RECIRCULATION AS A LEACHATE MANAGEMENT OPTION

by
Aslı Boduroglu
B.S. in C.E., Bogaziçi University, 1991

Submitted to the Institute for Graduate Studies in
Science and Engineering in partial fulfillment of
the requirements for the degree of
Master of Science
in
Civil Engineering

Bogaziçi University
1993

To my Mom, Dad and my sister, Ayşecan

TABLE OF CONTENTS

ACKNOWLEDGEMENTS

I owe a debt of profound appreciation to Prof.Dr. Kriton Curi for his guidance, motivation and patience as my thesis supervisor during the course of my research.

I thank the Ministerium of Baden Würtenberg for supporting me with a Scholarship during my stay in Stuttgart.

I wish to acknowledge gratefully to Dr. E. Thomanetz for generously providing advice and assistance at Universitat Stuttgart, Institut fur Siedlungswasserbau, Wassergüte und Abfallwirtschaft.

Special thanks go to Brigitte Haas, Young Kim, Werner Röder, Ellen Köhler and Darek Jung for offering their help and effective cooperation.

I also want to thank Dipl.-Ing. Wolfgang Dunz for his interest and his willingness to share technical knowledge.

Last but not least, I want to thank my Mom and Dad for having faith in me, for encouraging me and loving me no matter what(or when).

ABSTRACT

The results of a laboratory investigation on the behavior of municipal solid waste under the influence of leachate containment and recycle were used to illustrate the effects of leachate recycle on the rate of biological stabilization of refuse. Laboratory lysimeters were employed to simulate the behavior of the vertical center of a newly covered landfill. The mode of leachate recycle was practiced with two levels of simulated water addition. The availability of minimum amounts of moisture led to an acceleration of the conversions and interactions leading to biological stabilization.

ÖZET

Bu çalişmada, sizinti suyu tutulması ve resırkulasyon etkileri altındaki evsel kati atiklar üzerinde yapilan laborutuvar sonuçlari kullanilarak sizintı suyu resirkulasyonun atiğin biyolojik stablizasyon hizına etkileri açiklanmistır. Üzeri yeni örtülmüş dolgu çöplerinin düşey merkezinin davranışı simulasyonu icin laborutuvar lisimetreleri kullnilmiştır. Sizıntı suyu resirkulasyonu modu iki seviyeli su eklemesi simulasyonu ile elde edilmiştir. En düşük miktarda rutubetin bulunması dönüşüm ve etkileşimlerin ivmelenmesi ile biyolojik stabilizasyona yol açmaktadır,

LIST OF FIGURES

LIST OF TABLES

LIST OF SYMBOLS

A	Pre-exponential factor
a	Constant dependent upon the return period
b,c	Water production of organic matter
C_i	Correction coefficient
E	Evaporation in 10 days
E_a	The Arrhenius Activation Energy
f_c	Infiltration capacity at a constant rate
f_o	Maximum value of initial infiltration
f_p	The infiltration capacity
ET	Actual evapotranspiration
h	Thickness of soil layer
[H]	Hydrogen ion concentration
H	Side slope runoff
I_g ,I_s	Water from aquifers
J	Irrigation or leachate recirculation
K	Rate constant
K	Consolidation water arising during disposal
K	Rate constant for specific acid hydrolysis
K_O	Rate constant for neurtral hydroysis
K_{OH}	Rate constant for specific base hydrolysis
L	Total leachate production
L_I	Infiltration into an aquifer
[OH]	Hydroxyl ion concentration
P	Precipitation
Q	Direct runoff
Q_r	Quantity of waste for disposal
R	Surface runoff
R*	Runoff from external areas
RX	Organic molecule
S	Water added by sludge disposal
S	Sum of the moisture components of a landfill
t_d	Duration of leachate emissions

T	Mean air temperature
t	Time from the beginning of rainfall
U_s	Water content in soil
U_w	Water content in wastes
V_r	Volume available for waste disposal
W	Maximum evaporation in 10 days with no rainfall
α	Dehydration coefficient of agricultural soil
γ	Specific weight of water
γ_i	Specific weight of the dry soil
λ	Ratio of the difference in influx water to maximum volume of water
θ_{ms}	Water content in a dry basis

1. INTRODUCTION

The effect of uncontrolled infiltration of leachate into the environment is believed to be the biggest environmental impact a sanitary landfill can have. The leaching of miscellaneous pollutants through the solid waste tips has been pointed out over the last 20 years as a hazard associated with ground and surface water contamination [5,6,7,8,9,10,11]. These concerns have led to a variety of developments for control including the concepts of leachate containment and total landfill isolation. Methods are being defined to restrict leachate contamination. More frequently, landfills are sited in areas that are naturally impermeable by nature or areas which have been rendered impermeable through artificial means such that they are sealed at the top and the bottom of groundwater table. There has been a large movement toward relatively sophisticated, regionally based landfills that must comply with increasingly strict, local, state and federal regulations. In accordance with these approaches, various techniques have been proposed and implemented for the treatment and disposal of landfill leachates. Hence, landfill leachate management in an environmentally safe manner has become an essential aspect in the design of landfills.

The concept of leachate recycle in terms of an in situ treatment has been recognized as a management option both with respect to leachate quality-quantity and energy recovery. The practice of leachate containment-recycle mode has been determined to improve the homogeneity of the biochemical medium needed for anaerobic waste degradation and to reduce the time required for biological stabilization.

The aim of this study is to provide an investigation on landfill leachate and to determine the feasibility of leachate recirculation in providing leachate control and treatment.

The concepts of leachate generation, moisture distribution and water balance in sanitary landfills are introduced in Section 2. Landfill stabilization is analyzed in

Section3. The presentation of leachate analysis is considered in Section 4. Leachate treatment processes are included in Sections 5 and 6. In Section 7, current technology assessment is discussed. The investigations leading to the experimental work is introduced in Section 8.

The land is and will most likely be a 'final sink' for many waste residuals. For this reason, it is essential to continue encouraging the development of landfill management strategies which endorse an incorporation of present and emerging landfill technology in both a retrospective and prospective manner.

2. LEACHATE GENERATION

Once solid waste is placed in a landfill, a complex sequence of biological, physical and chemical events occur which result in gaseous and liquid emissions. Leachate is produced as water percolates through the solid waste, leaching soluble and insoluble components and degradation products from the refuse. This solution which contains water soluble salts, heavy metals and redox sensitive substances plus the products of decomposition of organic matter, may become a potential source of contamination if it joins a surface water or groundwater source before it is ameliorated of its impurities.

It has been noted that on the general, the more water that flows through the solid waste, the more pollutants are leached. Therefore, reducing the percolation to the refuse to a minimum account is regarded as a crucial point in the control of leachate production.

2.1 Factors Affecting Leachate Production

In the course of environmental protection, it is important to identify and quantify the processes generating leachates and to estimate how the nature of leachates change with time because leachate generation in sanitary landfill sites may continue for many years after a landfill has ceased operations.

2.1.1 The Amount of Leachate Production

The volume of leachate produced is known to be directly proportional to the amount of moisture present in and added to the landfill. This moisture is believed to result mainly from infiltration, small amounts added with the refuse at placement and some produced from biological decomposition [26]. However, in developing countries during the summer months(especially during the water melon season), the amount of moisture reaches as high as 60 to 80 percent and leaching starts immediately after the placement of the wastes in the dumping site [80].

It has been reported that in certain areas of Europe, infiltration rates could reach 50 percent of precipitation rates although typical annual average values have been between 10 and 25 percents. Using an annual average value of 0.9 mm/d , the leachate produced from a landfill having an area of one hectare would be nine m^3/day or 3285 m^3/ year. This value would be the leachate discharge rate to a groundwater system. However, it is important to keep in mind that the field capacity of the refuse is needed to be achieved before leachate discharge to groundwater begins and that infiltration rates can vary dramatically depending on the season, the site location and the condition of the landfill cover.

The strength of leachates produced in landfills is also found to be highly variable. However, based on generalizations, it is determined that peak strength of leachates occurs usually less than three years after placement and dissipates with time [26].

Studies conducted in West Germany on numerous controlled sites in different phases of operation (see Table 2.1.1.1) exhibit the significance of the effect of compaction on the amount of leachate production.

Table 2.1.1.1 Rainfall and Leachate in some Sanitary Landfills in the Federal Republic of Germany [1]

Landfill Compaction with steel compactors	Precipitation (mm/year)	Evaporation (%)	Leachate (m3/ha-day)
1	652	15.1	2.7
2	651-998	12.2-29.8	3.2-8.1
3	651-998	16.9-21.6	3.0-5.9
4	632	16.3-18.3	2.8-3.2
5	509	16.8	2.3
6	556-1057	15.6-19.6	2.6-5.1
7	770	3.3-7.2	0.7-1.1
8		22	3.8
9		38	6.7
Compaction with crawler tractor			
10	571	31.3	4.9
10	571	4.4	0.4
11	501-729	25-48.2	5.3-8.3
12	662	58.2	10.6
13	632	32.3	5.9
14	565-665	39.2-42.0	6.1-7.5
15	636	19.9-21.4	3.5-3.7

A different approach has been realized through the studies conducted in pilot scale cells in England. The results there indicated that the production of leachate is more dependent upon site slope and thickness of the cover material then on the density of compacted waste.

Table 2.1.1.2 Production of Leachate (expressed in percentage of rainfall) Measured in Experimental Cells (dimension of each cell:50x25x3.5m) [1]

Cell number:	1	2	3	4	5	6-b
Initial waste density (kg/m³)	670	950	650	na-a	1010	790
Thicknessof the cover material (cm)			30	30	50	50
Slope of the cover material (%)	0	0	0	0	2-c	0-d
Production of leachate	57,9	57,3	60,4	39,6	24,3	42,0-e
(yearly average as % of rainfall)						

a-not estimated with adequate precision
b-cell maintained with a water content to its field capacity
c-cover shaped to shed rainwater towards surrounding land.
d-cover shaped to hold rainwater within perimeter of cell.
e-during first year 75 percent was registered. The subsequent reduction is explained by the natural consolidation of the clay cover. Immediately after application this cover developed fissures during dry periods.

2.1.2 Seasonal Fluctuations

High leachate production rates are usually observed during winter and spring seasons, when an increase of more than 100 percent is assumed and to a great extent is caused by precipitation and the melting of snow.On the other hand, during the summer and autumn months significant reductions in the amount of leachate production are noted especially in places where evaporation is effective.

In most developing countries, as it is in Turkey, the leachate production pattern is quite different. Here, due to the production of fruits and vegetables during the summer months, leachate production occurs as soon as wastes are disposed even though precipitation does not take place. On the other hand, during the winter months a considerable reduction in leachate production is observed due to adsorption by ash wastes which are present in wastes in this season.

It is equally important to take into account the effects that seasonal fluctuations impart on the moisture regime in a landfill body where the principal source of moisture comes from precipitation. A part of this precipitation results in surface runoff, a part is returned back to the atmosphere in the form of evaporatranspiration and the remainder adds up to the soil moisture storage[26]. Diversion of all moisture except direct preciptation from the landfill body is considered to result in reduced leachate production[33]. A lack of moisture puts a limitation on the biological transformation by extending the period required for waste stabilization and allows higher organic contents to remain in the leachate for long periods of time. Therefore, in order to have a better insight of leachate production, the moisture components at a sanitary landfill should be examined.

2.2 Moisture Distribution in Sanitary Landfills

The water balance method, which enables the assessment of the pollution potential of a landfill has been developed through the interpretation of climatic parameters and assumptions based on the conditions prevailing in a landfill body(for example,the moisture capacity of the refuse and the capping layer, the percentage of precipitation that will infiltrate the cap).

In the design of landfills the attention is usually focused on excluding the moisture by means of a cap system or by retaining/recirculating leachate within landfills by means of the underliner. Because the moisture storage capacity of a landfill is quite difficult to quantify, knowledge on the moisture absorbtion and retention capacity of the stored refuse is available only to a small extent. However, it is known that the water retained within the refuse is a representation of a large volume of moisture in the landfill.

For studying the complete interchange of moisture between a landfill and its surroundings, the water balance is analyzed [38]. Accordingly, the water balance is expressed as:

$$\text{Water input} + \text{Water production} = \text{Water output} + \text{Water stored} \tag{2.2.1}$$

It can also be expressed as:

$$\text{Infiltration - Evaporation} = \text{Moisture storage} + \text{Leachate} \tag{2.2.2}$$

Here, infiltration is taken as the difference between precipitation on the landfill and runoff. Leachate is the water present at the base of the landfill or collected on the underliner. Moisture storage is referred as the water absorbed/held in the refuse and intermediate layers as well as water percolating downward or under upward evapotranspiration flow gradients. The moisture storage capacity or field capacity then could be defined as the water content that a porous material (refuse or soil cover for example) will absorb and store within its pores by capillarity. The measured storage capacities are found to range from 225 percent for fresh refuse (mainly paper and cardboard), to around 55 percent for one and five year old wastes after compression to high densities.

It should be noted that Equation 2.2.1, is based on the assumption that wastes are not in an over saturated condition of at the time of disposal. However, when the over saturated condition of the wastes arising from developing countries are concerned, Equation 2.2.2 need to be rewritten as:

Infiltration- Evaporation = +-Moisture storage + Leachate (2.2.3)

In this equation, the moisture storage is taken as a negative value when the wastes are over saturated.

In relation to the water balance, the water stored is observed to build up until the overall (field) moisture capacity is reached. Thereafter, any amount of water added at the top of the landfill would result in the production of an equal quantizy of leachate at the base.

2.3 The Water Balance for a Landfill

The investigations on the water balance could be carried out further to predict leachate production in completed landfills. A schematic representation is shown in Figure 2.3.1.

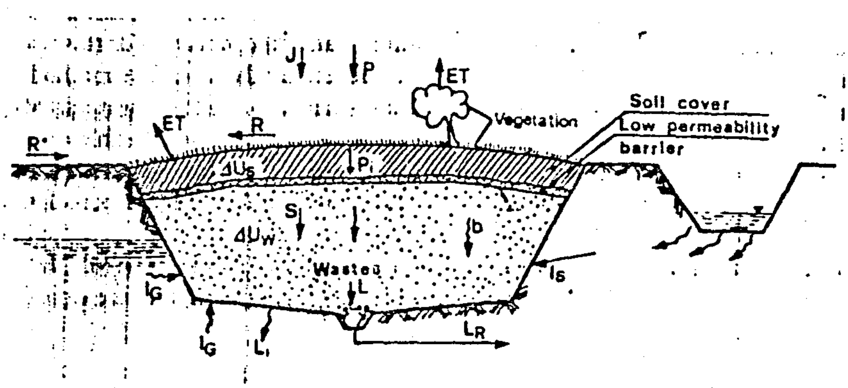


Figure 2.3.1 The Water Balance in a Completed Landfill [1]

where P = precipitation
J = irrigation or leachate recirculation
R = surface runoff
R* = runoff from external areas
ET = actual evapotranspiration
Us = water content in soil
Uw = water content in wastes (water generated or retained by wastes)
S = water added by sludge disposal
b = water production (if > 0) or (if < 0) caused by the biological degradation of organic matter
Is = water from a natural aquifer
Ig = water from a natural aquifer
L = total leachate production
L_I = infiltration into aquifers
L_R = leachate collected by drains

By synthesizing the above cited terms, the water balance model could be expressed as:

$$P - E \mp \Delta U_w - R = L_R \qquad (2.3.1)$$

2.3.1 Rainfall

Precipitation in the form of rainfall represents the single largest single contribution to the production of leachate. Rainfall provides the transport phase for leaching and migration of contaminants from a landfill body and the moisture needed for biological activity. Although some moisture is obtained from the input wastes, the major portion to leachate formation comes from the infiltration of rainfall. It has been determined that the peaks of light rainfall over a long lapse of time create the most critical situation of infiltration. On the other hand, there is believed to be little net infiltration when short bursts of heavy rainfall occur due to the quick saturation of the cover material. In the developing countries(esp. in Turkey), high amounts of leachate production are observed in the absence of precipitation owing to the over saturated conditions exhibited in the waste compositions.

In the calculation of the water balance, data regarding all forms of precipitation is considered. For example, the effect of thawing snow has been considered through the assumption of one cm fresh snow equalling one mm effective rainfall[1]. Information regarding the rainfall frequency data should be accurately gathered. The depth-duration frequency analyses of historic rainfall should be utilized to understand the isopluvial maps. Experience has shown that Equations such as (2.3.1.1) and (2.3.1.2) fit the intensity duration curves[4].

$$i = \frac{a}{(t_d + c)^b} \qquad (2.3.1.1)$$

$$i = \frac{a}{(t_d^b + c)} \qquad (2.3.1.2)$$

where i = rainfall intensity
t_d = duration
a = constant dependent upon return period
b,c = constants independent of return period

2.3.2 Surface Runoff

The part of the incident precipitation which runs off and is lost to the 'overland flow' before it has the possibility to infiltrate has been referred as surface runoff. The amount of runoff has been determined to be dependent upon factors such as the intensity and duration of the storm, the soil moisture content, the topography and the morphology of the soil. In the estimation of this runoff fraction, one of the two methods described below is undertaken.

The Rational Method:

The Rational Method which is applied commonly for this calculation is based on the general formula indicated below[4]:

$$R = c \times P \tag{2.3.2.1}$$

where R = runoff (mm/day or mm/month)
P = rainfall (mm/day or mm/month)
c = empirical coefficient

The empirical coefficient [c], is found to vary with the nature of the soil, the slope of the site and the presence-type of the vegetation cover[1]. However, the proposed values in Table 2.3.2.1 and Table 2.3.2.2 could be adopted in the calculation of the hydrological balance.

Table 2.3.2.1 Runoff Coefficients for the Rational Method[1]

Soil Cover	Slope	Soil Texture		
	(%)	Sandy loam	Loamy clay	Clay
Grassed soil	0\5	0,1	0,3	0,4
	5\10	0,16	0,36	0,55
	10\30	0,22	0,42	0,6
Bare soil	0\5	0,3	0,5	0,6
	5\10	0,4	0,6	0,7
	10\30	0,52	0,72	0,82

Table 2.3.2.2 Rational Equation Runoff Factors[4]

Description of area of surface	C factors
Lawns, sandy soil	
Flat, 2%	.05-0.10
Average, 2-7%	0.10-0.15
Steep, 7%	0.15-0.20
Lawns, heavy soil	
Flat, 2%	0.13-0.17
Average, 2-7%	0.18-0.22
Steep, 7%	0.25-0.35

The SCS Curve Number Method:

Infiltration is described as the flow of water through the soil surface. The maximum rate that water can enter the soil is defined as the infiltration capacity, f_p. As the soil becomes saturated, the infiltration capacity decreases to a constant rate, f_c[4].

$$f_p = f_c + (f_o - f_c)e^{-kt} \qquad (2.3.2.2)$$

where f_p = the infiltration capacity

f_c = infiltration capacity at a constant rate

f_o = maximum value at the beginning of the rainfall

t = time from the beginning of rainfall

K = rate constant dependent on the soil type

The values for K, f_c, and f_o are found to depend on the permeability of the top soil strata, type of vegetation growing on this strata, the slope of the site, and the moisture content of the soil at the moment the estimation is made.

In order to establish the standard SCS rainfall function, the relating equations need to be evaluated[4]. The rainfall runoff function which is expressed by:

$$Q = \frac{(P - Ia)^2}{(P - Ia) + S} \qquad (2.3.2.3)$$

where Q = direct runoff
P = total precipitation
S = maximum potential retention
Ia = initial abstraction
is empirically developed in order to estimate both Ia and S

$$Ia = 0.2S \qquad (2.3.2.4)$$

when Equations (2.3.2.3) and (2.3.2.4) are combined, the standard SCS Rainfall Function is obtained.

$$Q = \frac{(P - 0.2S)^2}{P + 0.8S} \qquad (2.3.2.5)$$

2.3.3 Evaporation and Evapotranspiration

From earlier studies, it is clear that the vegetation growing on the final cover causes a water loss to the atmosphere in the form of evapotranspiration. The amount of this water loss when compared with the loss from evaporation through the soil alone without any vegetation, has been determined to be greater. Hence, distinctions should be taken into account when landfills still in operation with intermediate cover are compared to the finished landfills with final covers.

In the calculation of evaporation from land without any vegetative cover, the degree of saturation in the soil layers place a critical role. Therefore, the evaporation rates for saturated siols are interpreted by taking the evaporation rates for shallow water basins and multiplying these values by the coefficients indicated below.

Table 2.3.3.1 The Coefficients Implemented for Saturated Soils[1]

Fine sand	1
Loams	0.9
Clays	0.75-0.95

Under the circumstances of having unsaturated ground without any vegetative cover, the Turc's formula is applied[1].

$$E = \frac{P+w}{\sqrt{1+\frac{[P+w]}{L}}} \qquad (2.3.3.1)$$

where E = evaporation in 10 days(mm)

P = rainfall in 10 days(mm)

W = 10 days of maximum evaporation from soil with no rainfall (from 1 to 10 mm)

L = heliothermic factor, given by:

$$L = \frac{(T+2)I_g}{16} \qquad (2.3.3.2)$$

where T = mean air temperature (^0C)

Ig = real solar radiation $(cal/cm^2.d)$

Although there are numerous empirical and semi-empirical formula to calculate the potential and effective evapotranspiration, The Thornwaite-Mather Method has been developed for determining the rate of effective evapotranspiration in European climates. The method is found to be applicable to landfills both under operation and completion (landfills which have been covered with soil and restored as agricultural land)[1].

$$PE_i = 16\left(\frac{10T_i}{I_t}\right)^a C_i \qquad (2.3.3.3)$$

where PE_i = potential evapotranspiration in the i-th month (mm/month)

T_i = monthly average temperature(^0C)

I_t = annual thermal indice

a = constant

C_i = correction coefficient

Furthermore, by subtracting the potential evapotranspiration (PE_i) from the rainfall (P_i) minus runoff (R_i) the monthly deficit/excess is obtained. A deficit occurs if P_i-R_i-PE_i <0 and is referred as the dry periods. In the same manner, the wet period exists when P_i-R_i-$PE_i>0$.

It has been observed that during the periods of water deficit (dry periods), the evapotranspiration is less than the potential evapotranspiration except where the ground is kept wet by an aquifer or where it is regularly irrigated. Under such conditions, the evapotranspiration is cited as effective evapotranspiration.

In the calculation of effective evapotranspiration, the progressive dehydration of agricultural soil is considered in the formula developed by Benfratello[1]:

$$\frac{d\alpha}{d\lambda} = \alpha^m \tag{2.3.3.4}$$

where m = resistance to desaturation coefficient

α = A / U= ratio between effective influx of water A into the soil to the maximum volume of water in mm that layer of soil exploited by roots could after water infiltrated under gravity

λ = (P-R-PE) / U = the ratio of the deficit to U

Moreover,

$$U = \theta_{ms} \frac{\gamma_i}{\gamma} h \tag{2.3.3.5}$$

where θ_{ms} = water content on a dry weight basis
h = thickness of the layer of soil in question
γ_i = specific weight of the dry soil
γ= specific weight of water

By integrating Equation 2.3.3.4 , α is equal to:

$$\alpha = \alpha_0 e^{\lambda} \quad \text{for } m = 1 \tag{2.3.3.6}$$

Hence, through the calculation of α and A, the difference; ßA between two conservative evapotranspiration, ET is then obtained as:

$$ET_{dry} = P - R + \beta A \tag{2.3.3.7}$$

2.3.4 Soil Moisture Storage

The maximum moisture a soil can hold against the gravitational force is defined as the field capacity[2]. Regardless of the waste type, the percolating water from the surface of the landfill is absorbed by the waste until the field capacity is reached. When the difference between the field capacity and the inital water content of the waste is taken, the result is simply the absorptive capacity of the waste. Therefore, an approximate evaluation of the time required for leachate before it reaches the drainage system can be made by considering the above mentioned values. Studies conducted in England on the field and absorptive capacities of solid waste are depicted in Table2.3.4.1.

Table 2.3.4.1 Water Content, Field Capacity and Absorptive Capacity of Solid Waste[1]

Type of Waste	Density *(density on a wet basis)	Inital Water Content	Field Capacity	Absoptive Capa
	(kg/m³)	(% weight)	(% weight)	(% weight)
Raw	282	14	101	87
Pulverized	370	39	72	33
Raw	523	37	75	38
Pulverized	504	26	42-75	16-49
Raw	624	28	59	31

*On a wet basis; weight water(kg)/ weight of waste(kg) as %

These values discussed in Table 2.3.4.1 relate to theoretical values, values reached by waste over a long period of time. However, in reality leachate flows through quicker than the established theoretical limits. For this reason, a 1:4 relationship between the quantity of water absorbed at the first appearance and that absorbed in a longer period of interval is acceptable[1].

Furthermore, it has been confirmed by a study conducted in England, that the first appearance of leachate is when the moisture content reaches values around 51 to 58 percents by weight on a dry basis[1].

Through the implementation of the time leachate appears, the quantity of waste that may be tipped daily to complete the landfill site could be determined as:

$$P = \frac{Q_R}{t_{pa}} = \frac{V_R \times r_{bt}}{t_{pa}} \qquad (2.3.4.1)$$

where P = daily capacity (t/days)

Q_r = quantity of waste for disposal (t)

t_{pa} = time before leachate first appears (days)

V_r = volume available for waste disposal (m^3)

R_{bt} = density of waste (t/m^3)

In the course of avoiding the production of leachate in the short term interval, a value greater than P would be an indication of completing the landfill before the field capacity is reached.

3. LANDFILL STABILIZATION

Most landfills receiving municipal solid waste are known to proceed through a series of predictable events whose importance and longetivity are determined by climatological conditions, operational variables, management options and control factors. When these events are followed by certain analyses (leachate and gas analyses), time dependent notions describing the 'phases of stabilitzation' in a landfill body emerge.

In order to establish the link between landfill age and leachate production, the phases associated with landfill stabilization need to be evaluated. The descriptive events of these phases are encountered at one time or another in landfills. In predicting the longetivity of these phases, factors such as the accumulation rate of moisture, the nutrient sufficiency and site specific conditions (refuse characteristics) are found to play an important role. However, it has been determined that in reality no landfill has a single 'age' but rather a several of ages associated with the different sections of the landfill body.

Studies conducted by Pohland et al. , 1986 have led to the description of these phases. Accordingly, landfills exhibit an initial lag or adjustment phase which lasts until sufficient moisture has accumulated; when 'field capacity' has been reached. Thereafter, waste conversion and stabilization is reflected by changes in leachate and gas quality as the stabilization proceeds.

3.1 The Phases of Landfill Stabilization

The following have been identified in terms of the major events occuring during each phase:

Phase 1: <u>Initial Adjustment</u>

* Intitial waste placement and preliminary moisture accumulation
* Initial settlement and closure of landfill area
* The changes in environmental parameters are first detected.

Phase 2: <u>Transition</u>

* Field capacity is reached
* Leachate is formed
* A transition from initial aerobic to anaerobic microbiological stabilization occurs
* Intermediates such as the volatile organic fatty acids first appear in the leachate.

Phase 3: <u>Acid Formation</u>

* With the continuing hydrolysis and methanogenesis of waste and leachate constituents, the volatile organic fatty acids become predominant.
* A drop in the pH level into the acidic range is observed
* Nutrients such as nitrogen and phosphorus are released and utilized in support of the growth of the biomass
* Hydrogen may be detected and affect the nature/type of the intermediary products being formed.

Phase 4: <u>Methane Fermentation</u>

* The intermediate formed products during the acid formation phase are converted to methane and excess carbon dioxide
* A rise in the pH level is observed
* The oxidation- reduction potentials are their lowest values
* Leachate strength is dramatically decreased in correlation to increases in gas production.

Phase 5: <u>Final Maturation</u>

* Following actice biological stabilization in the waste and leachate, a relatively 'dormancy' is present
* Nutrients may become limiting
* Gas production ceases
* The less available substrates are degraded

Furthermore, it has been detemined that there are certain indicator parameters, that could be being used to detect and describe the presence, the intensity and longetivity of each phase of stabilization. When leachate production commences, many of these could be applied to the analysis of leachate. For example, pH and Oxidation-Reduction Potential are physical-chemical parameters that are critical to the proper evaluation of the acid formation and methane fermentation phases; COD and BOD_5 are chemical and biological parameters, respectively and are both indicative of relative biodegradability; nitrogen and phosphorus are chemical parameters important to the determination of nutrient sufficiency and aerobic/anearobic conditions. Similarly, other parameters may reflect such factors as buffer capacity(alkalinity), potential inhibition(heavy metals), oxidizing potential(nitrates and sulfates), and ionic strength(conductivity).

Municipal landfill leachate and gas constituent concentration ranges encountered in literature and their relative significance to the degree of landfill stabilization[40]:

<u>Biochemical Oxygen Demand (BOD)</u>
Transition Phase: 100-10900 mg/L
Influence of dilution and aerobic solubilization of waste organics
Acid Formation Phase: 1000-57700 mg/L
Accumulation of biodegradable organic acids due to methanogenic lag
Methane fermentation Phase: 600-3400 mg/L
Conversion of biodegradable organics to gaseous end products (CH_4, CO_2)
Final Maturation Phase: 4-120 mg/L
Influence of high molecular weight organic residuals (humics, fulvics)

<u>Chemical Oxygen Demand (COD)</u>
Transition Phase: 480-18000 mg/L
Trending in a similar fashion to BOD
Acid Formation Phase: 1500-71100 mg/L
Trending in a similar fashion to BOD
Methane Fermentation Phase: 580-9760 mg/L
Trending in a similar fashion to BOD
Final Maturation Phase: 31-900 mg/L
Higher influence of residual organics than in BOD assay

<u>Total Organic Carbon (TOC)</u>
Transition Phase: 100-3000 mg/L
Beginning to appear as a result of aerobic solubilization
Acid Formation Phase: 500-27700 mg/L
Increasing rapidly: accumulation due to methanogenic lag
Methane Fermentation: 300-2230 mg/L
Conversion of volatile acids to methane decrese in aqueous carbon
Final Maturation Phase: 70-260 mg/L
Influence of higher molecular weight organics (humics, fulvics)
<u>Total Volatile Acids (TVA)</u>
Transition Phase: 100-3000 mg/Las Acetic Acid
Just beginning to appear as a result of all aerobic solubilization
Acid Formation Phase: 3000-18800 mg/L as Acetic Acid
Solubilization of organic polymers to monomers
Methane Fermentation Phase: 250-4000 mg/Las Acetic Acid
Conversion of fatty acids to fatty acid; fermentation of acetic acid to methane
Final Maturation Phase:
Essentially absent, methanogenic system undersaturated

<u>BOD/COD Ratio</u>
Transition Phase: 0.23-0.87
Increasing biodegradibility of organics due to solubilization
Acid Formation Phase: 0.4-0.8
High biodegradibility
Methane Fermentation Phase: 0.17-0.64

Decreasing biodegradibility due to methanation
Final Maturation Phase: 0.02-0.13
Low degree of biodegradibility

<u>Total Kjeldahl Nitrogen (TKN)</u>
Transition Phase: 180-860 mg/L
Acid Formation Phase: 14-1970 mg/L
May be low due to microbial assimilation of nitrogenous compounds
Methane Fermentation Phase: 25-82 mg/L
Low due to microbial assimilation of nitrogenous compounds
Final Maturation Phase: 7-490 mg/L

<u>Nitrate Nitrogen (NO_3-N)</u>
Transition Phase: 0.1-5.1 mg/L
Increasing due to oxidation of ammonia
Acid Formation Phase: 0.05-19 mg/L
Decreasing due to reduction to ammonia or N_2 gas
Methane Fermentation Phase:
Absent; complete conversion to ammonia or N_2 gas
Final Maturation Phase: 0.5-0.6 mg/L
<u>Ammonia Nitrogen (NH_3-N)</u>
Transition Phase: 120-125 mg/L
Acid Formation Phase: 2-1030 mg/L
Increasing due to nitrate reduction and protein breakdown
Methane Fermentation Phase: 6-430 mg/L
Decreasing due to biological assimilation
Final Maturation Phase: 6-430 mg/L

<u>Total Phosphate (PO_4-P)</u>
Transition Phase: 0.6-1.7 mg/L
Acid Formation Phase: 0.2-120 mg/L
Biological assimilation and metal complexation
Methane Fermentation Phase: 0.7-14 mg/L
Low due to bilogical assimilation
Final Maturation Phase: 0.2-14 mg/L

pH
Transition Phase: 6-7
Acid Formation Phase: 4.7-7.7
Low due to volatile acid accumulation
Methane Fermentation Phase: 6.3-8.8
Increasing due to volatile acid removal and bicarbonate dissolution
Final Maturation Phase: 7.1-8.8

Oxidation-Reduction Potential
Transition Phase: -80 to +80 mV
Acid Formation Phase: +80 to -240 mV
Decreasing due to the depletion of oxidation
Methane Fermentation Phase: -70 to -240 mV
Final Maturation Phase: +97 to+163 mV

Conductivity
Transition Phase: 2450-3310 μmhos/cm
Acid Formation Phase: 1600-17100 μmhos/cm
Increasing due to mobilization of metals
Methane Fermentation Phase: 2900-7700 μmhos/cm
Decreasing due to metals complexation with sulfides
Final Maturation Phase: 1400-4500 μmhos/cm

Chloride (Cl)
Transition Phase: 30-5000 mg/L
Biologically stable; good indicator of washout
Acid Formation Phase: 30-5000 mg/L
Stable; good hydraulic tracer
Methane Fermentation Phase: 30-5000 mg/L
Stable, good hydraulic tracer
Final Maturation Phase: 30-5000 mg/L
Stable, good hydraulic tracer

Sulfate (SO_4)

Transition Phase: 10-458 mg/L

Increasing due to aerobic oxidation

Acid Formation Phase: 10-3240 mg/L

Increasing initially due to aerobic solubilization then decreasing as anaerobic conditions are established

Methane Fermentation Phase:

Absent; complete conversion to sulfides

Final Maturation Phase: 5-40 mg/L

Reappearing due to oxidation

Sulfide (S)

Transition Phase:

Essentially absent

Acid Formation Phase: 0-818 mg/L

Beginning to appear and increasing due to sulfide reduction under anaerobic conditions

Methane Fermentation Phase: 0.9 mg/L

Low due to heavy metal precipitation

Final Maturation Phase:

Absent

Methane Gas

Transition Phase:

Essentially absent (aerobic metabolism)

Acid Formation Phase: <1%

Transition to anaerobic metabolism

Methane Fermentation Phase: 30-60 %

Suitable for energy recovery

FinalMaturation Phase: 0-<10 %

Decreasing due to substrate limitations and reversion to aerobic metabolism

Carbon Dioxide Gas

Transition Phase: 0-10 %

Product of aerobic decomposition of organics

Acid Formation Phase: 10-30 %

Increasing due to waste decomposition

Methane Fermentation Phase: 30-60 %

Decreasing due to <5% as methanogenesis increases

Final Maturation Phase: <40 %

Aerobic Metabolism

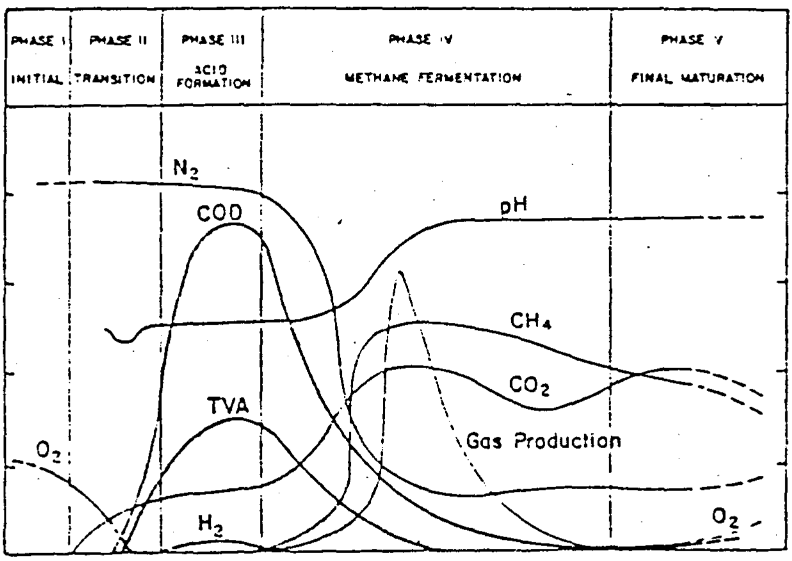


Figure 3.1 Phases of Landfill Stabilization

3.2 Anaerobic Decomposition

As it has been explained in the preceeding section, the anaerobic conditions prevail shortly after the placement of the refuse and the decomposition process takes place in the form of three-stage reactions:

(1) Hydrolysis

(2) Acetogenesis

(3) Methanogenesis

Complex Polymers
(proteins,polysaccharides)
1
Mono and Oligomers
(sugars,amino acids,peptides)
1
Propionate,Butyrate,etc.
(long chain fatty acids)
2
H_2,CO_2-3-Acetate
4-5
CH_4,CO_2

Figure 3.2.1 A Model for the Interactions Involved in the Conversion of a Polymeric Substance[21]

The microbial groups involved in the conversion process are as follows:

Group 1- fermentation bacteria

Group 2- hydrogen producing acetogenic bacteria

Group 3- hydrogen consuming bacteria

Group 4- CO_2 reducing methanogens

Group 5- aceticlastic methanogens

3.2.1 Hydrolysis

Hydrolysis, is an important reaction of organic compounds with water in aqueous environments such that it can be the rate limiting step. It is defined as a chemical transformation process, in which an organic molecule RX, reacts with water[22].

$$R\text{-}X \rightarrow R\text{-}OH + X + H \tag{3.2.1.1}$$

$$\text{alkyl halide} \rightarrow \text{alcohol} + \text{anion} + \text{hydrogen} \tag{3.2.1.2}$$

Although, it is regarded as a family of reactions, only some organic functional groups are potentially susceptible to the process. Thus, a correlation can be made between the resistant and susceptible functional groups:

Table 3.2.1.1 Types of Organic Functional Groups that are Generally Resistant to Hydrolysis[22]

Alkanes	Aromatic nitro compounds
Alkenes	Aromatic amines
Alkynes	Alcohols
Benzenes/Biphenyls	Phenols
Polycyclic aromatic hydrocarbons	Glycols
Heterocyclic polycyclic aromatic compounds	Ethers
Halogenated aromatics/PCB's	Aldehydes
Dieldrin/aldrin and related halogenated	Ketones
Hydrocarbon pesticides	Carboxyclic & Sulfonic acid

If the multifunctional organic compounds contain a hydrolyzable functional group in addition to the alcohol, acids, etc., they can be hydrolytically active.

Table 3.2.1.2 Types of Organic Functional Groups that are Potentially Susceptible to Hydrolysis[22]

Alkyl halides	Nitrites
Amines	Phosphonic acid esters
Amides	Phosphoric acid esters
Carbonates	Sulfonic acid esters
Carboxyclic acid esters	Sulfuric acid esters
Epoxides	

It is observed that hydrolysis of organic compounds is first order in the concentration of the organic species. The first order dependence plays an important role because it is an indication of having half life of RX (an organic molecule) independent of the RX concentration. The rate of disappearance of RX; -d[RX]/dt is directly proportional to the concentration of the compound, [RX]:

$$-[RX]/dt=k_T[RX] \qquad (3.2.1.3)$$

where k_T is the hydrolysis rate constant which can be expressed as:

$$k_T=k_H[H]+k_o+k_{OH}[OH] \qquad (3.2.1.4)$$

where k_H = rate constant for specific acid catalyzed hydrolysis

k_o =rate constant for neutral hydrolysis

k_o =rate constant for specific base catalyzed hydrolysis

[H]=hydrogen ion concentration
[OH]=hydroxyl ion concentration
As for the half life, the equation is presented as follows:

$$t_{1/2}=0.693/k_T \qquad (3.2.1.5)$$

The pH depence of the hydrolysis rate implicit in Equation (3.2.1.4), can be illustrated either with U-shaped or V-shaped pH profiles.

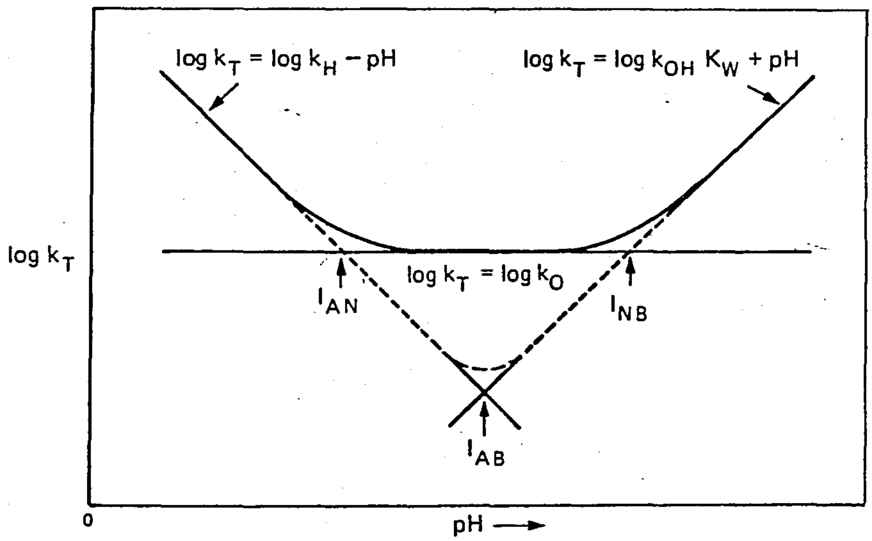


Figure 3.2.1.1 pH Dependence of k_T for Hydrolysis by Acid, Water and Base Promoted Processes[22]

** The transition points marked by 1,2,and 3 correspond to values of pH at which the acid or base catalyzed processes begin to make significant contribution to k_T .

With regard to temperature, it is found that the rate of hydrolysis increases with temperature. The Arrhenius Equation gives the quantitative relationship between the rate constant and temperature as:

$$k_T = Ae^{-E_a/RT} \tag{3.2.1.6}$$

where k_t =hydrolysis rate constant

E_a=the Arrhenius activation energy

R =the gas constant(1.987cal/deg-mol)

T =the temperature(°K)

A =pre-exponential factor (in the same units as the rate constant)

The values of E_a for hydrolysis of organics in water is found to fall in the range of 12-25kcal/mol. On the other hand, it is determined that in the vicinity of 300°K;

- a 1° change in temperature causes a 10% change in K
- a 10° change in temperature causes a factor of 2.5 change in K
- a 25° change in temperature causes a factor of 10 change in K

3.2.2 Acetogenesis

The acetogenesis process is the intermediate step in the anaerobic decomposition of MSW where the organic compounds are broken down primarily to organic volatile acids like acetic, propionic and butyric acids. These low molecular weight fatty acids are referred as volatile acids because they can be distilled at atmospheric pressure. An accumulation of volatile acids can have a destructive effect upon anaerobic decomposition if the buffering capacity of the system is exceeded and the pH falls to unfavorable levels.

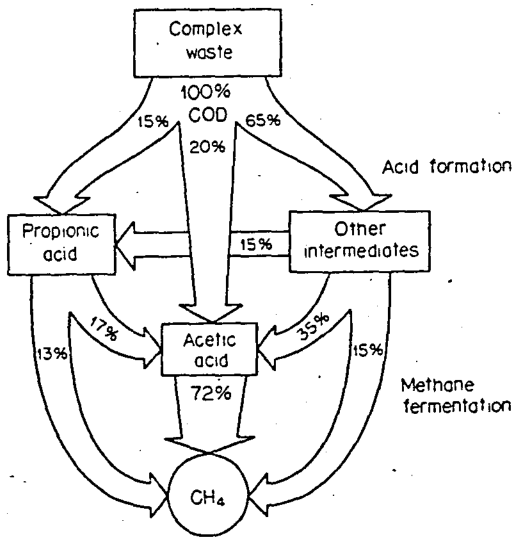


Figure 3.2.2.1 Pathways in the Conversion of a Complex Waste[32]

About 30 percent of the complex waste is converted to propionic acid before finally being transformed into methane gas. Acetic acid being the most abundantly produced volatile acid comprimises about 72 percent of the organic matter. The importance of acetic and propionic acid intermediates is then evident when it is considered that about 85 percent of the waste is converted to these two acids before methane gas is formed. If these acids are not converted to methane gas as rapidly as they are formed, their concentrations will increase and will result in lower pH values. In order to maintain the desired pH levels, buffering materials such as bicarbonates are utilized.

3.2.3 Methanogenesis

The methanogenesis is the terminal process in the anaerobic decomposition. The methanogenic bacteria capable of utilizing the H_2 produced by the acid formers reduce CO_2 to CH_4. About 70 percent of this methane is derived from the methyl group of acetate. By being one of the strickest anaerobes, the methanogenic bacteria require not only oxygen free conditions but also a redox potential lower than -330mV. They are more sensitive to environmental conditions, such as pH temperature and inhibitory compounds. They are known to exhibit a wide temperature spectrum with an optimum of 35° to 40°C for mesophilic species and 60°C for thermophilic species.

4. LEACHATE ANALYSIS

All landfills generate leachate. However, considerable deviations in leachate quality are encountered owing to seasonal/hydrological factors, age of landfills and the chemical nature of wastes accepted at the landfills. Nevertheless, establishing a uniform basis within a range of variations is possible by analyzing the sources of wastes. The majority of wastes disposed of in sanitary landfills are solid in nature. As shown in Table 4.1, five major source categories can be identified.

Table 4.1 Waste Source Categories and Corresponding Waste Types[40]

Source Category	Major Waste Constituents
Residential	Rubbish, food&garden wastes, plastics, glass, ash
Agricultural	Crop&animal wastes, food wastes, rubbish, chemicals
Commercial	Rubbish,food wastes, construction/demolition debris, ash
Municipal	Rubbish, ash, food wastes, sewage sludge
Industrial	Biological&chemical sludges, rubbish, ash, construction/demolition debris

Previous studies have confirmed the relative composition of municipal solid waste[40]. The results tabulated in Table 4.2, reflect the high potential for variance in the composition and relative proportions of wastes contributed from each source category. Nevertheless, from these analyses it can be presumed that food and garden wastes, rubbish, crop and animal residues will make up the organic compounds. Sewage sludges and certain industrial wastes will also contribute to the organic compounds. Hence, these wastes will furnish the moisture needed for leachate formation and biological activtiy leading to gas production. Different types of industrial sludges and residues, will constitute common sources of heavy metals. Ashes, construction/demolition debris will compose the inorganic constituents.

Table 4.2 Ranges of Composition of Municipal Solid Waste(MSW)*[40]

Component	Range	Typical
Food wastes	6|26	15
Garden wastes	0|20	12
Paper	24|45	40
Cardboard	3|15	4
Plastics	2|8	3
Rubber	0|2	1
Leather	0|2	1
Textiles	0|4	2
Plastic film		
Wood	1|4	2
Glass	4|16	8
Metallics		
Tin cans	2|8	6
Nonferrous metals	0|1	1
Ferrous metals	1|4	2
Dirt,ashes,brick etc.	0|10	4
Moisture	15|40	20

*Percent by weight, wet weight basis

Variations in MSW compositions are always expected. The MSW composition encountered in West Germany, (Table 4.3) and Turkey (Table4.4) could be cited as examples.

Table 4.3 The MSW Composition (1985) Encountered in West Germany[37]

Components	%by Weight
Organic materials	30.0
Paper	12.0
Cardboard	4.0
Packaging materials	2.0
Nonferrous metals	0.4
Ferrous metals	3.0
Glass	9.0
Plastics	5.0
Textiles	2.0
Minerals	2.0
Diapers	3.0
Problematic waste	0.4
Fine waste(<8mm)	10.0
Waste(8-40mm)	16.0
Miscellaneous	1.0

Table 4.4 The MSW Composition (1985) Encountered in the City of Istanbul[19]

Components	%by Weight
Organic materials	43.0
Paper-Cardboard	12.0
Textiles	5.0
Plastics	7.0
Glass	2.0
Metals	4.0
Ash	17.0
Miscellaneous	10.0

However, experience during the last 20 years has resulted in a good knowledge of the main parameters of pollution in leachate arising from MSW landfills.

4.1 Leachate Quality

It has been detemined that leachate quality is influenced in a complicated manner by the biological, chemical and physical processes which take place in a landfill body. Furthermore, leachate quality has been found to be variable owing to such factors like, the composition of solid waste, refuse moisture content, rate of water application and landfill design, operation and age.

The organic contaminants of leachate have been established as mainly soluble components of refuse or decomposition products of biodegradable parts of MSW. Previous works indicate that organic compounds detected in MSW landfills included organic acids, ketones, aromatic compounds, ethers, phthalates, halogenated aliphatic compounds, alcohols, amino aromatic compounds, nitroaromatic compounds, phenols, heterocyclic compounds, pesticides, sulfur substituted aromatic compounds, polyorganophosphates[9].

Volatile fatty acids produced during decomposition of lipids, proteins and carbohydrates make up the highest concentration of the class of organic compounds in leachate. The aromatic compounds (benzene,various xylenes, toluene...) are found frequently but in lower concentrations. Nonvolatile classes of compounds such as phenolic compounds arise from degradation by products of lignin.

A variety of heavy metals are frequently found in leachates including copper, zinc, cadmium, lead, nickel, chromium and mercury. The concentration of these heavy metals appear not to follow pattern of organic indicators like chemical oxygen demand (COD), or biochemical oxygen demand (BOD), nutrients or major ions. Their release is determined to be a function of pH, flowrate and concentrations of complexing agents

Table 4.1.1 Compounds Identified in Leachate[41]

Carboxyclic Acids	Phenols	Neutrals/Bases
Acetic	Phenol	Benzyl alcohol
Propionic	p-Cresol	2-Phenylethanol
1-Butyric	2,6 Dimethylphenol	x-Terpinol
Methylbutyric	2,4 Dimethylphenol	3-Methylindole
Valeric	C_3-phenols	Dimethyl phthalate
Methylvaleric		Methyloxindole
Hexanoic		Nicotine
Heptanoic		Diethyl phthalate
2-Ethylhexanoic		N-ethyl-p-toluene sulfonamide
Octanoic		Phenacetin
Benzoic		N-butyl benzene sulfonamide
Phenylacetic		Caffeine
Phenylpropionic		4 dimethylaminoantipyrine
Palmitic		Di-i butyl phthalate
Estearic		Dibutyl phthalate
		Butylbenzyl phthalate
		Di-i octyl phthalate
		Dioctyl phthalate

It is difficult to discuss in a general way the strength of leachates produced in landfills. . . However, generalizations can be made. Its peak strength occurs usually less than three years after placement and it dissipates with time particulary in the case of organic contaminants.

As a further illustration, a study conducted on two different types of leachate from a municipal waste tip could be given(Table 4.1.2). The first type of leachate was collected from a waste tip in the acidification stage. The organic load of this leachate was high and consisted mainly of free volatile acids (over 95% TOC). The second type of leachate was collected from a waste tip in the methane fermentation stage. The organic load was relatively low. The majority of the organics were identified as high molecular weight over 1000.

Table 4.1.2 Chemical characteristics of the Leachate Samples[11]

Parameters	Acidification Stage	Methane Fermentation Stage
	(Leachate 1)	(Leachate 2)
pH	5,70	7,00
TOC(mg/l)	20000	2100
COD(mg/l)	60000	7000
BOD(mg/l)	30000	50
HCO_3(mg/l)	17	12000

The organic load of the two types of leachate as observed from tha above table is completely different. Leachate 1 has very high TOC (Total Organic Carbon), COD (Chemical Oxygen Demand) and BOD (Biochemical Oxygen Demand) values where as with Leachate 2 these values are much lower. The low BOD value is an indication of the organic compounds as the degradation comes to an end.

Table 4.1.3 Composition of Organic Fraction of Leachate Samples in relation to the Fermentation Stage of the Refuse[11]

Acidification Stage		
(Leachate 1)		
Component	Concentration (mg/l)	TOC(mg/l)
Free volatile fatty acids		
Acetic acid	11000	4400
Propanoic acid	3760	1829
2-methyl-propanoic acid	520	284
Butanoic acid	9890	5395
2methyl-butanoic acid	360	206
3methyl-butanoic acid	320	188
Pentanoic acid	2510	1476
4methyl pentanoic acid	70	43
Hexanoic acid	5770	3581
Volatile amines		
Methylamine	6	2
Trimethylamine	83	51
t-butylamine	41	28
sec-butylamine	102	67
iso-butylamine	32	21
Alcohols		
Ethanol		
Organic Compounds		
Extractable with Hexane		
Aliphatic compounds	0.64	<1
Aromatic compounds	1,21	~1
Polar compounds	12,50	~7
High Molecular Weight		
Compounds		
Molecular weight >50000	/	100
Molecular Weight>1000-50000	/	160

Methane Fermentation Stage		
(Leachate 2)		
Component	Concentration (mg/l)	TOC (mg/l)
Free Volatile Fatty Acids		
Acetic Acid	BD*	
Propionic Acid	BD	
2-methyl-propionic acid	BD	
Butanoic acid	BD	
2 methyl-butanoic acid	BD	
3 methyl-butanoic acid	BD	
Pentanoic acid	BD	
4 methyl pentanoic acid	BD	
Hexanoic acid	BD	
Volatile amines		
Methylamine	BD	
Trimethylamine	BD	
t-butylamine	BD	
sec-butylamine	BD	
iso-butylamine	BD	
Alcohols		
Ethanol	BD	
Organic Compounds Extractable with Hexane		
Aliphatic compounds	0.09	<0.1
Aromatic compounds	0.38	<0.3
Polar compounds	4.38	2.5
High Molecular Weight Compounds		
Molecular weight > 50000	/**	252
Molecular weight > 1000-50000	/	420

*below detection limits

**not measured

A study conducted on the stability of organic matter in 12 landfill leachates collected from representative locations and various precipitation regimes under a range of temperatures revealed that the most stable group of organics with increasing age was a fulvic like material with relatively high carboxyl and aromatic hydroxyl group density [15]. Increasing stability was further noted for carbohydrates, hydrolyzable amino acids and aromatic hydroxyl compounds in this order.

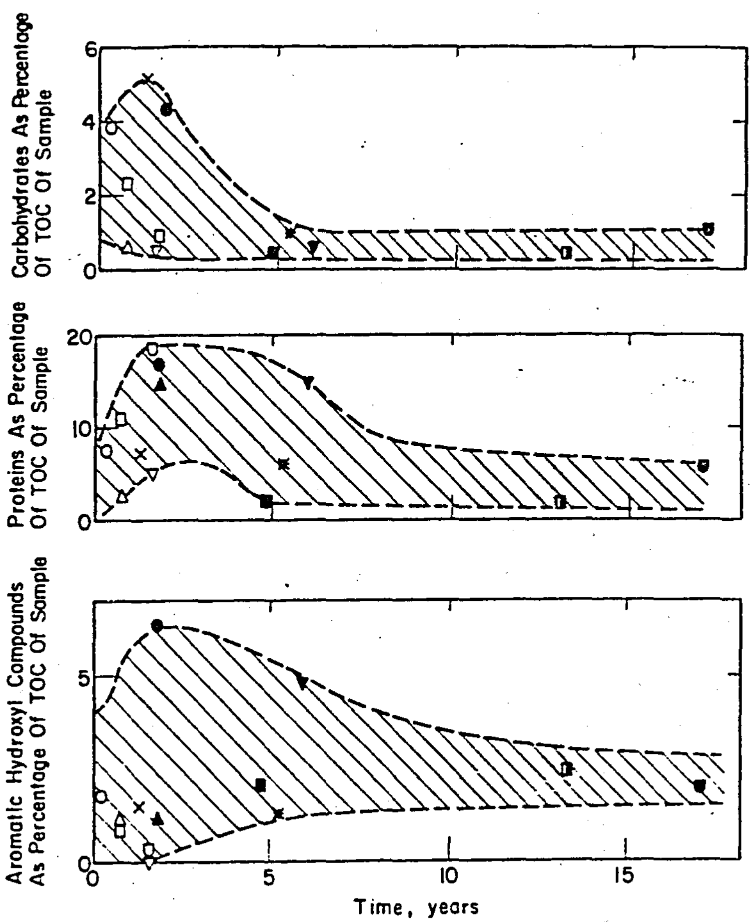


Figure 4.1.1 Stability of Organic Matter in Leachates[50]

Recognition of the significance of the parametes, their relationship to site or with time is important. The results of leachate anlyses from more than 15 landfills is presented as [27]:

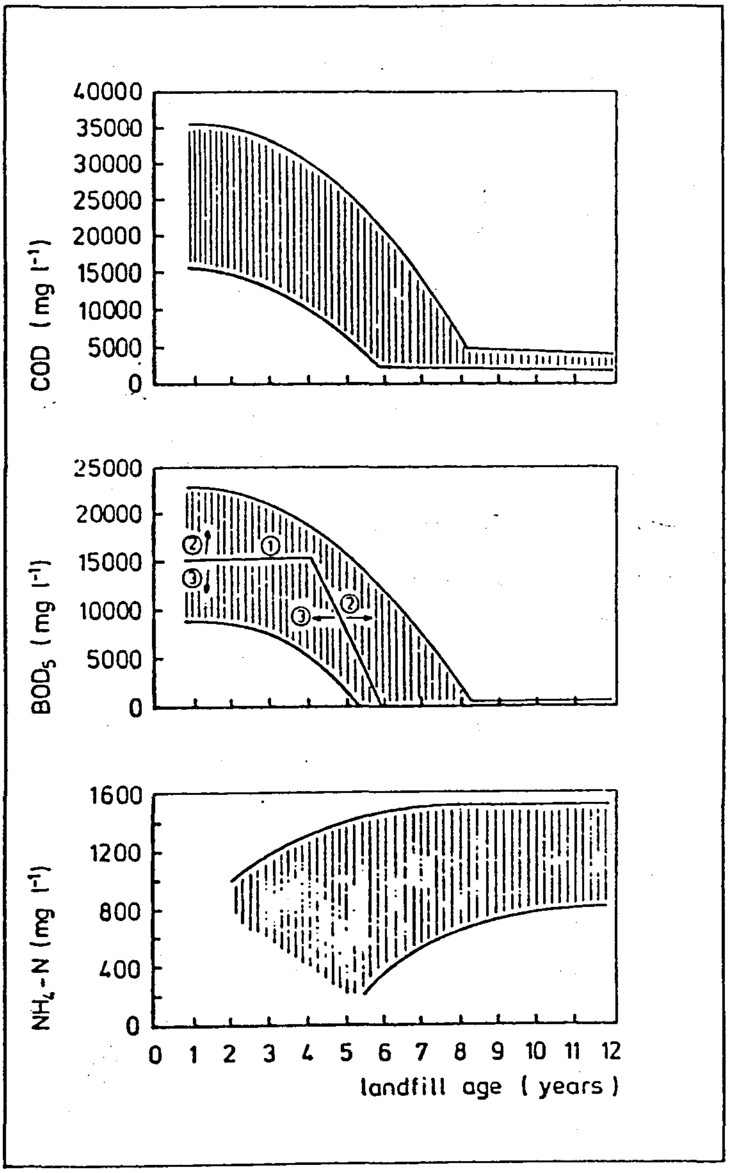


Figure 4.1.2 General tendencies of COD, BOD5 and NH4 versus Time

Table 4.1.6 Leachate Analysis Values for Parameters with Differences Between Acetic and Methanogenic Phase[27]

Parameter	Average	Range
Acetic Phase		
pH	6,1	4,5-7,5
BOD_5 (mg/l)	13000	4000-40000
COD (mg/l)	22000	6000-60000
BOD_5/COD	0.58	/
SO_4 (mg/l)	500	70-1750
Ca (mg/l)	1200	10-2500
Mg (mg/l)	470	50-1150
Fe (mg/l)	780	20-2100
Mn(mg/l)	25	0.3-65
Zn (mg/l)	5	0.1-120
Sr (mg/l)	7	0.5-15
Methanogenic Phase		
pH	8	7,5-9
BOD_5 (mg/l)	180	20-550
COD(mg/l)	3000	500-4500
BOD_5/COD (mg/l)	0.06	/
SO_4 (mg/l)	80	10-420
Ca (mg/l)	60	20-600
Mg (mg/l)	180	40-350
Fe (mg/l)	15	3-280
Mn (mg/l)	0.7	0.03-45
Zn (mg/l)	0.6	0.03-4
Sr (mg/l)	1	0.3-7

Table 4.1.7 Leachate Analysis for Parameters with no Difference between Phases

Parameter	Average	Range
Cl (mg/l)	2100	100-5000
Na (mg/l)	1350	50-4000
K (mg/l)	1100	10-2500
Alkalinity (mg $CaCO_3$/l)	6700	300-11500
Organic N (mgN/l)	600	10-4250
NH_4 (mgN/l)	750	30-3000
NO_3 (mgN/l)	3	0.1-50
NO_2 (mgN/l)	0.5	0-25
Total P (mgP/l)	6	0.1-30
CN (mg/l)	/	0.04-90
AOX (μgCl/l)	2000	320-3500
Phenol (mg/l)	/	0.04-44
As (μg/l)	160	5-1600
Cd (μg/l)	6	0.5-140
Co (μg/l)	55	4-950
Ni (μg/l)	200	20-2050
Pb (μg/l)	90	8-1020
Cr (μg/l)	300	30-1600
Cu (μg/l)	80	4-1400
Hg (μg/l)	10	0.2-50

4.1.1 Duration of Leachate Emissions

As discussed previously, great differences exit between the acetic and methanogenic phases. After changing to the methanogenic phase most values change to a more homogenous level of 3000 to 4000 mg COD/L. Much carbon is transferred from the solid phase into the gas phase. But the main carbon transfer from the solid phase is estimated to take place in a period of one century.

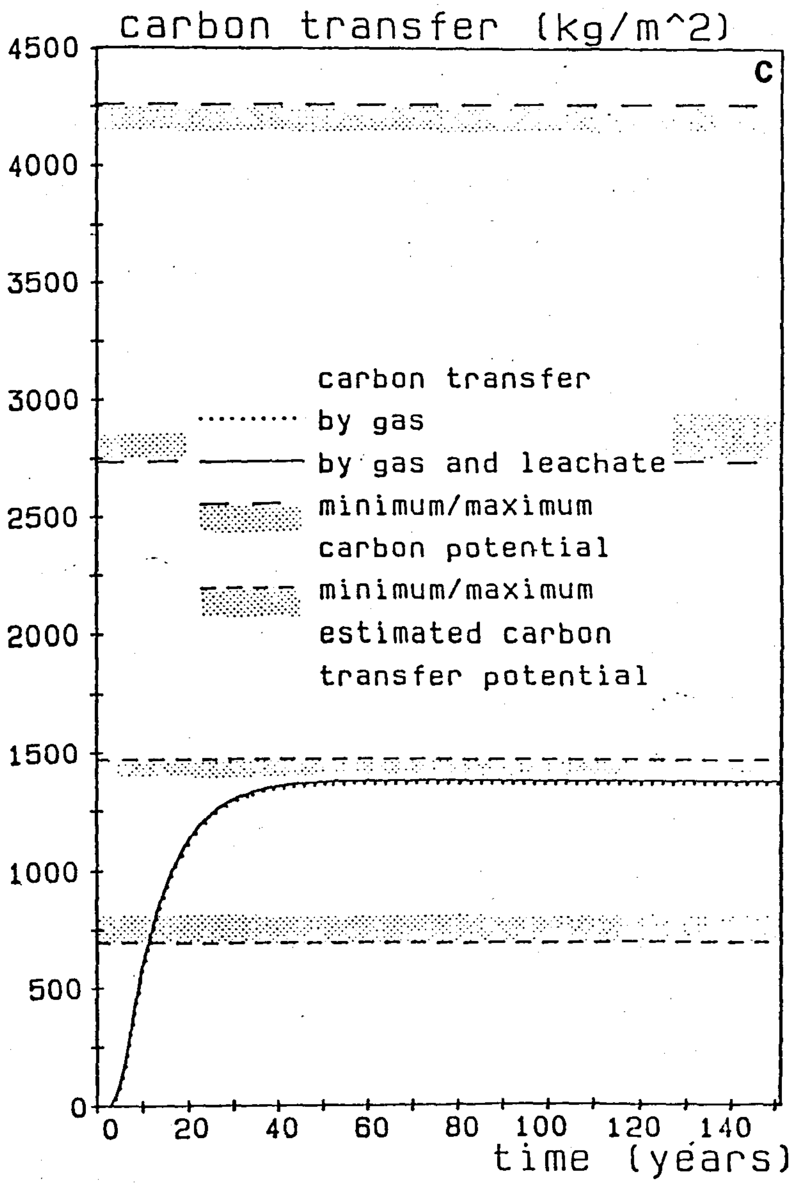


Figure 4.1.1.1 Estimated Carbon Potential and Transfer by Gas and leachate from a Refuse Column (Area= $1m^2$ and Height= 20m) [27]

In contrast to carbon transfer, the transfer of chlorides is based mostly on chemical processes[27]. Observations at landfills with a maximum age of 20 years show no decrease in chloride concentrations. And that the chloride emission potential of landfills as batch reactors is limited; therefore, a decrease in concentrations must occur.

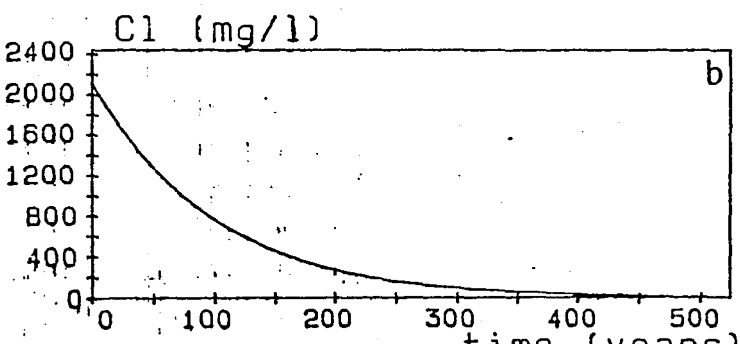


Figure 4.1.1.2 Estimated Chloride Concentrations of a Full Scale Landfill

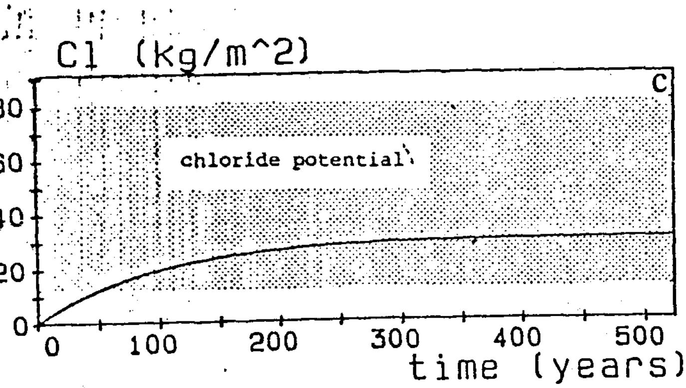


Figure 4.1.1.3 Chloride Transfer and Potential from a Refuse Coulumn (Area= 1 m^2 and Height= 20m)

The nitrogen transfer is a combination of carbon and chloride transport processes. Most of the nitrogen is converted from organic nitrogen into ammonia. As a result the ammonium concentration in Figure 4.1.1.4 is less defined when compared with the others . Differences to measured potential are much greater than for chloride.

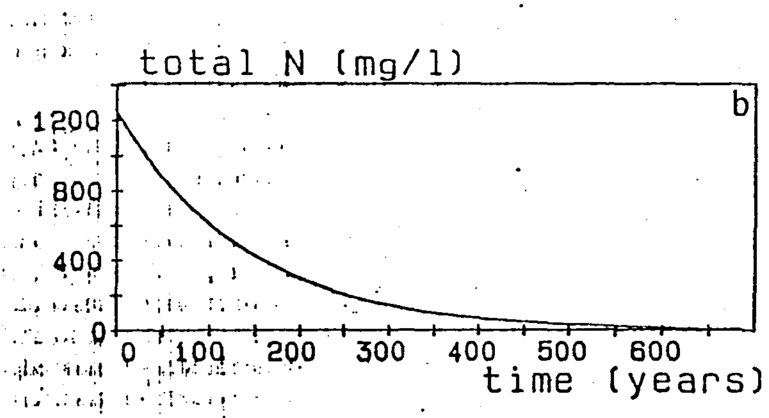


Figure 4.1.1.4 Estimated Nitrogen Concentrations of a Full Scale Landfill[27]

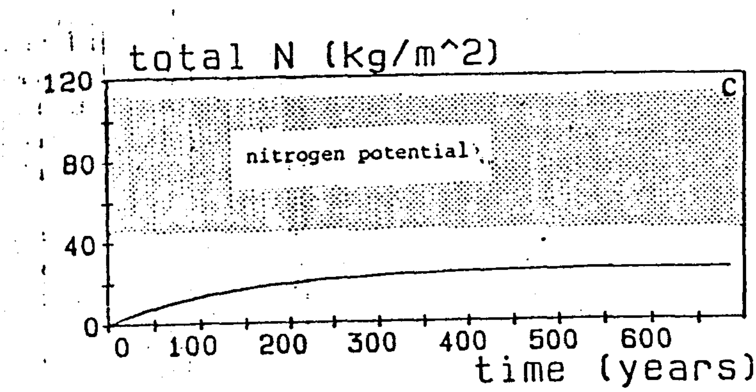


Figure 4.1.1.5 Nitrogen Transfer and Potential of a Refuse Column (Area= $1m^2$ and Height= 20m)

The relationship between measured initial concentrations and the potential in landfills is divergent. Accordingly, longer half-times must be estimated. As cited by an example, from a refuse column with an area of $1 m^2$ and a height of 20 m had the potential of organic chloride as 19.2 kg. The total organic chloride transfer by gas was 0.053 kg organic Cl/m^2 estimated to be in 100 years[27]. Similar relationships for metals are presented in Table 4.1.1.1.

Table 4.1.1.1 Metal Content of Refuse and Transfer Rates of a refuse Column (Area $= 1 m^2$, Height $= 20 m$, Leachate production $= 20\%$ Precipitation)

Parameter	Contents (mg/t dry refuse)	Transfer Rate (mg/t dry refuse/year)
Ni	15000	2,70
Cr	5000-100000	4,00
Cu	238000	1,10
Pb	399000	1,20
Zn	521000	8,00
Cd	3480	0,08
Hg	640	0,13

In order to reduce leachate emissions, the following operation effects are implemented:

1. Water addition to increase the moisture content
2. Leachate recirculation,
3. Seeding with digested sewage sludge,
4. Lime addition,
5. Loose refuse compaction,
6. Controlling the thickness of dumped layers,
7. Controlling the landfill build up speed

4.1.2 Leachate and Soil Interactions

Prediction of leachate migration requires an understanding of the various mechanisms governing the behavior of the soil and the leachate. It has been found out that the interaction of the leachate with the soil is a function of several factors including pH, total organic carbon content of the soil organic matter, the aerobicity/anaerobicity of the soil, nature and concentration of metal ions. Studies on the interactions of soils with leachates have been focused on the fate or biological treatment related to heavy metals, pesticides, organics and toxic substances.

In the evaluation of the interactions between heavy metals in the leachate and the soil, considerable emphasis have been placed on the fates of cadmium, nickel, lead, zinc and copper[44]. Results of the studies provide evidence that the above discussed metals are mobile in natural soils (sand,silt&clays). The relative mobility of these metals has been determined to be dependent on the pH, soil types, initial concentration of heavy metals in the leachate and in the soil,and in the cation exchange capacity. Metals have been found to be more mobile as the pH decreased. The clay content has been found to be important due to its cation exchange properties. Therefore, it is believed that the presence of high level of salts, iron and organics (TOC) will enhance the migration of metals due to a more rapid exhaustion of the soil cation exchange capacity.

Additionally, microbial activity has been believed to influence metal migration. Many biochemicals, including amino acids& aliphatic acids synthesized by microrganisms Form soluble complexes with metal ions. The microroganisms also impart a change on the pH levels during the biodegradation phases, compete for adsorption sites and prevent the flow due to clogging in the soil pores.

In a study conducted to determine the heavy metal migration in soil and leachate systems, four site specific soils and a characteristic liquid from an anaerobic digestion plant in Florida were used.

Table 4.1.2 Characteristics of the Soils[43]

Soil	Organic Content	Average pH	Cation Exchange Capacit
	g/kg		μeg/g
Hallandale Fine Sand(Sandy)	14,5	8,2	1,13
Plantation Muck Top Layer(Clayey)	705,2	7,1	4,53
Plantation Muck Middle Layer(Clayey)	670,7	7,2	4,09
Plantation Muck Bottom Layer(Sandy)	27,9	7,3	1,58

Table 4.1.2.2 Initial Concentration of Cations in the Soil[43]

Soil	Metal Concentrations (μeg/g)									
	Cu	Zn	Fe	Mn	Ni	Cd	Cr	Ca	Ag	Co
Hallandale Fine Sand	.031	.192	5,52	.185	.139	.056	.271	4,72	0	.006
Plantation Muck Top Layer	.173	.293	223.8	.442	.669	.058	.819	893.5	0	.203
Plantation Muck Middle Layer	.173	.327	1021.7	1,36	.422	.088	1,19	1185.5	.005	.370
Plantation Muck Bottom layer	.056	.177	16.67	.185	.139	.056	.271	4,72	0	.067

Batch experiments were carried out on the four types of soil and the characteristic liquid. The results of the batch experiments were utilized in the determination of chemical distribution coefficients.

Table 4.1.2.3 Intial Heavy Metal Concentrations in Leachate from an Anaerobic Digestion Plant[43]

Cation	Concentration (ppm)
Copper	0.68
Zinc	2,26
Iron	13,53
Manganese	1,8
Nickel	0.36
Cadmium	0.05
Chromium	0.26
Calcium	528000

The results were established through the implementation of the Lapidus-Amundson Linear Isotherm and the Langmur Nonlinear Isotherm. The solution procedures of these models however are not presented in this paper.

Table 4.1.2 Results of the Relative Mobility of the Cations[43]

Model	Most Mobile Cation	Least Mobile Cation
1.Lapidus-Amundson Model		
(Linear Isotherm)		
Hallandale Fine Sand	Manganese	Chromium
Plantation Muck Top Layer	Iron	Nickel
Plantation Muck Middle Layer	Manganese	Calcium
Plantation Muck Bottom Layer	Manganese	Chromium
2.Langmuir Model		
(Non-linear Isotherm)		
Hallandale Fine sand	Manganese	Nickel
Plantation Muck Top Layer	Copper	Cadmium
Plantation Muck Middle layer	Iron	Manganese
Plantation Muck Bottom Layer	Manganese	Nickel

With regard to their impact on contaminants, leachate oriented organics are also important through their effects on soil structure. Previous works indicate that the solubility of clays in acids is dependent on factors like the concentration of the organic acid present, the nature, and the duration of the acid/clay contact period and temperature[44].

Experiments carried out with laboratory columns and field cells reveal that an initial reduction in the permeability of soil is followed by noticeable increase in permeability. The weak acids are found to be more reactive than weak bases. The dissolution of aluminum and other ions is observed when they are exposed to weak acids.

Tests conducted in a similar fashion with neutral polar organics such as ethylene glycol, acetone and methanol along with neutral nonpolar organics such as xylene and heptane show that apronounced initial decrease in permeability is followed by a gradual increase in permeability.

Studies have also been carried out to evaluate leachate and soil interactions arising from toxic compounds such as arsenic, cyanide and halogenated organics. Arsenic has been found to be relatively immobile in soils and its complete immobilization is possible by fixation. In contrast, cyanide has been determined to be very mobile in soils. Furthermore, it has been observed to be more mobile in water than in avererage leachates. Its attenuation is found to be better accomplished at öow pH and in the presence of iron oxides and clays; especially kaolinite.

The halogenated organics including polychlorinated biphenyls(PCB's), polybrominated biphenyls(PCB's) and hexachlorobenzenes(HCB's) are known to be as carcinogens along with low solubility in water. In tests using typical soil and leachates these compounds have been found to be relatively immobile. They present a high potential for eventual migration due to their refractory characteristics.

4.1.3 Modelling of Leachate Organic Migration

Leachate organic matter is treated as a non-onservative material. The mass transport equation (Equation 4.1.3.1) for a non-reacting solute in a homogenous media can be modified to include the appropriate terms for a non-conservative material[42].

$$\frac{\partial C}{\partial t} = \frac{\partial}{\partial x_i}\left[D_{ij}\frac{\partial C}{\partial x_j}\right] - V_i\frac{\partial C}{\partial x_i} \qquad (4.1.3.1)$$

where C = concentration of the non-reacting solute
t = time
x_i = distance
D_i = hydraulic dispersion tensor
V_i = intestitial fluid velocity

During transport through soils, biochemical degradation and absorption are regarded as the reduction mechanisms in the organic mateer. Hence, sink terms are needed to be added to the Equation (4.1.3.1).

$$\frac{\partial S}{\partial t} = \frac{\partial}{\partial x}\left[D_{ij}\frac{\partial S}{\partial x_j}\right] - V_i\frac{\partial S}{\partial x_i} - r \qquad (4.1.3.2)$$

where r = sink term for solute reactions
s = substrate concentration.

The sink term is expressed as:

$$r = R + R_A \tag{4.1.3.3}$$

where R = sink term for biochemical reaction
R_A = sink term for adsorption.

Furthermore, when the microbial mass concentration reaches a steady state and low levels of COD is observed then the sink term R could be rewritten as:

$$R = k_2 S \tag{4.1.3.4}$$

where k_2 is the first order degradation coefficient.

Hence, the migration of organics at low concentrations relative to the initial non-degraded concentration can be expressed as:

$$\frac{\partial C}{\partial t} = \frac{\partial}{\partial x_i}\left[D_{ij}\frac{\partial C}{\partial x_j}\right] - V_i\frac{\partial C}{\partial x_i} - k_2 S - R_A \tag{4.1.3.5}$$

For various degradation phase, the equations of mass consrvation for leachate organics can be combined and solved through finite element methods. However, the solution procedures are not discussed within the scope of this paper.

5. LEACHATE TREATMENT PROCESSES

Most of the processes applied for industrial wastewater treatment have been examined for the treatmrnt of landfill leachates. The traditional aerobic and anaerobic biological processes together with physical and chemical processes have been employed for the removal of organic and inorganic contaminants. Moreover, the process performance in each case is related to the chemical nature of the leachate used. Biological treatment methods have been found to be superior to physical-chemical tecniques for the removal of biodegradable fractions. However, processes including reverse osmosis, activated carbon and ion exchange have been effectively to treated effluents coming from biological processes[40,46,47].

As shown in Table 5.1, there are a number of options available for leachate treatment. For the on-site applications leachate recycling has been recommended. In the case of external treatment of leachates, biological treatment has been considered as the most appropriate first step[46].

Table 5.1 Treatment options Available for Leachate Management[40]

In Situ Treatment	*External Treatment*	
Leachate Recycle	*Biological Treatment*	*Physical/Chemical Treatment*
	Activated Sludge	Precipitation/Coagulation
	Aerated Lagoon	Chemical Oxidation
	Stabilization Pond	Disinfection
	Fixed-Film Processes	Adsorption
	Anaerobic Filter	Ion Exchange
	Anaerobic Contact	Reverse Osmosis

As for the performance of the above discussed processes, the following has been summarized as:

Table 5.2 The Performance Capability of Leachate Treatment Processes[40]

Processes	BOD Removal	COD Removal	Fe Removal	Zn Removal	Ni Remo
Aerobic Biological	(%)	(%)	(%)	(%)	(%)
Activated Sludge	95	95	96-99	96-99	60
Combined Leachate&Sewage	94-99	92-98			
Aerated Lagoon	99	92-98	99		
Stabilization Pond	93-99	99	80-99		
Anaerobic Processes					
Attached Growth	85-98	75-95	80-99	80-99	10-80
Suspended Growth	85-98	75-95	80-99	80-99	10-80
Leachate Recycle	NA*	NA	NA	NA	NA
Physical-Chemical Proceses					
Coagulation		12	95-99	75-98	
Oxidation		10-50	99	90	
Reverse Osmosis		86-94			
Ion Exchange		40-70	40-80	20-96	14-96
Adsorption		75-99	65-95		

*Not available

Table5.2 Leachate Treatment Processes[48]

Biological	Anaerobic
	Aerobic
	Combination
Physical	Membrane Filtration
	Stripping
	Distillation
	Combination
Chemical	Oxidation
	UV Irradiation
	Ion Exchange
	Combination
Chem/Physical	Flocculation/Coagulation
	Adsorption
	Combination
Thermal	Volatilization
	Incineration

5.1 Biological Treatment

The important characteristics of leachate which have influence over biological treatment are related to the high concentration of organic and inorganic substances, irregular production, variations in the biodegradable fraction of organic substance and the low or negligible amounts of phosphorus. These characteristics of leachate impose difficulties in operation and the evaluation of the treatment processes when inconsistent strength and volumes are encountered. Hence, different treatment strategies are implemented to match the differing leachate volumes and strengths.

Furthermore, leachate may contain substances which have the capabiltiy of limiting biological treatment efficiency. the compounds which may affect treatability are as follows[46]:

1.Metals-Under the prevailing anaerobic conditions, the concentrations of metals are reduced to a great extent. The presence of heavy metals in leachate is dependent on the concentration and activity of bacteria. In the case of metal inhibition, simple pre-treatment could be applied to precipitatite and to remove the metals as hydroxides.

2.Carbon Compounds-Compounds including chlorinated solvents, cyanide compounds and phenols are attenuated inside the landfill body. As a prevention against the effects of high concentrations of toxic components of leachate, the chlorinated solvents could be volatilized. High concentrations of phenols and cyanide compounds could be biologically oxidized.

3.Ammonia-Very high concentrations of ammonia nitrogen in influents have been known to be tolerable in both aerobic (6350mg/l) and anaerobic (2700mg/l) systems.

4.Chloride-Anaerobic systems are known to be more sensitive to chloride concentrations than aerobic processes. It is known that up to 1000mg/l a profound effect on the gas production.

5.Sulphide-The precipitation of sulphides in the landfill reduces the concentration of sulphide in leachate. Anaerobic digestion is known to tolerate up to 200mg/l as soluble sulphide whereas aerobic processes can tolerate up to 10000mg/l of S^{-2}.

5.1.1 Aerobic Biological Leachate Treatment

The operation and evaluation of aerobic biological treatment processes are dependent upon controlling certain process variables. Carbon which is the limiting nutrient in biological treatment systems of primary interest due to its reflection of the rate of biomass utilization.

The kinetic parameters which are used in describing the growth of microrganisms in response to the availability of a limiting substrate include, maximum specific cell growth rate (μ_{max}), the cell decay coefficient (b), the saturation coefficient (k_s) and the cell yield (Y). The operational parameters are the mean cell residence time (Q_c), the volumetric organic loading rate (OLR), and the food to microrganisms ratio. According to the manipulation of these operationg variables process performance is evaluated by making comparisons of effluent organic concentrations to influent concentrations and to existing effluent.

5.1.1.1 Aerated Lagooning

Aerated lagoons are known to provide an efficient treatment system in the case of low leachate production. Mechanical or diffused aeration is used to provide oxygen and mixing. The process of degradation orf organic substances are carried out faster with respect to anaerobic processes. Although, not often practiced, aerated lagoons may employ biomass recycle to increase cell retention time (Q_c) in a manner similar tto the activated sludge process.

Studies conducted in 1970's by various researchers indicate that aerobic biological treatment of leachate would not be succesful at high organic loading and low retention periods without addition of nutrients[46]. Leachates with BOD_5 concentrations less than 1000mg/l were degraded down to values near 25 mg BOD/l in about 14 days at room temperature. Leachates with higher organic content (400-100000mg/l) could not be degraded down to values around 25 mg/l in a period of 80 days. In these experiments no phophorus was added to the leachates.

Studies conducted by Chian [15] determined that treatment in an aerated lagoon at retention periods ranging from seven to 85 days could remove 93 to 96 percent of the organic matter from a leachate having a COD of 58000mg/l.

Studies at laboratory scale were also done by Robinson et at., 1985. Simulated aerated lagoons with a volume of 20 liters were kept at constant temperature(10°C). The average influent concentration was around 2845mg/l of BOD. The results showed that sludge age $\geq$ 10 days, BOD_5 concentrations $<$25mg/l were achieved. When the detention time was shorter than 15 days and the temperature was changed from 10° to 5°C, higher BOD_5 concentrations were observed.

Full scale investigations on aerated lagooning has proven to be effective in terms of BOD, COD and Fe removal. Detention times ranging from 7 to 135 days provided 70-99 percent BOD_5 removal and 70-95 percent COD removal. 90 percent of BOD removal was achieved along with 92 to 99 percent Fe removal.

5.1.1.2 Activated Sludge

Activated sludge plants differ from aerated lagoons in the sense that the discharged sludge is recirculated. Also, the retention period is considered to be shorter than that for an aerated lagoon.

The effects of mean cell residence time (Q_c), on process performance have been investigated; six to 10 days of Q_c have been found to be sufficient for process performance. Retention times higher than 10 days resulted in effluent COD concentrations above 300mg/l. Residual COD was attributed to refractory organics such as humic and fulvic like substances[40].

The effects of temperature on the performance of bench-scale activated sludge units were evaluated by referring to effluent COD concentrations and percentage of removals at temperatures between 5° and 25°C. However, since the performance parameters are also dependent on Q_c and the influent concentration, the data collected were also compared with regard to these variables. For the influent organic strength, three categories were defined as follows:

Table 5.1.1.2.1 Leachate Organic Strength Categories[40]

Leachate Strength Category	Concentrations Ranges, mg/l COD basis
Low strength	<1000
Medium strength	1000-10000
High strength	>10000

The cell residence time was also grouped into three as follows:

$Q_c < 4$ days

$6 < Q_c < 10$ days

$Q_c > 12$ days

Accordingly, the trend has shown an expected increase in organic removal with increasing temperature. Mean residence time > 6 days at temperatures as low as 5°C has demonstrated a succesful operation.

The activated sludge process has also been investigated in its heavy metal and alkaline earth removal. The process was effective in removing the majority of heavy metals monitored. Particularly, zinc, iron, manganese and cadmium were removed by 95 percent or greater. Chromium and lead were removed in the range of 80 to 90 percent. Nickel removal was generally low. The alkaline earth metals were removed to a lesser degree than the monitored heavy metals. Calcium and magnesium were removed in the range of 3 to 99 percent. However, typical values were from 40 to 70 percent. Potassium and sodium removals were generally on the order of 20 to 40 percent.

Table 5.1.1.2.2 Summary of Heavy Metal and Alkali-Alkaline Earth Metal Removal from the Bench Scale Activated Sludge Process[40]

Heavy Metals	Cd	Cr	Fe	Pb	Mn	Ni
Influent concentration range, mg/l	0.04-0.4	0.1-1.9	240-2130	0.17-1.44	13-41	o.18-(
Removal range, %	85-99	75-98	96-99	82-98	90-99	39-75
Average removal, %	96	92	98	89	97	60
Alkali & Alkaline Earth Metals	Ca	Mg	K	Na		
Influent concentration range, mg/l	88-3730	35-660	200-1060	430-1350		
Removal range, %	64-99	3-90	8-46	0-35		
Average removal, %	90	52	27	16		

Nitrification of leachate in the activated sludge process has been studied in depth. For complete nitrification longer Q_c (10 days or longer) were needed for this growth of nitrifying bacteria. At lower Q_c, in the range of six to 10 days, 60-80 percent nitrification was accomplished. Air stripping of NH_3 resulted in 96 to 99 percent removal at 23°C and a pH of 8.5-8.8 [40].

5.1.1.3 Combined Treatment with Municipal Wastewater

The combination of leachate with larger volumes of municipal wastewater has been reviewed on only one pilot-scale study. The combined treatment of leachates was succesful in removing 98 to 99 percent of BOD_5 and COD reduction was observed to be 95 percent. However, greater air requirements as high as 400 and 800 percents in oxygen supply have been found to be necessary for succesful treatment at 10 and 20 percents leachate domestic wastewater fractions[40]. Solids production was also observed to be higher than in the range of 300 and 800 percents at 10 and 20 percents leachate to wastewater ratios, respectively. For sludge handling, L/D ratio should be less than 5 percent to have an appropriate treatment strategy.

5.1.1.4 Rotating Biological Contactors

Rotating biological contactors have been employed as an aerobic system for leachate treatment. In the process, the biomass is attached to circular blades which are rotated by a mast to which they are fixed. Hence, the rotational movement is a means of contact with atmospherical oxygen. this oxygen is used by the biomass present on the disk to aerobically degrade the organic substances.

It has been concluded through laboratory and full-scale investigations that rotating biological contactors show very high ammonia removal rates at loading ranging from 2-3gN/m^2 at temperatures of 18°-22°C[46]. Higher loading rates up to

17.5gN/m².d although led to nitrification rates greater than 90 percent could also result in high nitrite concentration in the effluent.

Table 5.1.1.4.1 Experimental Conditions during aTreatment by a Rotating Biological Contactor Process(* chemically pre-treated) [40]

Leachate *	Rotating Biological Contactor
InfluentBOD,mg/l	50
InfluentCOD, mg/l	400
InfluentTOC, mg/l	114
BOD/COD	0.13
COD/TOC	3.5
BOD loading, kg/m² day	0.78
COD loading, kg/m²day	6.2
COD removal, %	15
Temperature, °C	17
Reactor type	Plug flow reactor

5.1.1.5 Trickling Filter

The trickling filter mechanism is made up of a several meter high tanks filled with gravel or synthetic material such that leachate is sprayed over the surface of the filling material. The movement of the leachate is then through the filter donward. Leaching occurs from one filling element to the next such that the entire bed is not submerged and air passes through the free spaces.

The results of a study indicate that ammonia can be completely nitrified using a trickling filter[46]. However, the treatment of high strentgh leachates with trickling filters is believed to be diffcult due to clogging and scaling oxidation.

Table5.1.1.5.1 Experimental Conditions during a Trickling Filter Leachate Treatment Process[46]

Leachate*	Trickling Filter
InfluentBOD_5, mg/l	50
InfluentCOD, mg/l	380
InfluentTOC, mg/l	114
BOD_5/COD	0.13
BOD_5 loading, kg/m^{3}d	0.1
COD loading, kg/m^{3}d	0.9
COD removal, %	0.9
Temperature, °C	17
Reactor type	Continuously stirred tank reactor

*Chemically pre-treated leachate

5.1.2 Anaerobic Biological Leachate Treatment

Anaerobic biological treatment methods for the treatment of leachates have provided several advantages over the traditional aerobic processes studied. These advantages could be cited as methane production as a by product, much lower production of biological solids in the form of sludges or suspensions and no requirement of aeration[40,46,4,10,18]. Also, the generation of hydrogen sulphide from the biolgical reduction of sulphate provides an efficient precipitant for most of the metals.

However, a disadvantage of anaerobic treatment process is that the microorganisms are inhibiteed by low pH values and thus these inhibitors can lead to reduced growth rates and a net washing out of microbial cells from a completely mixed reactor system. Hence, preventions in the form of an anaerobic filter should be taken.

5.1.2.1 Anaerobic Lagooning and Digesters

Anaerobic lagooning method has been practiced through physical and physico-chemical processes (sedimentation, flocculation and precipitation) which prevent aeration of the liquid mass. Nitrogen removal is accomplished only by means of assimilation. Hydrolysis of organic nitrogen in ammonia is an important step for heavy metal reduction due to precipitation and embeshing in the biomass.

Contrary to anaerobic lagooning which requires an open tank, the anaerobic digester is composed of a closed container. Morever, the anaerobic digester is continously agitated by the re-introduction of biogas produced during degradation processes.

Boyle and Ham [51] showed that under anaerobic conditions for 10-12 days at temperatures between 23° and 30°C, greater than 90 percent removal of organic material could be accomplished. It was also noted that temperature is an important factor affecting the performance of anaerobic treatment. At 11°C, the efficiency dropped to 22 percent.

From further studies, it was concluded that the percentage removal of iron, zinc, nickel, cadmium, lead and chromium increased with increasing concentration of metals in the leachate and with increased hydraulic retention time[50].

In the experiments performed by Cameron and Koch (1980) on anaerobic digestion, the reactors were mechanically agitated and were kept at a constant temperature of 34°C-/+1°C. Retention periods of five, 10 and 20 days were studied and the percentages of removal ranged between 82-99 percent.

A study conducted on a full scale anaerobic lagooon, activated sludge oxidation and sedimentation. The anaerobic pond which was lined with PVC membranes had a volume of 900m³. The removal rates of organic substances was achieved in the order of 50-60 percent.

Bull et al .,[10] investigated on a full-scale, the anaerobic treatment of leachate with a COD value of 8000mg/l. Room temperature between 20-25°C were maintained. The treatment was shown to remove 95 percent of the soluble BOD and essentially all iron. 90 percent of the original nitrogen was converted to free ammonia.

5.1.2.2 Anaerobic Filter

Anaerobic filters have received attention with the recent advances in anaerobic treatment tecniques. New reactors like upflow anaerobic sludge blanker have been succesfully used to treat high strength leachates.

Advantages of upflow anaerobic sludge blanket reactors include high loading capacity, low sludge production, small plant size requirement compared to that of anaerobic systems and low nutrient requirements. Although the start up of the process has been identified as a high cost investment, the savings from excess biogas recovery has been believed to counter balance this advantage.

By a study performed in 1981[46], on an upflow anaerobic sludge blanket, efficiences in COD removal ranged from 70 to 92 percent at 20° and 30°C, respectively.

Investigations performed on batch laboratory tests showed that with methane granules in an upflow anaerobic sludge blanket, a good pretreatment of leachate was accomplished. These granules induced an immediate gas production without any need for an acclimatization period . Approximately 85 percent of organic carbon was removed and the gas composed of 80 percent methane and 20 percent of carbon dioxide.

COD removal rates obtained from laboratory and pilot-scale treatment plants could be presented as:

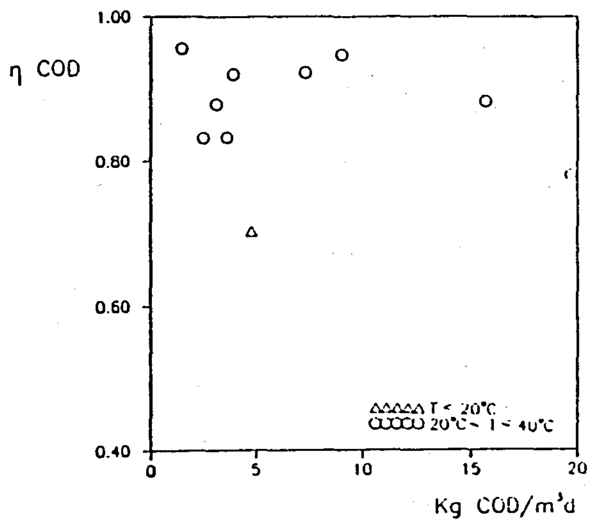


Figure 5.1.2.2.1 COD Removal Efficiencies Relative to Organic Load in Upflow Anaerobic Sludge Blanket Reactors[46]

5.2 Physical-Chemical Leachate Treatment Processes

Much of the emphasis on physical/chemical processes has been on the bench-scale investigations. These bench-scale investigations have included the application of chemical oxidation, precipitation, coagulation, ion exchange, adsorption, reverse osmosis and ionizing radiation. Full-scale investigations have been performed on chemical precipitation/coagulation, ammonia stripping and activated carbon adsorption.

Table 5.2.1 Bench-scale Research Performed on Leachate Treatment by Physical/Chemical Processes[40]

Process-oxidation, disinfection and precipitation
Process Description-oxidation and disinfection by diffusing ozone in batch test, precipitated by standard batch jar test
Research Objectives-effect of ozone oxidation and disinfection on organics, effect of lime additionon organics and metal removal
Leachate Source-lysimeter.

Process-coagulation, oxidation and precipitation
Process Description-standard batch jar test apparatus to simulate coagulation, precipitation and settling
Research Objectives-determine optimum dosage for organic, iron and color removal using $FeCl_3$ and alum as coagulants
Leachate Source-landfill.

Process-adsorption and ion exchange.
Process Description-continous upflow filters for both AC and glaucomite greensand
Research Objectives-effect of AC adsorption and greensand for metal removal
Leachate Source-landfill.

Process-disinfection
Process Description-batch reactor using NaOCl as disinfectant

Research Objectives-effect of NaOCl dosage and contact time on bacterial and viral inactivation
Leachate Source-lysimeter and landfill.

Process-ionizing radiation
Process Description-radioactive isotope of cobalt as gamma source for ionizing organic substrate
Research Objectives-effect of pH, aeration rate and dose rate on organic removal; determine MW distribution of ionized organics
Leachate Source-landfill.

5.2.1 Coagulation and Precipitation

Coagulation and precipitation method has been the most thoroughly studied physical/chemical treatment process for the removasl of organics and metals. Alum, lime, ferric chloride and polymers have been used as coagulants. However, none ofthe coagulants used have been succesful in removing more than 30 percent of the influent COD[53,40]. Alum has been referred as the most successful coagulant in dosages of 50 to 100mg/l and at a pH near 8.2, resulting in more than 25 percent COD removals. At a pH value of 7 with sodium sulfide and sodium hydroxide results revealed less than 10 percent COD removal at dosages greater than 2000mg/l.

For metal removal capabilties, ferric chloride, alum and lime have been tested. Iron and zinc were removed in the order of 90 percent. By using alum and ferric chloride at dosages less than 100mg/l succesfull removals were achieved. Sodium sulfide at a dosage of 1000mg/l indicated a 99 percent iron removal in the only one analysis conducted. Lime has shown to be capable of removing manganese, poatssium and sodium at high dosages.

Table 5.2.1.1 Heavy Metal and Alkali-Alkaline Earth Removal Data for Bench-scale Chemical Addition Processes[40]

Heavy Metals	Cd	Cr	Cu	Fe	Pb	Ni	Zn	Mn
Concentration Range, mg/l		.08-064	035-056	317-1000	0.10	73	0.4-30	0.7-25
Removal Range, %		30-53	21-96	0-99	20	4	0-90	28-99
Average Removal, %		40	40	84	20	4	86	66
Alkali&Alkaline Earth Metals	Ca	Mg	K	Na				
Concentration Range, mg/l	178	100-160	156-380	188				
Removal Range,%	0-6	0-60	Aug 27	43				
Average Removal, %	2	15	19	42				

In full scale experiments, lime was the only precipitant used for organic and metals removal. Lime doses up to 3600mg/l achieved about 50 to 70 percent organics removal. The removal of heavy metals especially in the case of Fe and Zn reached around 98 percent. The effects of using lime in heavy metal removals were similar to that for the bench-scale investigations. However, due to NH_3 stripping in full-scale studies, more than 40 percent COD removals were achieved.

Table 5.2.1.2 Performance and Design Parameters for Full-scale Physical/Chemical Leachate Treatment Facilities[40] (* percent removal)

Process	Precipitation	Coagulation	Chemical Addition	Precipitation	Adsorption
&		Adsorption		NH_3 Stripping	
Influent Quality					
BOD_5,mg/l	10000	100	/	11900	/
COD,mg/l	14000	/	2500	18500	11500
NH_3-N,mg/l	600	10	50	760	/
Cd,mg/l	0.05	/	/	0.08	/
Cr,mg/l	/	/	/	0.26	/
Cu,mg/l	/	0.56	/	0.40	/
Fe,mg/l	1000	20	750	333	330
Pb,mg/l	/	0.10	/	0.74	0.4
Ni,mg/l	/	/	/	1.76	/
Zn,mg/l	8	/	20	20	/
pH	6.0	7.6	6.0	6.7	5.6
Treatment					
Coagulation					
Flocculation					
Dosage,mg/l	1650	Alum		3000-Lime	
pH	/	/			
NH_3 Stripping	No	No	No	No	No
Effluent Quality					
BOD_5,mg/l (%R)*	/	28(72)	/	3930(67)	/
COD,mg/l (%R)	/	/	200(92)	6890(63)	/
NH_3-N,mg/l (%R)	/	1.7(83)	15(70)	6000(56)	/
Cd,mg/l (%R)	/	/	/	0.03(60)	/
Cr,mg/l (%R)	/	/	/	0.07(70)	/
Fe,mg/l (%R)	2.3(>99)	/	5(99)	3.2(99)	/
Pb,mg/l (%R)	/	0(>90)	/	0.17(77)	/
Ni,mg/l (%R)	/	/	/	0.61(65)	/
Zn,mg/l (%R)	0.03(>99)	/	2(90)	0.6(97)	/

5.2.2 Chemical Oxidation

Chemical oxidation of leachate has been studied using chlorine, sodium hypochlorite, potassium permanganate, ozone and calcium hypochlorite. The results indicated more succesfull organics removal than the chemical coagulation and precipitation processes. However, removal efficiences have been too low to be considered as a practical treatment method[40]. With dosages of 2000mg/l of NaOCl, Ca(OCl) and Cl_2 COD removals in the order of 10 to 30 percent were achieved. Retention times of three to four hours were necessary to obtain similar effects with ozone at lower dosages. With regard to the other oxidants, hypochlorites were superior to the other oxidants at exceedingly high dosages.

For the treatment of biologically treated leachates, only NaOCl has been tested. the application of this oxidant to a biological process effluent resulted in a reduction of COD in the order of 20 to 70 percent. The best removal (69 percent) was achieved at the lowest dosage (1600 mg/l). Higher pH values around 9.5 indicated better results.

Ozone treatment was succesfull in removing 82 to 99 percent of iron, copper and zinc though nickel was not removed. With dosages of 800 to 1000 mg/l chlorine was also succesful in remaining iron with 99 percent or higher efficiencies.

5.2.3 Chemical Disinfection

Ozone and sodium hypochlorite have been utilized to evaluate their performance for disinfection to raw and biologically treated leachate. Ozone treatment at 100mg/l decreased the bacterial density of raw leachate. Higher ozone dosages were necesaary for high strength raw leachates. The investigations using NaOCl were conducted to biologically treated and diluted leachate[40,4]. The effects of NaOCl dosage were studied for bacterial and viral inactivation of a leachate seeded with E.coli and T-4 coliphage. Dosages of NaOCl between 5 to 55 mg/l were used in a time interval of 2 to 60 minutes. The bacterial inactivation for NaOCl dosage of 1 to 20 mg/l at a contact time of 30 minutes. For viral inactivation, 48mg/l NaOCl at a constant time of 60 minutes were used to obtain 99 percent inactivation. It was estimated that higher NaOCl dosage and longer contact time was required for viral inactivation than bacterial inactivation.

5.2.4 Ionizing Radiation

For the treatment of landfill leachate, ionizing radiation has been performed on a medium-strength leachate with a TOC value of 2000mg/l. Radiation with a 5-kilocurie Co60 source, with an average dose of 0.6mrad/hr was given. Maximum TOC removal of 75 percent was obtained at pH 4 and a radiation of 20mrad/hr. At lower radiation doses, increased TOC removals were observed due to increases in aeration. The high molecular weight compounds were converted to low molecular weight compounds. As a result of obtaining low molecular weight compounds, ionic radiation has been viewed to hold promise as a pre-treatment method in the biological removal of organic materials[4,40].

5.2.5 Ion Exchange

For polishing biologically treated leachates, ion exchange resins (anionic-mixed) hacve been used. Ion exchange mechanism in both batch and continuous processes reduced 10 to 70 percent of organic constituents[40,4,48].

Table 5.2.5.1 Ion Exhange Performance Using Effluents from Aerated Lagoon and Activated Sludge Leachate Treatment Systems[40]

Process	Leachate Type	pH	Influent Concentation	Removal	Comme
			COD,mg/l	COD,%	
Anion Exchange	AL	8.8	500	6-59	Contin
Anion Exchange	AL	6.2	500	48	Contin
Anion Exchange	AS	5.0-7.7	180	36-68	Batch
Anion&Cation Exchange	AS	7.3	185	10	Batch

Cationic exchange has been studied using glaucnitic greensand(GG) for the removal of metals. Flow rates of 0.1 and 1.0l/min were used in a continous flow process. Low flow rates resulted in better removal of copper, lead and nickel over iron and zinc(see Table 5.2.5.2). Potassium and sodium were poorly removed. Also chromium, manganese, calcium and magnesium were 'fairly removed' at lower flow rates.

Table 5.2.5.2 Glauconitic Greensand Performance for the Removal of Metals from Leachate[40]

Parameter	Influent Concentration,mg/l		Removal,%	
	0.1l/min	1.0l/min	0.1l/min	1.0l/min
Cd	.006	.08	83	96
Ca	129	181	63	22
Cr	.03	.13	66	/
Cu	.38	.28	99	14
Fe	8.1	14	86	3
Pb	.13	.18	99	13
Mg	62	164	67	26
Mn	4.1	6.1	88	48
Ni	.07	.21	96	14
K	122	364	39	62
Na	275	585	36	0
Zn	.49	.78	67	20
pH	7.7-6.3	7.5-6.6		

5.2.6 Adsorption

Adsorption of organics and metals from leachates has been investigated through the implementation of activated carbon and peat. Much of the attention has been focused on granular and powdered carbons in batch and continous processes.

Activated carbon, in continous flow processes was capable for the reduction of 30 to 70 percent of organics at retention times varying from one to 15 minutes. the results indicate that chemically treated leachates were removed in lower efficiencies than biologically treated leachates.

Table 5.2.6.1 Freundlich Isotherm Parameters for Bench-scale activated Carbon Adsorption of Raw Leachate and Treated Leachate[40]

Leachate Type	COD/TOC	Co,mg/l-COD	X/M, mgCODorTOC/mgAC
Raw	/	5000	2.5
Biologically Treated	2.4-3.8	184-830	.261-.54
Chemically Treated	3.3-3.7	508-2990	.20-.80
Biologically&Chemically Treated	3.0-3.7	192-344	.15-.66

In the evaluation of the adsoption effect, the adsorption capacity (X/M) were utilized. Accordingly, X/M represents the ration between the absorbed COD and the absorbing activated carbon. Thus, the adsorptive capacity is a function of the effluent COD values. It has been determined that the dosage of activated carbon increase with an increasing slope of the isotherms.

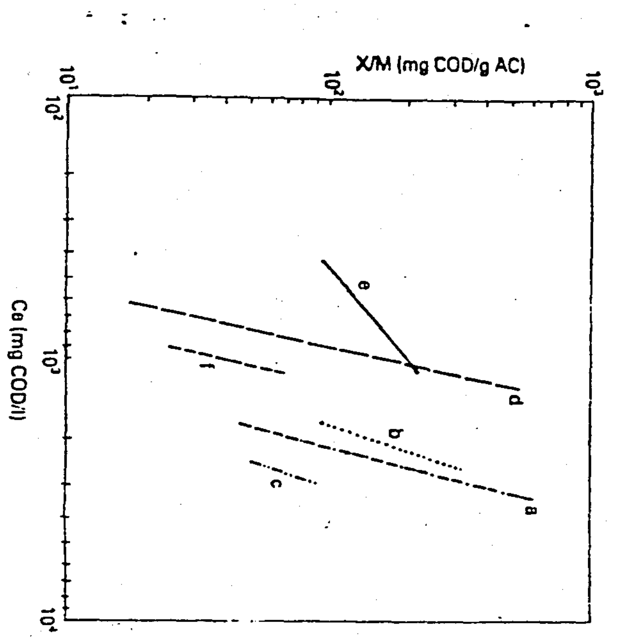


Figure 5.2.6.1 Activated Carbon Adsorption Isotherms for Different Substances[54]
a-leachate
b-after biological treatment with different AC dosage
c-raw leachate diluted 1:10

In conclusion, activated carbon adsorption has been determined to be not an effective process for volatile fatty acids and for very large molecules. The adsorption capacity may decrease if large molecules could block the pores. Hence, the best adsorption could be observed after biological treatment (elimination of volatiole fatty acids) and flocculation (reduction of large molecules).

Peat adsoprtion investigations were conducted in continous upflow columns filled with dried peat. At pH 7.1 to 7.8, the peat columns were capable of removing 86

Peat adsoprtion investigations were conducted in continous upflow columns filled with dried peat. At pH 7.1 to 7.8, the peat columns were capable of removing 86 percent of COD, 95 percent NH3-N and over 90 percent removing metals studied (except lead).

Table 5.2.6.2 The Performance of Peat for Adsoprtion of Organics and Metals from Leachate[40]

Parameter	Leachate Concentration	Removal
	(mg/l)	(%)
pH	4.8-7.1	4.8-7.1
COD	830-830	66-66
TKN	/-465	/-95
Ca	254-174	66-92
Fe	27-22	82-99
Pb	.03-.06	98-73
Mg	106-126	55-96
Mn	.52-.61	67-92
K	580-126	71-96
Na	1400-780	70-95
Zn	.43-.60	47-90

Table 5.2.6.3 Heavy Metal and Alkali-Alkaline Earth Metal Removal Data with Activated arbon Adsorption and Resin Ion Exchange of Leachate[40]

Heavy Metals	Cd	Cr	Fe	Pb	Mn	Ni	Zn	Cu
Concentration Range,mg/l	.03-.08	.07-.13	14-66	.18-.22	6.1-25	.13-60	.7-60	.24-
Removal Range,%	27-98	0	10-97	22-33	21-87	0-37	0-99	0-14
Alkali&Alkaline Earth Metals	Ca	Mg	K	Na				
Concentration Range,mg/l	15-181	15-164	63-380	200-585				
Removal Range,%	0-95	0-99	0-95	0-99				

5.2.7 Reverse Osmosis

Reverse osmosis has been one of the new developments in leachate treatment. It has received attention as both a raw leachate treatment step and a final polishing step (using biological, activated carbon or anion exchange treatment process) effluents. However, raw leachate treatment has led to membrane fouling due to solids colloidal particles and iron hydroxide[54].

Table 5.2.7.1 Reverse Osmosis Performance for the Removal of COD from Raw and Biologically Treated Landfill Leachates[40]

Process:	Leachate Type	pH	Influent COD,mg/l	COD Removal,%
Reverse	Raw	5.5	13000-18500	85-98
Osmosis	Raw	5.5	13000-18500	98-99
	AL	8.8	214	95
	AC	8.8	48	86
	IX	5.5	119-143	94-97

AS: Activated Sludge Effluent
AL:Aerated Lagoon Effluent
AC:Activated Carbon Effluent
IX:Anion Exchange Effluent

5.2.8 Ammonia Stripping

Ammonia stripping has been undertaken at only one landfill site. A pre-treatment of leachate was done with lime in order to raise the pH value to 10. Afterwards, air stripping was practiced. the detention time was taken to be 11 to 12 days at alarge lagoon. ammonia nitrogen removal was observed to be between 48 to 54 percent. the relatively poor stripping efficiency has been believed to be due to operational difficulties faced[40].

5.2.9 Vaporization

The vaporization of leachate has been discussed for some time in Germany as a process where leachate is separated into a clean water stream and a solid phase with the polluting materials. However, the separation is believed not to be an accurate one.

Table 5.2.9.1 Results of the Different Vaporization Experiments on Full and Pilot-scale Plants[54]

Parameter(mg/l)	1.In	1.Out	2.In*	2.Out*	3.In	3.Out	4.In	4.Ou
COD	4200-5510	227-508	4000-8160	800-22	1150-3000	45-460	5300-32600	500-
NH_4-N	475-1400	1-219	2174-1310	1242-20	220-1800	160-1300	1200-3200	480-

*Vaporization combined with stripping

5.2.10 Combinations

According to an investigation[55], the type and the number of facilities associated with leachate treatment in Germany is as follows:

Table 5.2.10.1 The Combined Leachate Treatment Facilities in Germany

Type	*Number*
2step reverse osmosis	11
1step biological treatment	9
2step biological treatment	4
Biological/precipitation/coagulation	1
Biological/ozone radiation	3
Biological/reverse osmosis/vaporization	1
Biological/2step reverse osmosis	1
2step reverse osmosis/vaporization	1
3step reverse osmosis/4step vaporization/	1
Biological/2step reverse osmosis/2step	1
Biological/H_2O_2UV	1

6. LEACHATE RECIRCULATION

Leachate recirculation is distiquished from the other treatment processes due to its in situ applicability, leachate is collected and recycled back on to or into a landfill under primarily an anaerobic treatment mode. Aerobic conditions prevail at the beginning and in the formation of humic substances during the final phases of landfill stabilization and may be important with regard to organic conversion and the possible re-mobilization of heavy metals, respectively. It has been reported that the practice of recycling leachate serves to improve the homogeneity of the biochemical environment needed for waste degradation and thereby, shorten the time required for waste stabilization by as high as 80 to 90 percent. Moreover, the cost of leachate recycle treatment currently suggests that it may be as much 25 percent of the costs of corresponding separate treatments.

A number of pilot scale investigations on leachate recycle process have been performed. The test cells used and research objectives are presented below [40]. The use of recycle has led to the acceleration of the anaerobic waste degradation and maximization of the rate, consistency and quality of gases produced.

TestUnit-3 m high column lysimeter(1)
Process Description-Six columns with different waste mixtures, organic and inorganic wastes
Research Objectives-Determine thr treatability of leachate from pulp and paper mill wastes through recycle-Evaluate organic and metal removal and gas production
Leachate Source-Pulp and paper mill waste.

Test Unit-15 m square by 3 m high test cell(2)
Process Description-Five test cells, control(no recycle), high initial moisture content, continous flow through of water, leachate recycle and biological sludge seeding with high initial moisture
Research Objectives-Determine the feasibility of leachate recycle for refuse stabilization, effect of once through moisture, effect of biological sludge seeding

Leachate Source-Municipal solid waste.

Test Unit-3m high column lysimeter(3)
Process Description-Four columns, control(no recycle), recycle, recycle with pH control at neutral pH, and recycle with pH control at neutral pH with biological sludge seeding
Research Objectives-Effect of pH control and biological sludge seeding on organic stabilization of waste
Leachate Source-Municipal solid waste.

Test Unit-5m square by 3m high test cell(4)
Process Description-Two cells; one sealed to prevent evaparation, other open to atmosphere to allow evaporation. Bothe received equivalent amount of water from rainfall
Research Objectives-Effect of evaporation on refuse stability by recycle. Evaluate organic removal and gas production
Leachate Source-Municipal solid waste.

Test Unit-3m high column lysimeter(5)
Process Description-Four columns; control(municipal solid waste only) and three with different quantities of placing wastes mixed with municipal solid waste
Research Objectives-Determine removal mechanisms of metal ions by studying chemical activity and chemical complexation
Leachate Source-Municipal solid waste with metal plating wastes.

Test Unit-1.8m high, with a 0.9m diameter test cells(6)
Process Description-Sixteen cells; combintaion of recycle, buffers, nutrient addtions to leachates
Research Objectives-Determine the effect of moisture, recycle, pH and nutrients on gas production and leachate quality
Leachate Source-Municipal solid waste.

Test Unit-1.5m by 1.5m square test cells(7)
Process Description-Four cells; recycle of leachate plus annual rainfall, no recycle, recycle of half the annual rainfall and presaturation followed by recycle of half the annual rainfall

Research Objectives-Detrmine the effects of moisture content and leachate recycle on gas production and leachate quality
Leachate Source-Municipal solid waste.

Test Unit-600m³ test fields, 15mX10mX4m(8)
Process Description-Three fields filled with compacted wastes, one sealed against evaporation and recycled, one with recycle, one without recycle
Research Objectives-Determine the effects of moisture content and leachate recycle on gas production and leachate quality
Leachate Source-Municipal solid waste.

Test Unit-1.6m deep by 5m² area test cells(9)
Process Description-Three test cells; simulated annual rainfall applied to each; one with leachate recycle, one with recycle of aerated leachate and one without leachatr recycle
Research Objectives-Determine the effects of leachate recycle on gas production and leachate quality, effects of leachate aeration and phophorus addition in situ biodegradation.
Leachate Source-Municipal Solid waste.

Although recycle and buffer additon considerably shortened the stabilization period, the effluent concentrations were similar in the comparative cells without recycle. The effluents observed also revealed a similar character to those obtained from anaerobic treatment processes, e.g., BOD,˜ 100 mg/l; COD,˜ 300mg/l to 500mg/l, TKN,˜ 100 to 300 mg/l, and Fe,˜540 mg/l[40].

It has been summarized that in general, recycling of leachate promoted the development of in situ biological, physical and chemical mechanism responsible for waste stabilization and/or leachate treatment. Residual BOD_5 and COD concentrations resulted through biological assimilation in the order of 30 to 500 mg/l and 70 to 800 mg/l, respectively.

Table 6.1 Summary of Leachate Character and Gas Results for the Pilot-Scale Leachate Recycle Studies[40]

(2)*	BOD5,mg/l	COD,mg/l	BOD5/COD	Fe,mg/l	Zn,mg/l	pH	CH4,%
(2)	40000	50000	0.8	1200	50	5.5	1
(2)	30000	50000	0.6	300	70	5.5	20
(2)	3000	5000	0.6	200	1	6.2	65
(2)	400	1500	0.3	50	1	6.5	65
(2)	30000	50000	0.6	800	80	5.5	20
(3)	2000	3500	0.6	450	15	5.8	/
(3)	30	70	0.4	4	0.2	6.8	/
(3)	35	240	0.15	3	0.4	7.0	65
(3)	40	170	0.2	9	0.5	7.0	65
(4)	90	350	0.3	39	0.2	6.7	/
(4)	70	300	0.2	29	0.4	6.6	55
(6)	/	120000	/	/	0.5	6.2	50
(6)	/	1600	/	/	0.2	5.7	65
(6)	/	36000	/	/	/	6.3	65
(6)	/	26000	/	/	/	6.8	70
(7)	36000	61290	0.6	/	/	6.3	55
(7)	38300	62690	0.6	/	/	6.5	45
(7)	39700	66310	0.5	7	/	6.3	45
(7)	35000	58330	0.6	7	/	6.3	50
(9)	/	/	/	/	/	/	/
(9)	33	600		42	0.14	7.1	/
(9)	20	618		50	0.19	7.0	/

*Reference numbers

As the degradation process progressed, the organic substrates in leachatesbecame more refractory, as indicated by low BOD_5/COD (0.15 to 0.4) ratios. As the volatile fatty acids decreased in number, the pH of recycled leachate eventually increased to a range of 6.5 to 7.0.

In the recycled leachate, the residual concentrations of heavy metals, especially of Fe and Zn were commonly reported to be 40 and 4 mg/l, respectively. The efficent removal of heavy metals was referred to chemical complexation by organic /inorganic ligands which were found to be present in leachate in a great extent and were able to form metal ligand precipitates, sulfides were also effective in the precipitation of heavy metals.

Leachate recycle resulted in a gas yield of 7.1 m^3/1000 kg dry waste with 55 to 65 percent CH_4 and 35 to 45 percent CO_2 gas compositions.

In the United States, leachate recycle as an in situ treatment have not been tested in full scale experiments. However, a demonstration project has been conducted in California (1983). Six field cells were constructed to evaluate the effects of adequate moisture content, pH buffering and nutrient availability through leachate recycle regime.

The average volume of each test cell was 10500m^3 with a refuse mass of 4825 metric tons. These cells were operated under variable conditions of water content, seed sludge, nutrients, and buffer. Only one of the six test cells were operated using leachate recirculation. The results indicated that cells to which moisture and buffer were applied , produced higher quantities of gas than control cells. Despite sporadic recirculation, the highest quantities of gas was produced from the recycle cells. Conclusions regarding stabilization patterns were difficult to establish.

A study conducted on leachate spraying on the top of refuse using a sprinkler system. Preliminary results indicated that the COD value of leachate is decreasing at a higher rate than in a non-recirculated control area. Because the landfill was not covered, no data regarding gas production was available.

Table 6.2 Summary of Available Information Concerning the Application of Leachate Recycle at Full-Scale Landfills in Germany[40]

Landfill	Stapelfeld	Flechum	Dorpen	Hattorf	Blankenhagen	Nauroth	Reinstet
Surface Area,ha	8	2.46	5.5	9.4	18	7	5.4
Refuse Received,m^3/a	/	17000	70000	100000	180000	120000	/
Leachate Produced,m^3/a	/	1000	570	1800	7630	/	/
LeachateQuality							
BOD_5,mg/l	820	/	/	390	1400	200	/
COD,mg/l	1680	/	/	930	2900	/	/
pH	7.4	/	/	6.95	7.65	7.4	/
Annual Precipitation,mm	750	810	700	/	650	1100	800

Several full scale landfills in Germany were operated (see Table 6.2) under leachate recycle regime. At one of these landfills, a two stage approach was undertaken. Leachate was removed from a newer landfill section was recirculated in an older stabilized section. Eight months of operation under this system indicated that the two stage approavh may be utilized to obtain consistent quantities of methane gas from a full scale landfill.

Table 6.3 Organic Character of Leachates Removed from a Full-Scale Two Stage Recirculation Process In Germany[40] (* data not given)

Old Field	BOD,mg/l	COD,mg/l
(Methane Stage)	60	1473
	64	1278
	59	1370
	/*	/
	63	1561
	67	1409
	/	/
	60	1273
New Field	**BOD,mg/l**	**COD,mg/l**
(Acidic Stage)	/	/
	1310	5903
	5320	10390
	2660	5559
	6000	16725
	/	/
	6340	11200
	11970	19300

In order to further understand the effects of leachate recycling, the following research project was undertaken by Heike Doedens and Cord Landwehr [32]. In the experiment, four air tight, temperature controlled, stainless steel test cells were constructed. The diameter of each test cell was 1.5m and 1.35 m waste was installed. The temperature was regulated to be constant at 35°C. The original water content was in the range of 24 to 31 percent under a packing density of 0.77-0.87t/m³.

Test cell1 was set to 660mm/year rainwater such that all leachate that was released was fed back into the cell. Test cell 2 was used as a control cell and received only 330 mm/year rainwater. In the third test cell, rainwater (330mm/year) and leachate were initially irrigated where the proportion of leachate was 40 percent of the totalamount. Test cell 4 was saturated with leachate at the beginning.

Intensive gas production was observed in test cell 3 shortly after the volume ratio of CH_4 to CO_2 was >1. COD values less than 20000mg/l showed a decrease in gas production. A further decrease to below 5000mg/l indicated insignificant gas production. During the recycling process, no increase in the concentration of chlorides was noted.

As a result of through soaking of the test cell at the start of the experiment, highest loads were emitted from the test cell4. Lowest emissions came from test cell3. Even though, leachate recirculation was practiced in test cell 1, the organic load was twice as high of test cell 3 which received half the amount of simulated rainfall.

Under comparable conditions, it was found out that the load emitted from test cell 2 (with no recycling) was two to three times as hgih as cell3 (without recycling).

Table 6- 1.1 Emissions from the Test Cells (without loads in recycle)

Test Cell	Loads Withdrawn during Test Period						
	kg COD	kgBOD	kgCl	gZn	gPb	mgCd	Gas Production, m^3/dry matter
1	5	1.5	1.4	1.9	0.04	3	50
2	7.8	3	1.4	1.2	0.38	1.5	114
3	2.7	1	0.9	0.5	0.023	0.55	111
4	20	10.3	3.2	3.4	0.59	11.7	105

The carbon turnover which has been regarded as a measure of stabilization in terms of gas production under the effect of recirculation did not provide any indication. Test cell2, had the highest gas production in the absence of recirculation. Hence, it was concluded that carbon recirculation lowers the quantity and quality of organics in the leachate. However, due to recirculation, low emissions of carbon in gas and leachate denoted a slower stabilization of waste.

6.2 Organic Carbon Content Stabilization through Landfill Leachate Recirculation

The purpose of this investigation was to study the feasibility of a leachate recirculation system in providing leachate treatment and to examine the effect of pH and nutrient control on both shredded and unshredded refuse [33].

Four simulated landfills (diameter 1m, height 3.7m) were constructed in such a manner that leachate came in contact only with polyvinyl chloride pipe or epoxy coated steel parts. The leachate was collected in a sump and was pumped back through a distributor. In each cell, the dry refuse was compacted to a height of 2.4m with a density of 237kg/m^3.

The four simulated landfills were operated as follows:

Cell1-Control cell, only simulated rainfall added, no leachate recirculation, unshredded refuse.
Cell2-Water recirculated to achieve approximately 70 percent moisture content, leachate pH control(pH=7), shredded refuse.
Cell3-Water recirculated to achieve approximately 70 percent moisture content, leachate pH control (pH=7), unshredde refuse.
Cell4-Water recirculated to achieve approximately 70 percent moisture content, leachate pH control (pH=7) and nutrient control, unshredded refuse.

To accelaerate the production of leachate in cells2,3 and 4 with leachate recirculation, 750 liters of tap water was added. After the initial production of leachate, only simulated rainfall equivalent to daily rainfall total was added to the four cells.

Leachate generated from the test cells were analyzed for 27 chemical/physical and biological constituents. The volatile acids, biochemical oxygen demand (BOD), chemical oxygen demand (COD) and total organic carbon (TOC) were particular of interest in the evaluation of the effects of leachate recirculation on landfill stabilization. The data collected through a 514 day period demonstrated that leachate recycle reduced the concentrations of readily biodegradable constituents emitted in the leachate considerably.

The following specific conclusiosn were drawn from the data obtained in the study.
** The recirculation of leachate in a simulated landfill is effective in establishing a rapid active anaerobic biological population within thefill.
** The characteristics of shredding and nutrient control with regard to leachate recirculation are not necessary nor of any advantage for the enhancement of organic carbon content stabilization.
** The recirculation of leachate with ph control leads to an increased rate of biological stabilization of the organic components of refuse.
** The ultimate effect of leachate recirculation is the reduction and control of leachate pollution.

6.3 Leachate Recycle as a Management Option

The present study reviewed and summarized the nature of leachate and gas production at landfills receiving municipal solid wastes and extended these concepts to the consideration of leachate recycle as a management option [28].

In the study, two lined cells each three meter square by 5.2 mdeep were employed. Shredded municipal solid wastes of inital moisture content of 33.5 percent were compacted to an as palced density of 319kg/m^3. The cell which was referred as the open cell was left open at the surface to receive incident rainfall whereas the other cell was completely enclosed to prevent gas collection and measurement. Moisture as tap water was added to the sealed cell in equivalent amounts of incident rainfall.

The experimental period was a total of 699 days. During this time , the cells reached field capacity, leachate was collected and recycled and anayses were carried out both on leachate and gas samples. Initially in both of the cells some leaching was observed. However, leachate did not appear regularly until 399 days after the waste was placed.

The data obtained was evaluated at the time when continous leaching and recycle was commenced to those at and after the onset of gas production. During initial (weekly) recycling, the obtained data revealed a somewhat erratic pattern especially for the sealed cell where the even distribution of simualted rainfall was not completely achieved. However, with the onset of daily recycle, these variations diminished.

The accumulation of volatile acids lowered the pH values until the organic acids were utilized and were converted to methane and carbon dioxide. this gas production indicated the behavior of normal bicyrbonate system and a neutral pH. A decrease both in conductivity and total alkalinity were observed at such a pH condition.

Gas production reached as high as 639 liters per day with a normalized methane composition of 60 percent. Furthermore, increased gas production was observed in the period of most rapid removal of organic pollutants.

It was concluded that for on site applications leachate recycle promises both with respect to leachate quality and quantity. For the ultimate leachate disposal, the leachate recycle provides assurances with respect to degree of stabilization and hence leads to the establishment of the residual leachate characteristics.

6.4 Codisposal of Municipal Refuse and Industrial Waste Sludge in Landfills

A two year old pilot investigation was conducted on the codisposal of heavy metal sludge with municipal refuse under leachate recycling practice. The effects of various metal loadings on the progress of waste stabilization was analyzed to assess assimualtive capacity [14].

Four simulated columns each having a diameter of 0.97m and a total heoght of 4.27m were constructed. The four columns were equipped with a leachate collection and management system. Column 1 was utilized as the control cell and received 400 kg of refuse. the rest of the columns (2,3 and 4) contained 33.6kg, 65.8kg and 135.2kg of metal plating sludge, respectively.

Table 6- 4.1 Characteristics of Metal Plating Sludge Codisposed with Municipal Refuse in Simulated Landfill Columns

Parameter	Analytical Results*
Moisture content, %	84.7
Zinc,mg/kg	317000
Chromium,mg/kg	21000
Nickel,mg/kg	400
Cadmium,mg/kg	13100

*Metals determined on a dry basis

Sufficient amount of tap water was added to attain the necessary field capacity. Thereafter, the occurence of rainfall and evaporation regulated the amount of moisture. Leachate was practiced generally at a frequency of one sump volume per week when leachate was obtained.

The operational features of the four landfill columns were as follows:

Table 6- 4.2 Operational Phases during Landfill Investigation

Operational Phase	Time,days	
A	0-200	Facile production of leachate and washout
B	200-380	Initial microbially mediated stabilization
B'	380-480	No leachate productiob or recycle
C	480-600	Post drought resumption of leachate production and stabilization
D	600-720	Terminal phase of leachate production and stabilization

Selected analytical methods were used in accordance with the stabilization patterns of the refuse. The chemical oxygen demand (COD) and total volatile acids (TVA) were performed to display the changes in organic strength. The heavy metals (Zn,Cd,Ni,Fe) were chosen to investigate the effects of heavy metal loadings in the leachate.

In the control column, COD and TVA were rapidly reduced in concentration, ewith the indication of a normal stabilization. Column 2 which contained the lowest metal sludge content exhibited a similar pattern in a rather delayed conversion. In contrast, COD and TVA analyses from columns 3 and 4 indicated a clear inhibition of the normal progress of refuse stabilization.

These varying results in COD and TVA concentrations suggested the interference of heavy metals with the normal progress of refuse stabilization. It has been also observed that this process of delay or inhibition occurred in a cyclic fashion with alternation periods oftoxicity and acckimation especially at higher metal contents. In the operational mode, this suggested that leachate with high metal contents would not be reduced in organic concentrations until the inhibitor was reduced below the toxic 'threshold.'

In most of the leachate samples, chromium which was among the prevalent metals in the sludge was found to be below detectable limits. The measured concentrations were below 0.1mg/l which suggested a high assimilative capacity within the landfill environment.

In contrast, leachate Fe prevailed in relative abundance arising from ferric oxide present in the sludge and also from metallic iron oxides resulting from corrosion of various refuse components.

With Cd and Zn, the effects of sulfide precipitation were easily observed. Both in columns 1and 2, inhibition did not result by heavy metal loadings. During the terminal phase however, increasing concentrations of these heavy metals were detected under high sludge contents.

Aromatic hydroxyl (ArOH) analyses were carried out in search of the apparent cause of increased mobility of the heavy metals during the later stages of stabilization.

During the intial phase of the experiment, relatively high release of ArOH in columns 1 and 2 were observed to drop to constant levels in the later stages of stabilization. In contrast, ArOH in the leachates of columns 3 and 4 remained at lower levels until the terminal phase where they became as high as the values of column1 and 2 during the intial phase. This divergence of ArOH concentrations was linked to the presence of heavy metals. The The interaction between himic like substances and heavy metals was observed to be reduced toxicity and increased mobilization of the heavy metals especially in the case of dissolved zinc and cadmium.

Therefore, it was established that landfill assimilative capacity for heavy metals involves mobilization of leaching (sulfide, carbonate or hydroxide precipitation), immobilization arising from reducing conditions, filtration and sorption oppurtunities during leachate recycle and re-mobilization through reaction with humic like substances.

7. CURRENT TECHNOLOGY ASSESSMENT

7.1 Leaching from Simulated Landfills

The aim of the study was to evaluate the chemical and pollutional caharacteristics of leachate from different heights of refuse columns containing similar refuse of the same age and operating under similar conditions [56].

Three simulated landfills were construcred. These were fills consisting of concrete cylinders 0.9m in diameter and were placed over a 15.2 cm thick concrete base. the heights of these cylinders (1,2,3) were 1.2,2.4, 3.6 , respectively. An exterior valve was placed in each to carry away the drainage water. Refuse sampling ports near the bottom and at 1.2 m intervals were prepared. The cylinders were coverd with thick plywood to exclude precipitation. Water as equivalent precipitation was applied at each 2 week intervals. The fill materials which were municipal refuse were compacted in the zones of refuse collection ports. Measurement of temperature indicated a maximum value at the start or a maximum attained with a short period. However, after the initial temperature rise, the refuse temperature dropped gradually to daily air temperatures.

Table 7.1.1 The Characteristics of the Simulated Landfills

Data	Cylinder1	Cylinder2	Cylinder3
Refuse quantity(lb)	472	1078	1829
Average Moisture Content of Refuse(%)	47.1	42.8	49.9
Depth of Refuse(ft)	2.6	6.4	10.1
Total Depth of Simulated Landfills(ft)	3.6	7.6	11.6
Volume of Cylinders(cuyd)	0.7	1.7	2.7

Table 7.1.2 Summary of Maximum Concentrations, Maximum Concentrations per foot Depth and Percent Reduction in Concentration at the end of the Test Period.

Substances Extracted	A 1	A 2	A 3	B 1	B 2	B 3	C 1	C 2	C 3
BOD_5	14760	26940	33360	5630	4630	3290	54.8	36.3	23.9
pH				/	/	/	/	/	/
Alkalinity($CACO_3$)	10630	16200	20850	4050	2540	2060	58.0	20.8	22.3
Ammonia Nitrogen	473	756	1106	181	119	109	52.9	21.3	25.1
Organic Nitrogen	288	664	1416	110	104	139	52.1	46.2	57.5
Total Iron	175	546	860	67	86	85	52.0	12.5	14.9
Magnesium	176	387	420	67	61	41	53.4	47.3	20.5
Potassium	1050	2535	3770	400	398	372	62.6	54.0	13.8
Sodium	584	1428	1439	223	224	142	71.2	61.9	39.5
Calcium	3030	4080	2790	1157	642	276	49.0	31.1	4.10
Chloride	951	2000	2310	363	314	228	76.4	62.1	36.6

As a conclusion, the following were summarized:

-Refuse beds with approxiamtely 45 percent absorbed nearly 13.5cm/m of water before any leaching occured.

-Under similar conditions of refuse material and age of the fill, deeper fills showed much lesser reductions in the concentrations of leached materials than shallower fills.

-The concentrations of pollutants from deeper fills decreased as the depth of fill increased.

Therefore, deeper fills pose less of a pollutional problem than do shallow fills which may leachate the contaminants in a shorter period, therby, excede the dilutional capacity.

7.2 Laboratory Study of the Behavior of a Sanitary Landfill

In this paper, the behavior of a simulated sanitary landfill, in the form of a laboratory lysimeter under a humid climate was observed. The laboratory lysimeter simulated the behavior of the vertical center of a sanitary landfill in question [57].

The lysimeter was constructed of 0.64 low carbon steel plate. Fiberglass lining was utilized in the interior walls to protect the steel walls against corrosion by the products of decomposition. The height of the lysimeter was 3.96m with a $1.83m^2$ cross-section.

The vertical walls were completely insulated by urethane board of fiberglass insulation, dead air pockets and heating tapes. Air temperature was maintained at the monthly average +-1°C.

Gas samples were obtained at five different locations in the tank at depths of 0.9,1.5,2.1 and 2.7m below the steel cover plate. Three times a week the gas samples were collected and analyzed.

The refuse placed in the lysimeter consisted of a typical domestic refuse(see Table7.2.1). The compaction of the refuse was conducted outside the lysimeter resulting in a 2:1 compaction ratio.

Table 7.2.1 The Composition of Refuse Used

Constituent	Dry Weight Basis,%
Corrugated paper boxes	23.38
Newspapers	9.40
Magazine paper	6.80
Brown paper	5.57
Mail	2.75
Paper food cartoons	2.06
Tissue paper	1.98
Plastic coated paper	0.76
Wax cartoons	0.70
Vegetable food waste	2.29
Citrus rinds&seeds	1.53
Meat scraps,cooked	2.29
Wood	2.29
Ripe tree leaves	2.29
Flower garden plants lawns	1.53
Lawns,grass,green	1.53
Evergreens	1.53
Plastics	0.76
Rags	0.76
Leather goods	0.38
Rubber composition	0.38
Paints and oils	0.76
Vacuum cleaner catch	0.76
Dirt	1.53
Metals	6.86
Glass,ceramic,ash	7.73
Adjusted moisture	9.05

Leachate was collected from the lysimeter on a weekly basis. The following analyses were performed when sufficient amount of leachate was obtained.

Table 7.2.2 Liquid and Gas Sample Analyses Performed

Liquid Sample Analyses	Gas Sample Analysis
BOD_5	CO_2
COD	CO
Chloride	H_2S
Copper	CH_4
Dissolved Oxygen	N_2
Hardness	O_2
Iron	
Nickel	
Nitrate	
Nitrogen	
pH	
phophate	
Sodium	
Sulfate	
Suspended Solids	
Total Solids	

During the test period, significant variations in pH were observed. In most of the time, the leachate was acidic and pH varied between 5.0 and 6.5 except for low and high peaks. the erratic pH periods were evaluated to correspond to periods of low leachate production. The periods of relatively consistent pH corresponded to periods of large leachate production.

During the early days of the experiment, the COD values reached 50000mg/l. After this initial peak, COD dropped to a minium of 1000mg/l and thereafter increased to 22000mg/l. For most part of the test period, it remained around 20000mg/l.

The maximum iron concentration (both ferric and ferrous) exceeded 1600mg/l. During the periods when leachate pH was greater than 5.5, the iron concentrations were low.

The first appearance of methane gas corresponded to the period of rapid temperature decline. It did not appear until 100 days had elapsed. Thereafter, the amount of methane generated increased and reached approximately 30 percent in the lower sampling ports.

7.3 Leaching of Pollutants from Sanitary Landfill Models

In this study, the concentrations and quantities of pollutants, leaching out of solid waste was evaluated together with the extent of leachate production and its long term effects on a groundwater pollution [58].

The typical observed residential refuse in Israel was filled into polyvinyl chloride columns of 250 cm height and an internal diameter of 20 cm. The internal insulation was provided by a three centimeter layer polyurethane foam. Leachate was drained through a tube which was connected to the bottom of the column.

The water application was done in a low and a high level. The low level was equivalent to 600mm/a such that 2/3 of it was applied during January and February and 1/3 during November, December and March of the experimental year. During the irrigation period, water was applied every two weeks. The high level of water application ws 100mm/a. 600mm was applied during the winter, distributed in the same manner as in the low level application and 500mm was distributed twice a month during the summer.

Table 7.3.1 The Components of the Refuse Used

Constituents	Dry Weight Basis, %
Food residues	54.7
Paper&cloth	30.3
Plastics	4.4
Glass	3.0
Metal cans	3.1
Leaves, wood	3.1

The following variations were tested in the columns:

Table 7.3.2 Variables Tested in a Model Landfill

Refuse Type	Filling Season	Compaction, g/cm^3	Water Application, ml/cm^2
Raw	Winter	0.62	85
Shredded	Winter	0.79	85
Shredded	Summer	0.73	85
Composted	Winter	0.50	85

The fastest decay was observed with the least dense columns filled during the winter. Correspondingly, the slowest decay was associated with the most compacted columns filled with during the winter. The decay of columns filled during the summer was observed to be slow.

The results indicated that when the refuse was not densely packed and not shredded, the column was better aerated and less acidic. The decrease in the concentrations of soluble components were faster. When the refuse was compacted or shredded, anaerobic conditons were observed.

Refuse filled during the summer, in the temperature range of 30° to 333°C stabilized faster than refuse filled during 10° to 15°C in winter. The composted refuse showed a similar behavior to that of shredded refuse.

7.4 Biological Treatabilty of Landfill Leachate

The purpose of this study was to evaluate a variety of leachate treatment schemes, developing design and operating criteria for most promising ones [51].

Preliminary laboratory scale tests results indicated the efficacy of biological processes in leachate treatment. Processes evaluated include anaerobic and aerobic biological treatment of leachate, aerobic treatment of combinations of leachate and domestic wastewaster in a simulated activated sludge treatment plant and aerobic polishing following an anaerobic treatment of leachate.

The characteristics of the leachates used in the study are summarized as:

Table 7.4.1 Leachate Characteristics as Utilized by Boyle and Ham

Batch	BOD(mg/l)	COD(mg/l)	pH
1	1550	2700	8.10
2	2900	5230*	6.75-7.45
3	8000	8640*	6.75-7.20
4	6940*	8300*	6.90-7.20
5+	8260*	11140*	5.90-6.40
6	6250	8370	7.40

*Filtered

+Diluted(1:1)

The anaerobic treatment or raw leachate provided more than 90 percent BOD reduction at detention times greater than 10 days at emperatures rannging from 23° to 30°C.

Anaerobic polishing of the anaerobic effluent produced BOD values acceptable with surface water discharge. Although, aerobic treatment of leachate proved also to be promising, foaming problems and poor solids liquid separation were the disadvantages observed.

Leachate combined at least five percent by volume with domestic wastewater in an extended aeration activation sludge plant volume provided satisfactory results. Leachate additions greater than five percent by volume resulted in increased oxygen uptake rates, poorer mixed liqour separation and substantial solids production.

7.5 Stability of Organic Matter in Landfill Leachates

Twelve representative leachate or heavily polluted groundwater samples were collected from solid waste lysimeters and from pilot/field scale solod waste landfills. The samples were collected anaerobically and were subjected to extensive organic analyses[15].

The soluble organics concentrated, separated and were analyzed by ultrafiltration, gelpermeation chromatography.

Samples from a recently installed waste lysimeters showed that the majority of the organics consisted of free volatile fatty acids. The second largest group was a fulvic like material of intermediate molecular weight, characterized by high carboxyl and aromatic hydroxyl group density. The high molecular weight compounds consisted of humic-carbohydrate like complex.

Analysis of samples collected from varying landfill ages revealed that the free volatile acids decreased with increasing age of the fill. Increasing stability was observed for carbohydrates, hydrolyzable amino acids and aromatic hydroxyl compounds with increasing age of the fill.

It was concluded that leachate from a recently generating fill is best suited for biological treatment processes, while organics in stabilized leachate are removed satisfactorily with physical-chemical processes.

7.6 A Convenient Parameter for Tracing Leachate from Sanitary Landfills

In this study, the cations sodium and potassium were used as tracers of leachate from a sanitary landfill area. Potassium proved to be a better indicator of leachate production due to its elevated concentration in the leachate produced by bacterial and chemical breakdown of the refuse [59]. The local hydrological/hydrgeochemical environment played an important role in the ratio of [K] in receiving waters downstream of the landfill to the [K] upstream.

As a conclusion, it was determined that [K] is a clear indicator of leachate pollution. However, when using K as a tracer, care should be taken to check for its possible contact with concrete, agricultural fertilizers etc. in addition to its production from leachate.

7.7 Containment and Treatment of the Love Canal Leachate

In this paper, the emerging problem solving efforts undertaken to identify, contain and treat leachate from the Southern Sector of Love Canal, Niagara Falls, New York, landfill site were discussed [60].

Table 7.7.1 The Characterization of Love Canal Leachate

Parameter	Concentration
pH	5.6
TOC,mg/l	4300
SOC,mg/l	4200
COD	11500
Oil&Grease,mg/l	90
Suspended Solids,mg/l	200
Dissolved Solids, mg/l	15700
Chloride,mg/l	9500
Sulfate,mg/l	240
Sulfide,mg/l	<0.1
Phosphate(total),mg/l as P	<0.1
Phosphate(inorganic),mg/l as P	<0.1
Sodium,mg/l	1000
Calcium,mg/l	2500
Iron,mg/l	330
Mercury,mg/l	<.0005
Lead,mg/l	0.4

Granular activated carbon studies were conducted to test the applicability of this technology to the Love Canal Leachate. The adsorption isotherms indicated that over 92 percent of the SOC (soluble organic carbon) could be removed in a properly designed carbon system. Dynamic column studies were utilized to confirm the carbon efficiency.

The collected leachate from the Love Canal was then treated with reactivated bituminous granular activated carbon. This carbon utilization method met all discharge standards including the propsed future pollution standards.

7.8 Trace Metals and Anaerobic Digestion of Leachate

Three bench-scale anaerobic digesters were operated to determine the efficiency of anaerobic treatment on leachate. The digesters were operated in the mesophilic range at 34°C. each was equipped with a mechanical mixer and a thermostatically controlled heater [52].

Table 7.8.1 Digester Performance of Various Feed Strengths

Unit	Detention Time(days)	Parameter	Batch1 (1-4 days)	Batch2 (5-25 days)	Batch3 (26-47 days)
1	5	BODeff,mg/l	2530	2165	1992
		BODrem,%	842	82.9	82.3
		CODeff,mg/l	/	/	8250
		CODrem,%	/	/	65
		Gas Production,m^2/kgBODrem	0.74	0.91	0.94
2	10	BODeff,mg/l	/	/	1005
		BODrem,%	/	/	91.1
		CODeff,mg/l	/	/	7234
		CODrem,%	/	/	68.9
		GasProduction,m^2/kgBODrem	/	/	0.91
3	20	BODeff,mg/l	594	415	541
		BODrem,%	96.3	96.7	96.9
		CODeff,mg/l	/	/	4875
		CODrem,%	/	/	79.3
		Gas Production,m/kgBODrem	0.71	0.87	0.89

The anaerobic digestion process was totally unaffected by the presence of heavy mateals and other toxicants in the leachate throughout the investigation of the digesters. Heavy metal removal by anaerobic digestion proved to be not as high as was with using aerobic biostabilization. However, greater than 85 percent for aluminum, barium, cadmium, mercury, nickel and zinc, 80 percent fro iron; 40 to 70 percent for chromium, copper, lead and manganese; 30 percent for calcium and less than 10 percent for magnesium, potassium and sodium.

7.9 Leaching of Cadmium from Pigmented Plastics in a Landfill Site

Leaching of cadmium from pigmented plastics disposed to landfill was investigated following concern over cadmium release to the environment [61].

The results of the experimental program indicated that cadmium pigmented plastic did not make a significant contribution to the release of cadmium in landfill leachates. It was observed that when the plastic used in the experiment was ground to a fine particle size, some leaching occurred but this was not likely to be encountered in practice.

7.10 Lysimeters for Simulating Sanitary Landfills

The various decomposing processes inside sanitary landfills were investigated by operating eight lysimeters. The lysimeters consisted of flexible walls which allowed settlements similar to that in nature [62]. Thermal insulation of the lysimeters were provided to protect the biological and chemical processes against outside temperatures. The inside diameter of each lysimeter was five meters and the total height reached to six meters. the amount of dumped refuse depending upon treatment and operation amounted to 60000-80000kg.

Table7.10.1 The Characteristics of the Lysimeters

Lysimeter	*Placing*	*Treatment*	*Material*
1	200m layer	Compaction	Refuse
	every 1.5 years		(without mixing)
	10 cm soil cover		
2	0.40m layer	Compaction	Refuse
	every 4 months		(mixing)
	no soil cover		
3	0.40m layer	Compaction	Refuse&sludge
	every 4 months		(mixing)
	no soil cover		
4	2.00m layer	Compaction	Refuse&sludge
	every 1.5 years		(mixing)
	no soil cover		
5	after 1.5 years	Aerobic	Refuse&sludge
	rottening sieved	Decomposition	(mixing)
	rest compacting		
6	after 1.5 years	Aerobic	Refuse&sludge
	rotttening	Decomposition	(mixing)
	compacting		
7	0.50m layer	Compaction	Refuse&sludge
	every 4 months		(without
	no soil cover		mixing)
8	2.00m layer	Compaction	Refuse&sludge
	every 1.5 years		(without
	10cmsoil cover		mixing)

It was found that the density of the compacted refuse differed with regard to the various types of operation.

Table 7.10.2 Density of the Compacted Refuse and Sludge

Lysimeter	Total Moist Mass	Density Required
	10^3kilograms	10^3kilograms/m^3
1.Refuse	57.9	0.78
2.Refuse wetted, mixed	58.0	0.81
3.Refuse &sludge mixed	83.6	1.90
4.Refuse&sludge mixed	93.5	1.80
5.Aerobic, sieved	41.7	2.20
6.Aerobic	77.0	1.50
7.Refuse&sludge ponds	74.1	0.90
8.Refuse&sludge ponds	66.0	0.84

It was also observed that the curves of accumulated outflow of the different lysimeters are similar to each other and that of commom landfilling. However, it was noted that the outflow occurs before the refuse has been saturated and that the rate of outflow may increase considerably after the operation of the landfill has come to an end.

7.11 Decomposition of Solid Waste in Test Lysimeters

In this investigation, eight test lysimeters containing 100 or 200 tons of refuse were monitored for a period of seven years for leachate and runoff rates, leachate and gas composition, refuse temperature and surface settlement [63].

Table 7.11.1 The Monitoring Procedures and the Performances of the Lysimeters

Cell 1	1.2m deep	Unprocessed refuse	Covered immediately	
Cell 2	1.2m deep	Shredded refuse	Covered immediately	
Cell 3	1.2m deep	Shredded refuse	Covered after 6 months	
Cell 4	1.2m deep	Shredded refuse	Not covered	High COD
Cell 5	1.2m deep	Unprocessed refuse	Covered with shredded refuse	High COD
Cell 6	1.2m deep	Unprocessed refuse	Not covered	Lowest %of leachate
Cell 7	2.4m deep	Shredded refuse	Not covered	High COD
Cell 8	2.4m deep	Unprocessed refuse	Covered immediately	Highest COD

Table 7.11.2 Summary of the Methane Content in the Cells

Cell1	Low CH_4
Cell 2	Highest CH_4
Cell 3	Highest CH_4
Cell 4	
Cell 5	Low CH_4
Cell 6	Low CH_4
Cell 7	
Cell 8	Low CH_4 than Cell1

-The shredding of the refuse increased the rate of decomposition
-The shredding led more quickly to methane concentration
-The application cover increased methane concentration
-The shredded refuse increased peak concentration of contaminants in the leachate.

7.12 Identification of Organic Compounds in Leachate from a Waste Tip

This paper describes the result of analyzing various organic compounds in the two main types of leachate [11].

The first leachate was collected at a nine year old part of a waste tip which was in the acidification stage. The organic contents of the leachate was high and was comprimised mainly of free volatile acids. Volatile amines and ethanol were observed to be present as 0.8 and 0.7 percents , respectively. The leachate was extracted with hexane and the extract was analyzed. In the extract alcohols, hydrocarbons, esters, terpenes and phthalates were identified. The high molecular weight compounds were found only in low concentrations.

Table 7.12.1 Chemical Characteristics of the Leachate Samples

	Acidification Stage (Leachate1)	Methane Fermentation Stage (Leachate 2)
pH	5.7	7.0
TOC,mg/l	20000	2000
COD,mg/l	60000	7000
BOD_5,mg/l	30000	50
HCO_3,mg/l	17	120000

The second leachate was collected from a waste tip in the methane fermentation stage. The organic load was low. Major portion of the organics were identified as high molecular weight compounds. Acids, amines and alcohols were not detected, meaning that the organic compounds consisted of end products of decomposition process.

Table7.12.2 Heavy Metals in the Leachate Samples

Heavy Metals	Acidification Stage	Methane Fermentation Stage
Fe,mg/l	1120	40
Mn,mg/l	53	0.24
Cd,mg/l	0.01	0.02
Cu,mg/l	0.65	0.20
Ni,mg/l	1.04	0.40
Pb,mg/l	0.17	1.0
Zn,mg/l	54	1.6

7.13 Biological Technology of the Treatment of Leachate from Sanitary Landfills

A laboratory investigation was carried out under both aerobic and anaerobic conditions. The anaerobic treatment showed that over 95 percent of the soluble BOD and all iron, were removed, 90 percent of the nitrogen was converted to free ammonia [10].

Table 7.13.1 Summary of the Results of Two- Stage Biological Treatment

Component,mg/l	*Leachate Feed*	*After Anaerobic Treatment*	*After Aerobic Treatment*
BOD5	6280	200	28
TOC	8000	3800	1850
Total Iron	40	<0.02	<0.02
Total Phosphorus	8	<0.02	<0.02
Kjeldahl N	870	850	300
Free Ammonia	300	770	180
Nitrate+Nitrite as N	/	/	48
pH	6.6	7.8	8.6

Table7.13.2 Summary of Performance of Two-Stage Biological Treatment

Component mg/l	% Removal by Anaerobic Treatment	% Removal by Aerobic Treatment	% Removal by Combined (Anaerobic/Aerobic)
BOD	96.8	86	99.5
TOC	53	51.3	77
Total Iron	>99.95	>99.95	>99.95
Total Phosphorus	>97	>97	>97
Kjeldahl N	2.2	65	66

With the above discussed performances, two complete schemes of leachate treatment were suggested for pilot plant evaluation. With each, anaerobic processing with spray irrigation or municipal sewer were to be performed followed by ammonia stripping and aerobic polishing if discharge to surface waters was required.

7.14 The Treatment of Leachates from Domestic Wastes in Landfills

Leachate with a medium organic content (COD:5000mg/l, BOD:3000mg/l) was treated under aerobic biological conditions in continous flow laboratory-scale reactors[64].

At 10°C with addition of phosphate (COD:P less than 100:1) solids retention time of 10 days were required for > 98 percent BOD and >92 percent COD removals. Reduction of temperature to 5°C resulted in adverse effects on settling of sludges with SRT values of less than 10 days. Removal of ammoniacal nitrogen was observed. However, no nitirfication took place at 5° or 10°C with Str values of 10 days. Increasing SRT units to 20 days resulted in erratic conversion to nitirte. But simultaneously denitrification and low pH values resulted in floating sludges and poorly classified effluents.

7.15 Landfill LeachateTreatment

In this paper, a combination of chemical/physical treatment plus biological treatment was is discussed. The treatment plant was designed to consist of lime treatment and sedimentation followed by activated sludge treatment and chlorination. To the chemical/physical section subsequently, equalization, air stripping of ammonia and nutrient addition were added [8]. The activated sludge units were tested on two occasions:

Table7.15.1 Treatment Performance after Acclimization of the Activated Sludge

Parameter	Parallel Operation	Series Operation
	% Removal	% Removal
Suspended Solids	97.4	71.1
Dissolved Solids	58.0	44.7
COD	94.9	94.6
BOD	99.1	94.3
Alkalinity, as CaCO3	87.5	71.6
Hardness, as CaCO3	75.4	41.9
Magnesium	78.6	62.0
Calcium	62.6	82.7
Chloride	39.2	73.7
Sulfate	/	/
Phophate	/	/
Ammonia Nitrogen	88.7	99.4
Kjeldahl Nitrogen	86.4	/
Sodium	37.3	8.3
Potassium	42.2	14.8
Cadmium	87.5	33.3
Chromuíum	75.0	75.0
Copper	77.3	25.0
Iron	99.2	99.3
Nickel	60.2	24.1
Lead	85.4	64.6
Zinc	97.4	95.3
Mercury	28.9	33

Overall the treatment sequence of chemical/physical treatment (lime precipitation, sedimentation, ammonia stripping and neutralization) followed by activated sludge process produced: - a 99 percent reduction in BOD -a 90 percent BOD/COD ratio to 0.16 --a 90 percent ammonia reduction --73.1 percent reduction of chromium --72.5 percent reduction of copper --99.2 percent reduction of iron -- 83.3 percent reduction of lead --27.4 percent reduction of mercury --57.4 percent reduction of nickel -97.3 percent reduction of zinc.

7.16 Moisture Transport in a Solid Waste Column

In this study, the theory of unsaturated flow through poeous media was apllied for the analysis of leachate flow. The results indicate that flow through relatively homogenous refuse can be described by fundamental transport process and that capillary diffusivity contributes very little to the moisture once the field capacity is reached [6].

7.17 Leachate Treatment with Nitrification of Ammonia

This paper presents the results of a two year old study on ammonia removal by biological nitrification. Activated sludge and biological filter plants were employed [65].

Leachate ammonia nitrogen concentrations ranged from 150 to 550 mg/l while TOC values were in the range of 200 to 500 mg/l. Nitrification results were satisfactory in both plants. Maximum ammonia removal rates 131 kg N/VSS.d were 131 N kg /VSS.d (at 13°C) and 309 mg N/m^2.d (at 16°C) in the activated sludge and trickling filter plants, respectively.

7.18 Pilot Testing for Combined Treatment of Leachate from a Domestic Waste Landfill Site

In this study, combined treatment of leachate and primary wastewater was undertaken. Leachate was combined with primary effluent in the pilot testing of 2,4 and 16 percent by volume [66].

Table 7.18.1 The Characteristics of the Leachate Used

Parameter	Concentration
COD,mg/l	1167
BOD_5,mg/l	373
pH	7.9
TSS,mg/l	274
NH_4-N,mg/l	71
Ca,mg/kg	246
Cd,mg/kg	<0.025
Cr,mg/kg	<0.120
Cu,mg/kg	<0.026
Fe,mg/kg	41.4
Mg,mg/kg	77.4
Pb,mg/kg	<0.08
Zn,mg/kg	0.180
Mn,mg/kg	3.24

Table 7.18.2 The Characteristics of the Primary Wastewater Used

Parameter	Concentration
COD,mg/l	238
BOD_5,mg/l	139
pH	7.3
NH_3-N,mg/l	34

The combination with leachate of 2,4 and16 percents by volume did not cause any instability to the activated sludge process. Sludge settlability was not hampered. Normal clarifier design and the activated sludge was satisfactory for the proposed combined treatment.

The addition of leachate did not impair COD removals. COD reductions of 69 and 71 percents were possible in a combined treatment of 16 and 4 percents by volume, respectively.

Ammonia concentrations were reduced by 80 percent. It was noted that leachate additions would not increase effluent ammonia concentrations adversely with 4 percent leachate volume addition.

Sludge production and oxygen uptake rate were considerably higher when leachate was combined with the wastewater. Oxygen availability needed to be increased. Leachate additions led to an increase of metal content in the sludge. However, threshold values for handling were not exceeded.

7.19 Fractionation of the Organic Matter of a Landfill Leachate before and after Aerobic or Anaerobic Biological Treatment

This paper discusses the amount and composition of organic matter in leachate resistant to aerobic and anaerobic biological treatment. Membrane ultrafiltration, gel permeation and high performance liquid chromatography were used to separate the soluble organics of a leachate collected from a mixed industrial and urban wastes tip. The organic compounds in the leachate were low and consisted of compounds below 500MW. However, volatile fatty acids were not present[17].

Table 7.19.1 Chemical Composition of Leachate Before Treatment

Parameter	Concentration
COD, mgO_2/l	1930
Mineral Carbon, mg/l	90
Organic Carbon, mg/l	640
Total VFA, mg/l	<20
pH	7.21
Kjeldahl Nitrogen, mg/l	310
NH_4, mg/l	295
Total P, mg/l	2
SO_4, mg/l	92
Cl, mg/l	1650
Na, mg/l	700
K, mg/l	215
Ca, mg/l	205
Mg, mg/l	95

The fractionation of the organic matter was performed before and after aerobic or anaerobic treatment and the biodegradibilities of the teratment processes were discussed. As a result, it was found out that large majority of the organics initially present in the leachate were very well degraded under both aerobic and anaerobic conditions and that the organic matter resistant to these processes consisted of the same molecules. These recalcitrant molecules were below 500 MW and consisted mostly of industrial chemicals. These organics were only biodegradable to aerobic conditions to about 50 percent.

Table 7.19.2 Molecular Weight Distribution of the Organic Matter of the Leachate before and after Aerobic Treatment

Before Treatment	Before UF	MW>20000	5000<MW<20000	500<MW<5000
COD,mg/l	1930	NM*	NM	NM
BOD_5,mg/l	1260	NM	NM	NM
TOC,mg/l	640	8.9	4.6	15.3
TOC,%	/	1.45	0.75	2.5
After Anaerobic Treatment				
TOC,mg/l	410	6.4	2.7	19.2
TOC,%	/	1.55	0.65	47
COD,mg/l	590	NM	NM	NM
BOD_5,mg/l	0	NM	NM	NM
After Aerobic Treatment				
TOC,mg/l	190	1.5	3.1	17.6
TOC,%	/	0.85	1.75	10.0

*Not measured

7.20 Metal Precipitation in two Landfill Leachates

Two solid waste landfill leachates were studied for the removal of heavy metals by hydrous ferric and manganese oxides. The partioning of cadmium, copper, zinc and nickel with iron and manganese fromleachate constituents at a pH 9 were studied. Batch type partioning experiments were conducted using sodium and calcium hydroxides for pH control [5].

In the experimental program leachate was aerated using fine bubble diffuser with rapid mixing. Then pH was adjusted to 9. Aeration was continued for 10 minutes after pH adjustment at which time the oxidation of iron and manganese apparently reached a state od equilibrium. The leachate in each container was then studied at 20 rpm for 20 minutes and settled with no agitation for 20 minutes. Samples were then analyzed for thier organic strengths.

Table 7.20.1 Performance of Partioning via Sorption

Heavy Metals	Removal Rate, %
Zinc	>95
Cadmium	89-90
Copper	70-75
Nickel	24-36

7.21 Aging Effects of Highmolecular Weight Organic Acids which can be Isolated from Landfill Leachates

The objectives of this work was to isolate organic acids from the anaerobic leachates of a landfill and to charaterize the isolated material[67].

Table 7.21.1 General Characteristics of the Anaerobic and Aerobic Leachate Waters and Effluents of the Model Aquifer

Sample	COD, mgO_2/l	BOD, mgO_2/l	DOC, mgO_2/l	Retention Time, days
A-Model aquifer $_{in}$ anaerobic leachate	7168	2099	812	200
A-Model aquifer $_{out}$ anaerobic leachate	50	22	nd*	200
B-Model aquifer $_{in}$ aerobic leachate	194	43	nd	250
B-Model aquifer $_{out}$ aerobic leachate	15	5	nd	250

*not detected

The horizontal sand beds having the water table about 110cm to 60 cm above the sealed bottom to be one meter broad and 2.2 m deep were constructed. Aerobic seepage water from a municipal landfill mixed with groundwater at a volume ratio 1:5 to 1:10. Aerobic leachate was utilized at a volume ratio of 1:8 for the second acquifer(B).

Analytical determinations were conducted on the isolated samples. The results indicated that the amount of isolated DOC increased from the anaerobic samples to aerobic ones. The isolated material showed an increase of the molecular weight with increasing age of the fill.

leachate was relatively more than the anaerobic leachate. Under the ageing aquifer conditions, the organic acids in the anaerobic leachate increased and were of the humic type refractory weights.

7.22 Biological Treatment of a Landfill Leachate Containing Phenoxy Herbicides and Chlorophenols

A landfill leachate containing phenoxyacetic acid herbicides and chlorophenols were traced under a new biological treatment process. A mixed microbial culture was developed by using a sample soil that had been repeatedly exposed to the phnoxy herbicides[68].

The mixed microbial population was capable of mineralizing the key phenoxy componants in a continous culture medium for 700 days. 99 percent of the phenoxes and 96.7 üercent of chlorophenols were removed. A substantial reduction in the toxicity of the treated leachates were observed.

Table 7.22.1 Toxicity test Results for Treated and Untreated Leachates

Test -24 hours	Untreated-EC_{50}**	Treated EC_{50}	%Reduction
Daphnia magna	2.4	19.9	87.9
Microtox	1.4	26.9	94.8

**Equivalent concentration where 50 % of the organisms survive for 24 hours.

In order to determine the kinetics of removal of individual components in the leachate, a laboratory scale activated sludge system was developed for which the influent leachate concentration was taken as 10 percent. The operating parameters of the activated sludge system were as follows:

Table 7.22.2 Operating Parametes of Laboratory Activated Sludge System

Parameter	Value
Cell Concentration, x	1.1g/l
Hydraulic Retention Time, Q_h	7.3h
Sludge Age, Q_c	32.6h
Recycle cell, X_r	10g/l

The activated sludge system was operated for over 500 hours for the system to reach the stability. As a result, complete degradation of the chlorophenols and phenoxies were observed. Further investigations were started to determine the effects of inhibition and the interactions between the leachate components.

7.23 The Assimilation of Organic Hazardous Wastes by Municipal Solid Waste Landfills

The co-disposal of 12 compounds from major organic classes (aromatic hydrocarbons, halogenated hydrocarbons, pesticides, phenols, phthalate esters) with municipal solid waste (shredded) was tested with a laboratory scale column and pilot sacle lysimeter. The goal of the study was to characterize the transport and the identification of the biotransformation products, leachate and gases were examined. The results provided the attenuating and assimilative capacity of MSW for specific organic compounds.

7.24 Two Dimensional Leachate Estimation through Landfills

A numerical model was developed to compute the time variation flow in landfills. The two dimensional moisture transport equation together with the boundary conditions were solved by a finite element method. the time varying boundary conditions took into account thr runoff, evapotranspiration and infiltration in an unsteady state condition. The model was applied to a landfill in New Yor[70].

7.25 Anaerobic Digestion of Landfill Leachate

A municipal landfill leachate from Kaushung City (Taiwan) was treated in a two phase anaearobic digestion under mesophilic temperatures with modes of batch and semi continous operations. 92 to 95 percent of COD removal was obtained for an influent COD value of 22750 mg/l at digestion times of 8to 20 days. At a digestion time of 11.1 days, and an influent of 37920mgCOD/l, the COD and the BOD removals were more than 90 percent [71].

7.26 Co-treatment of Landfill Leachate and Domestic Sewage in an Activated Sludge Plant

In activated sludge process, the suitability of landfill leachate together with dimestic sewage was investigated. Toxicity and biodegradibility of leachate from landfill was studied. Adaptation effects or domestic sludge addition was observed. The degradation of leachate took place only after the adaptation period. Leachates with high organic loads were also found to be applicable to activated sludge of a municipal treatment plant[72].

7.27 Evaluation of Serial Anaerobic /Aerobic Packed Bed Reactors for Treatment of a Superfund Leachate

Serial anaerobic/aerobic packed bed boireactores were used to illustrate the extent of the biodegradibility of the organic contaminants in leachate from a Superfund site. Over 90 percent of TOC (Total Organic Carbon) between 80 to 90 percent of specific volatile and non-volatile pollutants were removed. Organic and inorganic species belonging to aqueous and gas phases were identified. Mass balances for the anaerobic/aerobic systems were computed [73].

7.28 The Effects of External Carbon Loading on Nirification and Denitrification of a High Ammonia Landfill Leachate

A lanfill leachate containing high ammonia (170-240 mg/l), low BOD (10-60 mg/l) was investigated in a single sludge nitrification-denitrification system. Glucose, acetate, methanol and a brewery yeast waste were used as carbon sources for denirification. The four carbon sources supported denirification. However, glucose addition led to an erratic COD to NO_x-N value of 23:1 whereas complete denitrification was reached with COD to NO_x levels in the order of 5.9:1 to 90:1 [74].

7.29 Attenuation of Heavy Metals within Municipal Waste Landfill Sites

The goal of this investigation was to determine the extent of attenuation of toxic species disposed to landfills. Seven columns were prepared to examine the fate of eight metals ; cadmium, copper, iron, lead, zinc, mercury, nickel, chromium released in solution onto pulverized municipal solid waste [75].

Each column was 50 cm high and had a diameter of 10 cm. The columns were filled with 3.25 kg of fresh wet pulverized municipal waste and were packed to a density equivalent to 0.83 tons/m^3.

The columns which were initially flushed with nitrogen were irrigated at a rainfall rate of 800mm/year. The columns were arranged as three pairs and a single experiment. Each pair was evaluated as a control column and a column dosed with heavy metals. The 7th column had low dosage of heavy metals and was operated like a control cell for the months in the beginning.

During the first 18 days of the experiment, each pair of dosed columns recieved a solution of heavy metal while the control was irrigated with distilled water. The 18 day long dosing of metals was a simulation of the effect of depositing a waste high in metal content on the municipal waste.

The heavy metal solution consisted of eight metals: cadmium, chromuim, copper, iron, lead, mercury, nickel and zinc. All concentrations in the dosing solution was 1000 mg/dm^3 except for mercury.

Leachate samples were collected from the columns twice a week. Analyses revealed that only nickel migrated through the waste with a leachate concentration of 20 mg/dm^3. The other metals were precipitated or absorbed rapidly and were found in the top 0.2 m of 0.5m of waste. In these regions, fatty acid production was inhibited due to the immobilized ions leading to neutral pH values.

7.30 Stabilization Mechanism of Leachate from Semi-Aerobic Sanitary Landfills of Organics Rich Waste

A semi aerobic landfill in which leachate coleection pipes were utilized to form a pipe network was studied. The ends of the pipe network were left open to the atmosphere in order to receive oxygen supply by natural connection [76].

The landfill structure led to early stabilization of waste layer including early reduction of BOD and the waste layer settlement in early stage. The BOD removal capacity in the aerobic zone was between 0.2 and 0.4 mol O_2 $/m^3h$.

7.31 Effect of Moisture Regime and other factors on Municipal Solid Waste Stabilization

In the present study, the rate of solid waste stabilization under aerobic conditions were investigated. The purpose of the study was to evaluate the amount and the composition of gases generated during anaerobic decomposition of solid waste [77].

Eighteen 208 liters steel containers that were filled with 55 to 80.5 kg dry weight of solid waste were sealed and held under different environmental conditions for subsequent measurement of gas production and composition. The moisture content of the milled waste was around 28 percent calculated on a dry basis, water was added to have a moisture content of 36 percent.

The evalaution of the results were conducted through the employment of different factors. The effect of moisture content on the gas production was studied with the largest number of containers. The effect of temperature was evaluated by having one container at 26°C and the rest at 17°C. The effect of solid waste size was evalauted with two containers that contained unshredded soils and were torn by hand to smaller sizes. The solid waste density was also investigated in two containers with a densita higher and lower, respectively. Three containers were exposed to air to evaluate the effect of air addition on the gas composition.

Table 7.31.1 Results of Gas Production

Effect Studied	Container no	Initial Mositure Content (%)	Dry Density, (kg/m^3)	Average Gas Production ml/kg-d
Moisture Condition	1	36	334	0.29
	2	36	393	3.50
	5	36	345	5.63
	7	36	368	0.77
	9	60	399	0.43

	10	78	342	3.87
	15	99	375	12.9
	6	99	386	17.8
	13	99	436	15.8
	14	99	436	20.2

Effect Studied	Container no	Initial Mositure Content (%)	Dry Density (kg/m^3)	Temperature (°C)	Average Gas Production (ml/k
Temperature	11	36	356	26	1.07
	(1,7,9)*	36	367	17	0.50
Size	4	78	310	17	1.63
	18	78	356	17	9.03
	15	78	375	17	12.9
Density	12	36	421	17	1.23
	(1,2,5,7,9)*	36	368	17	2.12
	16	36	355	17	9.7
Open to Air	17	36	406	17	/
	3*	26	375	17	/
	18**	28	399	17	/

* Average values of the containers indicated

** Sealed but gas outlet left open.

The major gases detected were $C0_2$, H_2, and CH_4. The maximum rate of gas production was in the range of 6.0 l/kg dry weight during the testing period. It was demonstrated that increasing the moisture contents increased the rate of gas production. Similar effects were observed by increasing temperatures and decreasing solid waste size, decreasing solid waste density. H_2 gas was detected in the containers with the lowest moisture content whereas CH_4 was found in only one container with the shredded refuse.

8. THE EXPERIMENTAL WORK

8.1 The Municipal Solid Waste of Istanbul

The city of Istanbul, with a population approximately approaching to 10 million is among the fastest growing cities in the world. One of the main concerns regarding this rapid growth has been linked to the solid waste management strategies attained in Istanbul.

The amount of solid waste generated today from the residential and commercial districts of Istanbul has been reported to be higher than 10000 tons/day while in 1979 and 1986 this value corresponded to 3600 tons/day and 6000 tons/day, respectively [78,79]. This escalation has been attributed mainly to the rise in population and to the shifts in the socio-economic structure which have been imparting tremendous effects on the solid waste production. As an example, the per capita solid waste production in the year 1980 was 0.72kg/d. On the other hand, in the following years 1985 and 1986, this figure inflated to 0.1 kg/d and 1.1 kg/d, respectively.

Analogously, over the years significant transformations in the solid waste compostion have taken place:

Table 8.1.1 Summary of the MSW Composition Encountered in Istanbul (% by weight) [19]

Component	1980*	1983**	1984**	1985**	1986***	1987***
Food Wastes	60.8	59.0	51.81	43.25	42.70	44.62
Paper&Cardboards	10.15	13.12	10.73	11.72	10.20	11.98
Textiles	3.22	1.40	3.21	4.87	3.10	4.28
Plastics	3.05	4.04	4.83	7.13	6.22	11.87
Glass	0.65	0.55	1.36	1.71	3.61	3.42
Metals	1.43	6.49	4.82	4.34	4.56	2.28
Ash	4.54	10.40	18.45	16.84	20.15	14.12
Miscellaneous	4.54	5.00	4.79	10.14	9.46	7.43

*Average refuse analyses for 10 months

**Yearly average values

***Average values for the first six months.

The MSW composition of Istanbul has been observed to reflect a similar pattern to those encountered in Brazill[20] and Israel[58]. Compared with about 15 percent food wastes and 44 percent paper and cardboard in an average town in the U.S.,[40] the typical refuse in Istanbul contains about 50 percent food wastes and only 11 percent paper and cardboard.

Moreover, the high water content (about 70 percent) represents another variation from the MSW generated in the United States and Europe where the moisture contents are in the range of 20 and 35 percent, respectively.

As a result, these distinct characteristics of refuse affect the quality and quantity of leachate generated in the disposal sites. Based upon an understanding of the processes influencing leachate characteristics, management of generation and transfer rates can be executed by managing the moisture regime within the landfill body. It is known that, without moisture, the transport medium necessary for the conversions and interactions determining leachate and gas quality is subdued. However, once under control, the presence of moisture can be used as an advantage. When the moisture input of the MSW originating from Istanbul is analyzed, it is discovered that the dumped refuse especially during the summer months is somewhat in a saturated condition. Due to this situation, in the calculation of the moisture input, the classical water balance model need to be modified accordingly.

Studies supervised by Universität Stuttgart (Dr. E. Thomanetz),Insitut für Siedlungswasserbau, Wassergüte und Abfallwirtschaft on 'Sonderabfalldeponie Malsch' [13] led to a more detailed frame of reference which could also be defined accordingly to the situation in Turkey:

$$S=N+\ddot{U}+K+R+U-V-H-A \tag{8.1.1}$$

where S=sum of the moisture components of leachate in a landfill body.
N=rainfall
Ü=liquids arising from the wastes
K= consolidation water arising during disposal
R=reaction water arising from chemical/biological tranformations
U=underground water movement
V=evaporation
H=side slope runoff
A=surface runoff

After a thorugh inspection of the characteristics of MSW generated in Istanbul, steps toward effective gas and leachate management could be undertaken. Hence, leachate recirculation as an in situ treatment method is analyzed in this study to determine the extent of its applicability as a cheap, effective treatment mechanism.

8.2 The Proposal

Sanitary landfills are the most commonly employed method of solid waste disposal practiced in both metropolitan and rural communities. Yet, sanitary landfills remain poorly understood and imprecisely managed due to insufficincies in waste definition and perception of associated environmental variables. However, in the last two decades, the effect of leachate migration and the hazards identified with gas production have led to the development for control and treatment. In accordance with these approaches, various techniques have been enforced for the treatment and disposal of landfill gase and leachates.

The purpose of this study was to address two major areas of concern. The first was to investigate the feasibility of leachate recirculation in providing leachate control and treatment. The second was to establish an understanding on the rate of bilogical stabilization of Turkish solid waste and thereby, on the quality of leachate generated in a leachate recirculation system under mesophilic conditions.

8.3 Experimental Set-up

To accomplish the goals of the investigation, three laboratory lysimeters were constructed. The laboratory lysimeters were employed to simulate the behavior of the vertical center of a newly covered sanitary landfill.

Lysimeter design specifications were aimed at insuring proper simulation. Several elements that are integrated at the lysimeter boundaries were defined as follows:

*heat exchange across the vertical walls of the lysimeters must be a minimum.

*The lysimeters must be air-tight.
*Leachate recirculation mechanism must not allow a differencial moisture movement.
*the climate conditions 'at the boundary' soil cover interface must be similar to climate conditions observed in Istanbul.
*the cross-sectional area of the lysimeters must be large enough to insure the generation of reliable experimental data.
*the composition of the refuse must be representative of typical municipal solid waste of Istanbul.
*the density at placement must be in the range usually found in sanitary landfills.

The lysimeters were constructed of stainless steel. The top and bottom lids were tightened by a total of 32 stainless screws to have them in air tight conditons. The interior walls were not placed with any lining. However, geotextile filters were placed.

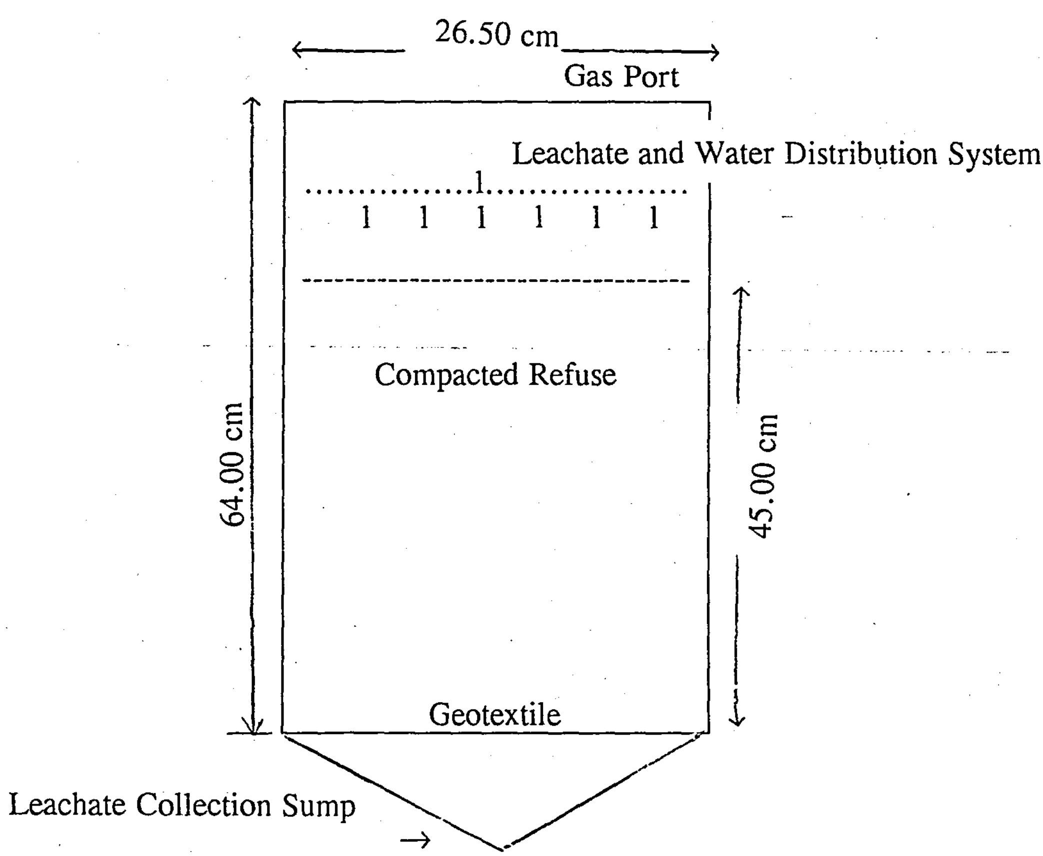


Figure 8.3 The Dimensions of the Laboratory Lysimeters

8.3.1 Leachate Collection

Leachate collection was faciliated by using glass storage bottles which were placed beneath each lysimeter. A notch controlled the drainge mechanism. Each day the notch in the system was pulled down for a few seconds and returned to 0-position after leachate collection was proceeded because leachate recirculation was not performed in a continous manner.

8.3.2 The Temperature Control System

The temperature control and water application system comprised the environment control system of the investigation. The temperature control of the lysimeters were accomplished by placing the lysimenters in a heating chamber. The heating chamber was monitored by an automatic digital thermometer which was set to 40°C at the start of the experiment. Throughout the experimental period, the same temperature was maintained.

Figure 8.3.2 The Heating Chamber

8.3.3 The Water Distribution System

Simulated rainfall which represented the excess of precipitation over evapoartion was added to the lysimeters. The yearly rainfall figures were converted to equivalent liters of tap water to be distributed periodically. The chlorine content of the tap water was analyzed and was determined to be in the range of 7.0mg/l. Water application was not practiced during the weekends and official holidays.

The water distribution system was supplied by a perforated teflon loop which acted as sprinkler underneath the upper lid of each lysimeter. A glass funnel mounted at the top of the lysimeters enabled the movement of water down into the perforated loops. Leachate collected from the storage bottles was recirculated in the same fashion.

Figure 8.3.3 The Water-Leachate Distribution System

8.3.4 The Gas Measurement

Three drum gas counters (BRAND), were connected to the gas ports of the lysimeters in order to measure the quantity of gases produced. The drum gas counter belonging to the third lysimeter(3) was replaced three times at the start of the experiment to insure air-tight conditions. Care was taken to obtain representative gas samples. Referring to the density differences (CO_2 is about 2.8 times as dense as CH_4) within the lysimeters, the gas mixture was thoroughly intermingled via a syringe-pump mechanism attached to the gas ports before gas samples were to be collected.

Figure 8.3.4 The Gas Drum Counters

8.3.5 The Composition of the Refuse

The synthesized refuse in the lysimeters were representative of 'typical' domestic refuse encountered in Istanbul. The refuse composition was patterned after the results of [79]. The shredded refuse was manually compacted into an approximate dry density and a height of 45.0 cm. The placement of 25 kg of refuse was done in layers to simulate the landfilling operations. The approximate composition of the refuse placed in the lysimeters is shown in Table 8.3.1. In each lysimeter, six liters of tap water were added to provide a representative baseline moisture content.

Figure 8.3.5 During the Compaction Process

Table 8.3.1 The Refuse Placed in the Lysimeters

Constituent	*Dry Weight Basis (%)*
Food Wastes	44.62
Paper&Cardboard	11.98
Textiles	4.28
Plastics	11.87
Glass	3.42
Metals	2.28
Ashes	14.12
Miscellaneous Materials	7.43

Figure 8.3.5.1 The Inner Diameter of the Lysimeters

Figure 8.3.5.2 A Representative Composition of the MSW Used

8.4 Experimental Procedures

8.4.1 Operation

The operation of the lysimeters was commenced with the addition of sufficient amount of tap water (twice the asumed amount) to increase the moisture content in the lysimeters. After the initial leachate production, simulated rainfall was added five days of the week on a continous basis. This was accomplished by converting the annual excess precipitation over evapoaration to equivalent liters of tap water and adding that amount to the water distribution system. Two levels of water application was used. The high level application was 700mm/a which corresponded to 80ml/d with the assumption of 25 percent evaporation taking place. Correspondingly, the low level equivalent in quantity to 350 mm/year was converted to 40ml/d of tap water.

Dry periods without any practice of tap water/leachate recirculation were coincided with the weekends and official holidays. The recirculation of leachate began starting from the 16th day of the investigation and was continued until the completion of the investigation.

The operational features of the three lysimeters were as follows:
Lysimeter 1- Control cell; only simulated rainfall added, no leachate recirculation, received 700mm/a rainwater (80ml/d tap water).
Lysimeter 2- Leachate released was recirculated; received 700mm/a rainwater (80ml/d tap water).
Lysimeter 3-Leachate released was recirculated; received 350mm/a rainwater (40 ml/d tap water).

8.4.2 Analytical Methodology

In order to evaluate the effects of recirculation on the stabilization of organic matter in the laboratory lysimeters periodical analyses on the collected leachate and gas samples were performed. These data were then compared with similar information available to establish trends in constituent concentrations.

Leachates generated in the laboratory lysimetres were analyzed for certain indicator parameters that could be used to detect and describe the effects of leachate recirculation on landfill stabilization.

Table 8.4.2.1 The Laboratory Analyses Performed on Leachate and Gas Samples

Constiuent	*Indicative Parameter*
pH	physical/chemical
COD-TOC	chemical/biological-(biodegradability)
Heavy Metals	potential inhibition
Conductivity	ionic strength
Chlorides-AOX	migration potential
Ammonia Nitrogen	oxidizing potential
Carbon Dioxide	phase stabilizations
Methane	phase stabilizations

The above mentioned anayses were selected on the basis of three criteria:
1.They represent the most common parameters in the literature used to characterize landfill leachate,
2.they provide sufficient data,
3.they are required specifically to define leachates.

The analytical methodology used is presented as follows:

Table 8.4.2.2 The Analytical Methodology Used during Laboratory Lysimeter Investigations

Indicator Parameter	**Procedure**
pH	WTW LF91, Präzisions pH meter 391
COD,mg/L	COD 1500 Tube Test, The Nanocolor Analytical System
TOC,mg/L	Neuenentwicklung eines TC/TIC/TOC Analytical System
Heavy Metals,mg/L	ICP Jarrell Ash 1100 Series Atom Corp
Conductivity,mS/cm	WTW LF 91, Conductivity meter
Chloride,mg/L	926 Cloride Analyzer
AOX,mg/L	MCI Total Organic Halogen Analyzer TOX-10
Ammonia Nitrogen,mg/L N	Ammonium 50 Tube Test, The Nanocolor Analytical System
CO_2,CH_4, %	Gas Chromatograph 2300 AS -Carlo Erba

8.5 Results

The production of leachate by municipal solid waste requires the establishment of a moisture content to be at a relevant field capactiy. Consequently, the particular leachate production is defined to be a function of the initial moisture contents of the various components as well as the water input pattern. Of concern is the leachate generation, induced by the water input. One of the following sources could be credited to this form of leachate production:

1. Refuse(MSW)-Most of the initially generated leachate is formed from the organic constituents of the refuse during compaction and placement procedure.
2. Channelling-As a result of non-uniformity within the refuse some of the distributed water may find a direct passage through the refuse to the collection sump.
3. An Advanced Wetting Front-Due to the movement of a wide band of a wetting front, considerable increases in leachate occur before the system reaches field capacity.
4. The Main Wetting Front-When the entire system reaches field capacity, the input water and the output leachate quantities evolve into a 1:1 ratio.

Information regarding the quantitative nature of leachate generated as a result of the products of decomposition is hard to establish due to the variable input waste characteristics. Nevertheless, it is substantiated that without mositure, the transport medium will not exist, the conversions and interactions will be restrained.

The relationship between the simulated rainfall added w/o leachate recirculation and the corresponding leachate production was analyzed. The correspondence between the input and output did not reach a 1:1 relationship while the observed pattren could be described as the transition between an advanced wetting front and the main wetting front. As a 1:1 correspondence between the input water and the leachate produced in the lysimeters is attained, the primary means of leachate production will be from the main wetting front. Once this relationship is established,this source will remain as the primary source during the life of the lysimeters.

The lysimeters were activated in May 1, 1993. Leachate first appeared in Lysimeter 1 on May 3, 1993 (the 2nd day of the experimental period). This was as a result of one or both of the sources presented on the preceeding page. Intially, the lysimeter was fed with a daily total of 160ml of tap water for a week; with the intention of accelerating the production of leachate. After May 10,1993 (11th day)only the presumed amount of simulated rainfall was distributed throughout the testing period. A clogging problem during the week of May 17,1993 led to constant leachate production.

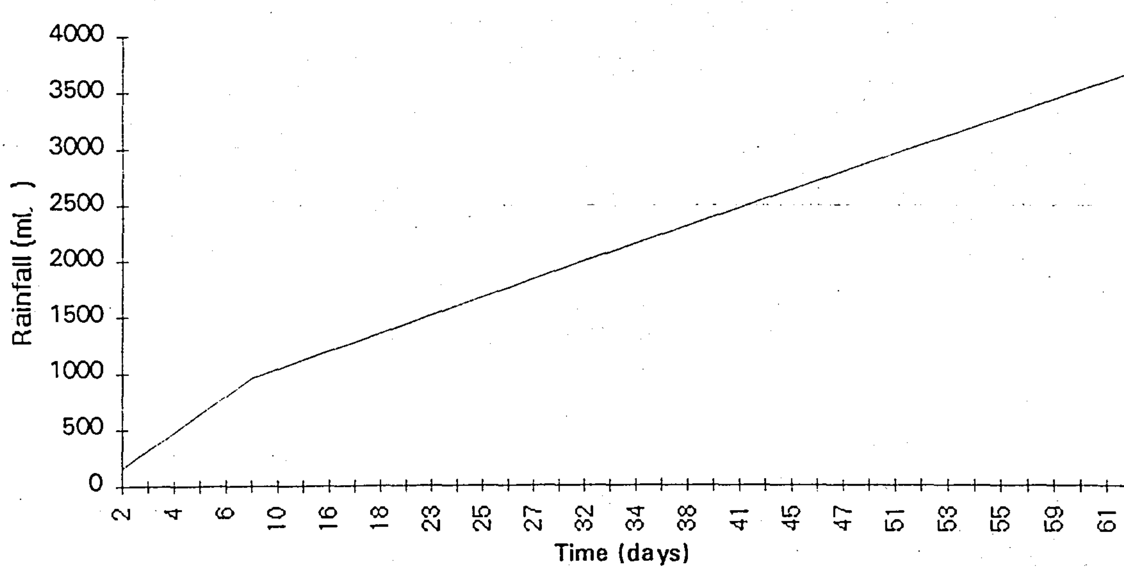


Figure 8.5.1 Simulated Rainfall in Lysimeter 1

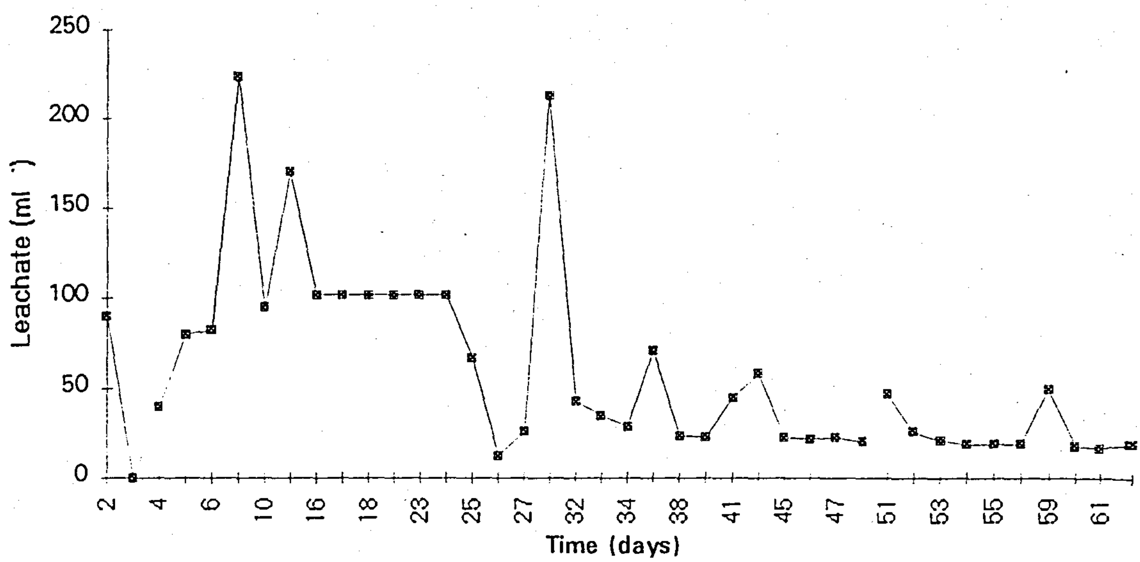


Figure 8.5.2 Leachate Volume in Lysimeter 1

The total quantity of water added is considered to increase the moisture content of the entire system within the lysimeters. Leachate recirculation was conducted in Lysimeters 2 and 3 to improve such a condition for accelerating the conversions and interactions leading to biodegradability of the refuse. However, the amount of leachate produced from these lysimeters (LR+rainfall) was significantly lower than the quantity of input.

Lysimeter 2 was initially treated in the same manner as Lysimeter 1 until the introduction of leachate recirculation. With the commence of leachate recirculation (May 17,1993) in the 16th day of the experiment the daily rainfall total was distributed together with the amount of recirculated leachate. The recirculated leachate on a daily basis comprised approximately 97 percent of the generated leachate. At the end of the testing period (preliminary results), it produced 1264 ml of leachate with a given input of 3680 ml tap water plus the recirculated amount of leachate (approximately,1227 ml).

Lysimeter 3 received 50 percent of the yearly precipitation. At the start of the experiment, it received a daily total of 80 ml tap water for a very short period of time. The leachate recirculation (May 17, 1993), in the 16th day of the experiment was performed in the same manner as in Lysimeter 2. With a total input of 1840 ml tap water and an amount of 1028 ml recirculated leachate, it produced 1060 ml of leachate.

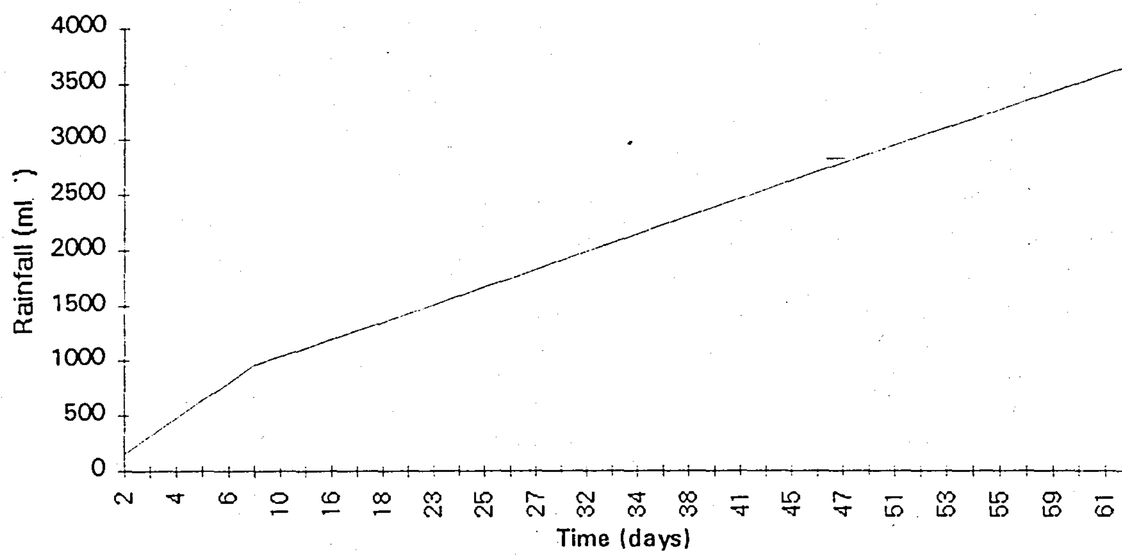


Figure 8.5.3 Simulated Rainfall in Lysimeter 2

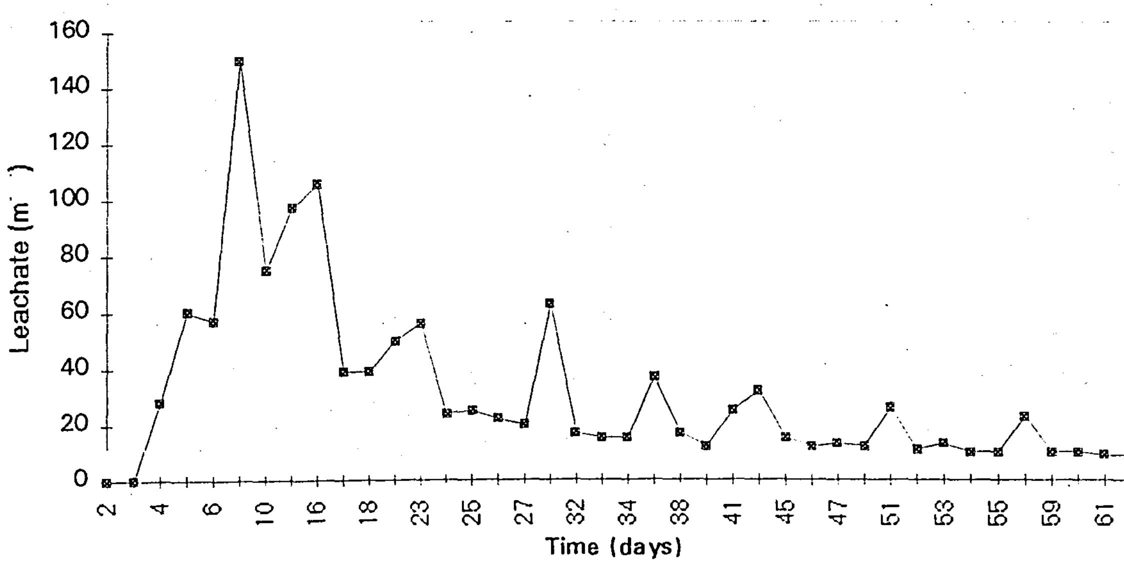


Figure 8.5.4 Leachate Volume in Lysimeter 2

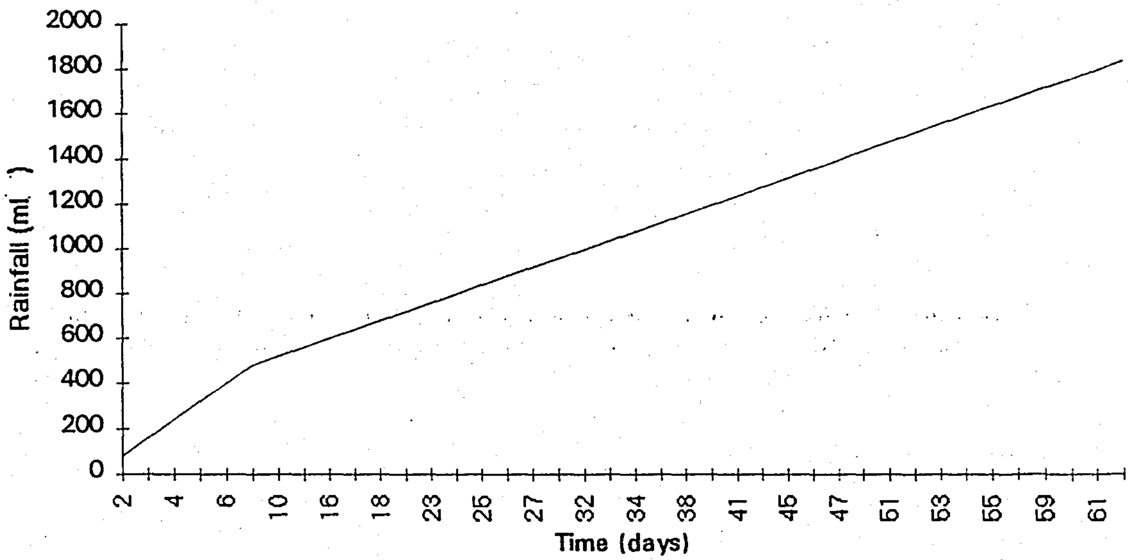


Figure 8.5.5 Simulated Rainfall in Lysimeter 3

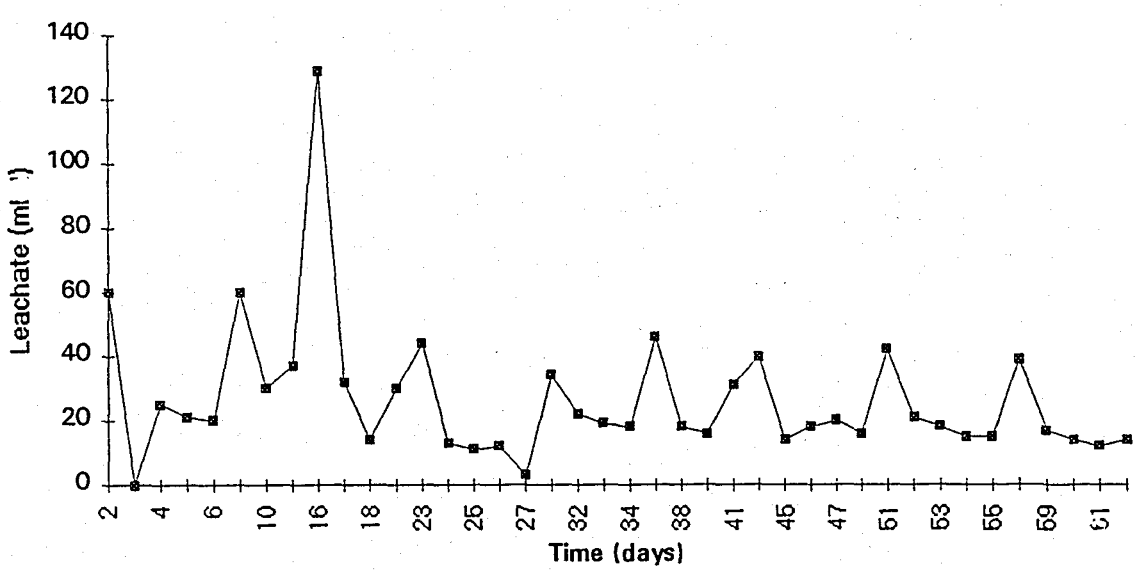


Figure 8.5.6 Leachate Volume in Lysimeter 3

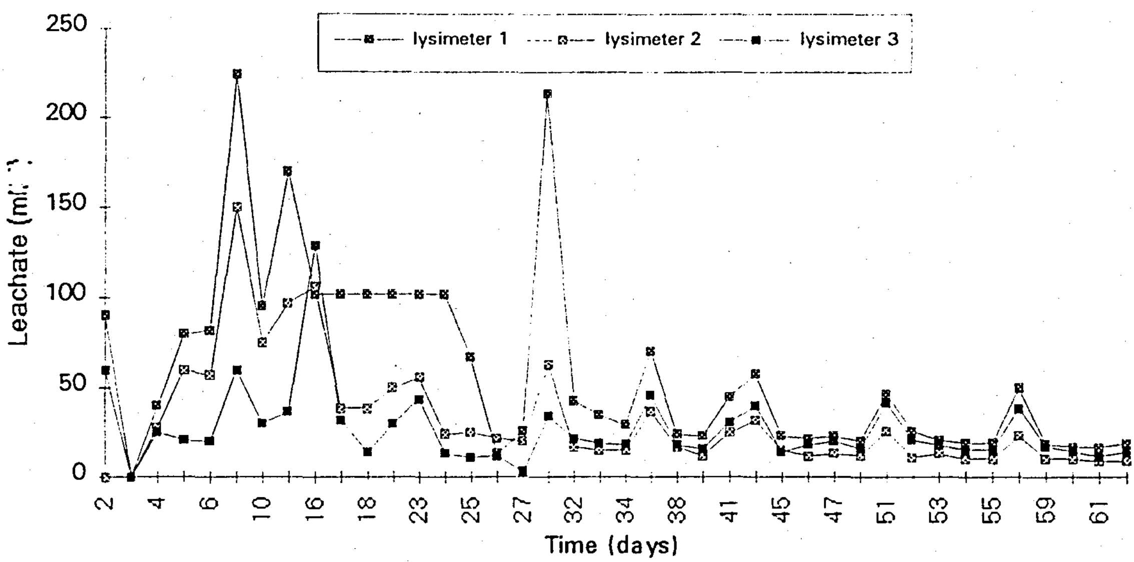


Figure 8.5.7 A Comparison of the Leachate Generation in the Lysimeters

In order to illustrate the changing nature of the leachates with time and to provide a basis for the interpretation, leachates from the lysimeters were correlated at the time when continous leaching (w/o leachate recirculation) commenced to those at&after the onset of gas (methane) production. This approach is believed to closely simulate longer term changes in leachate quality to be expected at conventional municipal landfills.

Table 8.5.1 Leachate Analysis Values for Parameters with Differences between Acetic & Methanogenic Phase

Acetic Phase	Lysimeter	Lysimeter	Lysimeter
Leachate Parameter	1	2	3
pH	6.6	6.3	7.0
COD,mg/l	35670	29855	15661
TOC,mg/l	11080	10799	7515
COD/TOC	3.24	2.70	2.12
Ca,mg/l	250	265	480
Mg,mg/l	97	99	148
Fe,mg/l	4.4	3.5	5.6
Mn,mg/l	5.2	4.4	2.0
Zn,mg/l	13.5	3.6	11.7
Methanogenic Phase	Lysimeter	Lysimeter	Lysimeter
Leachate Parameter	1	2	3
pH	7.8	7.9	7.9
COD,mg/l	5225	7150	2650
TOC,mg/l	6455	8220	3660
COD/TOC	0.80	0.87	0.72
Ca,mg/l	51	75	66
Mg,mg/l	272	410	341
Fe,mg/l	1.7	1.3	1.5
Mn,mg/l	<0.1	<0.2	<0.1
Zn,mg/l	1.5	1.2	0.7

Table 8.5.2 Leachate Analysis for Parameters with no Differences between Phases

Leachate Parameter	Lysimeter 1	Lysimeter 2	Lysimeter 3
Cl,mg/l	2044	2372	1462
Na,mg/l	1300	979	631
K,mg/l	1775	2238	2589
NH_4,mg/l	1028	886	1000
AOX,mg/l	3.0	/	/
Cd,mg/l	<0.02	<0.01	<0.02
Co,mg/l	<0.01	<0.01	<0.01
Ni,mg/l	0.19	0.26	0.19
Pb,mg/l	<0.25	<0.25	<0.25
Cr,mg/l	0.03	0.06	<0.02
Cu,mg/l	2.9	2.5	2.9

/not measured

The parameters COD and TOC were analyzed to illustrate the pollutional impact derived from the organic matter in the lysimeters. Initially, all of the three lysimeters produced high COD concentrations. This was due to the development of acid formation stage. As the biodegradation process proceeded, the values decreased to intermediate values. Lysimeter 1 produced leachate with the highest peak of COD concentrations. Comparetivelly, the COD load of Lysimeter 2 was higher than that of Lysimeter 3. Lysimeter 3, with the least amount of moisture, produced the lowest COD concentrations. COD values from Lysimeter 2 was slightly lower than that of Lysimeter 1.

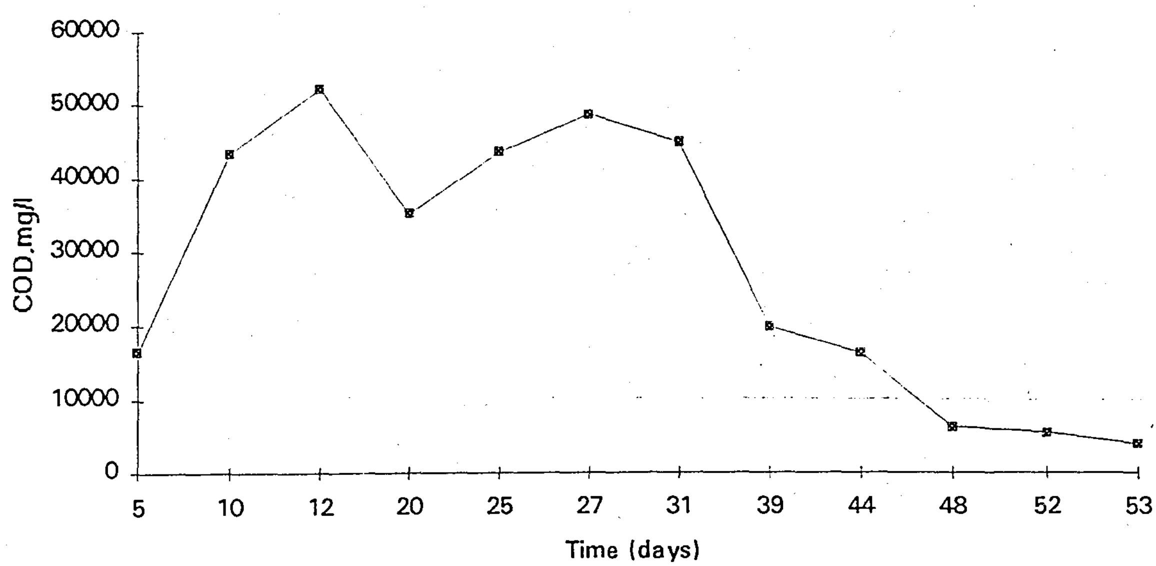


Figure 8.5.8 Chemical Oxygen Demand of Lysimeter 1

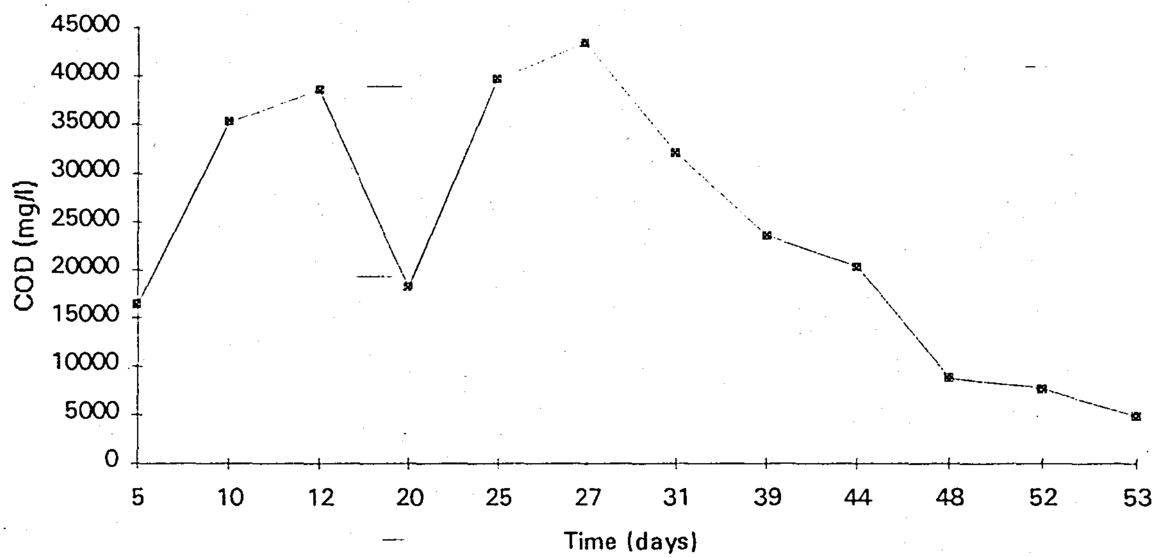


Figure 8.5.9 Chemical Oxygen Demand of Lysimeter 2

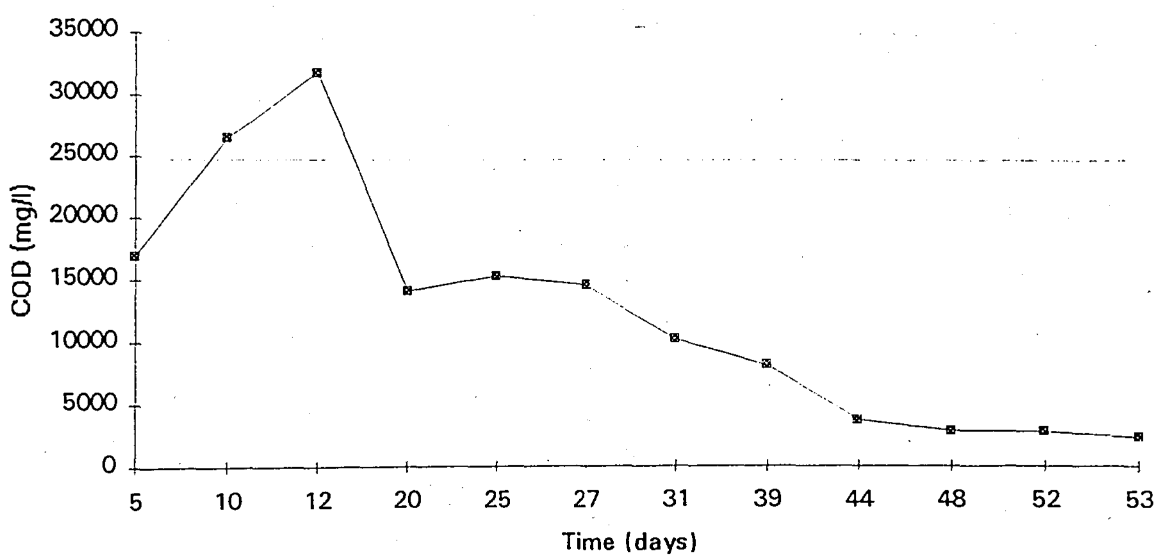


Figure 8.5.10 Chemical Oxygen Demand of Lysimeter 3

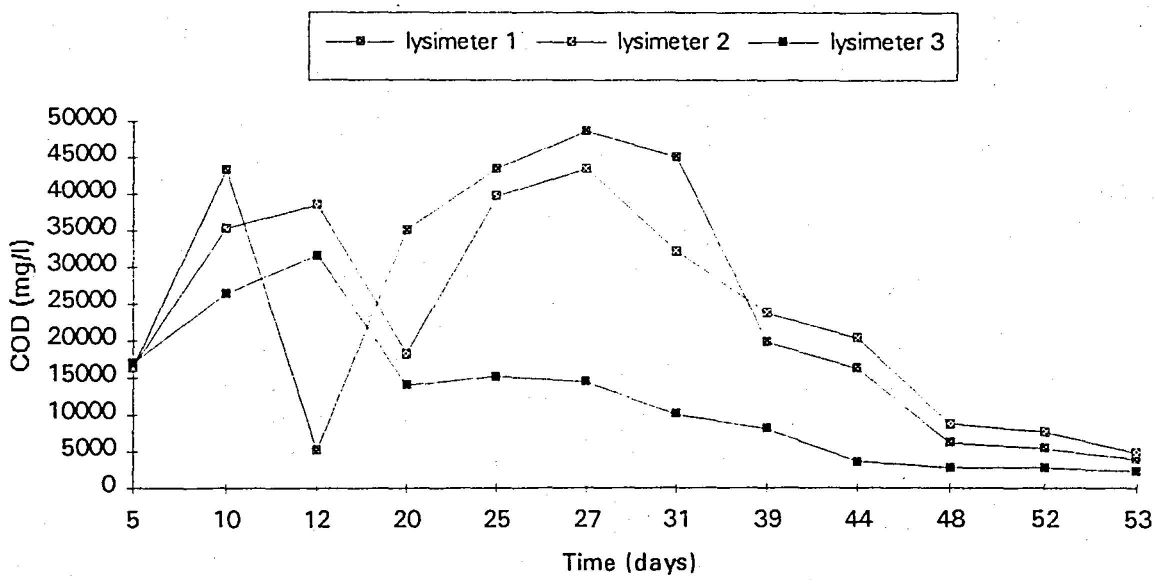


Figure 8.5.11 A Comparison of COD Values in the Lysimeters

The TOC values revealed a similar pattern to that observed in COD concentrations. Lowest emissions came from Lysimeter 3. The loads from Lysimeters 2 and 1 were observed to be in the same range.

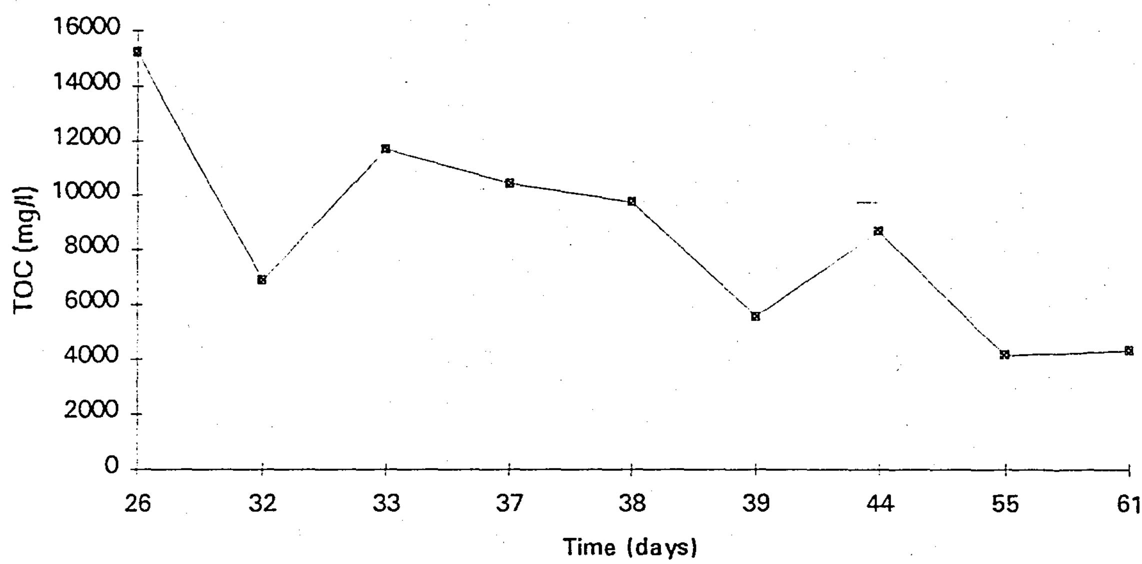


Figure 8.5.12 Total Organic Carbon of Lysimeter 1

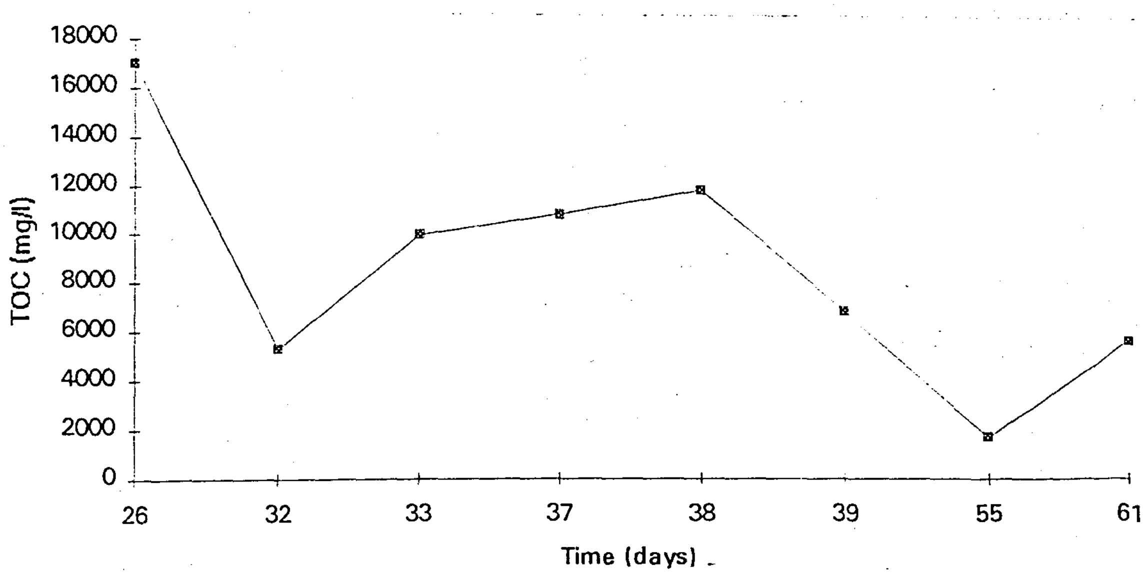


Figure 8.5.13 Total Organic Carbon of Lysimeter 2

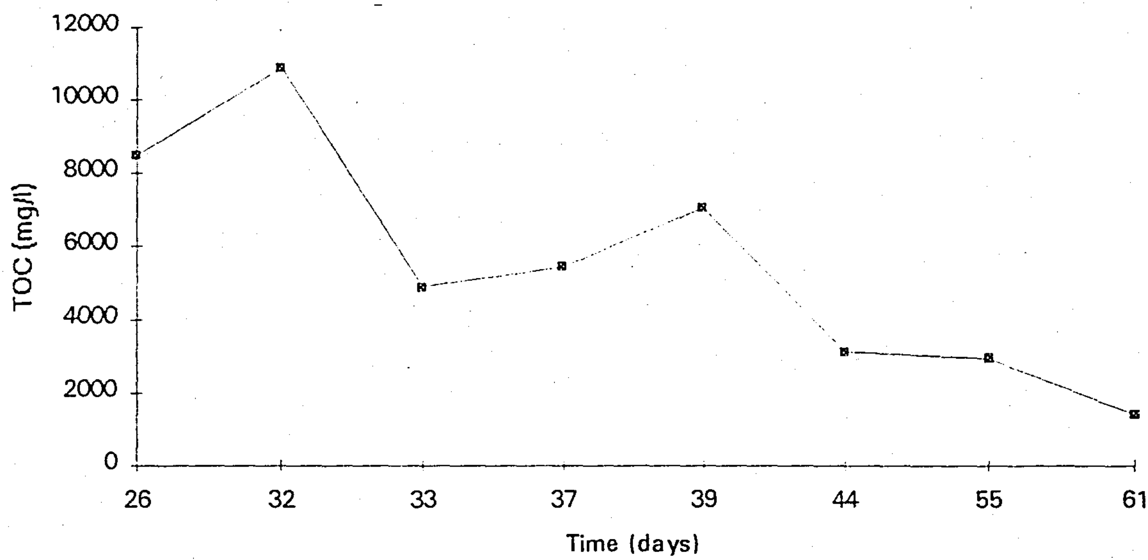


Figure 8.5.14 Total Organic Carbon of Lysimeter 3

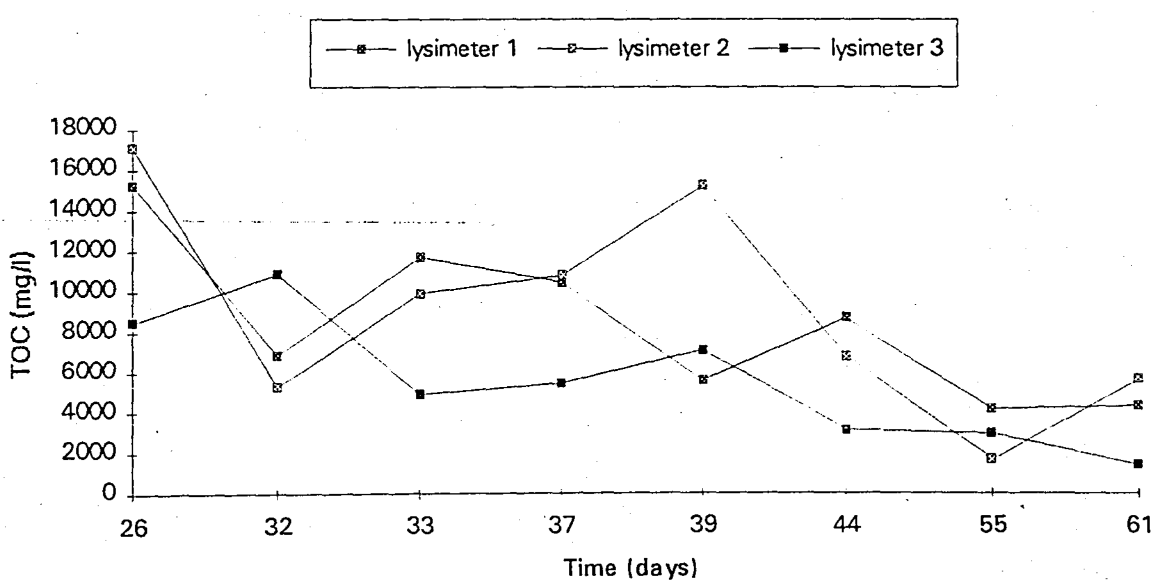


Figure 8.5.14 A Comparison of TOC Values in the Lysimeters

Furthermore, the COD/TOC ratio that existed in each cell was rewieved:

Table 8.5.3 The COD/TOC Ratios during the Phases of Stabilization

COD/TOC	Acid Formation Phase (2.1-3.4)	Methane Fermentation Phase(2.0-3.0)**	(0.4-2.
Lysimeter 1	2.62	2.86	0.8
Lysimeter 2	1.88	3.0	0.8
Lysimeter 3	2.33	3.0	0.7

**at onset of methane fermentation

* after methane fermentation

A comparison of the results with regard to COD and TOC values, indicated that all three lysimeters were performing in the expected manner; the stages of biological stabilization were proceeding accordingly. However, the end effects of leachate recirculation from Lysimeter 2 did not reveal an advantage over the control lysimeters. The data obtained from Lysimeter1 showed some variations from the results of Lysimeter 2. Furthermore, the concentrations in Lysimeter 3 reached the lowest values. This was established as a designation of the influence of only a sufficient moisture addition on the biological stabilization process.

The interrelationship between COD and specific conductance was evaluated. Specific conductance measurements were significant in the determination of the ionic strength in the lysimeters. There was a striking similarity in the curve shapes. As the anaerobic decomposition became dominant, the COD rose and the specific conductance increased. During the formation of CH_4, a reduction in specific conductance was correspondingly noted as a decrease in COD concentrations occrred.

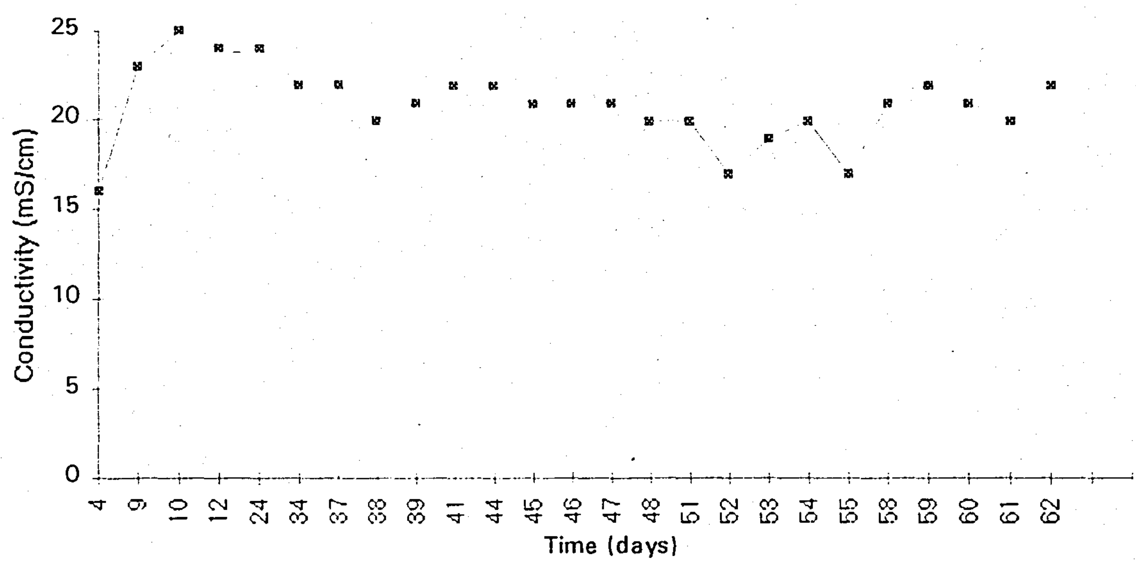


Figure 8.5.16 Leachate Specific Conductance of Lysimeter 1

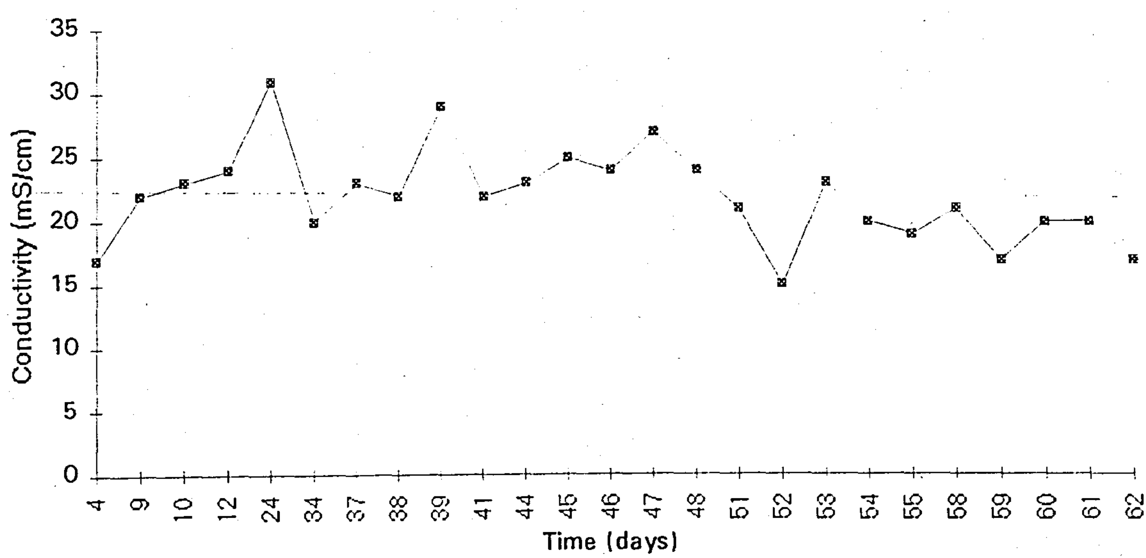


Figure8.5.17 Leachate Specific Conductance of Lysimeter 2

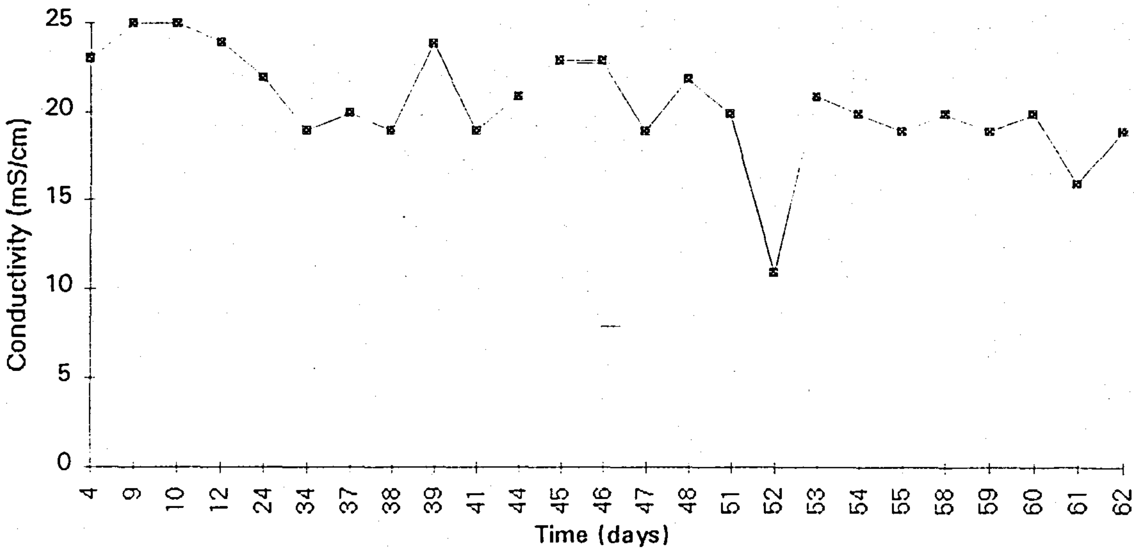


Figure 8.5.18 Leachate Specific Conductance of Lysimeter 3

After the commmencement of leachate recirculation, a sharp drop in the specific conductance of Lysimeter 3 was observed while Lysimeters 1 and 2 exhibited alike values.

The pH curves which generally vary as the inverse of the COD and specific conductance curves started from values in the range of 6, as the COD-specific conductance values reached a decreasing trend the pH values rose sharply to near 8.0.

Table 8.5.4 pH of Leachates from the Lysimeters

Time Since Placement of Refuse (days)	Lysimeter 1	Lysimeter 2	Lysimeter 3
5	6.7	6.3	7.0
10	6.8	6.7	6.9
47	7.9	8.0	8.0
51	7.7	7.8	7.8
54	7.7	7.7	7.7
58	7.7	7.7	7.8
59	7.8	7.9	7.9
60	7.9	8.0	8.0
61	7.8	7.9	7.9
62	7.8	7.9	7.9

Chloride emission potential of the lysimeters were limited. The concentrations remained relatively high as it is the case within a landfill body.

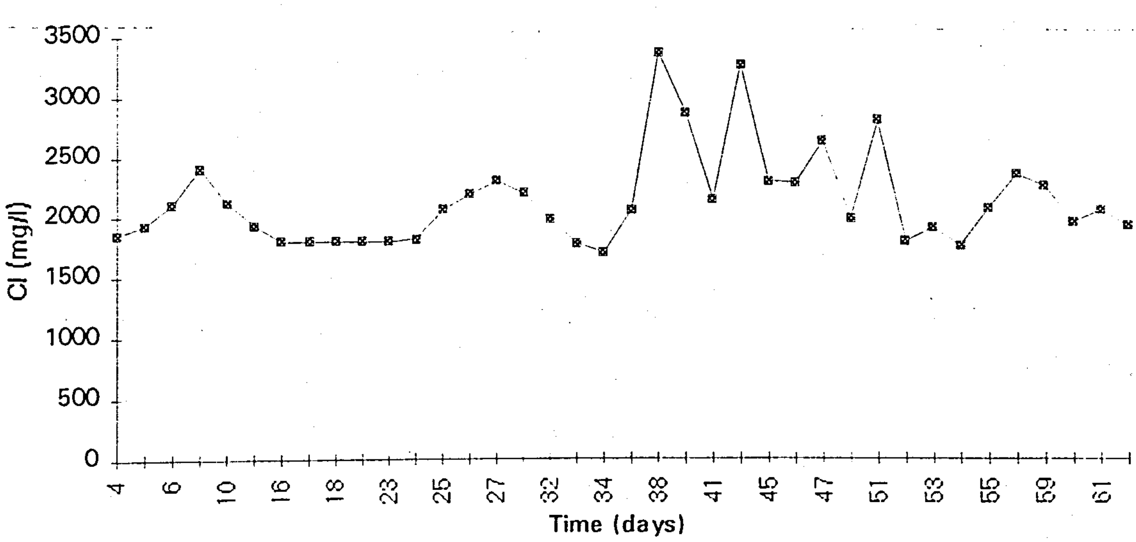


Figure 8.5.19 Chloride Concentration of Lysimeter 1

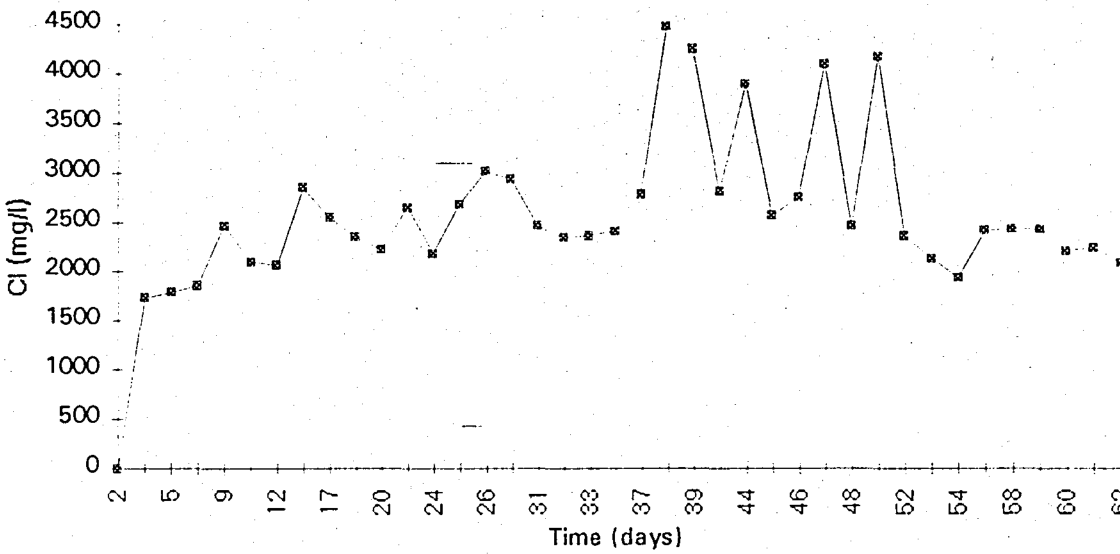


Figure 8.5.20 Chloride Concentration of Lysimeter 2

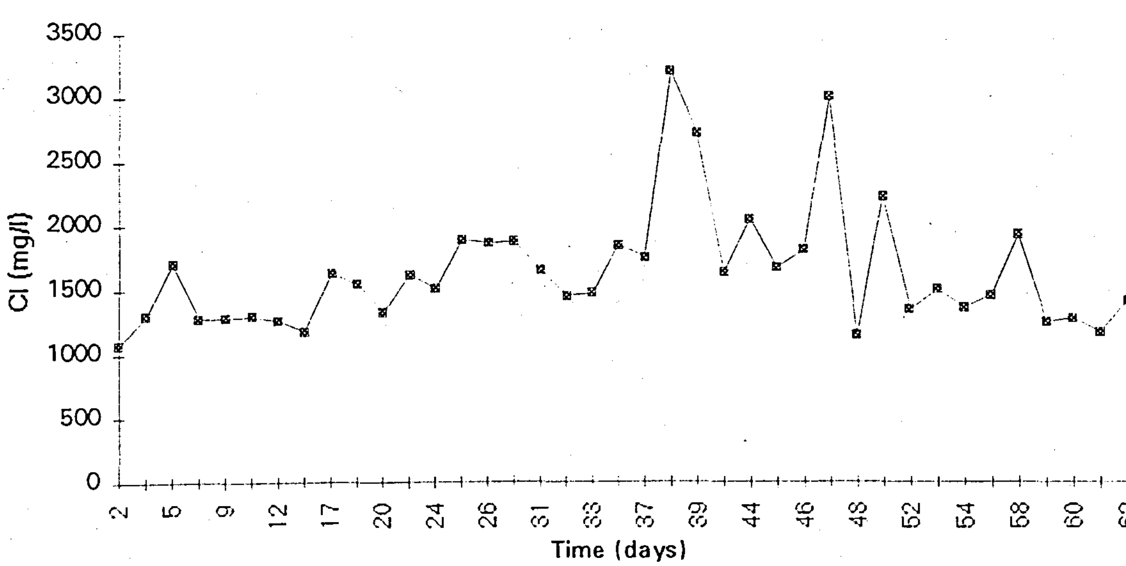


Figure 8.5.21 Chloride Concentrations of Lysimeter 3

The ammonium concentrations showed a low increase during the early stages of operation as expected. When the decomposition process advanced, relatively constant but high values were reached. There was not a significant difference in the concentrations of the leachates from the lysimeters.

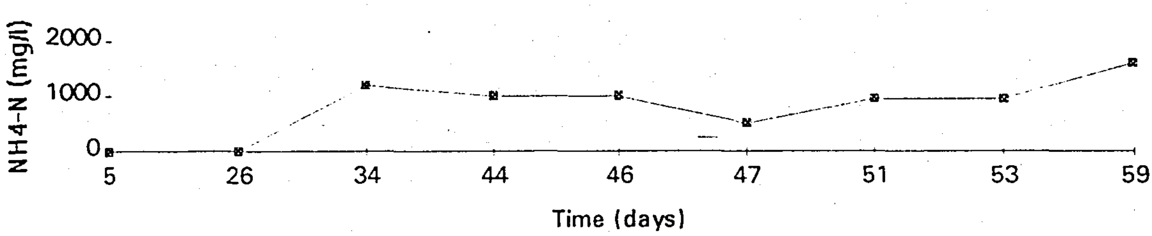


Figure 8.5.22 Ammonium Concentrations in Lysimeter1

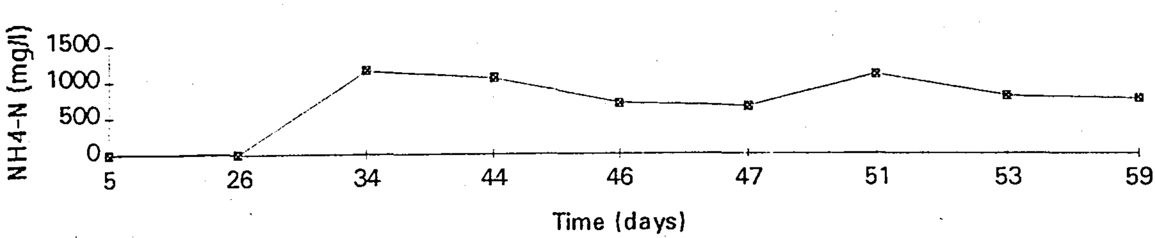


Figure 8.5.23 Ammonuim Concentrations in Lysimeter 2

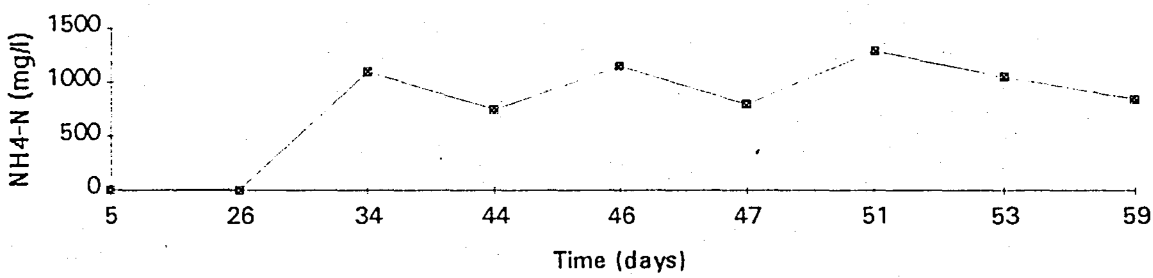


Figure 8.5.24 Ammonium Concentrations in Lysimeter 3

Because the nitrogen transfer is a combination of carbon and chloride transport processes, it was less defined when compared with chloride emissions.

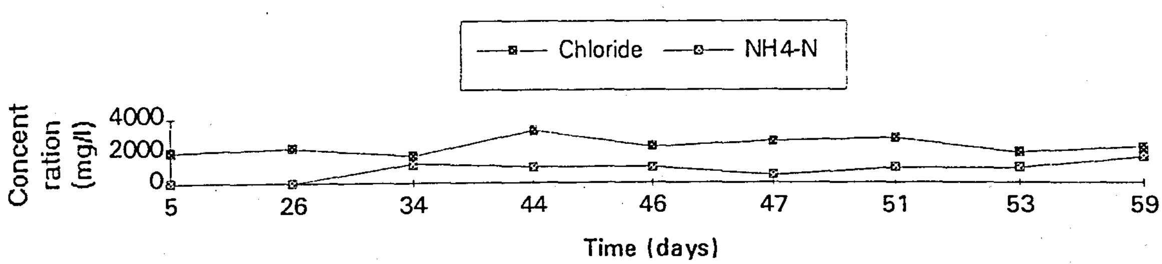


Figure 8.5.25 A Comparison of Cl-NH4 Concentrations in Lysimeter 1

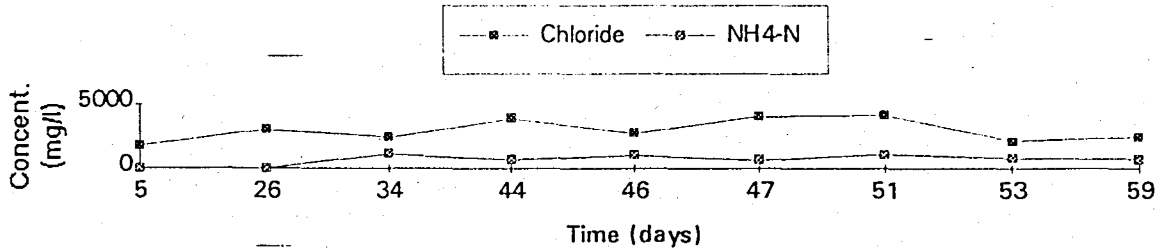


Figure 8.5.26 A Comparison of Cl-NH4 Concentration in Lysimeter 2

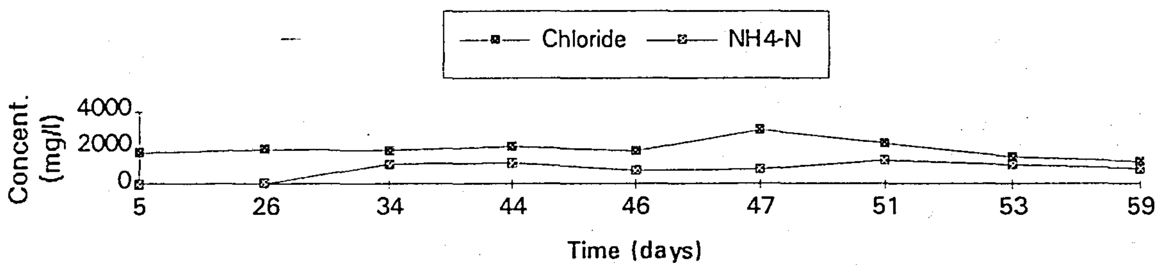


Figure 8.5.27 A Comparison of Cl-NH4 Concentration of Lysimeter 3

AOX analyses were conducted on the leachate samples of Lysimeter 1. The practice of recirculation of leachate in Lysimeter 2 and 3 did not enable the collection of necessary amounts for the analyses.

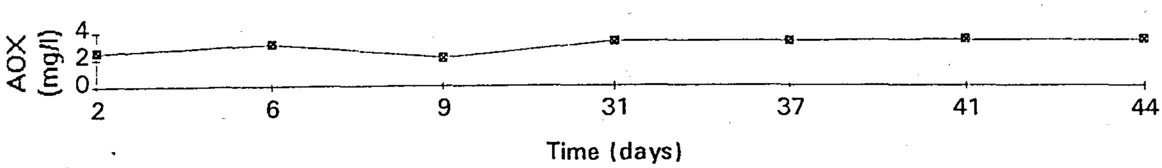
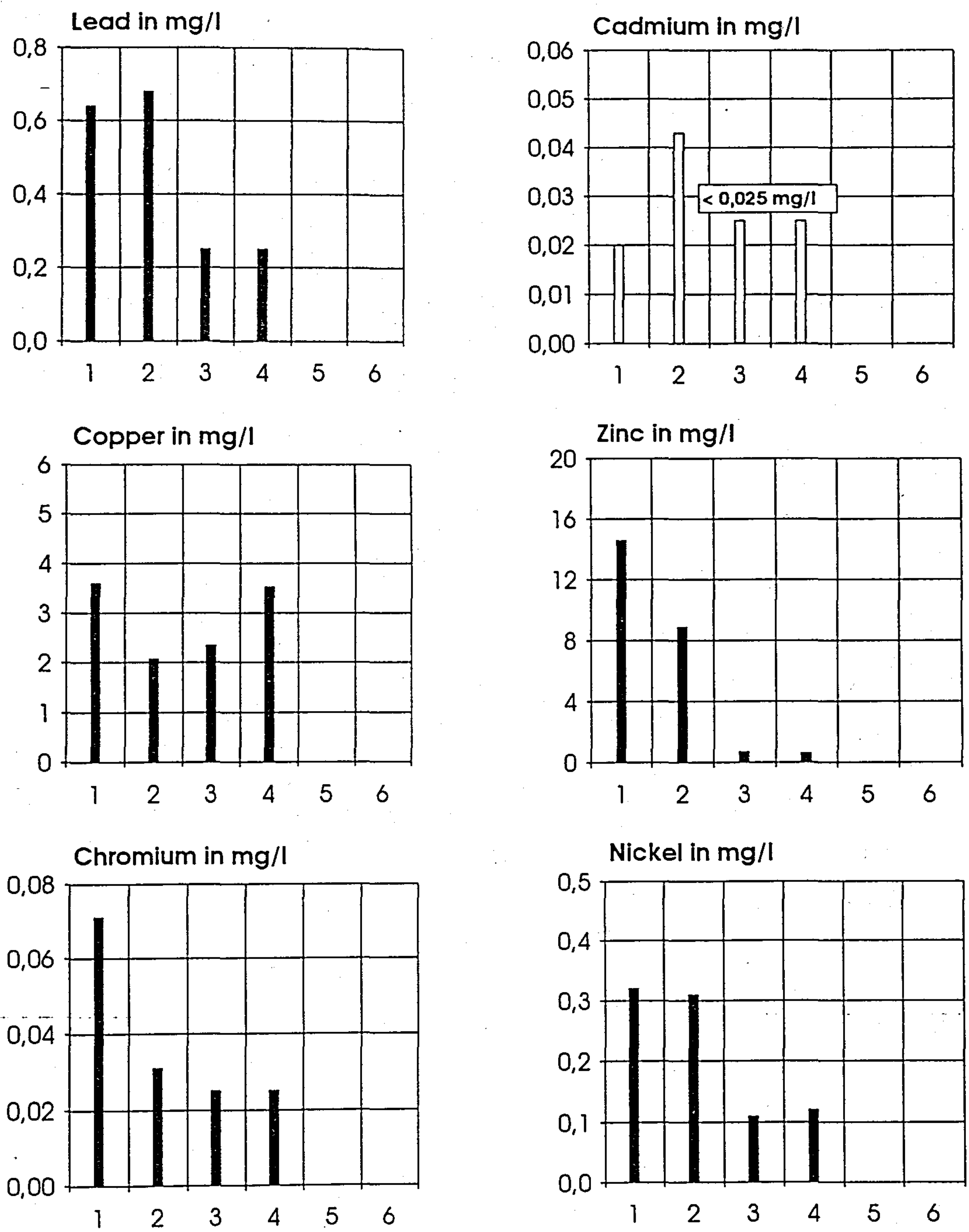
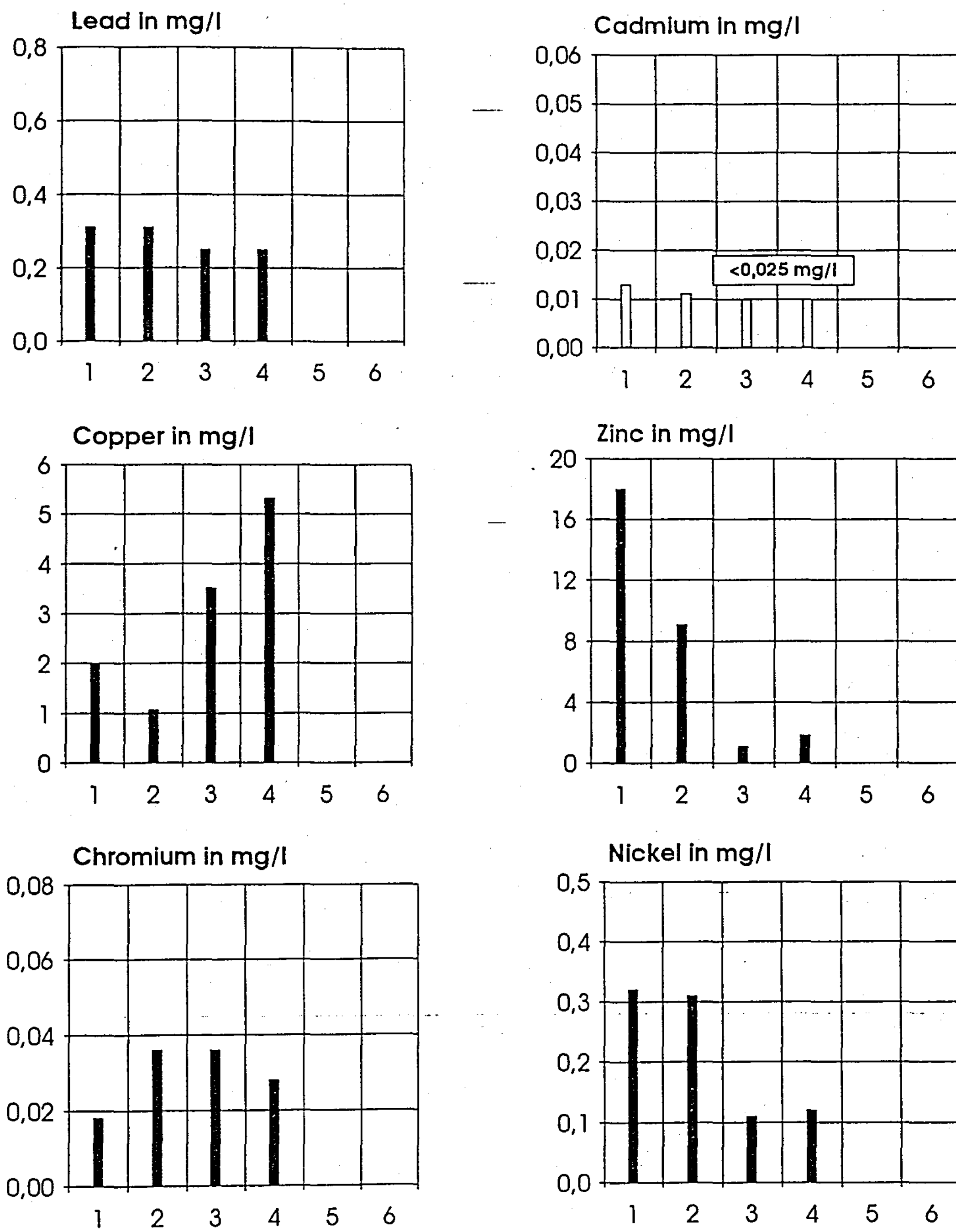


Figure 8.5.28 AOX Concentrations in Lysimeter 1



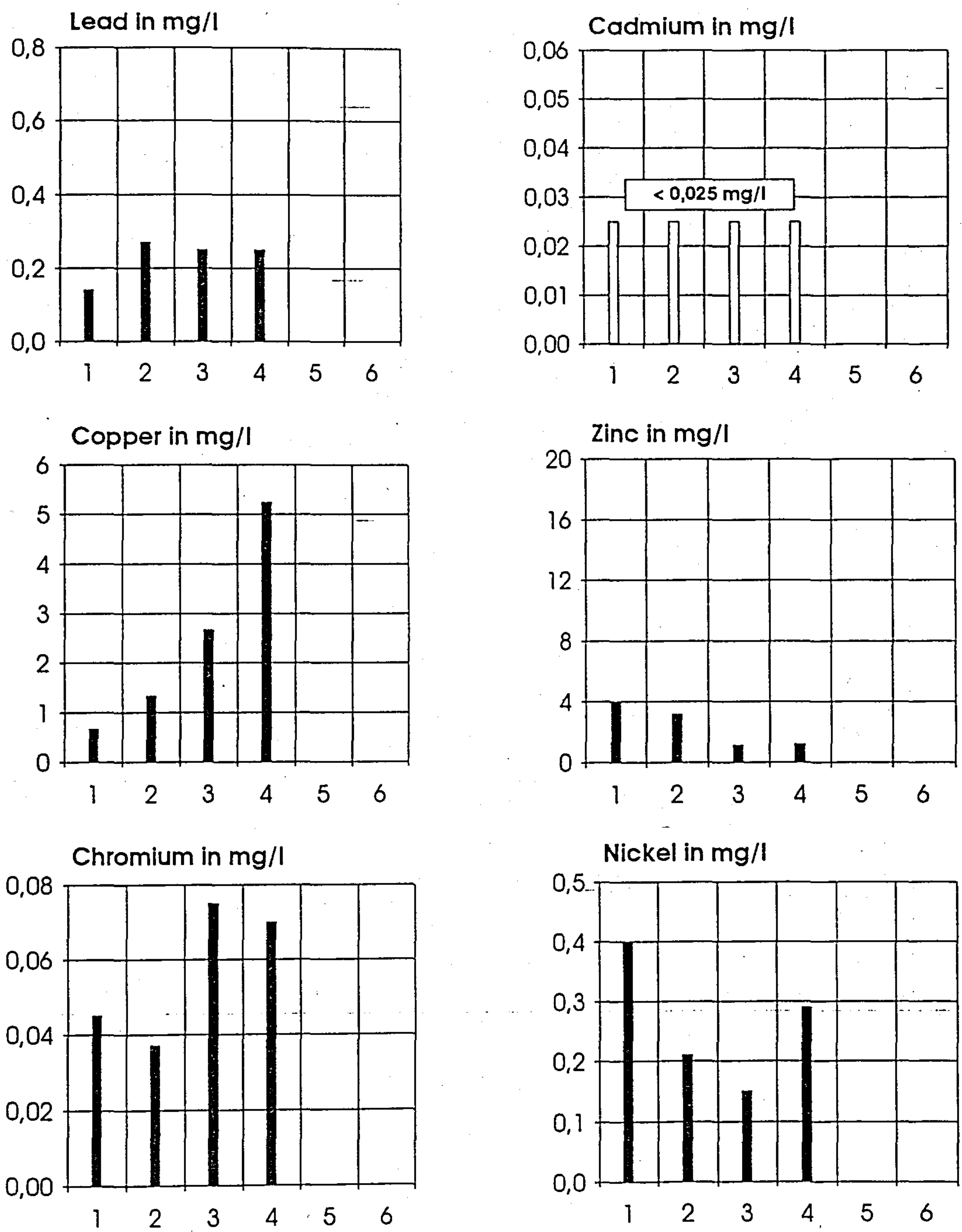
Lysimeter 3 (with leachate recirculation, aride conditions)
Heavy metal-concentrations in the leachate for four sampling times

Sample 1: May 6, 1993 **Sample 2: May 11, 1993**
Sample 3: June 21, 1993 **Sample 4: June 22, 1993**



Lysimeter 1 (without leachate recirculation, normal conditions)
Heavy metal-concentrations in the leachate for four sampling times

Sample 1: May 6, 1993 **Sample 2: May 11, 1993**
Sample 3: June 21, 1993 **Sample 4: June 22, 1993**



Lysimeter 2 (with leachate recirculation, normal conditions)
Heavy metal-concentrations in the leachate for four sampling times

Sample 1: May 6, 1993 Sample 2: May 11, 1993
Sample 3: June 21, 1993 Sample 4: June 22, 1993

Inspection of the data on heavy metal concentrations showed that there was no remarkable difference in the heavy metal contents of the lysimeters. The effects of sulfide precipitation were evident with Cd and Zn in all three lysimeters. It was considered that in the removal of precipitates, leachate recycle would play a significant role.

Table 8.5.5 Heavy Metal Concentrations in the Lysimeters

Lysimeter 1	Concentrations ,mg/l			
Cd	0.013	0.011	<0.025	<0.025
Cr	0.018	0.036	0.036	0.028
Cu	2.0	1.07	3.51	5.32
Ni	0.32	0.19	0.11	0.12
Pb	0.31	0.31	<0.25	<0.25
Zn	18.0	9.06	1.10	1.81
Lysimeter 2				
Cd	<0.01	<0.01	<0.025	<0.05
Cr	0.045	0.037	0.075	0.070
Cu	0.67	1.33	2.67	5.24
Ni	0.33	0.21	0.15	0.29
Pb	0.64	0.27	<0.25	<0.25
Zn	14.6	3.16	1.12	1.22
Lysimeter 3				
Cd	<0.02	0.043	<0.025	<0.025
Cr	0.071	0.031	<0.025	<0.025
Cu	3.60	2.08	2.34	3.53
Ni	0.33	0.16	0.11	0.15
Pb	0.64	0.68	<0.25	<0.25
Zn	14.6	8.86	0.70	0.63

The gas production in the lysimeters increased in accord with reduction in volatile acids and increase in pH. During periods of acid formation, little methane gas was formed. Thereafter, gas production accelerated. This accelerated gas production coincided with the period of rapid removal of readily available organics.

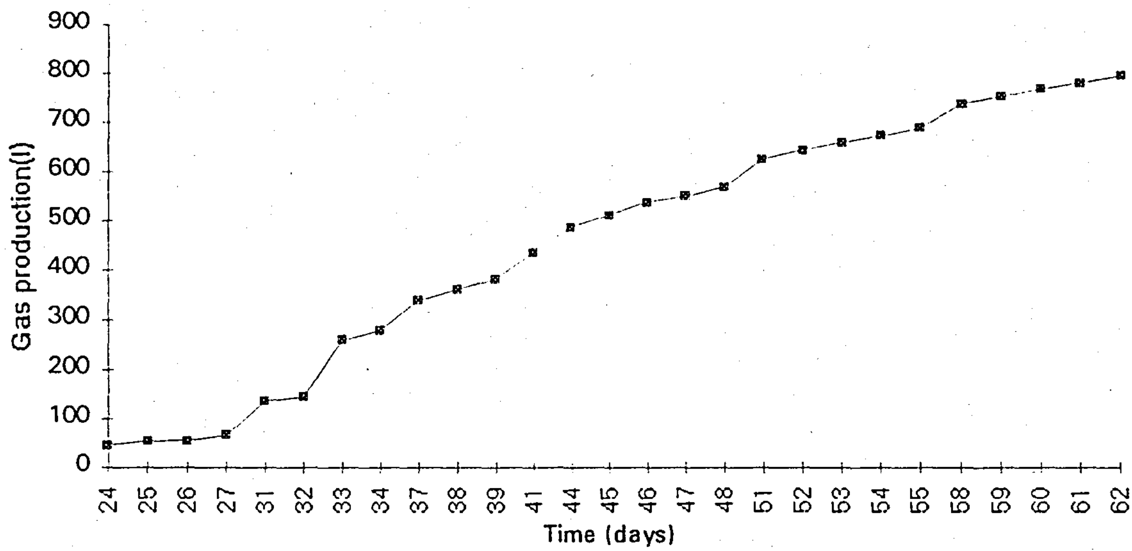


Figure 8.5.29 Gas Production in Lysimeter 1

The highest gas production was observed in Lysimeter 2. The gas drum counter for Lysimeter 3 was replaced three times to ensure accurate readings. Because gas measurements were resumed (May 13,1993) at a stage when both of the Lysimeters had produced approximately 40 liters, the gas yield from Lysimeter 3 reached as high as 748 liters. Gas production in Lysimeter 1 was initially parallel to the production in Lysimeter 2. After the introduction of leachate recycle, Lysimeter 2 generated higher quantities.

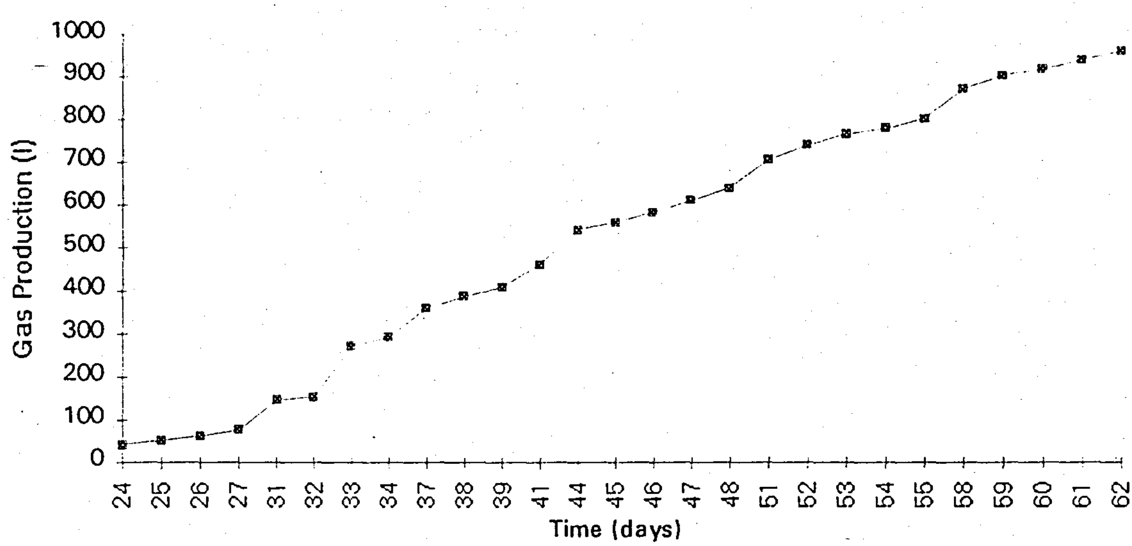


Figure 8.5.30 Gas Production in Lysimeter 2

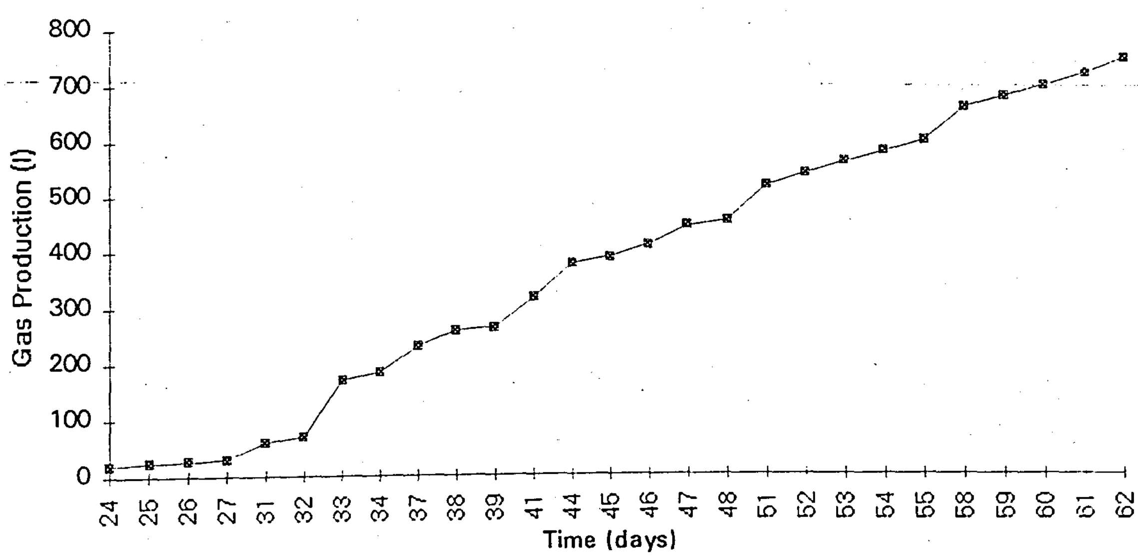


Figure 8.5.31 Gas Production in Lysimeter 3

Gas composition measurements were conducted to differenciate the phases of stabilization in the lysimeters. Carbon dioxide composition in Lysimeter 1 was sligthly higher than in the other lysimeters. The gas compositions were reported as 60 to 65 percent CH_4 and 35 to 40 percent CO_2.

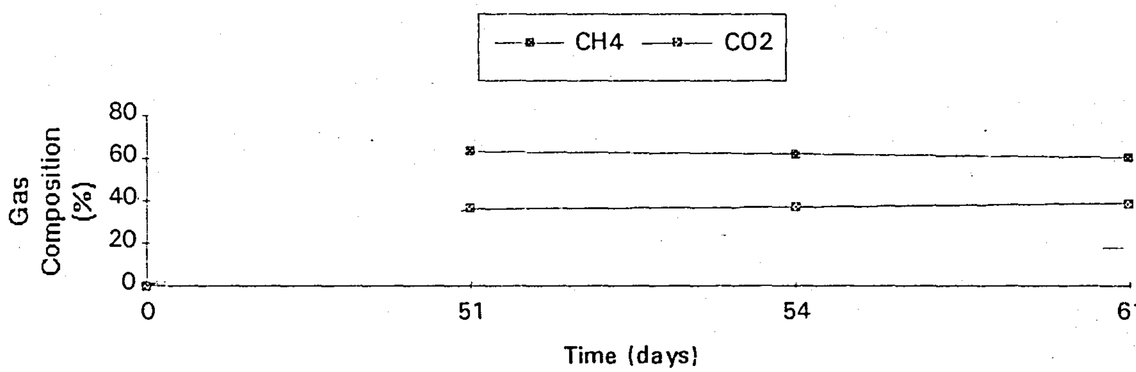


Figure 8.5.32 Gas Composition in Lysimeter 1

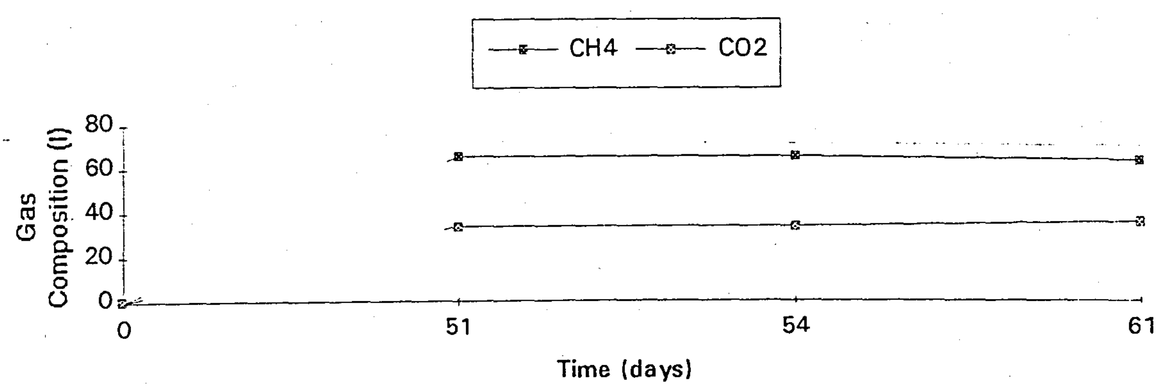


Figure 8.5.33 Gas Composition in Lysimeter 2

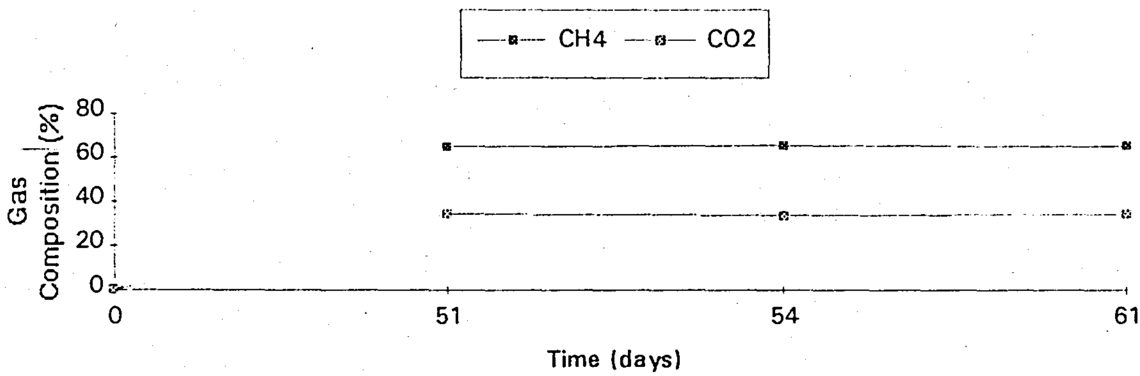


Figure 8.5.34 Gas Composition in Lysimeter 3

8.6 Discussion

The results obtained during the study demonstrated the following;

1.The mode of rainfall addition practiced was not found to be realistis in the sense that uniform amounts of rainfall in terms of tap water were added daily to the lysimeters. However, the daily uniform precipitation values were assigned to the lysimeters in order to provide an easy interptretation of data.

2.The laboratory lysimeters did not perform under their field capacity level because the mode of rainfall addition in terms of equivalent liters of tap water was not sufficient to simulate the natural conditions encountered in the sanitary landfills.

3.Leachate recirculation was not conducted in a continous manner. Therefore, a somewhat erratic display of leachate quality was observed.

4.Information regarding the movement of water/leachate within the lysimeters was limited. It was assumed that the majority of the wastes produced was as a result of the wastes. As the decompostion process accelerated, the absorption capacity of the refuse increased.

5.Interpretations on the moisture content and the decompostion rate of the refuse and as well as the prevailing temperature levels within the lysimeters was hard to establish.

6.The decompostion process was somewhat influenced in a negative manner due to maintaining keeping the system under the pressure of the produced gases.

7.Because the gas compostions were analyzed for CH4 and C02, the amounts of vapor content present within the lysimeters were not measured. However, with an installation of a gasometer to the system, more realistic values can be obtained. It is recommended that for the further study on these lysimeters a manometer connection would lead to a better understanding of the uncertainities faced.

8.7 Conclusion

Potential problems associated with the 'burial' of solid wastes have been reported as early as 1930's. Since then, deficiencies in waste definition and understanding of the impacts of environmental parameters on sanitary landfills have led to improper management of landfill leachate. At the same time, the potential for production of leachate has received considerable attention owing to the environmental consequences associated with its migration and thereby, representing a threat to groundwater systems. In response, rational and economically well grounded developments for control, containment and treatment of leachate have been proposed and initiated.

In accordance with these developments, leachate recycle as containment and recycle of leachate onto/into landfill body has been recognized as an in situ treatment mode. Studies conducted on leachate recycle showed that recycling induced a potential for increasing the rate of biological stabilization. It has been estimated that the normal 20 to 30 year period of organic stabilization could be 'shortened' to a few years with leachate recirculation. Therefore, leachate recirculation has been proposed to be adopted as a means of 'shortening' the time needed for biological stabilization and a management scheme with respect to leachate quality -quantity and energy recovery.

However, it has been determined that these advantages of leachate recirculation are accompanied by several disadvantages such as:

_Increased hydraulic loading via recycling may lead toan increased potential for groundwater contamination,

_Multiple administration of recycling may result in higher concentrations of salts and heavy metals in the leachate.

The results of this study have illustrated that biological stabilization was not enhanced to a great extent with leachate recirculation. Lysimeter 3 demonstrated that the availability of moisture to a certain amount is necessary for the acceleration of the conversions and interactions leading to biological stabilization. However, increased water addition within the lysimeter 2 did not reflect a significant benefit over the applied process. Yet, the phases of biological stabilization were observed to proceed as expected with the Turkish composition of solid waste. The high percentage of organic wastes showed that with the 'minimum' moisture input, the initially high organic loads could be reduced to the reported ranges encountered on leachate quality.

Although the gas production event has been referred to be unpredictable, daily recirculation has led to the production of high quantities of gas which could be cited as an advantage associated with energy recovery.

No potential inhibition was observed as far as heavy metal concentrations were concerned. The presence of sulfides were determined to be a significant factor in the precipitation mechanism. It was also established that the removal of precipitates is possible with the recycle regime. Had there been a co-disposal of heavy metal sludge with the refuse, the effects of leachate recirculation on the removal of heavy metals would have been described more adequately.

Within the frame of preliminary results, the leachate recycle scheme as a treatment mechanism did not prove to be feasible. It was concluded that minimum amounts of moisture within the landfill body lead to a steady form of biological stabilization and that the influence of recycling may only be a partial purification in terms of leachate treatment.

It has been decided that in order to establish an understanding on the behavior of Turkish solid waste with regard to biological stabilization, further investigations need to be undertaken on large scales. With the implementation of new recycling-reuse techniques, the composition of Turkish solid waste should brought to lower quantities in search of efficient leachate containment and treatment methods.

8.8 Recommendation

Upon the understanding of leachate generation and transfer rates, alleviating the problems occurring during the design and operation of leachate management schemes is important. For on site applications, the leachate recirculation process has been recommended as a method which could be implemented in predicting the ultimate behavior of landfill leachates. The co-disposal of industrial wastes with municipal refuse and the recirculation of the concentrate of treated leachate by reverse osmosis on or into the landfill have been presenting new management oppurtunities. Moreover, a two stage recirculation process, where leachate removed from a newer landfill section is recirculated in an older stabilized section has been providing a better basis to reduce the uncertanities within the landfill body.

The results presented in this study have laid the grounds for a more detailed investigation on the feasibility of leachate recirculation as a management option.

REFERENCES

[1] Canziani,R.,"Landfill Hydrology and LeachateProduction," in T.H. Christensen, R. Cossu and R. Stegmann (Eds.),*Sanitary Landfilling: Process, Technology and Environment Impact,* pp. 185-211, London: Academic Press, 1989.

[2] Dass, P., Tamke, G.R., and C.M. Stoffel, "Leachate Production at Sanitary Landfill Sites," *Journal of Environmental Engineering Division*, Vol. EE6, pp. 981-988, December 1977.

[3] Ranay, S., "Project Seeks to Boost Methane Production,"*World Wastes*, No. 6, pp.16-17, June 1992.

[4] Corbitt, A.R., *Standard Handbook of Environmental Engineering*, McGraw-Hill, New York, 1990.

[5] Mott, H.V., Hartz, K.E., and D.R. Yonge, "Metal Precipitation in Two Landfill Leachates," *Journal of Environmental Engineering,* Vol. 113, No.3, pp.476-485, June 1987.

[6] Korfiatis, G.P., Demetrocopoulos, A.C., Bourodimos, E.L., and E.G. Nawy, " Moisture Transport in a Solid Waste Column," *Journal of Environmental Engineering*, Vol.110, No.4, pp. 780-796, August 1984.

[7] Metry, A.A, and M.E. Harris, "A New Concept for Treating Leachate and Containment Runoff Waters from a Taconite Transhipment Facility," *Proceedings of the Thirty First Industrial Waste Conference*, Purdue Universtiy, Indiana, 4-6 May 1976, Vol.1, pp.227-239, Michigan, Ann Arbor science Publishers, 1976.

[8] Keenan, J.D., Steiner, R.L., and A.A. Fungarolli, "Landfill Leachate Treatment," *Journal of Water Pollution Control Federation*, Vol.56, No. 1, pp.27-33, January 1984.

[9] Reinhart, D.R., The Sources of Hazardous Pollutants Emitted from Municipal

Solid Waste Landfills, Gulf Coast Hazardous Substance Research Center, Beamont, 1991.

[10] Bull, P.S., Evans, J.V., Wechsler, R.M., and K.J. Cleland, "Biological Technology of the Treatment of Leachate from Sanitary Landfills," *Water Resources*, Vol. 17, No. 11, pp. 1473-1481, 1983.

[11] Harmsen,J., "Identification of Organic Compounds in Leachate from a Waste Tip," *Water Resources*, Vol.17, No.6, pp. 699-705, 1983.

[12] Robinson, H.D. and G. Grantham, "The Treatment of Landfill Leachates in On Site Aerated Lagoon Plants," *Water Resources*, Vol. 22, No.6, pp.737-747, 1988.

[13] Thomanetz, E., Bemerkungen zur Wasserbilanz der Sonderabfalldeponie Malsch, PR, Universitat Stuttgart, 1988.

[14] Pohland, F.G. and J. P. Gould, "Codisposal of Municipal Refuse and Industrial Waste Sludge in Landfills,", *Water Science Technology*, Vol. 18, No. 12, pp. 177-192, 1986.

[15] Chian, E.S.K., and F.B. DeWalle, "Characterization of Soluble Organic Matter in Leachate," *Environmental Science and Technology*, Vol.11, No.2, pp. 158-162, February 1977.

[16] Gibs, J., Schoenberger, R.J., and I.H. Suffet, "A Simplified Buffer Capacity Model for Sanitary Landfill Leachate, "*Water Resources*," Vol. 16, pp. 699-705, 1982.

[17] Gourdon, R., Comel, C., Vermande, P., and J. Veron, "Fractionation of the Organic Matter of a Landfill Leachate before and after Aerobic or Anaerobic Biolgical Treatment," *Water Resources*, Vol. 23, No. 2, pp.167-173, 1989.

[18] Peavy, H.S., Rowe, D.R. and G. Tchobanoglous, *Environmental Engineering*,

Mcgraw-Hill, Singapore, 1985.

[19] Curi, K., "La prise en compte du potentiel dechets a revaloriser au niveau d'une metropole: relexion sur les procede techniques de cette revalorisation," *Proceedings of Actes du Colloque Gestion des Rechets Managers dans les pays en developpement*, Paris, 9-11 September 1987, pp.199-205, Paris, Rexcop, 1987.

[20] Schmidell, W., Craveiro, A.M., Pers, C.S., Hirata, Y.S., and R.F. Verella, "Anaerobic digestion of Municipal Solid Wastes," *Water Science Technology*, Vol. 18, No. 12, pp. 163-175, 1986.

[21] Zinder, S.H., "Thermophilic Waste Treatment Systems," in T. Brock (Eds.) *Wiley Series in Ecological and Applied Microbiology*, pp. 257-275, New York: John Wiley and Sons, 1986.

[22] Harris, J.C., "Rate of Hydrolysis," in Lyman, Raehl, and Rosenblatt (Eds.), *Handbook of Chemical Property Estimation Methods*, pp. 7.1-7.14, Washington D.C., American Chemical Society, 1990.

[23] Sawyer, C.N. *Chemistry for Environmental Engineering*, McGraw-Hill, New York, 1978.

[24] Machie, R.I., White, B.A., and M.P. Bryant "Methanogensis, Biochemistry," in J. Lederberg (Ed.), *Encyclopoedia of Microbiology*, Volume 3, pp. 97-102, California: Academic Press, 1992.

[25] Singleton, P., and D. Sainsburg, *Dictionary of Microbiology and Molecular Biology*, Academic Press, California, 1978.

[26] Farquar, G.J., and J.F. Sykes, "Landfill Leachate Migration in Soil," *Proceedings of Leachate Managemnet Seminar*, Toronto-Ontario, 20-21 November, pp.157-195, Ontario, University of Toronto, 1980.

[27] Ehrig, H.J., "Leachate Quality ," in T.H. Christensen, R. Cossu and Stegmann (Eds.), *Sanitary Landfilling: Process, Technology and Environmental Impact*, pp.

213-229, London: Academic Press, 1989.

[28] Pohland, F.G.,"Leachate Recycle as a Management Seminar, Toronto-Ontario, 20 -21 November 1980, pp.39-62, Ontario, University of Toronto, 1980.

[29] Sulfita, J.M., Gerba, C.P., Ham, R.H., Palmisano, A.C., Rathje, W.L., "Decomposition Rates in Landfills," *Public Works*, No. 1, pp. 78-81, 1993.

[30] Dorau, W., "Deponiesckerwasser Behandlung," *Proceedings of Symposium von Umweltbundesamt*, Aachen, 9-11 April 1987, pp.55-70, Aachen, Erich Schmidt Verlag, 1987.

[31] Schultz, B., and J. Kjeldsen, "Screening of Organic Matter in Leachates from Sanitary Landfills Using Gas Chromatogtaphy Combined with Mass Spectrometry," *Water Resources*, Vol. 20, No.8, pp. 965-970, 1986.

[32] Doedens, H., and K.Cord Landwehr, "Leachate Recirculation," in T.H. Christensen, R. Cossu and R. Stegmann (Eds.), *Sanitary Landfilling: Process, Technology and Environmental Impact*, pp. 231-249, London: Academic Press, 1989.

[33] Titlebaum, M., "Organic Carbon Content Stabilization through Landfill Leachate Recirculation," *Journal of Water Pollution Control Federation*, Vol. 54, No. 5, pp.428-433, May 1982.

[34] Robinson, H.D., and M.H. Luo, "Characterization and Treatment of Leachates from Hong-Kong Landfill Sites," *Journal of IWEM*, Vol.5, pp. 326-335, June 1991.

[35] Ward, R.F., "Temperature Efeects:Methane Generation from Landfill Samples," *Journal of Environmental Engineering*, Vol. 109, No. 4, pp. 979-982, 1983.

[36] Stegmannn, R. "Leachate Treatment in Europe," *Proceedings of Leachate Management Seminar*, Toronto- Ontario, 20-21 November 1980, pp. 65-94,

Ontario:University of Toronto, 1980.

[37] Sattler, K.,and J.Emberger, *Behandlung fester Abfalle*, Vogel Verlag, Wurzburg, 1992.

[38] Blight, G.E., Bull, J.M., and J.J Blight, "Moisture Distribution in Sanitary Landfills," *Procedings of the Third International Landfill Symposium,* Cagliari, Italy, 14-18 October 1991, pp. 813-822, Cagliari, CISA, 1991.

[39] Baldi, M., Collivignerelli, C. and F. Conti," Leachate Production from an Industrial Waste Landfill: Approach Method and Experimental Results," *Proceedings of the Third Internal Landfill Symposium*, Cagliari-Italy, 14-18 October 1991, pp.833-846, Cagliari, CISA, 1991.

[40] Pohland, F.G. and S.R. Harper, Critical Review and Summary of Leachate and Gas Production from Landfills, Final Report. U.S. Environmental Protection Agency, Project No. CR-809997, Washington, D.C., 1985.

[41] Albaiges, J. Casado, F., and F. Ventura, "Organic Indicators of Groundwater Pollution by a Sanitarty Landfill, *Water Resources*, Vol. 20, No. 9, pp. 1153-1159, 1986.

[42] Sykes, J.F., Soyupak, S, and G.J. Farquar, "Modelling of Leachate Organic Migration and Attenuation in Groundwater Below Sanitary Landfills," *Water Resources Research*, Vol. 18, No. 1, pp. 135-145, February 1982.

[43] Wong, K.V., Sengupta, S., Dasgupta, D., Daly, E.L., Nemerow,N., and H.P. Gerrish, "Heavy Metal Migration in Soil lLeachate Systems," *Biocycle,* No.1, pp. 30-33, 1983.

[44] Pohland, F.G. and S.R. Harper, Critical Review and Summary of Leachate and Gas Production from Landfills, Final Report. U.S. Environmental Protection Agency, Project No. CR-809997, Washington, D.C., 1985.

[45] Vincent, F., Beaudoing, G.,and F. Colin, "Waste Behavior Modelling: A

Numerical Model to Describe the Flow,Transport and Biodegradation Processes," *Procedings of the Third International Landfill Symposium*, Cagliari-Italy, 14-18 October 1991, pp. 874-855 , Cagliari, CISA, 1991.

[46] Cossu, R., Stegmann, R, Andreottola, G., and P. Cannas,"Biolgical Treatment," in T.H. Chreistensen, R.Cossu and R. Stegmann (Eds.), *Sanitary Landfilling: Process, Technologhy and Environmental Impact,* pp. 251-283, London:Academic Press, 1989.

[47] Stegmannn, R. "Leachate Treatment in Europe," *Proceedings of Leachate Management Seminar*, Toronto- Ontario, 20-21 November 1980, pp. 65-94, Ontario:University of Toronto, 1980.

[48] Magnuson, A., "MSW Transportation," *MSW Management,* Vol. 3, No. 1, pp. 44-50, February 1993.

[49] Robinson, H.D., "On-Site Treatment of Leachates from Landfill Wastes," *Journal of IWEM*, Vol.4, pp. 78-87, February 1990.

[50] Chian, E.S.K., "Stability of Organic Matter in Landfill Leachates," *Water Research*, Vol. 11, pp. 225-232, 1977.

[51] Boyle,W.C.,and R.K. ham, "Biolgical Treatability of Landfill Leachate," *Journal of Water Pollution Control Federation*, Vol. 52, No.2, pp. 860-872, May 1974.

[52] Cameron, R.D., and F.A. Koch, "Trace Metals and Anaerobic Digestion of Leachate," *Journal of Water Pollution Control Federation*, Vol. 52, No. 2, pp. 282-292, February 1980.

[53] Thornton, R.J., and Blanc, F.C., "Leachate Treatment by Coagulation and Precipitation," *Journal of Environmeental Engineering Division*, EE4, pp. 535-543, August 1973.

[54] Ehrig, H.J., "Physicochemical Treatment," in T.H. Cristensen, R. Cossu and R. Stegmann (Eds.), *Sanitary Landfilling: Process, Technology and Environmental Impact*, pp. 285-297, London: Academic press, 1989.

[55] Thomanetz, E., "Stand der Technik bei der Deponiesickerwasser behandlung und Entwicklungen," *Proceedings of Zeitmasse Deponietechnik*, Stuttgart-Germany, 9-10 March 1993, pp. 1-9, Stuttgart, 1993.

[56] Quasim, S.R., and J.C. Burchinal, "Leaching from Simulated Landfills," *Journal of Water Pollution Control Federation*, Vol. 42, No.3, pp. 371-379, March 1970.

[57] Fungaroli, A.A., and R.l. Steiner , "Laboratory Study of the Behavior of a Simulated Landfill," *Journal of Water Pollution Control Fedration*, Vol. 42, No.3, pp. 371-379, March 1970.

[58] Raveh, A., and Y. Avnimelech, "Leaching of Pollutants from Sanitary Landfill models," *Journal of Water Pollution Control Federation*, Vol. 51, No. 11, pp.2705-2716, 1979.

[59] Ellis, J., "A Convenient parameter for Tracing Leachate from Sanitary Landfills," *Water Research*, Vol. 14, pp. 1283-1287, 1980.

[60] McDougall, W.J., Fusco, R.A., and R.P. O'Brien, "Containment and Treatment of the Love Canal Leachate," *Journal of Water Pollution Control Federation*, Vol. 52, No. 12, 2914-2924, December 1980.

[61] Wilson, D.C., Young, P.J., Hudson, B.C., and G. Baldwin, "Leaching of Cadmium from Pigmented Plastics in a Landfill Site," *Environmental Science and Technology*, Vol. 16, No. 9, pp. 560-565, 1982.

[62] Collins, H.J., and P. Spillmann, "Lysimeters for Simulating Sanitary Landfills," *Journal of Environmental Engineering Division*, EE5, pp. 852-863, 1982.

[63] Ham, R.K., and Bookter, J., "Decomposition of Solid Waste in Test Lysimeters,"

Journal of Environmental Engineering Division, EE6, pp. 1147-1171, 1982.

[64] Robinson, H.D., and P.J. Marris, "The Treatment of Leachate from Domestic Wastes in Landfills," *Water Resources*, Vol. 17, No. 11, pp. 1537-1548, 1983.

[65] Knox, K., "Leachate Treatment with Nitrification of Ammonia," *Water Resources*, Vol. 19, No. 7, pp. 895-904, 1985.

[66] Kelly, H.G. "Pilot testing for Combined Treatment of Leachate from a Domestic Waste Landfill Site," *Journal of Water Pollution Control Federation,* Vol. 59, No. 5, pp. 254-261, May 1987.

[67] Frimmel, F.H., and M. Weis, "Ageing Effects of High Molecular Weight Organic Acids which can be Isolated from Landfill Leachates," *Water Science Technology*, Vol. 23, pp. 419-426, 1991.

[68] McAllister,P.J., Bhamidimerri, S.M., Chong, R, and G.J. Manderson,"Biological Treatment of a Landfill Leachate Containing Phenoxy Herbicides and Chlorophenols," *Water Science Technology,* Vol. 23, pp. 413-418, 1991.

[69] Reinhart, P.R., and F.G. Pohland, "The Assimilation of Organic Hazardous Wastes by Municipal Solid Waste Landfills," *Journal of Industrial Microbiology*, Vol. 8, No. 3, 1992.

[70] Ahmed, S., Khanbilverdi, R.M., Fillos, J., Gleason, P.J., "Two Dimensional Leachate Estimation Through Landfills," *Journal of Hydraulic Engineering*, Vol. 118, No. 2, pp. 306-322, 1990.

[71] Lin, C.Y., "Anaerobic Digestion of Landfill Leachate," *Water S.A.*, Vol. 17, No. 4, pp. 301-306, 1992.

[72] Niininen, M., Kalliokoski, P., and T. Eskellinen, "Co-Treatment of Landfill Leachate and Domestic Sewage in Activated Sludge Plant," *Proceedings of International Conference on Environmental Pollution*, Lisbon- Portugal, 15-189

April 1991, pp. 314-321, Geneva, Inder Science Enterprises, 1991.

[73] Dienemann, E.A. Kossun, D.S., and R.C., Ahlert, "Evaluation of Serial Anaerobic/Aerobic Packed Bed Bioreactors for Treatment of a Superfund Leachate, *Journal of Hazardous Matter*, Vol. 23, No.1, pp. 21-42, 1991.

[74] Carley, B.N., and D.S. Mavinic, "The Effects of External Carbon Loading in Nitrification and Denitrification of a High Ammonia Landfill Leachate," *Journal of Water Pollution Control Federation*, Vol. 63, No. 1, pp. 51-59, 1991.

[75] Young, P.J., Baldwin, G., and D.C. Wilson, "Attenuation of Heavy Metals within Municipla Waste Landfill Sites, " *Proceedings of Hazardous and Industrial Waste Management*, Philadelphia, U.S.A., 7-10 March 1984, pp. 193-212, Baltimore, American Society for Testing and Materials, 1984.

[76] Matsuoto, T., Tanaka, N., and K. Koyama, "Stabilization Mechanism of Leachate from Semi-Aerobic Sanitary Landfills of Organics Rich Waste," *Proceedings of the Third International Landfill Symposium*, Cagliari-Italy, 14-18 October 1991, pp. 875-887, Cagliari, CISA, 1991.

[77] Chian, E.S.K., DeWalle, F.B., and E. Hammerberg, "Effect of Moisture Regime and other Factors on Municipal Solid Waste Stabilization," *Proceedings of Leachate Management Seminar,* Ontario-Toronto, 20-21 November 1980, pp. 73-85, Ontario, University of Toronto, 1980.

[78] Curi, K., and G. Kocasoy, "Waste Recovery with Sorting Technologies in the City of Istanbul," in K.J. Thome-Kozmiensky (Ed.), *Recycling in Developing Countries*, pp. 100-104, Berlin, Verlag E. Freitag, 1982.

[79] Curi, K., "Solid Waste Recycling in the City of Istanbul and Possible Application in other Developing Countries," in K.J. Thome-Kozmiensky (Ed.), *Waste Management in Developing Countries 1*, pp. 72-77, Berlin, Verlag E. Freitag, 1986.

[80] Curi, K., "Katı Atiklar," *Çevre ve Insan*, No. 9, pp. 7-17, September 1988.

[81] Curi, K., and Albukrek, M., "Wastewater Treatment by an Accelerated Anaerobic Process," *Proceedings of the Ninth Turkish, German, Polish Environmental Engineering Symposium*, Istanbul-Turkey, 5-7 October 1992, pp.51-74, Istanbul, Bogazici University, 1992.