded by the unconfined pool; water entering the pond at the extremely slow velocities characteristic of underground flow will be thoroughly mixed by wind action and convection currents. In addition, a pond has virtually infinite permeability compared to any porous medium, and will therefore "attract" flow through itself. For example, a cylindrical pit completely penetrating the surface aquifer will collect flow from a width of aquifer equal to twice the pit diameter.

The hydraulic conductivity of refuse submerged in the ground water is of some importance since it controls the flow of leaching water. No reliable measurements of the conductivity of combustible rubbish are known, but values are available from the British pilot plant studies (33). Initially the refuse had a conductivity of about 680 ft/day; after water had percolated through the bed at the rate of about 0.5 cu ft/sq ft/day for three months, the conductivity had fallen to about 100 ft/day. Other protracted experiments gave similar final values, and by comparison with Table XIII, it will be observed that the refuse remains quite permeable, although possibly somewhat less so than a coarse aquifer. Apparently the rate of flow through rubbish will be of the same order of magnitude as that through the aquifer itself. However, any difference between the permeability of the refuse and that of the surrounding aquifer will cause a corresponding difference in the flow velocity through each, and will thereby affect the rate of extraction of pollutants. If the ground water table is below the bottom of a dump, but close enough that the capillary fringe stands within the refuse, flow through the refuse will still take place, but at a rate which rapidly decreases with decreasing degree of saturation. Because of the high concentrations of pollutants which can enter the ground water even when it is moving slowly, the lowest permissible dumping elevation, when it is intended to control pollution by avoiding contact of refuse with the ground water, should be above the maximum anticipated level of the capillary fringe.

A complete history of well supply pollution from a garbage dump has been described by Rössler (59). About 650,000 cu yd of refuse were dumped near Krefeld, Germany, between 1913 and 1929. Some garbage was deposited in water standing in the bottom of an empty gravel pit. Nine years after the first dumping, pollution in the form of high salt and hardness concentrations began to appear in wells one mile downstream. The course of travel cannot be clearly traced because heavy drafts from several wells were influencing the direction of ground water movement, but wells up to five miles distant were seriously affected. Maximum chloride measured was about 260 ppm, compared to 40 ppm in the native water; up to 900 ppm hardness was obtained, the background being about 200 ppm. Small rises in iron, manganese, and ammonia were observed, but no contamination of wells with coliform bacteria took place, and the pollution by organic matter was very slight. For the wells nearest the dump, the pollution lasted 18 years.

A similar occurrence, also in Germany, was given in a review of ground water pollution examples by Lang and Bruns (80). The town of Schirrhof for 25 years had dumped ashes and household refuse into an empty sand pit which extended below the water table. The dump was covered and made into a park, and 15 years later the hardness in wells 2,000 feet downstream rose from 200 ppm to 1,150 ppm. Iron and manganese concentrations also increased and the water acquired an unpleasant taste. At the time of writing the condition had continued for several years with no signs of improvement.

Schlinker (74) has reported another German experience with water supply pollution due to refuse. In this case the source of pollution was not definitely known at first; well analyses were made throughout the area, and the direction of ground water flow was determined. Plotted contours of chlorides and hardness focused on an old refuse dump, whereas the permanganate demand contours surrounded an unsewered residential area. The maximum chloride concentration observed was 120 ppm against a background of 13 ppm; maximum carbonate hardness was 210 ppm, compared to 90 ppm in the unaffected water.

The flow velocity was about 2.5 ft/day, and the pollution had taken four to six years to reach the wells.

Examples of industrial waste pollution of ground water are common, but usually the data are not sufficient to permit quantitative evaluation of the dilution being afforded. The California Departments of Water Resources and Public Health investigated the travel of hydrocarbons and detergent in the ground water of the Mojave River near Barstow (75). There a tongue of pollution extended some 2-1/2 miles downstream from the waste discharge area. The depth to ground water was about 20 feet, and no pollution was found in wells with perforations deeper than 100 feet. Because of the low concentrations of the pollutants it is not possible to observe a clear pattern of dilution, but since the range of ABS detergent concentration throughout the zone of pollution was 0.05 to 1.10 ppm, the material was being diluted not more than about 20 times over the 2-1/2 mile stretch.

The Office of the Los Angeles County Engineer has maintained records of an occurrence of well pollution by water softener regeneration brine at Saugus, California. From about 1942 to 1949 some 50,000 gallons per day of brine were discharged to seepage basins. Wells downstream successively became affected with high salt concentrations. The last wells to be contaminated were those of the Saugus School, 2,000 feet from the basins, which became unusable in 1946. After 1949, when the brine was being hauled away, the salt concentrations decreased with considerable fluctuation, and in 1953 all wells had been restored practically to their original condition. The maximum total dissolved solids concentration recorded for the school wells was about 5,300 ppm, compared to about 15,000 ppm in the seepage basins; the dilution obtained 2,000 feet of travel was therefore probably no more than three.

The "Montebello Incident" which occurred in 1945 is a spectacular example of rapid travel of a pollutant over a long distance. A quantity of chlorinated phenols was discharged within a period of one week to the Alhambra, California, sewerage system and reached the Tri-Cities Sewage Treatment Plant mixed in a sewage flow of 12 million gal/day. The plant effluent was discharged to the Rio Hondo where it combined with an equal flow of rising ground water. About a dozen wells belonging to the South Montebello Irrigation District and the Montebello Land and Water Company became polluted with the chlorphenols within two weeks after their arrival at the treatment plant. The wells were all located within a few hundred feet of the Rio Hondo, but were more than four miles downstream from the point of discharge. In this case the dilution which occurred during passage of the water through the surface aquifer could not have been very great, since the distance of underground flow was short and the entire source stream was for a time contaminated with the pollutant. But the concentration of the material in the surface water could have been only of the order of a few parts per billion, and even so the well supply became impotable; treatment with chlorine dioxide was necessary to break down the pollutant. That a substance in such small concentrations can have drastic effects demonstrates the necessity of exercising extreme caution in the operation of refuse dumps with respect to the nature of any liquid and leachable solid wastes which might be permitted.

Changes in Pollutants During Travel

Filtration: While coliform and other bacteria have been found in appreciable concentrations in the liquid leached from household refuse, there is a large body of evidence demonstrating that such organisms disappear completely in the ground within a short distance of their source. In pilot plant studies performed by the British government (33), the leachate from refuse was passed through sand and gravel filters of various lengths. Even coarse gravel removed over 90 percent of the bacteria in 24 feet, and the most effective filter used--24 feet of sand of about 0.5 mm effective size--reduced bacteria concentrations by a factor of 10^4.

A comprehensive study of pollution travel accompanying the injection of sewage effluents into a thin aquifer has been performed by the University of California (81). The reduction in coliform bacteria with travel distance was very rapid, and although the concentrations of bacteria in the injection water were over 10,000 per ml, organisms were encountered no farther than 100 feet from the source. The effective size of the aquifer material was about 0.4 mm. Orlob and Krone (70) used the same injection well in their studies of coliform movement through porous media and observed bacteria removals of the order of five to ten percent per foot of travel, even at rather high interstitial flow velocities (about 20 ft/day).

The movement of bacteria is limited not only by the filtering action of the aquifer, but also by the natural die-away which occurs among organisms out of their usual habitat. Orlob and Krone measured the die-off of coliforms in ground water, and obtained rates ranging from 20 to 70 percent per day. Since the velocity of ground water travel in most natural situations rarely exceeds a few feet per day, the combined effects of filtration and die-away serve to make the travel of bacterial pollution for more than a few hundred feet extremely unlikely.

Adsorption and Ion Exchange: Soil minerals, especially clays, have in varying degrees the property of taking cations from solution and adsorbing them on their particle surfaces. Montmorillonitic clays are most effective in this respect, while kaolinic clays are generally the least surface active. The maximum amount of cation which a given soil can adsorb is known as the exchange capacity of the soil; it is usually expressed as milliequivalents of cation per 100 gm of dry adsorbent. Exchange capacities not only depend on the crystal structure of the mineral and on its total surface area, but also vary somewhat among the different cations which may be adsorbed. Typical exchange values for clays would be about 50 meq/100 gm for an average material up to 100 meq/100 gm for Bentonite (a montmorillonitic clay). Sand and silt have very small exchange capacities and the adsorption properties of soils are due to the clay fraction.

Ions and other materials are adsorbed according to some isotherm; the amount of adsorbed substance increases with its concentration in the surrounding solution, up to the exchange capacity. Where two or more ions are present the proportions of each adsorbed depend on the relative concentrations in solution, but there is also an order of preference for adsorption. Divalent ions are more readily adsorbed than monovalent ones, for example, and if calcium and sodium are present at comparable solution concentrations the adsorbed material will be mostly calcium. Thus if a water high in hardness were passed through a soil previously saturated with sodium (from prehistoric contact with the ocean), the hardness would be exchanged for sodium. While this process might occur in waters which had become hard through carbon dioxide pollution near a refuse dump, the probability of obtaining effective softening is not great because of the limited and isolated distribution of clay in permeable aquifers.

Ion adsorption may have a profound effect on the permeability of soils, through the action of ions in the flocculation and swelling of the clay components. A case of deliberate reduction of the permeability of a clay pond lining has already been described in Section II. The same effect is important in the reclamation of saline soils by leaching (82); soils initially containing high concentrations of cations are flocculated and permeable. However, if the percentage of sodium among the adsorbed cations is high, leaching with low-salt water will desorb much of the sodium, and the soil will deflocculate; its permeability may decrease by possibly one-hundred fold or more, and any further leaching or draining will be greatly hampered. To avoid this occurrence the soil may be leached with salty water which is gradually diluted with fresh water having a low sodium percentage and a correspondingly high proportion of calcium and magnesium. Permeability effects might arise during the flow of polluted waters from refuse. Because of the relatively low salt concentrations which would normally be involved, a polluted water high in sodium might tend to deflocculate the aquifer and thus impede its own spreading. However, this would require the coincidence of a uniformly clayey aquifer surrounding the dump, and polluted water with a high sodium ratio.

Biological Oxidation: Included among the substances leached from a refuse fill will be organic matter subject to continued biological breakdown. If dissolved oxygen is present, the stabilization will proceed aerobically at a fairly rapid rate, and will be essentially complete in a matter of days or weeks. Since the dissolved oxygen concentration at saturation with air is only about nine parts per million, whereas the BOD of refuse percolate may be greater than 20,000 ppm, the mechanisms of gas transfer from atmosphere to solution will limit the stabilization rate unless rapid dilution with oxygenated water should somehow be provided. Experiments have shown that BOD is effectively removed by passage through sand in comparatively short distances, even though much of the organic matter is in true solution. In Reference 33 it is reported that the BOD of the liquid from wet pit refuse was reduced from 310 ppm to 75 ppm by 24 feet of sand. It may therefore happen that the greater part of leached organic matter is trapped within the dump site to be oxidized gradually as oxygen becomes available. As oxygen is consumed it will be replaced in solution by an approximately equal volume of carbon dioxide, which could thereupon dissolve hardness from the aquifer.

Ammonia and organic nitrogen in refuse percolate are quickly converted to nitrate when dissolved oxygen becomes available for the process. While some of the reduced nitrogen leaving the refuse may be taken up by organisms growing in the aquifer, it will be released sooner or later as the organisms die away. Since nitrate nitrogen concentrations greater than 10 ppm are generally considered undesirable in domestic water, the release of organic and ammonia nitrogen in concentrations of several hundred parts per million could constitute a serious problem.

Summary of Needed Information

The nature of the movement of pollution in ground water is rather completely understood, but the flow pattern in any region depends to a great extent on local geological conditions. While basic research in this field is not considered necessary to a satisfactory evaluation of the ground water aspects of refuse disposal, it would seem advisable, in considering the suitability of any proposed disposal site, to obtain information which would define the risk of downstream pollution. Included in such data should be the velocity and direction of ground water flow, the location of present and probable future downstream wells, the probable future ground water levels, and the stratigraphy of the immediate site neighborhood.

In permitting the operation of a refuse dump in an area where there is a possibility of ground water pollution, it would seem essential to require the regular monitoring of at least one well located at a short distance downstream from the site, in order that remedial measures may be taken in time should pollution occur. In addition, it would be prudent to establish beforehand what remedial steps are likely to be effective in case they should be necessary. A promising procedure is to confine the polluted water by pumping wells within and downstream from the dump, and in this connection the necessary spacing and pumping rate of such wells should be determined.

VII. RECOMMENDATIONS

Summary

The principal processes involved in the introduction of undesirable substances to the ground water through the agency of refuse dumps are considered to be infiltration and percolation, refuse decomposition, gas production and movement, leaching, and ground water travel.

The amount of water which enters a refuse fill from the surface will be governed by the rate of water application, the nature of the refuse cover, and climatic conditions. In Southern California refuse dumps having unirrigated soil covers do not seem to transmit surface-applied moisture in quantities sufficient to cause noticeable pollution. In areas of higher precipitation the percolation of rainwater through refuse has definitely been observed, and estimated water balances indicate that heavy irrigation and occasional years of high precipitation may produce substantial percolation through permeable dump covers even in Southern California. The appearance of percolation even under such adverse conditions may be delayed for years because of the high moisture-retaining capacity of combustible rubbish, and because the heat liberated in the decomposition of the refuse may accelerate evaporation. Research is needed to determine what necessary and sufficient measures will limit percolation to an acceptable quantity.

Refuse contains mineral and organic substances in quantities capable of seriously damaging underground water supplies. The organic matter of refuse will undergo aerobic and anaerobic biological decomposition, and produce large volumes of carbon dioxide and methane gas. Carbon dioxide can seriously degrade ground water by dissolving calcium, magnesium, iron, and other substances which are undesirable at high concentrations. At present it is not possible to estimate reliably the rate at which carbon dioxide will be produced in buried refuse, and basic research is indicated to obtain quantitative information.

Decomposition processes and rates are believed to be closely connected with the circulation of air and gases of decomposition through the refuse fill. Gas movement can occur by displacement when there is a net production or uptake in the refuse, and by convection due to differences in gas density. Density variations may arise when portions of the refuse atmosphere receive heat liberated in the decomposition process, and when the gas produced has an average molecular weight different from that of air. However, the most effective transfer mechanism is molecular diffusion; the diffusivity of a porous medium is relative independent of the particle size, so that gases may diffuse readily through some materials of low permeability. Therefore, in preventing carbon dioxide from moving into the underground, methods which involve its removal to the atmosphere by encouraging draft or ventilation may be more effective than coatings to decrease permeability of the disposal pit surfaces. The testing of methods to limit gas penetration are of prime importance; they should be supplemented with basic studies to determine quantitatively the nature and extent of underground gas movement.

Available information on leaching indicates that percolate from household refuse can contain high concentrations of organic matter and mineral salts. The substances most likely to prove objectionable if percolate enters a ground water supply with inadequate dilution are hardness, iron, nitrate, and total dissolved solids. Sulfides and organic matter may also cause problems if the leached substances are pumped out too soon after entering the ground water. While the general magnitudes of the quantities of leachable substances in refuse are known, it has not been established how their rate of extraction depends on the rate of percolation. This type of data is necessary to permit setting maximum limits on the amount of percolation which may be permitted to pass through a dump site.

The extent of possible movement and dissemination of polluting materials in the ground water itself has been demonstrated by various tracer experiments, and by many actual pollution incidents, some of which were directly due to refuse. The lateral dispersion and consequent dilution of carried materials is more effectively accomplished by nonuniformities in the water-bearing strata than to random velocity variations through the pores of the sand. Information concerning the velocity and direction of the ground water motion in any proposed dumping area will give some indication of the pollution pattern to be expected downstream from the dump being leached. However, in view of uncertainty as to the future development of downstream areas, it would seem preferable to avoid serious pollution at the dump itself, rather than depend upon the dilution during flow to diminish any quality impairment.

There are apparently only three basic mechanisms by which solid refuse can impart undesirable qualities to the ground water: (1) direct horizontal leaching of refuse by the ground water, (2) vertical leaching by percolating water, and (3) the transfer of gases produced during refuse decomposition by diffusion and convection. The results of direct leaching are now approximately known, and are such that direct contact of refuse with the ground water could be tolerated only in full view of serious downstream water quality deterioration. More research will be necessary to determine precisely what precautions must be taken to avoid ill effects from vertical leaching, but it is fairly certain that pollution by this means can be averted at any stage of percolation up to the time the pollutants have actually reached the ground water level. This is true because infiltrating moisture moves at an appreciable rate only when driven by overlying dampness, and if penetration from the surface is curtailed by means of a well-maintained impervious coating or layer, the moisture profile will become essentially frozen. (Perched water entering the refuse through the pit wall is a possibility that probably can be foreseen from consideration of the permeability of the strata intersecting the pit, and of the probable adjacent land use with respect to water application.) The remaining possible pollution mechanism--gas transfer--may be very costly to abate once refuse has been deposited, and it is therefore recommended that research priority be given to obtaining information on quantities of gas production, mechanics of gas flow around refuse fills, and on proposed methods for diverting refuse gases out of the ground.

Recommended Research Projects

The values at stake are enormous, both in terms of the storage capacity and concomitant security of the ground water reservoirs and in terms of the potential land reclaimed for re-use through landfilling and savings to the community in reduced costs for refuse disposal. Studies should be designed not just to minimize the hazard, but, if possible, to eliminate completely the risk of exposure of the ground water to damaging pollution.

The following specific projects are recommended, and are listed in order of priority.

1. Gas Production Quantities. The objective would be to measure the volume, composition, and production rate of gases from decomposing refuse. In order to obtain results which may be transferred to field conditions, it will be necessary to carefully control the decomposition process and to accurately measure the volumes, and for this reason, the work could be carried out most effectively in laboratory scale units. The procedure would be to decompose selected refuse in closed containers under various representative conditions of moisture, temperature, and air admission. The gases produced would be collected or somehow measured, and analyzed at regular intervals.

The data obtained would permit assessment of the gas quantities to be expected in a full-scale dump of any description and would provide a basis of design for prevention and control measures. The cost of a satisfactory laboratory study is estimated at $20,000 to $25,000, and one to two years would be required for completion.

2. Observation of Gas Movement. It is suggested that a revealing and yet simple way of following the spread of gas out of a fill would be to sample the soil atmosphere at selected points beginning when the dump is first put into operation. In this way the change in composition as the refuse gas replaced the soil air would indicate the characteristics and magnitude of gas motion. Samples under the dump and around the edges could be taken from well points which might be moved whenever they were engulfed by the advancing gas in order to be constantly measuring a changing atmosphere. Within the refuse samples could most easily be taken through tubing of plastic or other non-corrosible material placed in the refuse at the time of filling. The results would indicate to what extent and for what length of time measures to restrict gas production and movement should be applied.

Because of the difficulty of placing well points in cobble-filled ground surrounding gravel pits, the installation cost would be high; it is estimated that $25,000 to $30,000 would be required for satisfactory sampling coverage in the first year. In subsequent years it might prove desirable to reduce the number of sampling points and analyses.

3. Control of Gas Movement. Since the only known mechanisms of gas transfer are by mass flow and diffusion, measures taken to control outward gas movement from a dump must be based on adjustment of the permeability and diffusivity, or of the gradients of pressure and concentration. For minimizing the diffusivity and conductivity of dump sides and bottom, the use of some coating or layer is indicated, and the questions to be resolved are those of cost and durability. Research in this respect would involve placing the proposed layers in representative sections of operating dumps, with provision to detect the subsequent transmission of gases. Alternately, in order to isolate the dump area being treated experimentally, the tests might be carried out in small pits specially excavated.

Possible techniques for relieving or reversing gas pressure and concentration gradients include the encouragement of ventilation through the use of gravel chimneys, relatively porous soil covers, and forced draft by means of blowers. There is a good possibility that the rate of refuse stabilization in a fill could be greatly accelerated by supplying oxygen at a rate sufficient to maintain an aerobic state. The use of blowers and piping to introduce air at the bottom of the refuse might be tried in conjunction with ventilating arrangements. Whatever the procedures tested, they should be provided with appropriate control determinations so that any benefit can be quantitatively estimated, and data should be taken in such a way that a balance can be made on the gas produced.

Costs would depend heavily on the scale or the experiment, but probably about $50,000 would be necessary for the installation, and $15,000 to $20,000 per year after the initial work for continued sampling and analyses.

4. Leaching Studies. Present available information on leaching does not demonstrate the effect of flow rate on the rate of extraction. This data is necessary to establish permissible percolation rates, and it is recommended that pilot plant-type experiments be performed for such purpose. The data should be taken for a period of years, if necessary, in order that it be applicable to dumps which are gradually filled, and therefore contain refuse which has been leached to various degrees. Of particular interest would be the results at the very low flow rates at which vertical leaching in Southern California may be expected to occur.

The cost would be of the order of $10,000 for the installation and $25,000 to $35,000 per year for operation, sampling, and analyses.

5. Monitoring of Field Moisture. The imminence of vertical leaching can be detected by monitoring the moisture content of the buried refuse. Downward moisture movement could be investigated conveniently by installing tensiometers or other moisture detecting devices in a fill as it is being placed. The coverage of the fill area with detecting devices should be ample to allow for inevitable non-uniformities of moisture movement. The likelihood of percolation when the fill

surface is irrigated could also be investigated; in this case the tensiometers should be installed in an isolated small pit to which any surface water applied will be confined.

The cost of installation would be about $10,000 to $15,000. About $10,000 per year would be needed for operation when active experiments were underway, and about $5,000 per year for monitoring only.

6. Prevention of Percolation. While studies of infiltration capacity and precipiation intensity would be useful in estimating the amount of percolation to be expected in any refuse fill, a more immediate need is developing techniques which will maintain percolation at a minimum regardless of future weather conditions and land use. The most promising method appears to be sealing of the pit bottom with clay or oil, with provision for drainage and removal of percolate by pumping. The work could probably be performed in cooperation with a public agency, which could provide space in an operating fill and possibly could construct the experimental installation. The operating cost would thus be about $10,000 to $15,000 for one year.

APPENDIX A

HEAT BALANCE IN REFUSE COMPOSTING

Analyses:	% Dry Weight Beginning	 End
Ash	24	38
Carbon	35	31
Water	98	65
Temperature	14°C	68°C
Average Temperature	58°C	

Assumed Reaction:

$$C_6H_{10}O_5 + 6O_2 \longrightarrow 6CO_2 + 5\,H_2O$$

Necessary Data:

Heat of Combustion of Starch	4,100 calories/gm
Heat of Vaporization of Water	540 calories/gm
Specific Heats	
Nitrogen	0.25
Carbon dioxide	0.20
Refuse solids	0.33
Water	1.0
Mole Weights	
C	12
N_2	28
CO_2	44
H_2O	18
$C_6H_{10}O_5$	162

Basis of Balance: One gram of dry material.

Percent Carbon Destroyed:

$$1 - \frac{(31)\,(24)}{(35)\,(38)} = \quad 43\ \%$$

Weight Carbon Destroyed: $(0.43)\,(0.35) =$ 0.15 gm

Net Water Loss:

$$1 - \frac{(65)\ (24)}{(98)\ (38)} = \qquad 58\ \%$$

Water Produced: $(0.15)\ (5/6)\ (18/12) =$ 0.19 gm

Total Water Exhausted: $(0.98)\ (0.58) + 0.19 =$ 0.76 gm

Average Water Content: $(0.98)\ (1 - 0.58/2) =$ 0.70 gm

Solids Destroyed: $0.76 \quad 1 - \frac{(62)\ (24)}{(76)\ (38)} =$ 0.37 gm

Average Solids Content: $1.0 - 0.37/2 =$ 0.82 gm

Oxygen Consumed: $(0.15/12)\ (6/6) =$ 0.0125 moles

Carbon Dioxide Produced: $(0.0125)\ (44) =$ 0.55 gm

Nitrogen Exhausted: $(0.0125)\ (79/21)\ (28)$ 1.32 gm

Heat Liberated

Heat of Reaction: $(0.15)\ (162/62)\ (4100) =$ 1390 cal

Heat Losses

Warming Refuse:

Solids $(0.82)\ (0.33)\ (68-14) =$ 15 cal

Water $(0.70)\ (1.0)\ (68-14) =$ 38 cal

Exhaust Heat:

CO_2 $(0.55)\ (0.20)\ (58-14) =$ 5 cal

N_2 $(1.32)\ (0.25)\ (58-14) =$ 15 cal

Total Heat Loss: 483 cal

Unaccounted Heat Loss

$1390 - 483 =$ 907 cal

$=$ 65 %

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COLORADO RIVER BASIN REGIONAL WATER POLLUTION CONTROL BOARD (NO. 7)
82-380 Miles Avenue, Indio (P. O. Box 277)

SANTA ANA REGIONAL WATER POLLUTION CONTROL BOARD (NO. 8)
3691 Main Street, Riverside

SAN DIEGO REGIONAL WATER POLLUTION CONTROL BOARD (NO. 9)
3441 University Avenue, San Diego 4