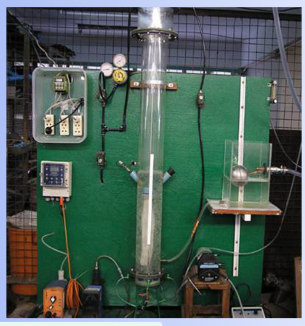
ARRPET

State of the Art Review
Landfill Leachate Treatment

Sustainable Solid Waste Landfill Management in Asia



Tongji University

Asian Institute of Technology

State of the Art Review

Landfill Leachate Treatment

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DISCLAIMER

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Preface

The objective of the Asian Regional Research Programme on Environmental Technology (ARRPET) funded by Swedish International Development Cooperation Agency (Sida) is to conduct regional research on environmental concerns in Asia which includes wastewater, air pollution, solid and hazardous wastes. ARRPET is coordinated by the Environmental Engineering and Management Program of the Asian Institute of Technology (AIT), Thailand and involves National Research Institutes (NRIs) in eight countries.

This publication "State of the Art Review Landfill Leachate Treatment" is intended to provide environmental researchers and technologists a background on the existing scenario of municipal waste landfill leachate treatment practices in Asia as well as in industrialized nations. As the first phase of ARRPET project on Sustainable Solid Waste Landfill (SWLF) Management in Asia, AIT and one of the NRIs under the project, the National Engineering Research Center for Urban Pollution Control, Tongji University (TU), China have jointly studied and reviewed the existing practices of landfill leachate management.

This report presents an extensive literature review concerning landfill practices, leachate generation, characterization and development of leachate treatment systems, which are biological, physical and chemical. An overview of successful treatment systems is provided to the reader that can be adopted for leachate treatment. Treatment practices and sequences are critically examined with case studies from operating plants that have successfully overcome problems and constraints encountered due to the varied characteristics of leachate generated from different landfills.

This publication is expected to enable the practicing environmentalists to gain an in-depth knowledge on leachate treatment practices, which may be applied to solve problems emerging during operation of leachate treatment plants. The upgradation of existing dumpsites and landfills in Asia would require installation of leachate treatment facilities for which this review could be a path-lighter. Landfill leachate management is a key issue in integrated solid waste management practices for the preservation of the deteriorating environment for the posterity and protection of our ecosystem.

On behalf of our respective institutes, we would like to thank Sida for enabling this opportune research and to AIT and TU for the affable atmosphere to carry out this study. We look forward to the adoption of sound leachate management practices in the developing Asian countries.

In conclusion, we express our sincere gratitude to the following experts for critically reviewing this report and providing their valuable suggestions prior to its publication:

- Mr. Tang Jia-fu, Deputy Director of Environment Sanitation Department, Shanghai, China; and
- Mr. Biswadeep Basu, Project Manager, Environment, Southeast Asia Technology Co., Ltd. (SEATEC), Bangkok, Thailand.

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Executive Summary

Landfill leachate treatment is an integral part of municipal solid waste management that in turn has a skin-blood relationship with urban infrastructure. To ensure the protection of our ecosystem, environmental health, and foster sustainable development, the waste generated by the increasing urban population requires treatment and disposal in an environmentally sound manner. The municipal solid waste (MSW) from the urban habitat is disposed off in dumpsites (a crude form of disposal) or in sanitary or engineered landfills. The constituents of the MSW undergo biological and chemical degradation after disposal resulting in emissions of landfill gas and discharge of leachate, which is a highly polluted form of wastewater when discharged into the environment, and would cause potential damages to environmental health and the ecosystem.

As the developing countries are beginning to adopt modern solid waste management practices with the upgradation of the existing dumpsites and unsanitary landfills, leachate management would require a highly specialized approach in dealing with the constituent pollutants in it. This report intends to highlight the solid waste landfill management with respect to leachate generation, composition, characteristics and factors affecting them. It attempts to compare the above factors for different landfills in Asia and other developed regions thus providing an understanding of the factors involved in the production of leachate.

The first three chapters provide an overview of solid waste landfill management, leachate generation, composition and characteristics and leachate management. Leachate generation depends on a host of different factors, which contribute to the characteristics of the generated liquid from a landfill. Primarily these characteristics depend on the phases of degradation of the waste in a landfill, which follow a sequence of time. These characteristics also depend upon the landfill methods, composition, characteristics and age of the disposed waste, regional and seasonal variations and filling techniques (filling height, density, stabilization and pre-treatment, leachate collection system and the linings used for the landfill).

Chapter four deals with leachate treatment systems that can be effectually adopted such that the effluent can be safely discharged into the environment without any negative impacts to the ecosystem and environmental health. It discusses extensively biological treatment options. The processes described here are aerobic options (activated sludge processes, sequence batch reactor, rotating biological contactors, lagoons and trickling filters); anaerobic options (upflow anaerobic sludge blanket reactors); and nitrification and denitrification for the removal of excessive nitrogen, which is toxic to the environment. Different systems are taken into account in the study with comparisons of the successful ones found in literature.

Physical treatment systems discussed include activated carbon adsorption of organics and heavy metals and its applications. Membrane filtration is stressed as a recent development that has revolutionized the trend in leachate and wastewater technology with compact and efficient methods. Another physical system is evaporation suitable only in relatively dry tropical climates and arid regions.

Chemical treatment methods that are widely applicable for leachate management are coagulation and precipitation, chemical oxidation and reduction, and ammonia stripping. Chemical oxidation entails different agents using ozone, ultra-violet radiation and

hydrogen peroxide. A combination of the methods is often used for effective oxidation. The presence of a large quantity of ammonia nitrogen in leachate requires ammonia stripping.

The natural leachate treatment systems described in this report are leachate recirculation in landfill bioreactors and use of reed beds for polishing treated effluent. The other treatment systems available are co-treatment with municipal wastewater and combined treatment systems in which biological methods are used in conjunction with physical and chemical methods. A review of different combinations is provided based on the characteristics of the leachate. The use of membrane bioreactors is discussed to provide a synopsis of the recent trends in the leachate treatment systems. It can be seen from the descriptions that a single method cannot suffice but a flexible treatment sequence is required which can be easily adapted and modified when the characteristics of the leachate generated change over time based on the degradation phases of the waste in the landfill.

The fifth chapter describes case studies of leachate treatment plants in industrialized countries, which have successfully tackled a variety of problems generated by the changing characteristics of the leachate in different phases. These case studies provide a summary of the most common treatment sequences used and the efficiencies of each.

As a conclusion, the last chapter highlights the recent trends in leachate treatment systems that begin from the upstream of municipal waste generation to the downstream remedial measures, which include the natural systems, analysis of the constituents for deciding the treatment sequences and their modification based on the changes in characteristics.

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List of Abbreviations

AOX	Adsorbable Organic Halogens
ASP	Activated Sludge Process
BAC	Biological Activated Carbon
BOD	Biochemical Oxygen Demand
CEC	Cation Exchange Capacity
COD	Chemical Oxygen Demand
DO	Dissolved Oxygen
DOC	Dissolved Organic Carbon
EL	Engineered Landfill
FS	Fixed Solids
GAC-BFB	Granulated Activated Carbon Biological Fluidized Bed
HRT	Hydraulic Retention Time
MBP	Mechanical-Biological Pretreatment
MF	Microfiltration
mg/L	Milligrams Per Liter
MLSS	Mixed Liquor Suspended Solids
MLVSS	Mixed Liquor Volatile Suspended Solids
MSW	Municipal Solid Waste
mV	Millivolt
MW	Molecular Weight
MWW	Municipal Wastewater
N	Nitrogen
NF	Nanofiltration
OD	Open Dump
ORP	Oxidation-Reduction Potential
P	Phosphorus
PAC	Powdered Activated Carbon
PL	Pretreated Waste Landfill
RBC	Rotating Biological Contactor
RO	Reverse Osmosis
SBR	Sequencing Batch Reactor
SRT	Sludge Retention Time
SS	Suspended Solids
SVI	Sludge Volume Index
TEQ	Toxicity Equivalent
TOC	Total Organic Carbon
TS	Total Solids
UV	Ultra-Violet
UF	Ultrafiltration
VFA	Volatile Fatty Acid
VOC	Volatile Organic Compound
VS	Volatile Solids

Chapter 1

INTRODUCTION

Landfill leachate can broadly be defined as the liquid produced from the decomposition of waste and infiltration of rainwater in a landfill. It contains heavy metals, salts, nitrogen compounds and various types of organic matter. Generation of leachate occurs when moisture enters the refuse in a landfill, dissolves the contaminants into liquid phase and produces moisture content sufficient to initiate liquid flow. This leachate is a high-strength wastewater that has a major impact and influence on landfill design and its operation. Leachate varies from one landfill to another, and over space and time in a particular landfill with fluctuations that depend on short and long-term periods due to variations in climate, hydrogeology and waste composition (Keenan *et al.*, 1984). Generally, leachate possesses high concentrations of ammonia and organic contaminants (measured in terms of chemical oxygen demand COD and biochemical oxygen demand BOD), halogenated hydrocarbons and heavy metals. In addition, leachate usually contains high concentrations of inorganic salts (mainly sodium chloride, carbonate and sulphate) and is dependent on the composition of landfilled waste.

The main environmental problems at landfill sites are the infiltration of leachate and its subsequent contamination of the surrounding land and aquifers. Improvements in landfill engineering are aimed at reducing leachate production, collection and treatment prior to discharge (Farquhar, 1989). Therefore, there is a need to develop reliable and sustainable options to manage leachate generation and treatment effectively. While designing a treatment system, the process train must include techniques or unit operations to treat leachate emanating from the landfill over a longer period.

Current treatment practices advocate leachate minimization by operating landfills as dry as possible, but this poses a problem of long-term landfill stabilization. However, the other alternative of operating the landfill as wet as possible by leachate re-circulation addresses the problem of leachate treatment by reducing the organics. However, this method does not prove effective in treating hard COD or refractory compounds and nitrogen, thereby cannot meet municipal discharge standards.

High removal efficiency cannot be achieved by either biological or physical/chemical treatments separately. Two reasons for the poor removal efficiency of individual treatment systems are the larger fraction of higher molecular weight organic material that are difficult to remove and biological inhibition caused by the presence of heavy metals. Physico-chemical treatment is required to remove the metals and hydrolyze some of the organics while biological treatment is necessary for the stabilization and degradation of organic matter.

The following chapters discuss the leachate generation and composition with comparisons from different landfills. Various leachate treatment schemes are described with a view to provide an insight into the systems being practiced as biological treatment to remove the organics and the nutrients. Physical and chemical treatments mentioned therein provide a glimpse of the schemes that can be used for the removal of the

inorganics as well as the refractory components that bypass the biological system. Various treatment options and combinations are provided to enable the development of techniques for implementation within the Asian context.

Case studies of treatment plants highlight the treatment technologies used in the developed countries that could be adapted to the Asian circumstances based on the characteristics of the leachate.

Consequently, a combination of biological and physico-chemical processes can be employed to achieve high removal efficiencies**,** which depend on the characteristics of the leachate. Thus, the objective of this review is to present an in-depth analysis of biological, physical, chemical and land based-treatment systems as well as combined treatment technologies and their roles in leachate treatment. Finally, possible future directions in landfill management and treatment have been presented.

Chapter 2

SOLID WASTE LANDFILLS

2.1 Solid Waste Management Practices

Safe and reliable long-term disposal of solid waste is the most important component of solid waste management. Landfilling is the most economical and environmentally acceptable method of disposal of solid waste throughout the world (Tchobanoglous *et al.*, 1993). Even with the implementation of waste reduction, recycling and transformation technologies, disposal of solid waste in the landfill remains an important component of the solid waste management strategies.

Landfilling practices in developing countries differ from that of the developed countries, which follow advanced landfilling practices such as sanitary landfills, and engineered landfills as opposed to open dumping practiced in the developing countries. Therefore, concerns with the landfilling of solid waste are related to:

1. uncontrolled emission of landfill gases that might migrate off-site, cause odor and other potentially dangerous conditions while contributing to the green house effect in the atmosphere;
2. uncontrolled discharge of leachate that might percolate into underlying groundwater aquifers or contaminate the surface water;
3. breeding and harboring of disease vectors in improperly managed landfills and dumpsites; and
4. associated health risks and environmental impacts with the release of trace gases from hazardous materials in the commingled waste.

2.2 Overview of Municipal Solid Waste Landfills

A landfill is any form of waste disposal land, ranging from an uncontrolled rubbish dump to a full containment site engineered with high standards to protect the environment. Figure 2.1 shows a cross-section of a typical engineered landfill, which has provisions for containing leachate and landfill gases.

There are several types of landfills with or without engineering measures, leachate and landfill gas management and day-to-day operation measures, which are cited in table 2.1.

The landfill is the most economical form of solid waste disposal that minimizes adverse environmental effects, associated risks and inconveniences, thereby allowing the waste to decompose under controlled conditions until it eventually transforms into a relatively inert and stabilized material (Robinson and Maris, 1983). Most landfills can be operated satisfactorily for at least some period in their lifetime and absence of any significant negative environmental impact makes this method cheap and effective in preventing pollution by leachate discharges.

In warmer climates, increase in leachate production after precipitation is rapid (Lema *et al*., 1988) due to rainfall exceeding the amount which can evaporate during the drier periods. Hence, generation of leachate needs to be controlled and effective treatment options require to be identified to avoid its negative impacts.

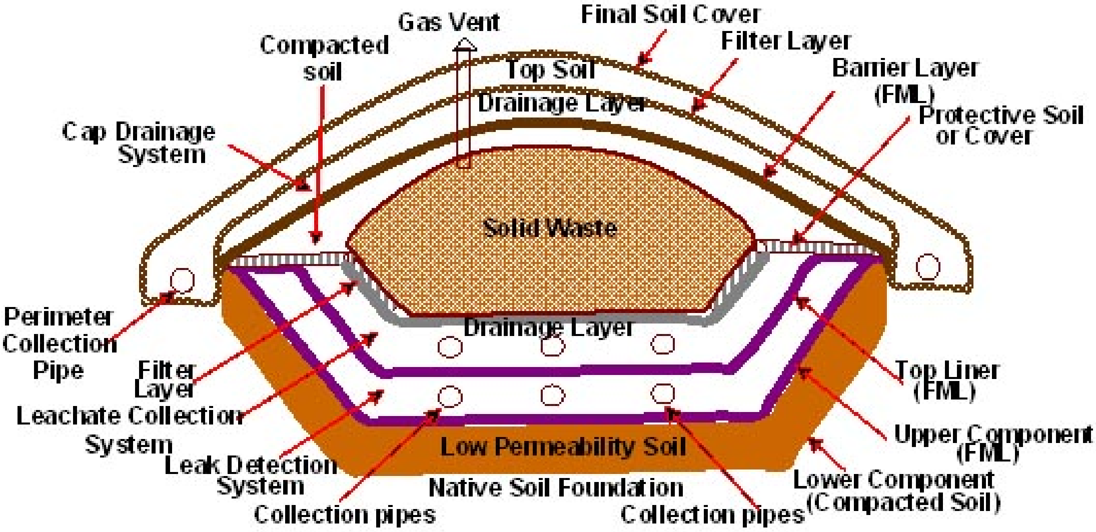


Figure 2.1 Cross section of a typical engineered landfill

Table 2.1 Types of landfills

Type	Engineering measures	Leachate management	Landfill gas management	Operation measures
Open Dumps	None	Unrestricted release of contaminants	None	Few, mostly scavenging
Controlled Dump	None	Unrestricted release of contaminants	None	Recording and placement of waste with compaction
Engineered Landfill	Infrastructure and placing of liners	Containment and some level of leachate management	Passive ventilation or flaring	Registration and placement of waste with compaction and daily use of soil cover
Sanitary Landfill	Proper siting and infrastructure; liner and leachate collection	Containment and leachate treatment (biological and physico-chemical)	Flaring	Registration and placement of waste with compaction and daily use of soil cover, and final top cover
Controlled Contaminant Release Landfill	Proper siting and infrastructure with low permeable liner; Low permeable final top cover	Controlled release of leachate based on assessment and proper siting and treatment	Flaring or passive ventilation through top cover	Registration and placement of waste with compaction and daily use of soil cover and final top cover
Landfill Bioreactor	Proper siting and infrastructure with liner and leachate recirculation system	Controlled leachate recirculation for enhanced degradation and stabilization of wastes and leachate	Landfill gas recovery	Registration and placement with compaction, daily cover, closure, mining and material recovery

Source: Joseph et al., 2002

A common practice in controlling leachate generation is to control the water permeation into the landfill by waste compaction to reduce the infiltration rate. Further, by designing impermeable covers and growing plants on the surface soil above the waste, infiltration can be minimized. The landfill characteristic is controlled by moisture content, solid waste characteristics, pH, redox potential, temperature, etc. Presence of moisture is necessary for biological conversions within the landfill and to reduce its stabilization that occurs with insufficient moisture. Degradation processes within the landfill are also temperature dependent. The pH and redox potential set the conditions for the different phases of degradation taking place while redox potential also controls the biological processes. Thus, microbial composition within the landfill effectively contributes to its stabilization.

2.3 Waste Degradation Processes

After the initial period of waste placement in a landfill, microbial processes proceed under anoxic conditions. Hydrolytic and fermentative microbial processes solubilize the waste components during the acid fermentation phase producing organic acids, alcohols, ammonia, carbon dioxide and other low molecular weight compounds as major products. This process occurs at a low pH (typically around 5) and is enhanced by the presence of moisture within. After several months, methane fermentation stage occurs. Methanogenic leachate is neutral in pH and possesses moderate organic compounds that are not easily degradable and are fermented to yield methane, carbon dioxide and other gaseous products (Harmsen, 1983; Farquhar, 1989). The potential reactions in a solid waste landfills are given in figure 2.2.

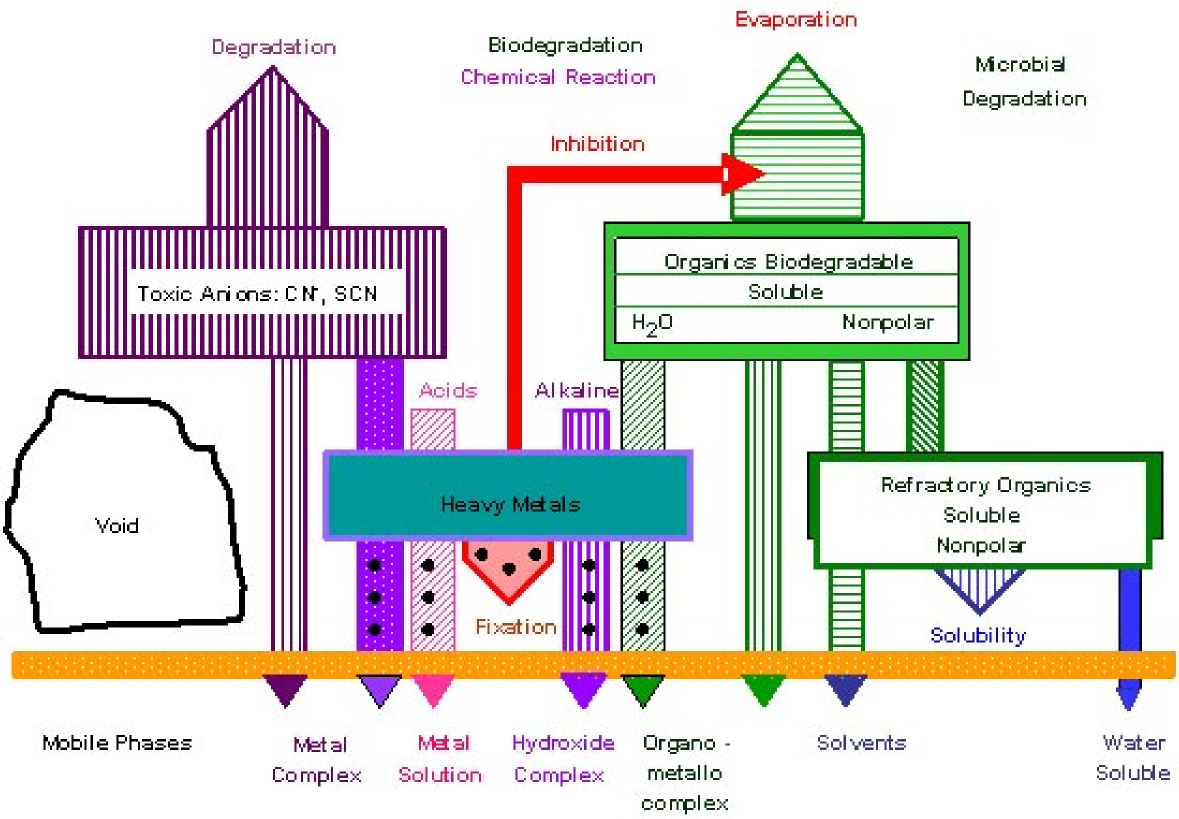


Figure 2.2 Potential reactions in solid waste landfills

Biodegradable waste constituents are converted into intermediates and products. This is primarily by initial hydrolysis to intermediate substrates supporting acidogenesis and by subsequent utilization of the products of acidogenesis as precursors to gas formation during methanogenesis of the five degradation phases (Pohland and Harper, 1985; Pohland and Kim, 1999). Figure 2.3 represents the typical concentrations of significant parameters in five different degradation phases, which are as follows:

Phase I: Initial Adjustment

This phase begins with the initial placement of waste and preliminary accumulation of moisture. Initial subsidence is observed after the closure of each landfill area. Changes in environmental parameters are detected, which reflect the onset of stabilization pattern trending in a logical manner.

Phase II: Transition

Moisture content exceeds field capacity of the waste and leachate is formed. A transition from initial aerobic to anaerobic microbial stabilization occurs. The primary electron acceptor shifts from oxygen to nitrates and sulfates with the displacement of oxygen into carbon dioxide. A trend toward reducing conditions is established. Intermediates such as the volatile organic fatty acids that are measurable first appear in the leachate.

Phase III: Acid Formation

Intermediary volatile organic fatty acids become predominant with the continuing hydrolysis and fermentation of waste and leachate constituents. A sharp decrease in pH occurs with a concomitant mobilization and possible complexation of metal species. Nutrients (nitrogen and phosphorus) are released and utilized to support the growth of biomass commensurate to the prevailing substrate conversion rates. Hydrogen may be detected and can affect the nature and type of intermediary products being formed.

Phase IV: Methane Fermentation

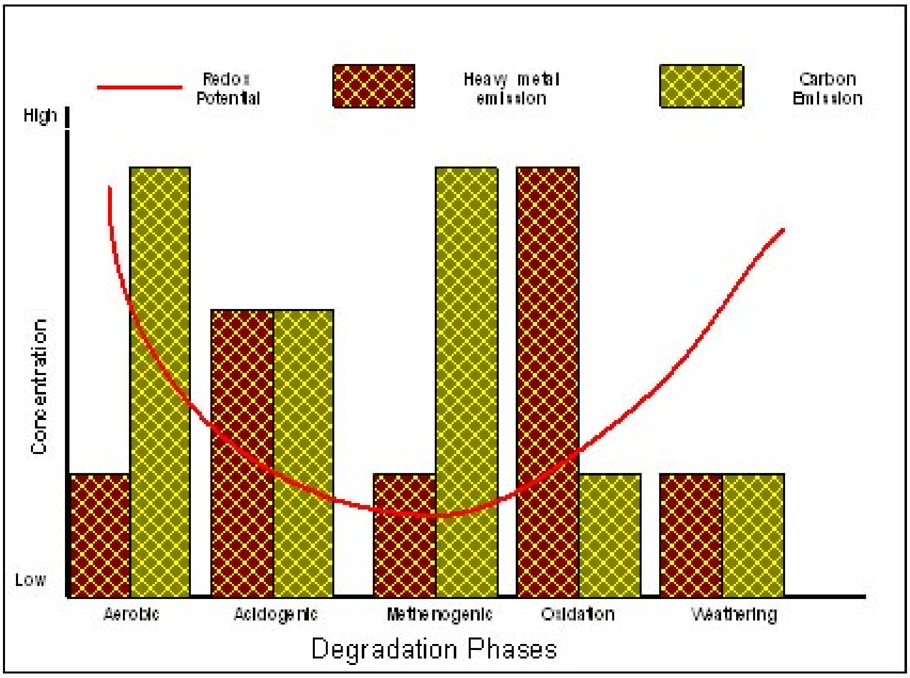
Intermediary products appearing during the acid formation phase are converted to methane and excess carbon dioxide. The pH returns from a buffer level controlled by Volatile Fatty Acids (VFA) to one characteristic of the bicarbonate buffering system. Oxidation-reduction potentials are at their lowest values. As nutrients continue to be consumed, complexation and precipitation of metal species proceed. The organic strength of the leachate is dramatically decreased with increases in gas production.

Phase V: Final Maturation

In this phase, the relative dormancy follows active biological stabilization of the readily available organic constituents in the waste and leachate. Nutrients may become limiting and measurable gas production ceases. Natural environmental conditions become reinstated and oxygen and oxidized species may slowly reappear with a corresponding increase in oxidation-reduction potential. More microbial resistant organic materials may be slowly converted with a possible production of humic-like substances capable of complexing with heavy metal and remobilizing them.

The phase distinction in all the landfills do not show a clear trend as described in the proposed phase degradation model since there are certain regions in the landfill, which are dominated by a particular degradation phase. Hence, the leachate generated is a

combination of the products of different microbial and physicochemical processes within the landfill.



(*Source: Pohland and Harper, 1985*)

Figure 2.3 Changes in significant parameters during different phases of landfill stabilization

2.4 Landfill Leachate

During the percolation of rainwater and moisture through municipal solid waste (MSW) in a landfill, the liquid medium absorbs nutrients and contaminants from the waste and exudes as leachate from the landfills or dumps posing a hazard to the receiving water bodies. This leachate contains many substances which depend on the types of waste disposed into the landfill, and may be toxic to life or may simply alter the ecology of the stream or watercourse if not removed by treatment.

Depending on the geographical and geological nature of a landfill site, leachate may seep into the ground and possibly enter groundwater sources. Though natural processes within the soil layers can remove a part of the contaminants from the leachate, groundwater contamination can be hazardous as drinking water sources may be affected.

The simplest method of leachate treatment is discharge into public sewer, which is only possible if the sewer is located in the proximity to the landfill or alternatively, leachate may be collected and transported to the public treatment plants. There is considerable difference between the characteristics of leachate and domestic wastewater and the volume of leachate discharge is limited. Depending on leachate characteristics, it may be necessary to pre-treat it prior to discharge into wastewater treatment plants so that it does not upset the biological process or cause any operational and maintenance problems in the treatment train. In designing a treatment scheme for leachate, it is necessary to determine whether the leachate effluent meets sewer or water body discharge standards.

Chapter 3

LEACHATE GENERATION AND COMPOSITION

3.1 Leachate Generation

Landfill leachate is a high strength wastewater generated as a result of percolation of the rainwater and moisture through the waste resulting in the solubilization of nutrients and the contaminants within a landfill. Generally, the quantity of leachate is a direct function of the amount of external water entering the landfill. Figure 3.1 presents the various water inputs and its movement in the landfill leading to leachate production. Figure 3.2 gives the equations for leachate generation and water balance.

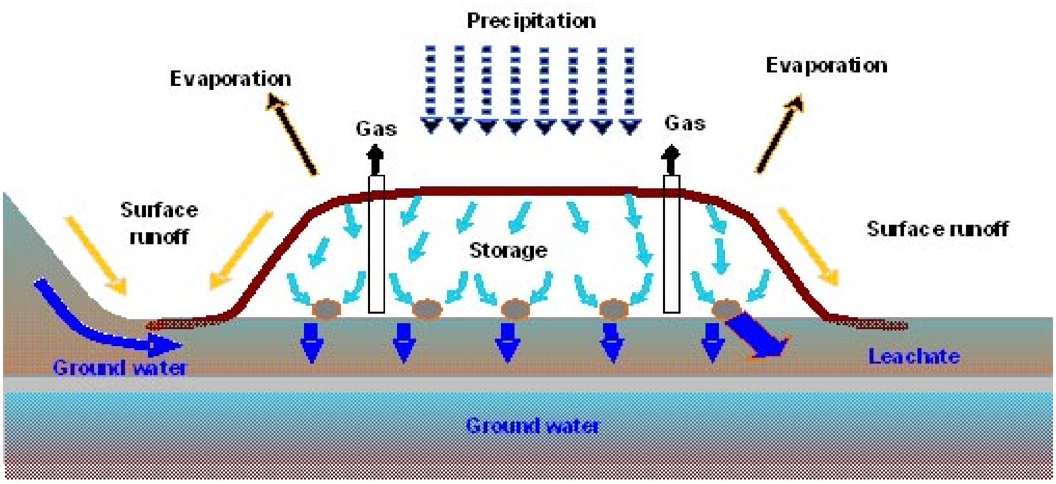


Figure 3.1 Movement of water in the landfill

Leachate Generation

$W = R - E$

W = Quantity of moisture either lost or retained in the waste [mm];
R = Precipitation [mm]; and
E = Evapo-transpiration from the landfill [mm].

Leachate Formation and Water Balance

$\Delta MC = P - RO - ET - PER$

ΔMC = Change in the amount of moisture stored in a unit volume of landfill [cm];
P = Quantity of net precipitation (incident precipitation less runoff) per unit area [cm];
RO = Quantity of runoff per unit area [cm];
ET = Quantity of moisture lost through evapo-transpiration per unit area [cm]; and
PER = Quantity of water which percolates through the cover per unit area of soil cover [cm].

Water balance

$\Delta MC = W_{sw} + W_c + W_p - W_{lfg} - W_v - W_{evp} + W_{leach}$

ΔMC = Change in the quantity of moisture stored in the landfill [kg/m³];
W_{sw} = Moisture in the incoming solid waste [kg/m³];
W_c = Water in the cover material placed on the waste [kg/m³];
W_p = Water from precipitation and other outside sources (less runoff) [kg/m³];
W_{lfg} = Water utilized in the formation of landfill gas [kg/m³];
W_v = Water lost as saturated vapor with the landfill gas [kg/m³];
W_{evp} = Moisture loss due to evaporation [kg/m³]; and
W_{leach} = Water leaving the landfill (control volume) as leachate [kg/m³].

Figure 3.2 Equations for water balance and leachate formation

The components, which contribute to leachate formation, follow a certain pattern based on the physical processes taking place within the landfill that can be summarized as follows:

- Precipitation on the landfill initiates runoff while some amount infiltrates into the surface.
- Some of the infiltration evaporates from the surface and/or transpires through vegetative cover while some part is retained as the soil moisture.
- The remainder of the infiltration percolates within eventually forming leachate at the base of the landfill.
- Percolation may be augmented by infiltration of groundwater.

3.2 Leachate Composition and Characteristics

The variation in leachate quality can be due to a variety of reasons based on the four aspects of landfills, which are:

- Waste type - its grade of decomposition with possible seasonal variance in its deposal;
- Landfill environment - waste degradation phase, humidity, precipitation, temperature, etc.;
- Filling technique - waste compaction, landfill cover and height of landfill layers; and
- Sampling - method of analysis and point of sample collection.

These aspects are interlinked and a combination of the first three categories contributes to the overall variance in leachate quality and characterization, whereas the fourth aspect affects the analysis.

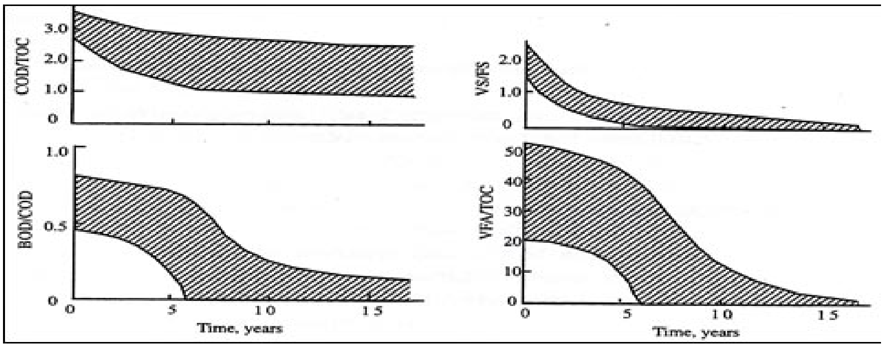
MSW landfill leachate in Asia (except Japan) is characterized by 60 - 90% organic waste and 3 - 18% plastic (Agamuthu, 1999). Due to the Asian food style, the concentrations of potassium, sodium and chloride are found to be high. A pre-treatment is recommended prior to the biological treatment. Mature landfills of 10 years or above ideally have leachate with typical concentrations of sodium (100 - 200 mg/L), potassium (50 - 400 mg/L) and chloride (100 - 400 mg/L). At the same time, leachate generated from mature landfills in Asia are characterized by sodium (1,500 – 5,640 mg/L), potassium (400 – 1,940 mg/L) and chloride (875 – 2,900 mg/L) contents. Leachate characteristics of different landfills surveyed in Asia and United States of America are presented in table 3.1.

xsTable 3.1 Leachate characteristics in developing and developed countries

Parameters	China[1]		Thailand[2]			India[3]		Sri Lanka[4]	Malaysia[5]		Hong Kong[6]		USA[7]	
	Shenzhen xiaping	Datianshan	Phitsanulok	Pathum-thani	Nonthaburi	Okhla, New Delhi	Perugundi & Kodungaiyur Chennai		Sabak Bernam	Taman Beringin				
Age (Years)	2	10	1	9	20	9	16	<1	7	16	6	10	1	16
Alkalinity (mg/L)	-	-	300-4700	6,620	1140-5800	8.4	-	450-3700	1200-1550	3750-9375	10,700-11,700	3230-4940	800-4,000	2,250
pH	7.8	-	7.1-8.3	8.1	8.1-8.5	-	7.3-9.3	5.4 - 7.7	8.0-801	7.8-8.7	8.1-8.6	7.6-8.1	5.2-6.4	-
Chloride (mg/L)	-	-	-	2530	3600-4200	16,000	119-5856	-	420-1820	875-2875	2320-2740	522-853	600-800	70
SS (mg/L)	250	385.63	1,950	12.5	150-746	-	-	1,875	111-920	420-1150	40-53	3-124	-	-
TS (mg/L)	-	-	6,700	848	767-2,155	-	-	4,568 – 6,786	-	10,300-13,680	-	-	100-700	-
COD (mg/L)	13040	3670.1	4900-11,000	3200	8800-17,600	23,306	72-5100	-	1250-2570	1960-5500	2,460-2,830	641-873	10,000-40,000	400
BOD (mg/L)	3220.5	1205.5	3000-7150	280	800-1800	1848	3-207	5000-15,000	726-1210	562-1990	-	-	7,500-28,000	80
TKN (mg/L)	-	-	-	1,256	154-2540	450	-	-	-	104-1,014	2,219-2,860	889-1,180	-	-
NH_3-N (mg/L)	2090	845.01	150-1250	-	-	745	-	-	3-8	2-47	1,190-2,700	784-1,156	56-482	-
Ni (mg/L)	0.39	-	0.02-1.56	0.25	0.29-0.66	0.17	0.026-1.05	-	-	0-0.6	-	-	-	-
Cd (mg/L)	0.01	0.126	0.037	0.002	0.001	-	0.001-0.05	-	0-0.001	0-0.15	-	-	-	<0.05
Pb (mg/L)	0.08	3.25	0.03-0.45	-	0.06	0.72	0.009-0.646	-	0-0.03	0-3.45	-	-	-	1.0
Cr (mg/L)	0.046	0.269	-	0.07	0.06-1.16	16.9	0.001-0.898	-	-	0.04-0.70	-	-	-	-
Hg (mg/L)	-	-	0.50-1.70	-	-	0.4	0.002-0.018	-	-	-	-	-	-	-

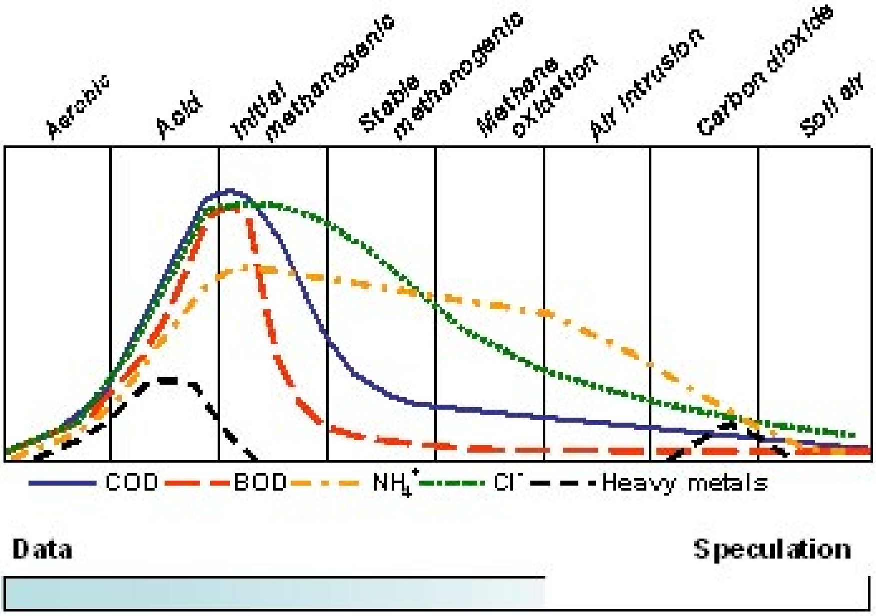
Sources: 1.NRI China (2003); 2. Pollution Control Department (2000) and Sivapornpun (2000); 3. NRI India (2003); 4. NRI Sri Lanka (2003); 5. Agamuthu (1999); 6. Robinson and Luo (1991); 7.Qasim and Chiang (1994)

The changes in the BOD/COD, COD/TOC, VS/FS and VFA/TOC ratios of leachate depend greatly on the age of the landfill (Chian and DeWalle, 1976; Kylefors, 1997). Figure 3.3 represents the trend in the variation of leachate over the age while figure 3.4 shows the trend of leachate quality over the lifetime of a landfill with its different phases as shown in figure 2.3.



(TOC – Total Organic Carbon; VS- Volatile Solids; FS – Fixed Solids) *Source: Kylefors, 1997*

Figure 3.3 Variation of leachate characteristics over time in a landfill



(COD – Chemical Oxygen Demand; BOD – Biochemical Oxygen Demand; NH_4^+ - Ammonia Nitrogen; Cl^- - Chlorides)

Figure 3.4 Trends in leachate quality over lifetime of a landfill

During the first few years (< 5 years), the landfill is in acidogenic phase and the leachate generated is generally referred to as young or carbon-based leachate due to the higher concentration of organic carbon. Landfill greater than 10 years old are generally in the methanogenic phase and the leachate generated is referred to as old or nitrogen-based (Mavinic, 1988). Ehrig (1998) conducted an extensive study on leachate characterization of 15 landfills ranging from 0 to 12 years in Germany and found the characteristics of

leachate in the acidogenic and methanogenic landfill leachate. Table 3.2 gives the characteristic of leachate present in acidogenic and methanogenic phases. In the former phase, the organic content (BOD and COD) is higher than in the latter. The pH has been found to increase with decrease in biodegradability when the landfill changes from the acidogenic to methanogenic phase.

Table 3.2 Leachate characteristic in acidogenic and methanogenic phases in a landfill

Parameter	Unit	Acidogenic Phase		Methanogenic Phase	
pH		6.1	4.5 - 7.5	8	7.5 - 9
BOD_5	mg/L	13,000	4000 - 40,000	180	20 - 550
COD	mg/L	22,000	6000 - 60,000	3000	500 - 4500
BOD_5/COD		0.58	-	0.06	-
SO_4	mg/L	500	70 - 1750	80	10 - 420
Ca	mg/L	1200	10 - 2500	60	20 - 600
Mg	mg/L	470	50 - 1150	180	40 - 350
Fe	mg/L	780	20 - 2100	15	3 - 280
Mn	mg/L	25	0.3 - 65	0.7	0.03 - 45
Zn	mg/L	5	0.1 - 120	0.6	0.03 - 4

Source: Ehrig, 1998

Ammonium concentration in the leachate also varies with the age of landfill: young leachate has a high COD (> 5000 mg/L) and low nitrogen content (< 400 mg N/L); and old leachate has high concentrations of ammonia (> 400 mg N/L) and recalcitrant compounds, and a low biodegradable organic fraction (BOD_5/COD = 0.1). Table 3.3 presents the general leachate characteristics with age and suitability of treatment options in terms of biodegradable, intermediate and stabilized landfill leachate.

Table 3.3 Relation between landfill age, leachate characteristics and treatment

Landfill Age (years)	< 5 (young)	5 to 10 (medium)	> 10 (old)
Leachate Type	I (Biodegradable)	II (Intermediate)	III (Stabilized)
pH	< 6.5	6.5 - 7.5	> 7.5
COD (mg/L)	> 10,000	< 10,000	< 5000
COD/TOC	< 2.7	2.0 - 2.7	> 2.0
BOD_5/COD	< 0.5	0.1 - 0.5	< 0.1
VFA (% TOC)	> 70	5 - 30	< 5
Process	**Treatment Efficiency**		
Biological Treatment	A	B	C
Chemical Oxidation	B-C	B	B
Chemical Precipitation	B-C	B	C
Activated Carbon	B-C	A-B	A
Coagulation-flocculation	B-C	A-B	A
Reverse Osmosis	C	A	A

A=Good; B=Fair; C = Poor

Source: Amokrane et al., 1997

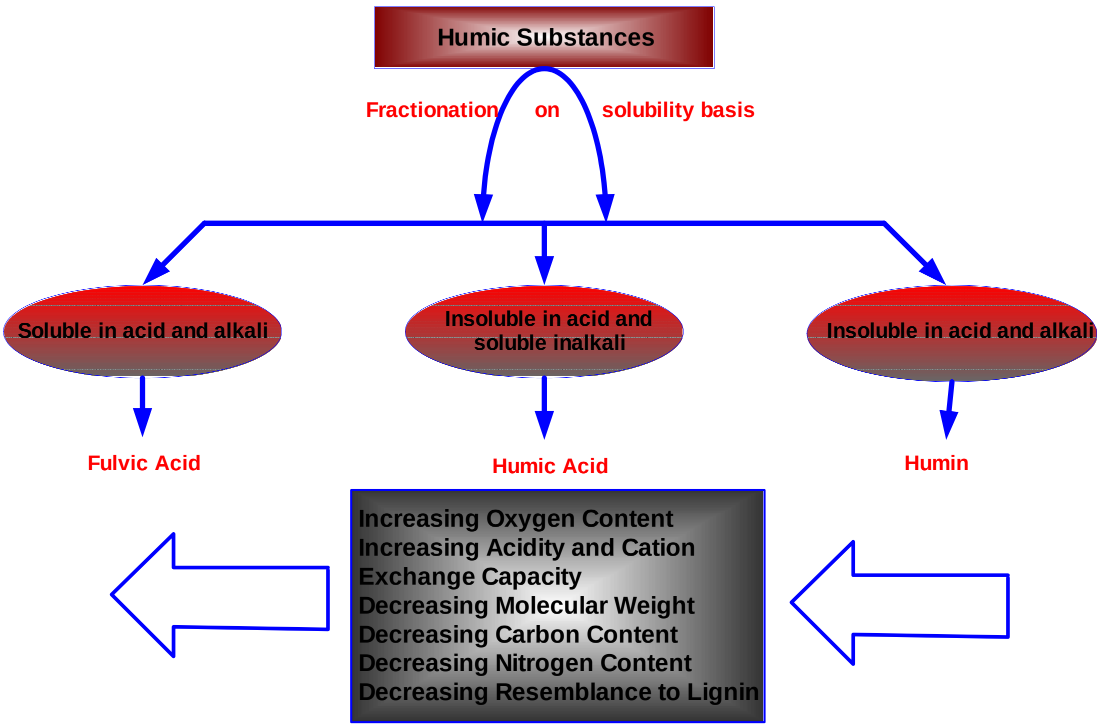
Table 3.3 aids in determining the appropriate leachate treatment options. Effectiveness of combined process for the treatment of a leachate produced at specific landfill age has not been considered. Individual treatment options cannot be a long-term solution for leachate as they are not effective in treating that generated at different periods of time and do not adapt to changing characteristics.

The solid components of leachate are mainly composed of water-soluble substances. The concentration of suspended solids in leachate is generally low. Organic matter present in the leachate varies and is dependent on the waste composition and degree of degradation. In general, the organic component consists of:

- low molecular weight (MW) alcohols and organic acids;
- medium molecular weight fulvic acid; and
- high molecular weight humic substances.

Low MW organics are composed of easily degradable VFA, which contribute to 90% of the total organic fraction. The most frequently occurring fatty acids are acetic, propionic and butanic acids. Medium MW compounds have a molecular weight between 500 and 10,000 Da comprising of fulvic acid and humic fraction present in the leachate. This group is dominated by carboxylic and hydroxylic groups, which are difficult to degrade, and hence are termed refractory. The high MW organic fraction varies from 0.5% in methanogenic landfills to 5% in acidogenic landfills. These compounds are more stable and possibly originate from cellulose or lignin.

Thurman and Malcolm (1981) reported that humic substances (hydrophobic acids) accounted for about 50 - 90% of the dissolved organic carbon (DOC) present in the leachate whereas Imai *et al*. (1995) indicated that they only contributed to about 30% of the DOC. This implies that non-humic substances (hydrophobic neutrals and bases, hydrophilic acids, neutrals and bases) may be more important than humic substances with regard to the refractory characteristics of leachate. The characteristics and fractionation products of humic substances present in leachate are shown in figure 3.5.



Source: Kylefors, 1997

Figure 3.5 Fractionation and characteristics of humic substances

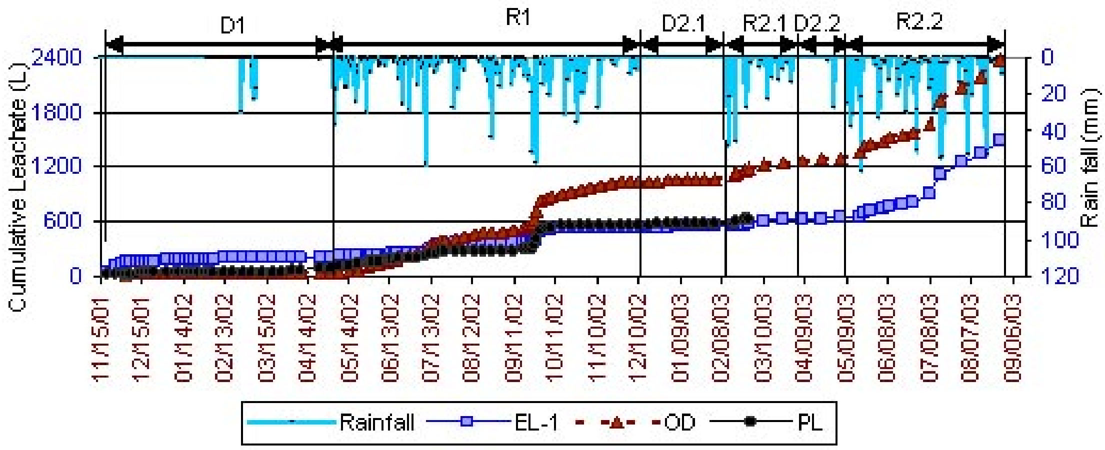
The effectiveness of a treatment process can be related to the removal of specific organic fractions present in the leachate. Both fulvic and humic substances are less influenced by a biological treatment. The accumulation of high MW humic carbohydrates was found to affect bacterial flocculation (Chian and DeWalle, 1976). Humic substances remain unaffected by activated carbon treatment, whilst low molecular weight fulvic-like substances could easily be removed by it.

3.3 Factors Affecting Leachate Composition

The composition of leachate is influenced by various factors - seasonal variation, landfill age, waste composition, geological characteristics and landfilling techniques. These factors interlinked with one another have potential to influence the leachate quality, thereby producing an integrated effect on its composition. Some important factors are discussed in the sections below.

3.3.1 Climatic variation

The influence of climatic conditions over changes in leachate quality and quantity should be considered while designing a landfill for disposal of MSW and proposing a treatment scheme for leachate. Rainfall acts as a medium of transportation for leaching and migration of contaminants from a landfill. It also provides the required moisture for methane production and biological activity. Normally, leachate is produced when the moisture content in the landfill exceeds the field capacity. The quantity is found to be comparatively low during the dry season. When the rainfall begins, initially it takes some time to reach the field capacity, which once reached, leachate production increases. In hot and humid climates, leachate production is much higher than that experienced in hot arid regions. On the other hand, high rainfall leads to increased leachate production and reduced leachate strength due to dilution. Figure 3.6 shows the influence of the rainfall in leachate production in Thailand based on the landfill lysimeter experiments for different conditions. EL represented an engineered landfill filled with fresh municipal solid waste comprising 59% organics, 24% plastics, 7% glass, 5% leather and rubber, 1% ferrous metal and moisture content 47 % (wet basis). EL had a top cover with clay liner of 10 cm gravel layer, 20 cm barrier layer (clay) and 40 cm cover soil comprising of sand, silt and clay mixture. PL depicts the pre-treated waste landfill of composted waste additionally pre-sorted and mechanically shredded having moisture content of 21% (wet basis). The top cover specification and the design parameters are the same as EL.



Lysimeter types (EL- Engineered Landfill; OD- Open Dump; PL- Pretreated waste Landfill).

Figure 3.6 Relationship between rainfall and cumulative leachate production from landfill lysimeters

The OD cell represented an open dump filled by mainly organic waste with 64% wet garbage, 22% plastic and 14% paper. The OD was designed without a top cover and only a 25-mm sand layer to avoid direct contact with ambient environment.

The total rainfall accounted for was nearly 3,294 litres at the end of August 2003 into each lysimeter on cumulative basis and in the OD cell nearly 70% of the rainfall formed the leachate. Since the OD cell does not have run-off similar to other lysimeters, the remaining 30% accounts for evaporation and storage. The EL cell produced less than 35% of the total rainfall as leachate. In addition, it could also be noted that during the prolonged dry periods D1 leachate generation ceased or reduced drastically in all lysimeters due to lack of moisture above field capacity. This could be an important point to be noted from the leachate management point of view in tropical Asia where 70 - 90% landfills are open dumps. This would be mainly because of the structure of the cover layer made out of compost and sandy loam soil mixed loosely.

Several mathematical models have been developed to predict leachate production on the basis of hydrological factors. The quality of leachate produced may be regarded as proportional to the volume of water percolating through the landfilled waste. Reduction in the quantity of water entering a landfill cell is therefore of great importance in reducing the rate of leachate generation (Tatsi and Zouboulis, 2002). Figure 3.7 shows the variation in the strength of leachate in terms of COD with rainfall. The dilution ffect is clearly indicated in the figure. Researchers have measured the temporal variation in leachate production to be 2-45 L/s which depends largely on the precipitation over the landfill (Martin *et al.*, 1995). The influence of seasonal variation in the leachate quality and quantity varies from place to place and is also influenced by other factors. It is necessary to consider the hydrological factors and leachate quality data while proposing a treatment scheme to avoid problems of environmental deterioration caused by direct disposal.

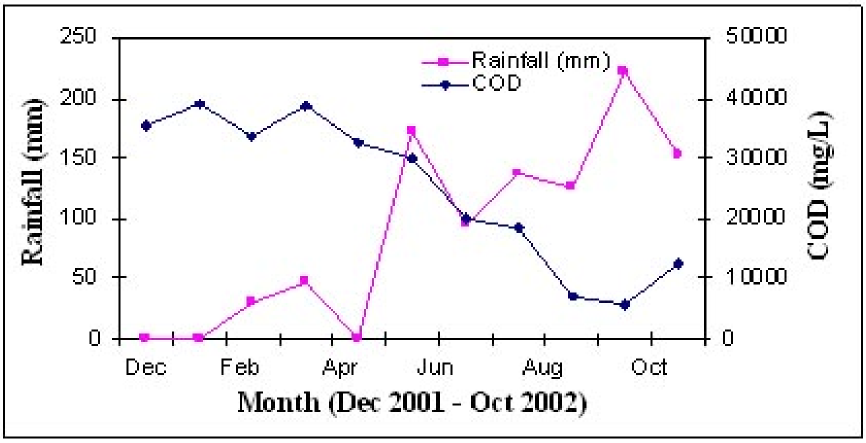


Figure 3.7 Variation in COD with rainfall in Thailand

3.3.2 Landfill age

Sampling and analysis of leachate are important for assessing changes in its quality over a period of time. The differentiation of landfill age can be made based on dominating degradation phase and the composition of the resultant leachate. The climatic variation of leachate quality and quantity depends on its age. The BOD/COD ratio that depicts the biodegradability of the leachate has a decreasing trend with age from a readily biodegradable ratio of 0.5 to a higher fraction of poor degradability with a value of 0.1 or less (Table 3.2). Table 3.4 represents few of the analytical parameters of pollution in a young and stabilized landfill leachate.

Table 3.4 Pollution parameters in young and old leachate

Parameter	Young Landfill Leachate		Stabilized Landfill Leachate	
	Mean	**Range**	**Mean**	**Range**
BOD_5/COD	0.54	0.41 - 0.97	0.26	0.05 - 0.64
pH	6.20	4.88 - 6.7	7.90	7.30 - 8.80
SO_4^{2-}/Cl^-	0.73	0.08 - 3.4	0.06	0.02 - 0.15
ORP (mV)	-22.60	-53 to -11	154.00	110 -218
VS/TS	0.48	0.21 to 0.66	0.28	0.22 - 0.37

Source: Tatsi and Zouboulis, 2002

The fresh leachate (from a young landfill) possesses high proportion of organic material, which can be removed by biological processes. Owing to biological reactions, fresh leachate is expected to produce an acidic pH and an unpleasant smell. The decrease in VS/TS ratio from fresh to old leachate also suggests that stabilized leachate is less amenable to biological treatment as the landfill age increases. The redox potential and pH increases with the stabilization process. A color change from light yellow to dark brown or black is also observed due to iron oxidation from ferrous to ferric iron, resulting in an increase in turbidity. The variation in the quality of leachate from a landfill in Taiwan composing of ten different units, each of which was closed every year is expressed in table 3.5.

Table 3.5 Variation of COD and BOD and their ratio with landfill ages from one of the landfills in Taiwan

Age (year)	1	2	3	4	5	6	7	8	9	10	11
BOD(mg/L)	25,000	10,000	290	260	240	210	190	160	130	100	80
COD(mg/L)	35,000	16,000	1850	1500	1400	1200	1200	1150	1100	1050	1000
BOD/COD	0.71	0.60	0.17	0.17	0.16	0.16	0.14	0.13	0.10	0.08	0.08

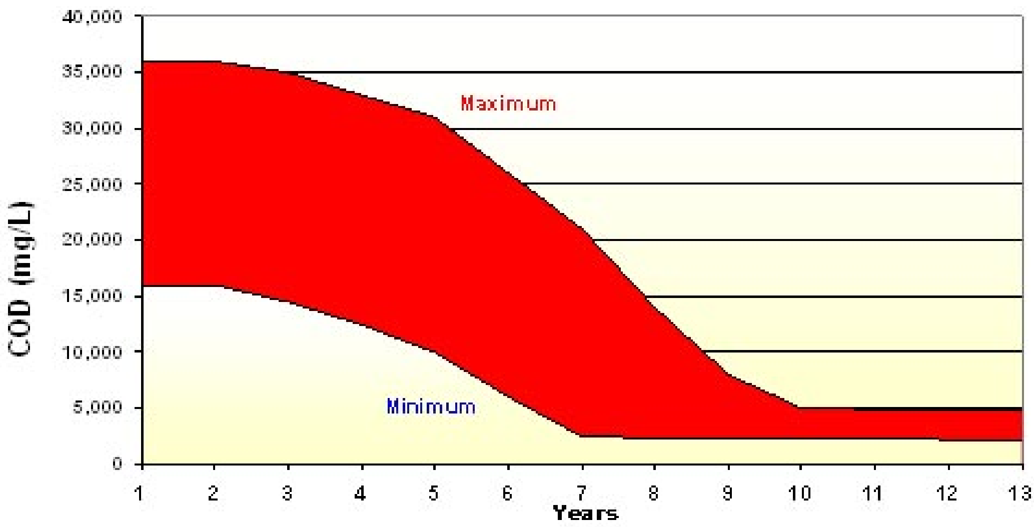
Source: Ragle et al., 1995

The variability of leachate composition with age of the landfill varies from place to place. Though the composition of the leachate follows a similar trend, the variability is influenced also by various other parameters. The leachate quality in the temperate region differs from that of tropical region and within the tropical region, there can be variations. When the leachate from two landfills in China, namely Shenzhen and Nanchang landfills were analyzed, the composition differed with the increase in age. The data given in table 3.6 indicates that young leachate from Shenzhen had a COD of around 20,000 - 60,000 mg/L whereas that of Nanchang was found to be 4,000 - 12,000 mg/L. The pattern is similar for old leachate too. This fact could be attributed to the nature of the waste landfilled as Nanchang lies in the interior area in the temperate zone with a colder climate and hence has more of clinkers and ash in the waste landfilled. Shenzen on the other hand, is a coastal city probably having a higher food waste due to the consumption pattern, presence of seafood industries, business establishments and tourist influx. Figures 3.8 (a) and (b) show the variation of COD and NH_4-N with the age of the landfills. There is a decreasing trend in the emission of these two parameters as the landfill matures owing to the stabilization of the waste within.

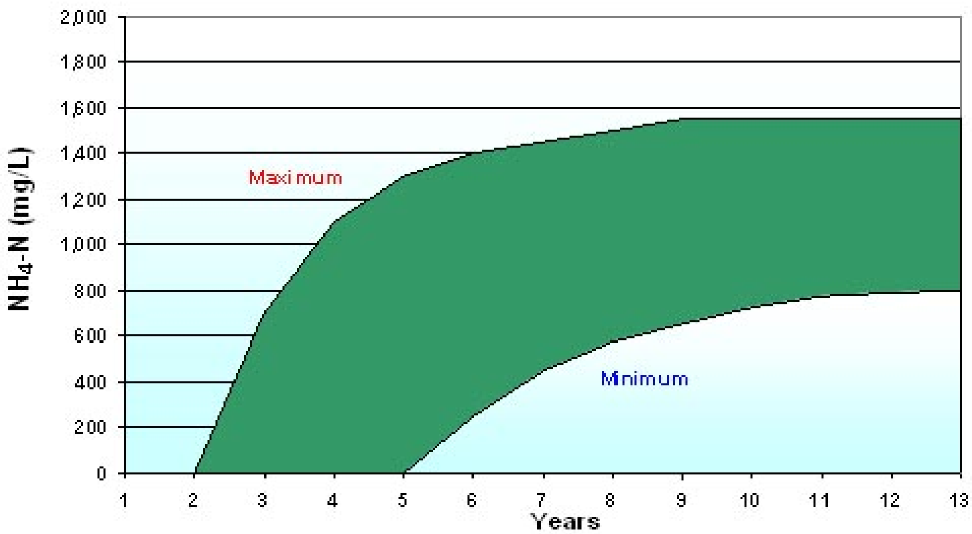
Table 3.6 Variation in leachate quality in Shenzhen and Nanchang landfills, China

Items	Shenzhen (Guangduang)		Nanchang (Jiangxi)	
	0 - 5 years	**> 5 years**	**0 - 5 years**	**> 5 years**
COD (mg/L)	20,000 - 60,000	3000 - 20,000	3785 - 12,000	1000 - 8000
BOD (mg/L)	10,000 - 36,000	1000 - 10,000	1900 - 7030	600 - 6000
NH_4^+-N (mg/L)	400 - 1500	500 - 1000	200 - 300	400 - 500
TP (mg/L)	10-70	10 - 30		
SS (mg/L)	1000 - 6000	100 - 3000	100 - 600	100 - 600
pH	5.6 - 7	6.5 - 7.5	6.0 - 7.0	6.5 - 8.0

Source: Zhao et al., 2000



(a) *Source: Ehrig.H.J. 1983*



(b) *Source: Ehrig.H.J. 1983*

Figure 3.8 Variation of (a) COD and (b) NH_4-N with landfill age

3.3.3 Composition of the waste

The leachate quality is significantly affected by the composition of refuse. Organic material present in the waste mainly comprise of kitchen waste while the inorganic constituents consist of plastic, glass, metal, ash, silt etc. The leachate composition depends upon the ratio of organic and inorganic components present in the waste disposed in the landfill. It is estimated that approximately one-half of the municipal solid waste is typically composed of cellulose and hemicellulose (Fairweather and Barlaz, 1988; Barlaz *et al*., 1989), which are considered readily degradable in the environment. The organic content leached is as a result of hydrolysis and degradation of higher molecular weight organic compounds by the microorganisms present in the waste. However, it has been shown that readily degradable refuse components can sometimes persist for

surprisingly long periods in landfills owing to several environmental factors that limit the microbial growth (Suflita *et al*., 1992; Gurijala and Suflita, 1993).

The composition of municipal solid waste in larger cities of China is different from the smaller cities. The organic components contribute about 31 - 36% whereas inorganic components contribute to about 60% of the total solid waste. Comparison between different regions in China shows that there is more organic compound in South China than that in North China because the temperate region in the north uses coal as fuel and for domestic heating. The effect of landfill composition on the leachate quality for four different cities in China is shown in table 3.7.

Table 3.7 Landfill composition and leachate quality in major Chinese cities

Composition	Parameters	Shanghai	Guangzhou	Shenzhen	Hang Zhou
Refuse (%)	Organic matter	55.33	60.17	40.00	58.19
	Inorganic matter	11.13	17.12	15.00	24.00
Leachate (mg/L)	COD	1500 - 8000	1400 - 10,000	3000 - 60,000	1000 - 5000
	BOD_5	200 - 4000	400 - 2500	1000 - 36,000	400 - 6000
	TN	100 - 700	150 - 900	-	80 - 800
	NH^{3}-N	60 - 450	130 - 600	400 - 1500	50 - 500
	SS	30 - 500	200 - 600	100 - 6000	60 - 650
	pH	5 - 6.5	6.5 - 7.8	6.2 - 8.0	6 - 6.5

Source: Li et al, 2002

Thus, the composition of waste significantly influences the leachate quality. Other factors that influence the leachate are the moisture content, nutrients and organic loading.

3.3.4 Geological characteristics

As the leachate percolates through the underlying strata, many of the chemical and biological constituents originally contained in it are removed by filtering and adsorptive capacity of the material composing the strata. In general, the extent of this action depends on the characteristics of the soil and especially the clay content. With this potential, it can allow the leachate to percolate into the landfill for elimination or contamination thereby affecting the leachate quantity. The soil particle size, the type of soil in the underlying ground and cover material are factors that further influence leachate production and strength.

3.3.5 Enhanced stabilization and mechanical-biological pretreatment of waste

In order to reduce the time required for leachate treatment, it is necessary to enhance waste stabilization. The factors affecting waste stabilization as proposed by Kylefors (1997) are:

- lack of water;
- lack and poor distribution of nutrients;
- accumulation of degradation products; and
- small contact area of waste, water and microflora.

Stabilization can be accomplished by:

- pre-treating the waste by size reduction, mixing and pre-composting; and
- using flow systems to influence the environmental conditions within the landfill.

Reduction of the particle size (pretreatment) can be achieved by mechanical milling and sorting. This effectively increases the contact area between the waste and the microorganisms for treatment. It will also improve the homogeneity of the waste within the landfill and ease compaction, despite the fact that the aerobic degradation phase can be shortened and anaerobic phase is prolonged.

Stabilization can also be attained by a continuous flow with the re-circulation of leachate and abstraction of gas within the fill. Kylefors (1997) reported that leachate re-circulation affects landfill stabilization by:

- removing the waste products after degradation from the liquid phase.;
- allowing addition and distribution of microorganisms and nutrients with the landfill; and
- maintaining homogeneous conditions within the fill.

Pretreatment of municipal solid waste using mechanical biological pretreatment (MBP) reduce reactional behavior and toxic affects of the waste. Zach *et al.* (2000) reported that gas generating potential decreased by 90% at the end of 20 weeks of composting. Because of MBP, the strength of the generated leachate decreased by about 90% in terms of BOD_5, COD and total nitrogen content as found by Leikam and Stegmann (1999). They also claimed that the volume of waste decreased by 60% as compared to untreated waste thus it saves on the investment for leachate and landfill gas management, and the cost involved in the landfilling of waste. Similarly, in a pilot scale study, Norbu *et al* (2003) reported that pile settlement for the bulk waste reduced by as much as 52% with aeration of the waste within a period of 2 weeks.

A lysimeter study to simulate engineered landfill, open dump and landfilling of pretreated waste at the Asian Institute of Technology, Thailand found that tropical seasonal variation influenced leachate generation, waste stabilization and settlement pattern of waste. Leachate generation was highest during the rainy season and the least during the dry periods and hence the stabilization of the waste was maximum during the wet season and minimal in the dry season. It can be concluded that leachate collected during the rainy season could be stored and recirculated into the landfill during the drier periods to enhance stabilization. A significant finding is that the lysimeter landfill with pretreated waste had minimum concentrations of COD (8-fold) and TKN (4-fold), and minimum discharge loads COD (25-fold) and TKN (5-fold) as compared to the engineered landfill cell (Kuruparan *et al*, 2003). Thus, pretreatment of the waste plays a major role in the generation of leachate and its composition.

3.3.6 Filling technique

During the filling of the municipal solid waste in a landfill, various factors influence the leachate quality and quantity considerably. In an engineered landfill, waste is placed above a series of layers, which are organized to help collect the generated leachate. These bottom layers prevent leachate percolation into the underlying soil layer, which results in contamination of soil and ground water aquifers. Once the waste has been placed, the layers above the waste have arrangements for the collection of landfill gas. These layers help prevent infiltration of rainwater into the waste. Figure 3.9 shows the layers in an engineered landfill. The arrangement of leachate collection pipes is placed in the primary and secondary barrier layers as clearly indicated in figure 3.10. Encasing of the pipes by

geotextile filter prevent clogging of the collection pores in the pipes. Figure 3.11 shows the network of leachate collection pipes with a cross section of the arrangement. The pipes are placed above the liner from where the leachate is collected in a tank.

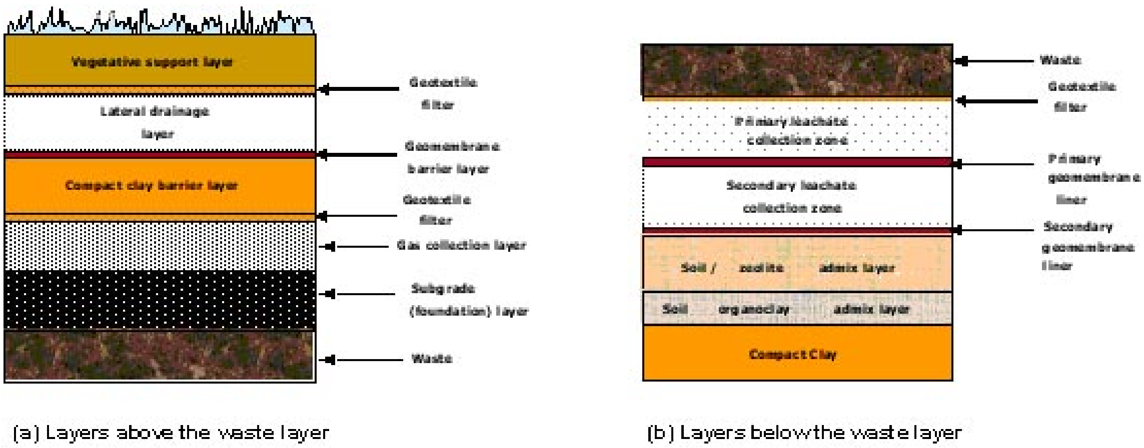


Figure 3.9 Schematic of an engineered landfill

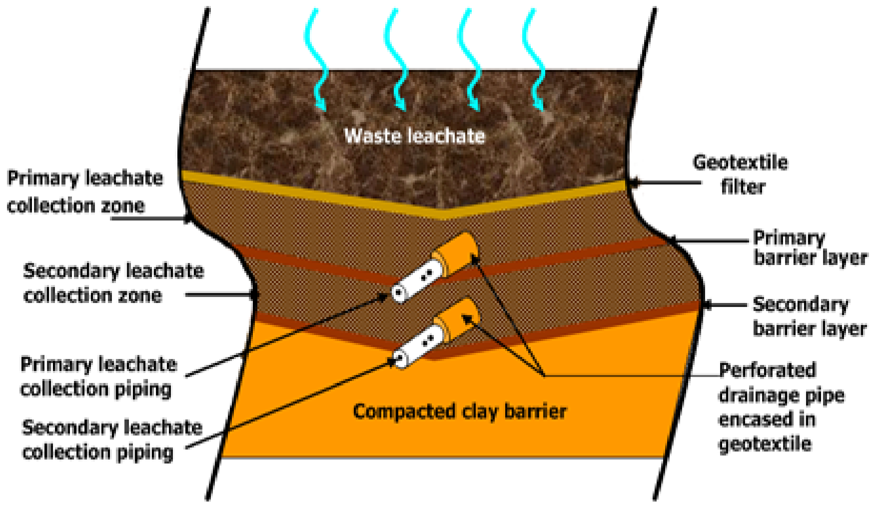


Figure 3.10 Schematic of waste placement, leachate collection and barrier system in a landfill

The other factors affecting the leachate composition related to filling of waste are given below.

Filling height

The surface to volume ratio of the waste in landfill has an impact over the infiltration, heat transfer and gas exchange occurring within the landfill. It is expected that an increase in landfill height may limit the affect of seasonal variation in the leachate composition and can preserve the heat from the microbial action to enhance further

degradation. However, aerobic conditions can be hindered due to limitations in gas transfer.

Another important consideration is the rate at which the height of the landfill is reached. Kylefors (1997) suggested that an increase in height of 4 m per year would increase the BOD and COD concentrations to an extent in which the transition from acidogenic to methanogenic conditions would be delayed.

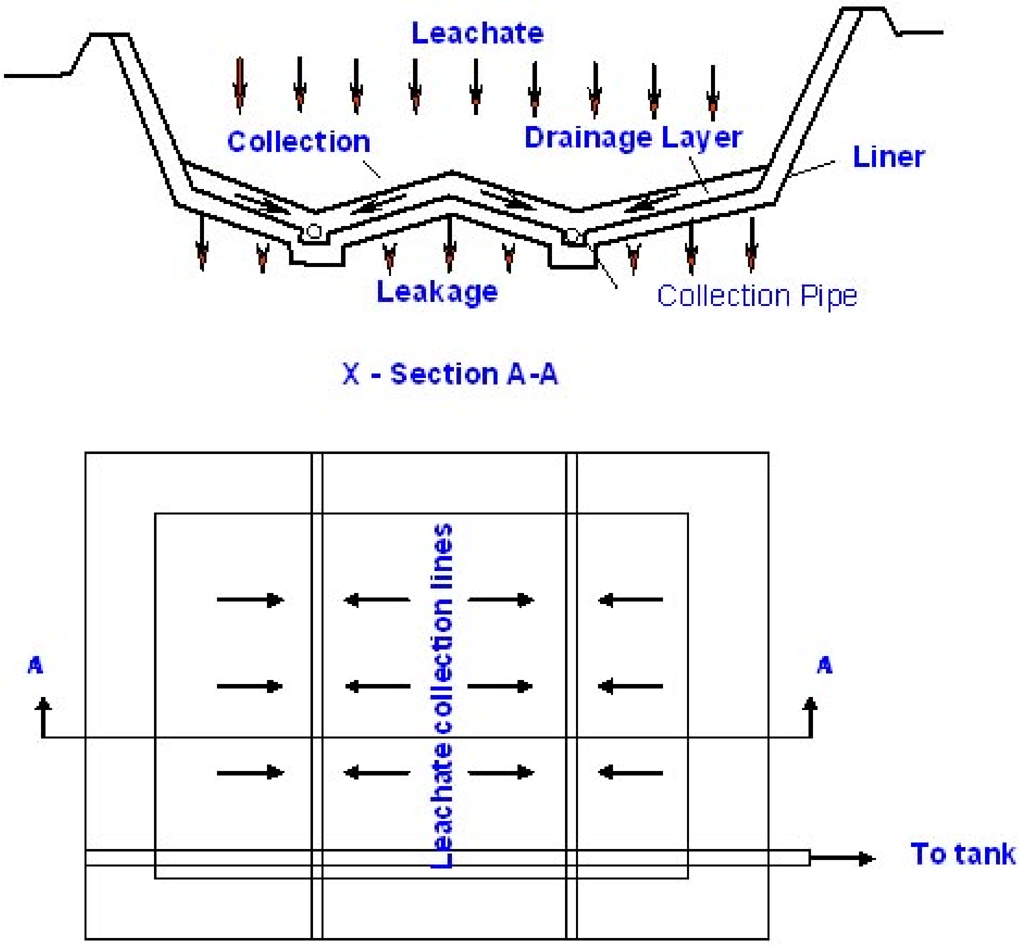


Figure 3.11 Arrangement for leachate collection in a landfill

Density

Low-density waste has a larger volume of air infiltrating through the landfill that promotes aerobic degradation. This enhances decomposition of the easily degradable waste fractions and elevates temperature within the landfill, which in turn can improve the conversion of waste, thus producing a better quality leachate. A prolonged aerobic phase can also lead to a drought condition within the landfill and reduce degradation rates. The reduction in degradation can also be attributed to the compaction achieved in the landfill. Pretreatment of the waste helps in compaction but the plastic content restricts the compaction to 400 -500 kg/m³ as cited by Ranaweera and Tränkler (2001) for Phittsanulok landfill in Thailand. Chang (1993) observed that simple shredding of MSW assisted in increasing the density from 420 – 810 kg/m³.

Bottom liners and top covers

The bottom liners are selected to prevent seepage of leachate into the aquifers and water courses, whilst top covers aid in maintaining moisture within the fill as well as limit infiltration, thus slowing down the degradation process. The liners are as shown in figure 3.9.

Chapter 4

LEACHATE TREATMENT

4.1 Introduction

Disposal of municipal solid waste in Asia is generally by open dumping and most disposal sites do not have a proper leachate treatment system. If present, varied treatment processes are used; majority of them are not properly designed to manage the quantity and characteristics of the generated leachate. Therefore, the objective of leachate management for solid waste disposal system should be to develop a highly cost effective treatment system with low area requirement and to identify the most significant factors affecting it. Based on this, it should also be able to set suitable criteria and prepare guidelines for proper leachate management in MSW dumpsites to reduce contamination and environmental impacts.

Leachate treatment is dependent on the quality and quantity of the leachate input, discharge limits or removal efficiency requirements, quantity of residual products and their management, site location and economics.

Generally, high organic and ammonia loads are the key factors in leachate treatment. The feasibility of treating leachate to COD levels less than 1,000 mg/L is uncertain, since at these values COD is primarily composed of humic and fulvic acids. The effect of these substances on aquatic life of receiving waters is dependent extensively on specific cases. While developing a treatment sequence for leachate to be discharged into a river, Robinson *et al.* (2002) reported that COD levels of about 500 mg/L comprising humic and fulvic acids did not adversely affect aquatic life.

High ammonia concentrations and phosphorus deficiency in leachate hamper the efficiency of biological treatment. A general consensus among researchers is that high nitrogen levels are also hazardous to receiving waters and need to be removed prior to discharge. This is generally carried out through biological nitrification-denitrification processes for young leachate and through physico-chemical processes in the stabilized leachate.

Leachate varies widely in quantity and in composition from one place to another (Kennedy *et al.*, 1988). Such a variable nature along with other factors make the applicability of a treatment method highly dependent on leachate characteristics and tolerance of the method against changes in leachate quality (Henry *et al.*, 1982).

The success of treatment depends also on the characteristics of the leachate and age of the landfill. Biological and physical/chemical treatment processes, land treatment systems and co-treatment with municipal wastewater can accomplish leachate treatment. Table 4.1 compares the performance of physical, chemical and biological treatment on different aged leachate.

Several wastewater treatment processes have generally been used to treat leachate (Amokrane *et al.*, 1997). The major biological treatment processes comprise aerobic and anaerobic while physical and chemical treatment processes comprise oxidation, coagulation-flocculation, chemical precipitation, activated carbon absorption and membrane processes.

Table 4.1 Comparative performance of various treatment processes for leachate treatment

Treatment Process	Young <5years	Medium 5-10 years	Old >12 years	Metals	VOC	Nitrogen	Priority Pollutants	Solids
Physical								
Natural evaporation	A	A	A	A	A	A	A	A
Floatation	D	D	D	B	B	D	B	A
Air stripping	D	D	D	D	A	A	B	D
Filtration	D	D	D	A	D	D	D	A
Membrane processes	A	A	A	A	B	A	A	A
Chemical								
Coagulation/ precipitation	C	B	C	A	D	C	D	A
Chemical Oxidation	C	B	B	D	D	D	A	D
Ion exchange	C	B	B	A	D	B	D	A
Carbon adsorption	C	B	A	D	A	D	A	D
Biological								
Aerobic suspended growth	A	B	C	A	A	B	B	B
Aerobic fixed film	A	B	C	A	A	B	B	B
Anaerobic suspended growth	A	B	C	A	A	B	B	B
Anaerobic fixed film	A	B	C	A	A	B	B	B

VOC- Volatile Organic Compounds; A- Good; B- Fair; C- poor; D – Not applicable

Source: Qasim and Chiang, 1994

4.2 Leachate Management

Leachate management is a complex task due to the highly variable nature of waste landfilled, type and design of the landfill, its age, and climatic and seasonal variations in different regions. Hence, it is difficult to recommend treatment options merely based on leachate age but it would be necessary to consider each scenario as a unique case. Treatment systems in recent years are sophisticated, reliable and are able to consistently treat leachate to keep in line with the specific discharge standards (Robinson, 1999). Although many full-scale leachate treatment systems use aerobic biological processes followed by reed bed polishing, majority of the European countries carry out co-treatment of diluted leachate with the municipal wastewater (Gierlich and Kollbach, 1998).

Usually biological treatment is insufficient to reduce the organic load. Hence, residual organics are removed by adsorption on activated carbon or oxidation of refractory organics to carbon dioxide, water and easily degradable constituents. However, elevated salt content in the leachate is unaffected by biological treatment, hence reverse osmosis (RO) can be adopted. RO concentrates the effluent to be recycled into the landfill while the permeate can be discharged into receiving water bodies.

Therefore, a combination of treatment options from existing plants and experiences demonstrates that there are no technical barriers for the treatment of landfill leachate for different discharge standards (Robinson, 1999).

4.3 Biological Treatment Processes

Majority of leachate treatment schemes that have been successfully installed in landfill sites, are anaerobic biological processes. The drawbacks generally experienced in biological treatment originate from operational problems such as foaming, metal toxicity, nutrient deficiency and sludge settling (Qasim and Chiang, 1994). Among the various biological treatment processes, Sequencing Batch Reactors (SBRs) has been to be proved a reliable and robust method for leachate treatment to meet specified effluent consent values. SBRs are often installed in combination with reed beds to provide consistently high effluent quality, which can be safely discharged into sensitive surface watercourses (Robinson, 1999). In general, the biological processes for leachate treatment can be divided into three broad categories: aerobic, anaerobic and anoxic processes.

4.3.1 Aerobic treatment

Conventional aerobic systems consist of either attached or suspended growth systems. The advantages and disadvantages of each system is case specific. Aerobic systems range from aerated lagoons, activated sludge and sequence batch reactors while attached growth processes include trickling filters and rotating biological contactors. In order to achieve good treatment efficiencies in activated sludge processes, the loading rate should not exceed 0.05 kg BOD_5/kgTS.d. Trickling filters are not generally used for treatment when the leachate contains high concentrations of organic matter (or precipitate-forming inorganic compounds) because of large volume sludge production causing clogging of the filters.

In attached growth systems, such as up-flow fixed film bioreactors, rotating biological contactors (RBCs) and trickling filters, microorganisms are established on an inert support matrix to aerobically degrade the contaminants. One promising methodology includes the use of active supports such as activated carbon, which adsorbs the contaminant and slowly releases it to the microorganisms for degradation. The microbial population may be derived either from the contaminant source or from inoculums of organisms specific to a contaminant. Nutrients are often added to the bioreactors to support the growth of microorganisms.

In maintaining a biological system, nutrients in the form of nitrogen, carbon and phosphorus are generally provided by the wastewater. However, in certain instances, these sources may need to be supplemented in order to maintain the conventional nutrient balance of

BOD_5:N:P = 100:5:1. Mavinic (1998) found that in a 10L aerobic SBR batch experiment BOD_5:N:P ratio of 100:3.2:1.1 was more successful in leachate treatment. When the nutrient ratio deviated from this optimum ratio, there was an increase in sludge production and BOD_5, SS and metal concentration in the effluent.

Activated sludge process (ASP)

The general operation conditions and parameters used for an activated sludge process to treat landfill leachate are given in table 4.2 (Qasim and Chiang, 1994).

Despite the variations in the leachate quality which depends on the source and time period from a single source, various researches conducted on bio-kinetic studies by Zapf–Gilje and Mavinic (1981); Wong and Mavinic (1984); Robinson and Maris (1985) and Gaudy *et al.* (1986) indicated a consistency in results (Qasim and Chiang, 1994).

Table 4.2 Operational conditions for activated sludge processes

Optimum Parameters	**Values**
MLVSS	5,000 - 10,000 mg/L
Food/Micro-organism	0.02 - 0.06 per day
Hydraulic Retention Time	1 - 10 days
Solids Retention Time	15 - 60 days
Nutrient requirements	BOD_5: N: P = 100: 3.2: 0.5
BOD and COD removal	90 - 99 %
Metal Removal	80 - 99 %

Dzombak *et al.* (1990) studied the extended aeration treatment of leachate. The BOD/COD ratio was below 0.1, characteristic of old leachate and contained mainly refractory organic compounds. They investigated different mean-cell residence times from 15 - 60 days and observed that a maximum COD removal of 40% could be achieved with mean-cell residence time of 60 days. This suggested that leachate from young landfills with organic matter containing high volatile acids could be more easily treated with an ASP than old leachates.

Doyle *et al.* (2001) performed sludge characterization in nitrification process for ammonia removal in an old landfill leachate. Although most researchers (Knox, 1985; Robinson and Maris, 1983; Strachan *et al.*, 2000) reported poor settleability of sludge (possibly due to high ammonia and low BOD:N ratio) in the activated sludge treatment of leachate, Doyle *et al.* (2001) reported good sludge settling with sludge volume index (SVI) ranging between 20 - 30 mL/g. A well-settled sludge generally exhibits a SVI of 80 - 150 mL/g. This was probably due to the presence of a high nitrifying fraction in the sludge. Further, the ability of the sludge to settle well indicates enhanced removal efficiencies and hence improved effluent quality.

Sequencing batch reactors (SBR)

Sequencing batch reactors are commonly used in biological treatment process for leachate. Several studies have been done to find its applicability of SBR for leachate. Doyle *et al.*

(2001) studied high-rate nitrification in SBR for a mature leachate from a domestic landfill. It possessed an average ammonia concentration of 880 mg/L while the average BOD_5 and COD concentrations were 60 and 1100 mg/L respectively. After nitrification with a hydraulic retention time (HRT) of 5 hours, treated leachate was discharged into sewer lines for denitrification. SBR studies in 6-hour cycles were conducted at an average temperature of 22±2 ºC with the sludge from nitrogen removal in a domestic wastewater plant. The reactor was operated at dissolved oxygen (DO) level of 2 - 4 mg/L and pH of 7. An average specific nitrification rate of 880 mg N/g VSS.d and maximum volumetric nitrification rate of 5.9 g NH_4-N/(L.d) could be achieved while mixed liquor suspended solids (MLSS) progressively increased to 13,000 mg/L at an infinite sludge retention time (SRT).

Hosomi *et al*. (1989) evaluated SBR treatment of leachate containing high nitrogen and refractory organic compounds. The advantages of SBR with nitrification-denitrification were found to be:

- Less likelihood of damages due to scale formation;
- Easier maintenance;
- Varying aerobic and anoxic cycles could effectively treat a wide range of pollutant loads;
- Unlikely sludge bulking; and
- Certain non-biodegradable halogenated organic compounds could be degraded biologically through nitrification and denitrification.

Chemical oxidation by ozone has been proven effective in the breakdown of high MW compounds that increased the biodegradability of organic substances. A comparative study of pre-treatment with and without ozone in SBR activated sludge process was performed (Hosomi *et al*.,1989). A reduction in pH to 5.2 could cause inhibition of nitrification. The adoption of the ozone pretreatment was ineffective in BOD removals with effluent of 1 - 3 mg/L. Ozonation as a pre-treatment was effective in removing COD and TOC with efficiencies of 50.8% and 43.9% respectively when compared with removal of 37.6% and 27.5% respectively without ozonation. A fractionation of molecular weight by Sephadex G-50 gel indicated that the SBR was effective in removing low molecular weight compounds while ozonation-SBR combination was effective for higher molecular weight compounds.

Yalmaz and Ozturk (2001) investigated the use of SBR technology for the treatment of high ammonia leachate by nitrification-denitrification and anaerobic pre-treatment. The study was carried out in two phases to evaluate SBR technology for the treatment of high ammonia leachate, and to investigate the feasibility of using leachate as a carbon source for denitrification. SBR was further tested to treat anaerobically pretreated leachate by an up-flow anaerobic sludge blanket reactor (UASB). The reactor was maintained at pH 7, hence loss of nitrogen via ammonia stripping was considered negligible with the main mechanism for nitrogen removal being nitrification-denitrification. The SBR achieved a 90% nitrogen removal when anaerobically pre-treated leachate was used with $Ca(CH_3COO)_2$ as a carbon source. The study revealed that young landfill leachate with a COD/NH_4-N greater than 10 was effective as a carbon source for denitrification. Although a two stage combination of biological treatment (UASB and SBR) were used in the scheme, the effluent did not meet the

discharge standards and required post-treatment in the form of physical-chemical processes (reverse osmosis or ozonation). This suggests that the most effective means of treating leachate is a combination of physico-chemical and biological treatments.

Aerated lagoons

An aerated lagoon is a simple, economic, robust, automatic and reliable process developed in the UK since 1980s. It has been used in different leachate plants and has proved successful with acceptable effluent quality under different conditions. Aerated lagoons for leachate treatment are in operation in different places (England, Ireland, Scotland and Wales). An extended aeration system has several advantages in terms of leachate treatment when compared to the activated sludge process. They are efficient for treating leachate with high ammonium concentration - 1000 mg/L and above. In comparison to the ASP, they have a longer HRT (days instead of hours) and are resistant to variations in influent quality and flow. Flocculation and settlement of microorganisms can affect nitrification process in the suspended growth systems. Despite slower settlement of sludge in an extended aeration system which acts as a settlement chamber, a well clarified effluent quality could be achieved.

Bryn Posteg was the first full-scale leachate treatment plant constructed in the UK. The landfill is an engineered one on the site of former lead mine at an altitude of nearly 400m where high rainfall rates make leachate management an important issue. An aerated lagoon SBR system is used to treat leachate at rates up to 150 m^3/day (Robinson, 1992). With over 18 years of operational experience and complete compliance to the consented discharge limits, the plant continues to operate well.

Trickling filters

The trickling filter (TF) consists of a bed of highly permeable media, a water distributor and an under drain system. The microorganisms attached to the filter medium degrade organics in wastewater. The filter media may be rocks, plastic or wood. The filter bed is normally circular with varying depths from 0.9 - 2.5m and an average of 1.8 m. It is generally believed that TFs can achieve better nitrification compared to activated sludge operations. In trickling filters, simultaneous removal of carbon and nitrogen is possible.

Usually leachate is treated by several physical/chemical wastewater treatment technologies. In most cases, COD and BOD are treated first while nitrification is done by a separate sophisticated system. Since the additional costs of these systems can be very high, use of a readily available media for in-situ nitrification should be considered primarily to avoid extra costs. Clabaugh (2001) has studied the possibility of removing ammonia nitrogen from bioreactor landfill leachate using TF biofilm technology in four laboratory scale reactors filled with four different types of packing media. The media consisting of pine wood chips was found to be the most efficient support for ammonia removal biofilms. The effects of varied concentration loading and hydraulic loading, and nitrification inhibitors were also studied. Changes in ammonia concentration did not have a significant impact on the ammonia removal rates (77 - 87) % in the reactor. The rates showed a strong dependence on hydraulic loading rate with the lowest producing the highest removal rates.

Rotating biological contactor (RBC)

The leachate from the waste disposal has a large quantitative/qualitative fluctuation due to the weather, composition of waste, quantity, etc. To treat the leachate efficiently to the required water quality level, biological contactor oxidation process can be adopted to treat organic pollutants (BOD, COD) especially with low concentrations or significant fluctuation of load. Stable treatment efficiency could be obtained through the RBC. Control over the operation and maintenance of RBC process is easy as the biological contactor oxidation does not require return sludge, and sludge bulking does not occur.

During an investigation by Siegrist *et al.*, (1998) to study nitrogen loss in a nitrifying rotating contactor to treat ammonium rich leachate without organic carbon. It was found that extensive loss of nitrogen (up to 70%) occurred in the nitrifying RBC treating the high ammonium concentration. DOC was less than 20mg/L, which suggested that the heterotrophic denitrification could be excluded.

Nitrification rate reached 3 - 4 g NH_4-N/m^2.d at a pH of 7.0 - 7.3 in the first two of three RBC compartments. An increasing partial pressure of oxygen and ammonium concentration favored nitrogen removal over ammonium oxidation. The reduction of nitrite in the aerobic biofilm layer close to the surface might therefore be coupled with ammonium oxidation that takes place in the deeper or temporarily anoxic layer of the biofilm. Henderson *et al.* (1997) also found that RBC could be effective in treating methanogenic landfill leachate.

4.3.2 Nitrification and denitrification process

The two main difficulties faced by the researchers in biologically treating leachate are:

1. It contains high nitrogen concentration with low COD:N ratio (Robinson and Maris, 1985); and
2. High ammonia concentration causes toxicity and problem arises due to phosphorous limitation (Keenan *et al.*, 1984).

During the acidogenic phase of the landfill, leachate contains large amounts of readily biodegradable organic matter in the form of VFA. As the landfill matures and enters the methanogenic stage, VFA are converted to biogas and thereby the content of readily biodegradable organic compounds decreases. In acidogenic and methanogenic stages, leachate contains high ammonium concentrations that affect treatment. Nitrification and denitrification processes play significant roles and hence have been given greater emphasis.

Ammonia toxicity occurs at a concentration of 31 - 49 mg/L (Cheung *et al.*, 1997). Complete removal of ammonia could only be achieved when the N:BOD_5 ratio does not exceed 3.6:100. Furthermore, sludge settling is adversely affected when ammonia concentrations exceed 200 mg/L (as N) in the mixed liquor (Robinson and Maris, 1985). Hence, it is important to remove nitrogen and its compounds from leachate by a pre-treatment prior to the biological processes.

There are various treatment options for the removal of oxidized and non-oxidized nitrogen from leachate. The most feasible options are air stripping of ammonia at high pH, ion

exchange of ammonia or nitrate and biological nitrification and denitrification. Biological nitrification-denitrification is one of the most economical processes for N-removal. Successful application of this system is dependent on microbial population, composition, concentration of the leachate and a variety of physical and chemical parameters (Table 4.3). The process essentially consists of oxidation of ammonia to nitrates and reduction of the nitrates to nitrogen gas. Nitrification is performed by the autotrophs and denitrification by the heterotrophs. Biological nitrification is preferred in the absence of inhibitory substances which interfere with the microbial ammonium oxidation process (Doyle *et al*., 2001).

Table 4.3 Operational and environmental conditions for nitrification-denitrification processes

Parameter	**Unit**	**Nitrification**	**Denitrification**
Substance transformed		NH_4^+	NO_3^-
End product		NO_3^-	N_2
Intermediate product		NO_2^-	NO_2^- ; N_2O
pH		7.5 to 8.6	6 to 8
Alkalinity	mmol of HCO_3^-/mg of N	Consumption - 0.14	Production - 0.07
Oxygen	mg O_2/L	> 2 (aerobic)	< 0.5 (anoxic)
Organic material	mg COD/mg of N	-	3
Phosphorus content	mg of P/g of N	> 4	> 11
Production of sludge	g/g of N	0.17	0.14
Temperature	An increase by 10 °C gives about twice the specific rate		

Source: Kylefors, 1997

Denitrification processes generally occur in anaerobic activated sludge, anaerobic filters and anaerobic lagoons. The carbon dosing in the denitrification stage is often an important parameter to be considered since a low dose with respect to nitrite to be denitrified can result in decreased ammonia removal. On the other hand, an overdose can increase the BOD and COD of the effluent and hence would require a post-aerobic stage to remove it, which becomes uneconomical. Moreover, the choice of carbon source is also important as an incorrect dosing by acetic acid can drastically lower the pH, affecting the denitrification process. Since the denitrifying bacteria become acclimatized to the carbon source, it would be essential to use the same carbon source instead of changing it after the start-up. Table 4.4 reflects the sources of carbonaceous substances with their potential to reduce nitrates. Endogenous respiration is not frequently used as it results in weak kinetics and requires larger volumes.

Methanol is generally used as an organic carbon source prior to denitrification; however, dosing should be monitored to prevent hydrogen sulphide formation and inhibition (Reeves, 1972). For denitrification processes, the choice of carbon source is based on the rate of nitrate reduction and economics.

Table 4.4 Nitrate reduction capability of various carbon substrates

Carbon substrate	Nitrate reduction at 20 °C ($mg\ NO_3$-N / g VS. h)
Ethanol	5.1
Acetate	4.9
Propionate	5.1
Methanol	2.5
Butyrate	5.1
Municipal wastewater	3.3
Endogenous respiration	1.5

Source: Kylefors, 1997

An increase in COD during start-up of a nitrification process is an indication of nitrite build-up in the system. Though ammonium is not oxidized in the COD analysis, nitrite gets oxidized and hence conversion of ammonium to nitrite in the process results in an increase in the COD. This nitrite build-up in the aerobic stage does not pose a problem unless concentrations become toxic to nitrification. Several studies (Bae *et al*., 1997; Aberling and Seyfried, 1992) have reported that conversion of nitrite directly to nitrogen gas in the denitrification process is more economical. This phenomenon could further be studied since the nitrification process could be inhibited once ammonia is oxidized to nitrite and then denitrification takes place via nitrite as opposed to nitrate. However, this approach would require external carbon source. Various studies on nitrification and denitrification processes are summarized in table 4.5.

Attached growth systems offer the advantage of reduced susceptibility to toxicity and variations in the environmental conditions. This is important while treating landfill leachate with high nitrogen content as there may be the risk of substrate as well as product inhibition in the nitrification process especially when nitrite accumulates. In high strength leachate, nitrification inhibition can be observed due to high ammonia content. Addition of activated carbon in the form of powdered activated carbon (PAC) is known for its ability to enhance biological treatment efficiency and remove refractory compounds to enhance nitrification. Removal of organic matter in PAC systems is a combination of adsorption and biodegradation. Activated carbon in conjunction with activated sludge increases the removal efficiency by adsorbing non-biodegradable, toxic and inhibitory organics. Many researchers suggest that a combination of PAC with a biological system would be more effective than biodegradation or adsorption alone. The system provides an attachment surface for microorganisms and protects them from shock loading of toxic and inhibitory compounds, while microorganisms also have the ability to regenerate carbon.

Horan *et al.* (1997) suggested that removing ammonia prior to organics in a granulated activated carbon biological fluidized bed (GAC-BFB) process could improve ammonia removal efficiencies to 90% while treating high ammonia landfill leachate though organic removal remained unchanged. This illustrated that the organic matter contributes to recalcitrant COD. As GAC-BFB displayed low suspended matter removal efficiency, adoption of a membrane system could prove more advantageous.

Table 4.5 Nitrogen and organic removal via nitrification-denitrification process

Process used	Experimental conditions	Results obtained
Four stage nitrification process (Illies, 1999)	• Methanol as carbon source; • Ammonia increased stepwise from 200-2300 mg/L; • HRT of denitrification and nitrification was 1.5-1.7 h and 3-3.4 h respectively; • SRT - 20 days.	• Ammonia removal better with acclimatization; • Ammonia removal of 90% could be achieved
Plug-in bench scale pre-denitrifying activated sludge process(Turk and Mavinic, 1989)	• Methanol with sodium acetate and phosphoric acid as C and P source; • SRT: 10-30 d sustainable at 10-13 d.	• 40% COD reduction during denitrification; • 63% high denitrification; • No nitrite toxicity effect; • SRT of 4 days was unsuccessful.
Suspended carrier biofilm process (Welander *et al.*, 1998)	• 60% plastic media, pH 7-7.8, HRT 4 d for nitrification; • 40% plastic media stripping at 30 rpm for denitrification; • HRT of denitrification and nitrification was 1.5-1.7 and 3-3.4 h respectively.	• Improved nitrification rate in the presence of plastic carrier media. Max. nitrification - 24 g N/m^3.h, denitrification - 55 g N/m^3.h; • COD removal - 20%; • Total N removal - 90%
Microorganism attached activated carbon fluidized bed process (Imai *et al.*, 1993)	• Methanol and KH_2PO_4 as C and P source; • Aerobic and anaerobic beds arranged in series with recycle of effluent from aerobic to anaerobic; • Leachate biodegradability < 0.1, Total N - 214 mg/L.	• Refractory compounds removal – 60%; • Total N removal – 70%; • Effective recycle ratio – 4
Granular activated carbon biological fluidized bed (Horan *et al.*, 1997)	• Reactors used in series to treat organic matter and ammonia, respectively.	• Ammonia removal – 70% • Organic removal – 20% • Ammonia removal when ammonia removal operation was done prior to organic removal-90%
Anaerobic filter with 2-stage activated process (Bae *et al.*, 1997)	• Initial ammonia concentration: 1400-1800 mg/L. • Initial COD concentration: 4000-7000 mg/L.	• Effluent COD- 150-200 mg/L; • Nitrification via nitrite more effective than nitrate similar to results obtained by Turk and Mavinic, 1989; Aberling and Seyfried, 1992.

Whilst biological processes are able to remove readily biodegradable organics, the non-biodegradable matter remains untreated. Biological nitrification on the other hand is generally difficult to achieve in landfill leachate due to large amounts of inhibitory substances present in it. Majority of physical processes are effective in ammonia stripping but have minimal effect on removal of organics.

4.3.3 Anaerobic treatment

The most common anaerobic treatment is the methanogenic degradation where the organic matter is completely degraded to methane and carbon dioxide. Anaerobic degradation follows a sequence where the interaction of several different microorganisms performing hydrolysis, fermentation, acetogenesis and methanogenesis is required (Kylefors, 1997). The degradation processes can be divided into different phases. The organisms occurring in a phase utilize the products of the preceding phase. To achieve good methane formation, the pH should be between 6 and 8 since anaerobic bacteria can be inhibited at highly acidic conditions. Further, both acid and methane fermenting bacteria are inhibited by heavy metal toxicity thus reducing the biological growth rate.

Anaerobic processes generally occur in attached film reactors. These reactors are insensitive to variations in loading, can retain biological solids irrespective of the waste flow and maintain a sufficiently high solid concentration over an extended period. It has been reported that removal efficiencies in anaerobic filters are higher than in anaerobic digesters maintained at the same hydraulic retention time (Pohland and Kim, 1999). The main advantages of anaerobic treatment over aerobic treatment are:

- lower energy requirement as no oxygen is required and thus reduces the operational cost;
- low sludge production as only about 10 - 15% of organics is transformed into biomass;
- biogas production (85 - 90%) favors the energy balance with a low nutrient requirement making it appropriate for treating leachate;
- possibility to treat leachate with high organic concentration without dilution used for aerobic processes, reduces space requirements, size of the plant and capital cost;
- valuable substances such as ammonia-nitrogen can be retained;
- anaerobic microorganisms seldom reach endogenous phase, important for the treatment of leachate with variable volume and strength;
- destruction of pathogens at thermophilic temperature ranges if it is intended to be used as fertilizers;
- elimination of odour problems;
- anaerobic sludge is highly mineralized than aerobic sludge, which increases its value as fertilizer if toxic metals are removed; and
- anaerobic sludge tends to settle more easily than aerobic sludge where addition of flocculants is required.

However, the major drawbacks of the systems are:

- working temperature above 30°C requires efficient kinetics;
- complexity of start-up period with strict operating conditions;
- apparently lower performance of anaerobic methods in elimination of heavy metals when compared to aerobic treatment systems; and
- need for complementary treatment to achieve high purification rates and acceptable effluent quality.

Cameron and Koch (1980) experimented anaerobic digestion at temperatures from 29 - 38ºC. The initial acclimation in the system was supplemented by adding lime to maintain pH and phosphorus to maintain BOD: N: P proportion. This process could reduce 65 - 80% BOD and 40 - 85% heavy metals from the leachate.

Mendez *et al.* (1989) conducted a treatment of young landfill leachate using anaerobic digestion. The COD removal efficiency was 65% with an HRT of 8 days. This study revealed that the COD removal efficiency of young landfill leachate is higher than old landfill leachate due to low percentage of refractory organic compounds.

Upflow anaerobic sludge blanket reactor (UASB)

The UASB technology combines well-mixed attributes of the contact system with an internal gas solids separation and clarification mechanism. The mixing within the reactor results from the gassing, which occurs as the organic components are distributed within the biomass bed at the bottom. The reactor contains no mechanical components but has a baffle arrangement to separate the gas, liquid and solid phases.

UASB reactor has achieved widespread acceptance as a high-rate partial treatment process for high organic strength wastewaters throughout the world. It is difficult to design a treatment process for leachate having a high organic strength and low suspended solids. In the past however, utilization of this treatment technique was thought to be inappropriate because it required a longer retention time and was often unreliable and sensitive to shock loading and toxic constituents. However, many of these drawbacks have been proved baseless since anaerobic reactors have performed with high efficiency.

As a pre-treatment, high rate anaerobic processes (such as UASB) have been found to be efficient in the treatment of municipal landfill leachate with COD higher than 800 mg/L and BOD/COD ratio more than 0.3 (Kettunen *et al.*, 1996). In particular, UASB reactors have exhibited superior performance compared to the other processes at high volumetric loading rates and with toxic and organic shock loads. However, effluents from UASB reactors cannot meet the discharge standards as they contain some amount of ammonia nitrogen and relatively high COD. Therefore, post-treatment by an aerobic process is needed to remove ammonia nitrogen and residual COD.

4.3.4 Summary of treatment process

Biological treatment takes the form of anaerobic, aerobic or anoxic processes. Each process is designed to remove wastewater constituents depending on the characteristics and nature of the water to be treated. For leachate, the main difficulty is the treatment of refractory compounds. The review of biological processes highlighted large space, energy and volume requirements necessary for sequence batch reactors, their advantage being immunity to shock loading and minimal operator input. A summary on aerobic processes, anaerobic processes and nitrification-denitrification is presented in tables 4.6, 4.7 and 4.8.

Anaerobic processes are difficult to control with longer SRT and HRT requiring larger volume reactors. Nitrification-denitrification processes are also difficult to control. These processes are susceptible to nitrification and/or denitrification inhibition if operational conditions are not maintained. Another disadvantage is the accumulation of nitrite or nitrate if either nitrification or denitrification is incomplete. This accumulation will result in high concentrations of nitrite and nitrate, which either accumulates in the reactors (causing shock to the biological process) or get discharged into the receiving waters.

Table 4.6 Treatment efficiencies of aerobic treatment systems

Processes	HRT (day)	Temperature (^{O}C)	COD loading (kg/m^3-day)	Initial COD (mg/L)	pH	BOD/COD Ratio	COD removal (%)	Initial NH_4-N (mg/L)	NH_4 removal (%)	Reference
Fill-and-draw batch process	1-5	23-25	0.5-1.7	3000-9000	6.0-8.0	0.5-0.8	30-90	-	-	Boyle and Ham, 1974
	10	22	1.66	16,000	7.6-8.4	0.4	97	TN: 280	92-95	Cook and Foree, 1974
	5	22	3.32	16,000	8.0	0.4	47	TN: 280	58	Cook and Foree, 1974
SBR	1	20	0.1	100-150	-	-	36-38	100-330	99	Hosomi, *et al*., 1989
	0.5	25	-	5295	9.1	0.4-0.5	60-68	872	-	Dollerer and Wilderer, 1996
	3.2	-	0.69	2200	6.8-7.1	0.46	95	35	>99	Zaloum and Abbott, 1997
	20	-	0.62	12,400	-	0.4	91	179	>99	Zaloum and Abbott, 1997
	8.5	20-25	-	1690	-	0.05	-	616	>99	Fisher and Fell, 1999
Aerated lagoon	34	0-20	<1.0	5600	-	0.7	97	130	93	Robinson and Grantham, 1988
	-	-	-	34,000	-	0.6	99	600	99	Robinson *et al*., 1992
Activated sludge process	20	10	1.2	24,000	6.0-7.5	0.5	98	790	>99	Robinson and Maris 1985
	20	10	0.06	1200	-	0.2	41	370	90	Robinson and Maris, 1985
	0.3	-	-	250-1200	-	-	85-90	-	-	Schuk and James, 1986
	31	15-18	0.4	12,500	-	0.6	93-96	-	-	Avezzu *et al*., 1992
RBC	2.9	-	2.8	9300	-	0.7	86	-	-	Vicevic *et al*., 1992

Note: SBR – Sequencing Batch Reactor; RBC – Rotating Biological Contactor

Table 4.7 Treatment efficiencies of anaerobic treatment systems

Processes	HRT (day)	Temperature (^{O}C)	COD loading (kg/m^3-day)	Initial COD (mg/L)	pH	BOD/COD Ratio	COD removal (%)	Reference
Anaerobic digestion	12.5	15	0.7	8400	6.9-8.1	0.7	73	Boyle and Ham, 1974
	5-20	23	0.4-2.2	2700-12,000	6.9-8.1	0.6-0.8	87-96	Boyle and Ham, 1974
	12.5	10	0.7	8300	6.9-8.1	0.8	22	Boyle and Ham, 1974
	5-20	29-38	0.2-1.3	20,000-30,000	5.0-5.3	0.5	65-80	Cameron and Koch, 1980
Anaerobic pond	86	20-25	-	6280	6.6	0.7-0.8	95	Bull *et al.* 1983
Anaerobic filter	2-4	21-25	1.5-2.9	13,780	7.3-7.7	0.7	68-95	Henry *et al.*, 1987
	0.5-1.0	21-25	1.3-3.1	3750	7.0-7.2	0.3	60-95	Henry *et al.*, 1987
	0.5-1.0	21-25	1.4-2.7	1870	7.1-7.9	-	88-90	Henry *et al.*, 1987
	17	37	3.8	9000	-	0.7	83	Wu *et al.*, 1988
UASB	0.3-0.5	33-35	15-25	25,000-35,000	7.4-7.8	-	80-85	Jans *et al.*, 1992
	1.0-3.2	28-32	3.6-20	11,500-33,400	-	0.7	66-92	Blakey *et al.*, 1992
	0.6	15-20	5-15	2800-13,000	-	-	73-93	Garcia *et al.*, 1996
	0.5-1.0	-	1.2-19.7	4800-9840	-	0.86	77-91	Kennedy and Lentz, 2000
USB/AF	2.5-5.0	30	1.3-2.5	17,000-20,000	-	-	80-97	Nedwell and Reynolds, 1996
AnSBR	1.5-10.0	35	0.4-9.4	3800-15,900	7.4-8.0	0.54-0.67	65-85	Timur and Ozturk, 1999

Note: USB/AF = Upflow hybrid sludge bed/fixed bed anaerobic system; AnSBR = Anaerobic sequencing batch reactors

Table 4.8 Summary of nitrogen removal in landfill leachate

Wastewater (WW) source	COD (mg/L)	BOD (mg/L)	TKN (mg/L)	External carbon source	Carbon removal technique	Nitrogen removal technique	Removal (%)		Reference
							C	N	
Domestic	750	110	60 - 80	-	BNR activated sludge	Ext nitrification-fixed film	92.00	-	Hu *et al.*, 1989
Landfill leachate (LF)	-	-	260.0	Methanol+ sodium acetate	-	Nitrification-denitrification	-	-	Turk and Mavinic, 1989
Potato starch WW	3000	990	1050	Acetic acid	Anaerobic	Nitrification - denitrification via NO_2^-	-	-	Aberling and Seyfried, 1992
Domestic leachate	1100	60	880	-	None required	Nitrification	-	-	Doyle *et al.*, 2001
Domestic leachate	311	46	2200	Methanol	None required	4-stage Bardenpho process	-	> 90	Illies, 1999
Co-disposal leachate	1000	100	340	-	AS + biological filter	AS + biological filter	-	-	Knox, 1985
MSW LF leachate	1400	85	600	Acetic acid + Methanol	SCBP biofilm	SCBP biofilm	20.00	90.00	Wetlander *et al.*, 1998
MSW LF leachate	125	27	253	Methanol	Ozone + SBR + AS	SBR + AS + ozone	>50	95.00	Hosomi *et al.*, 1989
Co-Disposal LF leachate	131	12	214	Methanol	MAACFB aerobic	MAACFB anaerobic	60.00	70.00	Imai *et al.* 1993
Landfill leachate (Raw)	2450	185	744	(Heavy metal precipitation)	-	-	-	-	Horan *et al.*,1997
Landfill leachate (treated)	2100	110	720	-	GAC-fluidized bed	GAC-fluidized bed	60.00	70.00	Horan *et al.*,1997
Landfill leachate	16,500	9500	1800	Leachate + $Ca(CH_3COO)_2$	UASB + SBR	Nitrification-denitrification	-	90.00	Yalmaz and Ozturk, 2001
UASBR effluent	1200	145	890	Leachate + $Ca(CH_3COO)_2$	SBR	Nitrification-denitrification	-	90.00	Yalmaz and Ozturk, 2001
Landfill leachate	2427	320	1274	Methanol	Nitrification-denitrification	Nitrification-denitrification	-	-	Strachan *et al.*, 2000
Landfill leachate	1878	695	445	Methanol	Nitrification-denitrification	Nitrification-denitrification	-	-	Strachan *et al.*,2000

4.4 Physical Treatment

There are several physical processes used for leachate treatment, which include activated carbon adsorption, filtration, pressure-driven membrane filtration processes, reverse osmosis and evaporation. These processes generally cannot be applied successfully to remove the organic material from raw leachate. Therefore, Pohland and Harper (1985) suggested that reverse osmosis, activated carbon (PAC and GAC) and ion exchange could be used more successfully as a post-treatment for leachate after biological treatment. Although each of the processes is coupled with a biological system, they have a limited application and therefore can be even more effective when physico-chemical treatment is used as pre- and post- treatment for biological systems.

4.4.1 Activated carbon adsorption

Granular activated carbon (GAC) in combination with biological pretreatment is the leading technology for the treatment of landfill leachate for the removal of COD, adsorbable organic halogens (AOX) and other toxic substances. Adsorption is the process by which molecules with particular characteristics of size and polarity are attracted and held to the adsorbing surface. More than 130 different types of organics have been identified on spent carbon from leachate treatment plants. GAC is used to remove AOX and COD, both of which are not the primary focus of biological treatment systems and may therefore be found above discharge consent levels from such treatment systems. With particularly dilute leachate, it may be operated with a plate separator or pressurized sand filter to remove suspended solids from the flow to ensure that they do not block the carbon filter. It is necessary to ensure that no substances are present in the leachate that would damage the carbon prior to selecting a system. Activated carbon has been used as a final polishing step after biological treatment, ultra-violet (UV)-oxidation, sedimentation and other physico-chemical treatment methods.

Fettig *et al.* (1996) studied the treatment of landfill leachate by pre-ozonation and adsorption in activated carbon columns. The data evaluation revealed that biodegradation took place inside the activated carbon beds. Therefore, total removal of ozonated leachate in activated carbon columns was found to be higher than the removal due to adsorption processes.

The adsorption of organics and metals from leachate is generally carried out by powdered or granular activated carbon or peat. This process can be adopted as a post-treatment for biologically or chemically treated leachate and can attain a COD removal efficiency of 30 - 70%. Pohland and Harper (1985) reported that removal efficiencies were lower for chemically treated leachate than for biologically treated leachate.

A review of physical-chemical processes done by Qasim and Chiang (1994) indicated that adsorption by activated carbon was more effective in organic removal from raw leachate than chemical precipitation with COD removal efficiencies of 59 - 94%. However, activated carbon proves disadvantageous for large quantities without sustainable high COD removal efficiencies. Further, the effectiveness of carbon adsorption on the removal efficiency of COD and TOC in young leachate containing high volatile fatty acid content is dependent on the magnitude and proportion of the high and low molecular weight free volatile acid fractions in the leachate.

Cook and Foree (1974) as cited in Qasim and Chiang (1994) reported that activated carbon treatment removal efficiencies were enhanced by a biological pre-treatment whilst Chian and DeWalle (1977) obtained a 70% COD removal by activated carbon on a low biodegradable (BOD/COD <0.1) leachate. Thus activated carbon coupled with a biological system was effective in treating stabilized leachate containing a large quantity of refractory compounds.

4.4.2 Membrane filtration

A membrane is defined as a material that forms a thin wall capable of selectively resisting the transfer of different constituents of a fluid and thus effecting a separation of the constituents. Membrane filtration can be defined as the separation of solid immiscible particles from a liquid or gaseous stream based primarily based on size differences. The principle objective of membrane manufacture is to produce a material of reasonable mechanical strength that can maintain a high throughput of a desired permeate with a high degree of selectivity (Visvanathan *et al*., 2000). Figure 4.1 illustrates the separation capability of these membrane processes based on size exclusion.

The classification of membrane separation processes are based on particle and molecular size. The processes such as reverse osmosis (RO), nanofiltration (NF), ultrafiltration (UF) and microfiltration (MF) do not generally require the addition of aggressive chemicals and can be operated at ambient temperature making these processes both environmentally and economically attractive alternatives to the conventional operating units.

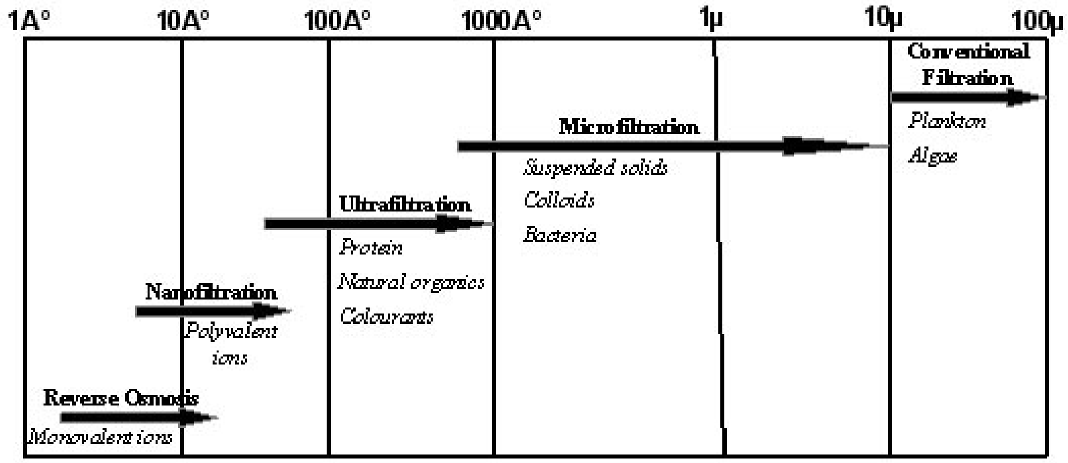


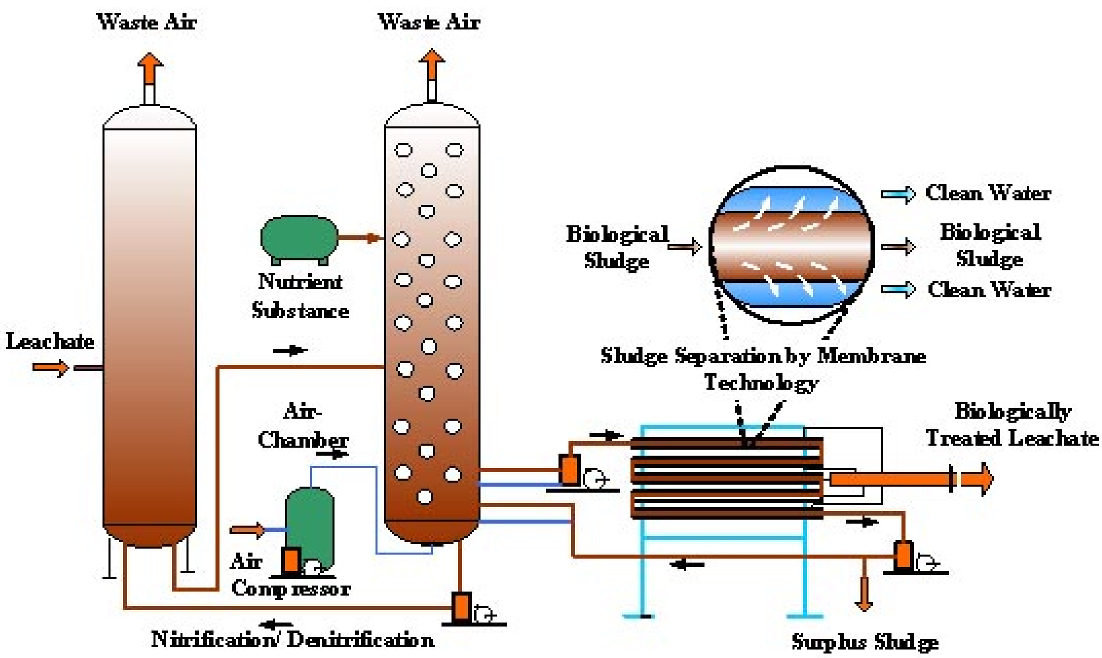
Figure 4.1 Selected filtration processes for water treatment and effective size range

RO membranes can remove more than 99% organic macromolecules and colloids from feedwater and up to 99% of the inorganic ions. Due to high rejection ability, reverse osmosis membranes retain both organic and inorganic contaminants dissolved in water with rejection rates of 98 - 99% thus being useful for purifying of liquid wastes such as leachate. Permeate generated from the reverse osmosis unit is low in inorganic and organic contaminants which meet the discharge standards.

RO technology was reported as the most effective in COD removal among different physical-chemical processes evaluated. The removal efficiencies are dependent on the

choice of membrane material. Chian and DeWalle (1977) reported 50 - 70% removal of TOC with cellulose acetate membranes while the use of polyethylamine membranes increased efficiency to 88%. The presence of VFA appeared to hamper filtration with an increase in pH from 5.5 - 8, reducing permeability of TOC in permeate. Reverse osmosis further offers the advantage of almost complete total solid removal and is effective as either a pre-treatment or a polishing of a biologically or ion exchange treated effluent. However due to biofouling of the membranes, biological pretreatment of leachate is necessary.

A typical RO unit commonly used in Europe for the removal of salts after biological treatment of the leachate is depicted in figure 4.2.



Source: Gierlich and Kollbach, 1998

Figure 4.2 Typical reverse osmosis plant for reduction of salts in leachate treatment.

Membrane filtration is less effective in treating young or acidogenic leachate. The efficiency of different membrane technologies in treating methanogenic leachate is presented in table 4.9.

Table 4.9 Removal efficiency of moderate to high concentrations of pollutants using nanofiltration, ultra filtration and reverse osmosis

Parameters	Reverse osmosis removal (%)	Nanofiltration removal (%)	Ultrafiltration removal (%)
COD	95 - 99	80 - 90	25 - 60
NH_4(N), pH = 6.5	90 - 98	80 - 90	< 20
AOX	95 - 99	70 - 90	30 - 60
Chloride	90 - 99	40 - 90	< 40

Source: Kylefors, 1997

Although nanofiltration and reverse osmosis are quite effective in leachate treatment in terms of organic, inorganic, nitrogen and AOX removals, the disadvantages of membrane treatment system are its susceptibility to fouling and short lifetime. Colloidal material as well as metal precipitation can cause fouling and clogging in the membranes. Fouling leads to an increase in osmotic pressure and hydraulic resistance, thus increasing the energy uptake. To minimize the fouling effect, pH can be adjusted from 4 - 7.5.

4.4.3 Evaporation

As cited by Ehrig (1988), leachate can be separated into a clear liquid and a solid phase bearing the pollutants through evaporation. Practically, this is difficult as the solid phase or the condensate with volatile or chlorinated organic compounds and ammonia requires further treatment. Concentration and nitrogen recovery with the evaporation technology is possible. Evaporation is a simpler technology with easier application and less complicated technical difficulties. Evaporation is also a cost-effective option if used in tropical countries applying solar radiation that however would risk volatilizing a number of VOCs.

Nevertheless, the problems concerned with evaporation of raw leachate as cited by Cossu *et al*. (1992) are:

- formation of foam due to high organic content;
- encrustation and corrosion, causing equipment damages;
- fouling on the heat-transfer surface;
- need for post-treatment for the removal of ammonium and halogenated organic materials; and
- high energy costs.

4.4.4 Summary of physical treatment process

Since membranes cannot retain volatile fatty acids, acidogenic leachate is poorly treated using membrane systems. It is therefore more practical to treat young leachate with activated sludge processes. However, the disadvantages of activated sludge processes are poor retention of solids, and large area and volume requirements. A coupling of a membrane and activated sludge process to form a membrane bioreactor (MBR) may be more viable as the membrane ensures total solids retention. For moderate to strong methanogenic leachate, a good removal of several substances including metals can be achieved using this combination. The MBR technology can achieve high treatment efficiency with an excellent effluent quality.

4.5 Chemical Treatment

A wide scope of chemical treatment is available for leachate. The advantages of chemical methods in general include immediate start-up, easy automation, insensitivity to temperature changes and simplicity of plant and material requirements. However, these advantages are out-weighed by the disadvantages of large quantities of sludge generated due to the addition of flocculants and chemicals with high running costs. Thus, chemical and physical treatments are merely used as pre- or post-treatment of leachate to complement biological processes. The various chemical processes used in leachate treatment are coagulation, precipitation, oxidation, reduction, stripping, ion exchange, etc., which are described below:

4.5.1 Coagulation and precipitation

Coagulation and precipitation involves the addition of chemicals to alter the physical state of dissolved and suspended solids to facilitate removal by sedimentation. This treatment is effective on leachate with high molecular weight organic material such as fulvic and humic acids. Since these components are generally difficult to degrade biologically; physical-chemical processes prove to be more effective with approximately 60% reduction of COD for methanogenic leachate.

Lime as a precipitating agent can reduce color up to 85% and remove metals through precipitation. Chian and DeWalle (1977) and Ho *et al.* (1974) reported that precipitation using lime could remove organic matter with molecular weight greater than 50,000 Da. This particular fraction is present in low concentrations in young landfills and absent in older landfills. Therefore, lime treatment is most effective for medium age landfill leachate. Easily biodegradable fatty acids are resistant to coagulation/precipitation and hence should be treated biologically.

Alum, ferric chloride, lime and polymers used in coagulation/precipitation are effective in removing only 30% of BOD and COD from either raw or biologically treated wastewater. They also have minimal effect in the removal of color, iron and other metals. It is expected that greater COD removal can be achieved if pre-treatment by ammonia stripping is implemented since this will enable the removal of volatile organic compounds in the leachate.

4.5.2 Chemical oxidation

Chemical oxidation technologies are useful in the oxidative degradation or transformation of a wide range of pollutants present in drinking water, groundwater and wastewater treatment (Venkatadri and Peters, 1993). Generally, chemical oxidation processes are incorporated into treatment sequences to treat constituents of wastewaters that are resistant to biodegradation or create toxicity in biological reactors. Chemical oxidation processes are widely used in leachate treatment. A variety of chemical oxidants are used for leachate treatment which includes hydrogen peroxide, ozone, chlorine, chlorine dioxide, hypochlorite, UV radiation and wet oxidation.

While selecting an oxidizing agent, it is necessary to consider the oxidizing potential of particular oxidant. The oxidant with a greater oxidizing potential is more powerful in the oxidation process. A summary of selective oxidants and their oxidizing potential is presented in table 4.10.

The oxidation potentials presented in table 4.10 clearly illustrate that ozone has a higher oxidizing potential among other oxidation agents that are suitable for application in water technology. However, hydroxyl radicals exhibit a stronger oxidation behavior than ozone. Since oxidation processes are energy intensive and expensive, their application is limited. Moreover, as oxidation processes are dependent on the stoichiometry, a large amount of oxygen is required for higher organic concentrations (Webber and Smith, 1986). Chlorine, chlorine dioxide and hypochlorite compounds are not used for oxidation as toxins can be produced.

Table 4.10 Oxidation potentials of selective oxidants	
Oxidant	**Oxidizing Potential (V)**
Fluorine (F)	3.03
Hydroxyl radical (OH^-)	2.80
Oxygen atom (O)	2.42
Ozone (O_3)	2.07
Hydrogen peroxide (H_2O_2)	1.76
Permanganate ion (MnO_4)	1.67
Chlorine dioxide (ClO_2)	1.15
Chlorine (Cl_2)	1.36

Ozone

Chemical oxidation with ozone is an innovative technology for the treatment of effluents and leachate that are contaminated with organic chemicals because of its capability to completely convert the organic contaminants to carbon dioxide.

Due to its strong oxidizing ability, ozone is effective and practical as a pre-treatment to remove refractory species and as a polishing step to treat organics or increase the biodegradability of refractory compounds without a significant reduction in COD. However, if used in excess, ozone can be hazardous since one of the oxygen present is an unstable atom with a strong tendency to break away and attach itself to other substances. This atom tends to oxidize most substances and hence proves destructive requiring air pollution clean-up caused by the formation of oxides of nitrogen (photo-chemical smog).

The oxidation potential of ozone is sufficient for the direct degradation of organic substances. Hydroxyl radicals can be formed in the presence of initiators such as hydroxide ions, hydrogen peroxide or humic substances present in the leachate. Since leachate has a high humic content, chemical oxidation with ozone may alone prove advantageous as the initiators are already present in the leachate and technical difficulties of combining ozone with another initiator can be eliminated. The oxidation of organic compounds by ozone is a zero order reaction, i.e. the reaction rate is constant until about 20% of the initial amount is left (Kylefors, 1997).

Effectiveness of perozone in reducing COD was around 70% (Steensen, 1997). Most organics display a fast reaction with the ozone molecule even with a small quantity. However, compounds such as halomethanes require an advanced oxidation by hydroxyl radicals. Hence, the feasibility of oxidation by perozone in terms of economics, reaction time and operating conditions should be considered. Table 4.11 summarizes the various application of ozone in leachate and sludge treatment.

Hydrogen Peroxide

To facilitate degradation and oxidize organic compounds (humic acids), the oxidizing potential of hydrogen peroxide is insufficient. However, hydrogen peroxide in the presence of a suitable catalyst, usually iron salts or UV-radiation (Steensen, 1997) can form hydroxyl radicals, which have a greater oxidation potential than hydrogen peroxide or ozone individually. The economic feasibility of adopting hydrogen peroxide in the

process of chemical oxidation is low, as a high peroxide concentration of about 120 - 250 g/m^3 is required to remove 1 kg of COD (Steensen, 1997).

UV-Radiation

UV-radiation is generally coupled with hydrogen peroxide or ozone to form an oxidation complex. UV on its own does not oxidize certain organic compounds present in the leachate though it is a good disinfectant. When decomposition of dioxins in a landfill by advanced oxidation processes was studied, O_3/H_2O_2 and $UV/O_3/H_2O_2$ processes were tested to evaluate their performances of decomposing dioxins contained in a landfill leachate. The data suggested that the $UV/O_3/H_2O_2$ process was more useful for the removal of total dioxins, but the O_3/H_2O_2 process was effective in reducing toxicity equivalent (TEQ), a parameter that expresses the actual toxicity of dioxins (Nakagawa *et al*., 1999).

Table 4.11 Application of ozonation in leachate and sludge treatment

Experiment	**Results**
Investigation of physico-chemical treatment for leachate (Bjorkman and Mavinic, 1997)	• Recirculation after ozonation improved biodegradability and minimized foaming problem; • Concentration of 100mg/L O_3 was effective in reducing the COD.
Chemical oxidation of leachate (Steensen, 1997)	• Pre-treatment using biological processes prior to ozonation is required to ensure nitrification; • As a post treatment perozone (O_3 and H_2O_2) is more effective; • pH reduction increases the oxidation potential of perozone.
Ozonation of leachate (Gierlich and Kollbach, 1998)	• Ozone effectively reduces 80% ammonia; • Ozonation is effective when biological treatment is used as a pre-treatment.
Investigation of various treatment processes for leachate (Chian and DeWalle, 1976)	• COD removal is ineffective for treating a young landfill leachate as VFA present in it are resistant to O_3.
Ozonation of young leachate (Qasim and Chiang, 1994)	• Ozone is ineffective for young landfill leachate whereas 22% COD reduction could be achieved for old landfill leachate.
Ozonation of leachate (Ho *et al*., 1974)	• Leachate of COD 7000mg/L was treated with ozone at the rate of 25.7 mg/L with detention periods of 1 and 4 hrs. COD reduction was found to be 5 and 37% respectively. • Iron removal was 80 and 95% respectively
Peroxide and ozone treatment for color removal (Siegrist *et al*., 2001)	• Perozone was effective at pH 2-4; • Increase in ozone time to 60 min removed 53% color; • Waiting time of 5 min improved COD removal from 27-31%.
Sludge ozonation (Ahn *et al*, 2001)	• Sludge mass reduction and improvement in settleability; • Solubilization and mineralization increased with ozone dose.
Biodegradability of ozonated sludge in aerobic and anaerobic systems (Yeom *et al*., 2001)	• Sludge solubilization increased with ozone dosage 0.5 g/g SS and decreased at higher doses; • Ozonated sludge in both aerobic and anaerobic system was 2 to 3 times more degradable than raw sludge.

As oxidation with ozone breaks down non-biodegradable organics into easily biodegradable organic acids, biological post-treatment can be more feasible. Biological post-purification further offers the advantage of reducing the ozone demand. Another possible strategy for leachate treatment could be to repeat the ozone-biological treatment

sequence several times. The effectiveness of this approach can be investigated and based on the results; a minimum re-circulation ratio can be evaluated.

Sludge disintegration has commonly been used as a pre-treatment for sludge digestion. The digested sludge has the advantage of controlling and reducing sludge bulking in conventional activated sludge processes and thus provides an internal carbon source for biological nutrient removal.

4.5.3 Chemical reduction

According to Kylefors (1997), chemical reduction can be accomplished by activated catalytic methods using aluminum, zinc and iron. Reduction reactions render some inorganic constituents non-hazardous and are more easily removed by post-treatment processes. The reaction involves the addition of chemical reducing agent under controlled pH. This reaction raises the valence state of one reactant and lowers the valence state of the other. It is effective in removing oxidizable inorganics such as cyanides, ammonia and some metals (Fe, Mn, Se) as well as reducing of metals (Cr, Hg, Pb, Ag, Ni, Cu and Zn) which may be present in the leachate.

Reactions are carried out in closed reactors with rapid mix agitators and are monitored by reduction potential probes. The reaction processes are exothermic and can be very violent. For this reason, they are conducted with dilute concentrations and as batch processes.

4.5.4 Ammonia stripping

Air stripping of ammonia involves passage of large quantities of air over the exposed surface of the leachate, thus causing the partial pressure of the ammonia gas within the water to drive the ammonia in the liquid to the gas phase.

Ammonia stripping can be undertaken by water falling through a flow of air as in stripping towers or by diffusion of air through water in the form of bubbles. Stripping towers are more efficient since there is a better contact between the gas and liquid phases when dispersion of liquid takes place in the form of fine droplets. Since ammonia stripping is mass transfer controlled, the surface area of the liquid exposed must be maximized. This can be achieved by creating fine droplets with the help of diffusers or sprayers. The process is further subjected to careful pH control and involves the mass transfer of volatile contaminants from water to air.

The process entails the existence of ammonium ions in equilibrium with ammonia and hydrogen ions in the leachate. The equilibrium is as illustrated below.

$$NH_4^+ \Leftrightarrow NH_3 + H^+$$

The formation of free ammonia is favored when the pH is above 7 resulting in an equilibrium shift to the right. At pH greater than 10, over 85% of ammonia present may be liberated as gas through agitation in the presence of air (Reeves, 1972). Ammonium hydroxide (NH_4OH) is formed as an intermediate at pH between 10 and 11 in the reaction. The bubbling of air through ammonium hydroxide solution results in the

freeing of ammonia gas. This process is subject to temperature and solubility interferences since ammonia is highly soluble in water and its solubility increases at low ambient temperatures.

To review the effectiveness of ammonia stripping as a pre-treatment option for landfill leachate, Cheung *et al.* (1997) investigated airflow rate and pH as critical parameters for the optimization of ammonia stripping in a stirred tank. In the study, varying airflow rate of 0, 1 and 5 mL/min and varying lime dosage of 0 and 10,000 mg/L for pH adjustment was evaluated. The study revealed an enhanced ammonia removal (86 - 93%) at airflow rate of 5 mL/min and pH greater than 11. It was realized that effectiveness of the process was also dependent on area: volume (A/V) ratio of the tank and leachate quality. With 24-hour detention time and increased A/V ratios, ammonia removal efficiencies as compared with previous studies by other researchers were 40 - 53% for A/V = $23m^{-1}$ (Cheung *et al.*, 1997) and 19% for A/V = $1.8m^{-1}$ (Smith and Arab, 1988 as cited in Cheung *et al.*, 1997). This indicated that the mass transfer governed the mechanism for ammonia stripping and it was further revealed that ammonia desorption into air bubbles was less significant than the air-water interfacial area. The provision of air to the system promotes air bubble formation and turbulence at the air-water interface, which aids in increasing the surface area for ammonia removal. Thus, an indefinite increase in airflow rate could greatly enhance ammonia stripping efficiency over a short detention time. The practicality of this approach depends on the power mixer efficiency and mass transfer rate, which should be optimized to render the process cost-effective.

Ammonia stripping has also the advantage of precipitating organics and heavy metals present in the leachate. The concurrent COD and phosphorus removal via lime precipitation is independent of airflow rate. The change in color of the raw leachate from dark brown to pale yellow after precipitation indicated the removal of the organic fractions that contributed to the color (humic substances). Chian and DeWalle (1976) mentioned that the minimal reduction in COD (20%) could be attributed to lime precipitation, as the molecular weight greater than 50,000 Da contributing to some amount of COD fraction was removed. However, an increase in lime dosage did not prompt a concomitant increase in COD precipitation. Phosphorus could be removed by calcium hydroxide precipitation.

4.5.5 Summary of chemical treatment processes

Physical/chemical or biological processes can remove ammonia nitrogen. However, physical/chemical processes such as stripping, precipitation and ion exchange have several disadvantages such as odour, high chemical costs and excess sludge production.

Ho *et al.* (1974) performed studies on various chemical treatment options for landfill leachate. In their study, chemical precipitation, coagulation, oxidation and adsorption on activated carbon were evaluated. The study revealed that precipitation by lime and sodium sulfite and coagulation by alum and ferric chloride were effective in color removal though organic content could only be marginally reduced. This suggests that precipitation and coagulation should be combined with a biological process for an effective treatment of leachate. Nevertheless, the disadvantages of coagulation and precipitation are the high doses of chemicals required and the large amounts of sludge produced.

Chemical oxidation was investigated using ozone, calcium hypochlorite, chlorine and potassium permanganate. Although all oxidizing agents produced less sludge and were effective, calcium hypochlorite was the most promising. The disadvantage of using calcium hypochlorite was increased chloride concentrations in the treated effluent.

4.6 Natural Leachate Treatment Systems

Natural leachate treatment is distinguished from conventional systems based on the source of energy that predominates. In conventional systems, forced aeration, mechanical mixing and chemical addition are the inputs for the pollutant degradation. However, natural systems utilize renewable energy sources such as solar radiation or wind. These systems are land intensive whilst conventional systems are energy intensive. Typical natural systems for leachate treatment include leachate recirculation, irrigation, wetlands and aquatic systems.

4.6.1 Leachate re-circulation (landfill bioreactors)

Landfill leachate normally contains elevated concentrations of dissolved methane gas, which can trigger explosive atmospheres if discharged into sewer lines a distance away from the landfill.

Moisture addition by means of rain infiltration and leachate recirculation is critical to the stabilization of landfill waste, enhancement of gas production, improvement of leachate quality, reduction of long-term environmental consequences and liability of waste storage and improving economic viability of waste storage. The landfill effectively acts as an uncontrolled anaerobic filter and promotes methanogenic conditions for the enhancement of organic degradation (Knox, 1985; Strachan *et al*., 2000).

The *in situ* treatment of leachate by recycling it to the landfill reduces the time required for biological stabilization of the readily biodegradable leachate constituents and increases the rate of bio-stabilization of the leachate. Re-circulated leachate reduces the stabilization time from 15 - 20 years to 2 - 3 years (Pohland and Harper, 1985). It can be suggested that by managing the moisture content within the landfill, the rate and characteristics of the leachate generated can be controlled to dilute inhibitory and refractory compounds. Further, seed nutrients and buffers can be added to supplement the biological activity within the landfill and thus create an engineered bioreactor in the landfill. Though this is effective in removing the organic constituents in the leachate, the landfill bioreactor has been ineffective in treating elevated ammonia concentrations.

Pohland (1972, 1975), Leckie *et al.* (1975, 1979) and Pohland *et al.* (1990) performed leachate recirculation studies. The results indicated that rapid decline in COD could be achieved due to the rapid development of active anaerobic methane forming bacteria in the landfill enhanced by recirculation of leachate and seeding with municipal sewage sludge. The COD reduction showed a similar trend as reduction in BOD, TOC, VFA, phosphate, ammonia-nitrogen and TDS.

However, a full-scale study of leachate re-circulation conducted by Robinson and Maris (1985) concluded that the re-circulated leachate still contained high concentrations of ammonia, chloride and COD, thus requiring dilution of leachate prior to direct discharge.

Although leachate re-circulation is beneficial, it cannot be an ultimate solution as a point is reached when the landfill becomes saturated and excess leachate flow should be removed (Barber and Maris, 1992 as cited in Strachan *et al*., 2000). Therefore, leachate recirculation alone cannot be a treatment option for its surface water discharge. It can be suggested that a combination of leachate recirculation with aerobic biological treatment may be more effective since nitrification would take place in an aerobic process followed by denitrification within the landfill.

4.6.2 Reed beds

A reed bed system (root zone treatment) can be designed to treat leachate. The wastewater to be treated passes through the rhizomes of the common reed in a shallow contained bed of permeable medium. The rhizomes introduce oxygen into the bed and as the effluent percolates through it, microbial communities become established on the roots and degrade the contaminants. Nutrients such as nitrogen and phosphorus may also be removed directly as the reeds utilize them for their metabolism. Reed beds cannot be used as a primary treatment for leachate since they are inadequate at removing ammonia even from a sewage having a low concentration of 30 mg/L. Furthermore, the accumulation of heavy metals within the bed may affect rhizome growth and bed permeability. Therefore, reed beds are generally used as a polishing step for leachate treatment (Robinson *et al*., 1992).

4.7 Co-treatment with Municipal Wastewater

The combined treatment of small volumes of leachate with large volumes of municipal wastewater was believed to be more effective as demonstrated by Pohland and Harper (1985). An advantage of co-treatment of leachate with domestic sewage is that leachate contains excess nitrogen while sewage contains excess phosphorus, which eliminates the need for addition of nutrients. However, the main disadvantage is the high concentrations of organic and inorganic components contributed by both young and old leachate as well as the dilution of the refractory compounds.

The variability of leachate composition and heavy metal content may result in a shock loading and possible toxicity to the microbial population would pose a problem for the discharge of sewage sludge as a fertilizer (Lagervist, 1986; Linde, 1995 as cited in Welander *et al*., 1998).

Boyle and Ham (1974) and Lema *et al*. (1988) found that high strength leachate with COD > 10,000 mg/L can be treated at a level of 5% (v/v) without seriously affecting the effluent quality and treatment process. Beyond 5% (v/v), leachate addition resulted in excessive solids production, increased oxygen uptake and poor settling of biomass.

In a review of co-treatment of leachate in municipal wastewater (MWW) treatment plants, when researches conducted by various researchers (Chian and DeWalle, 1977; Raina and Mavinic, 1985) were summarized, a doubt arose at the feasibility of the process.

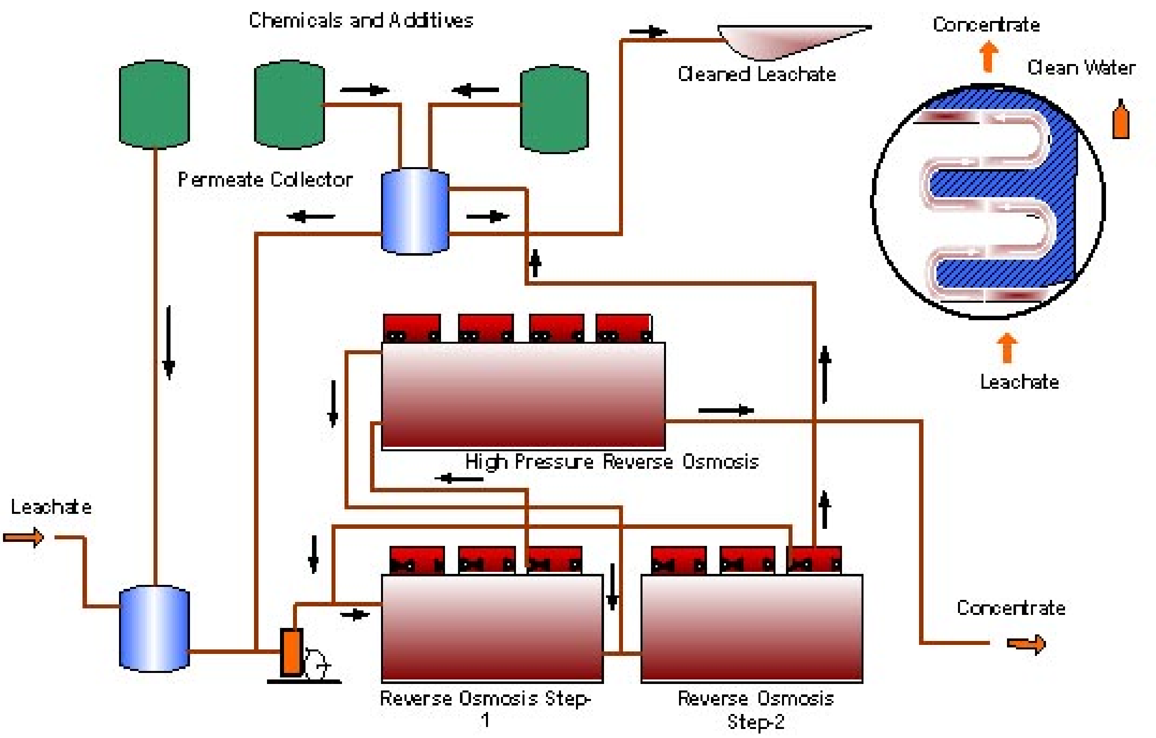
It was also found that this practice is not a long-term feasible solution for leachate treatment as its characteristics are variable while that of municipal wastewater remains consistent. The mixture of MWW and leachate will have a variable characteristic, which can affect the performance of the MWW treatment plants, thus resulting in wastewater treatment plants being ineffective, and may pose operational problems due to equipment damage, scaling and malfunctioning.

4.8 Combined Leachate Treatment Systems

Physical-chemical treatment processes for leachate from young landfills are not as effective as biological processes, whereas they are extremely efficient for stabilized leachate. COD/TOC and BOD/COD ratios, absolute COD concentration and age of the landfill are necessary determinants in the leachate characteristics for selection of adequate treatment options. In treating leachate, the sequences should be able to meet either the standards for discharge in receiving water bodies or an acceptable limit for discharge into water treatment works. Various combination of treatment systems have been applied for leachate. Figure 4.3 shows a typical leachate treatment train used in Europe. Few of the common combined treatment processes are explained below:

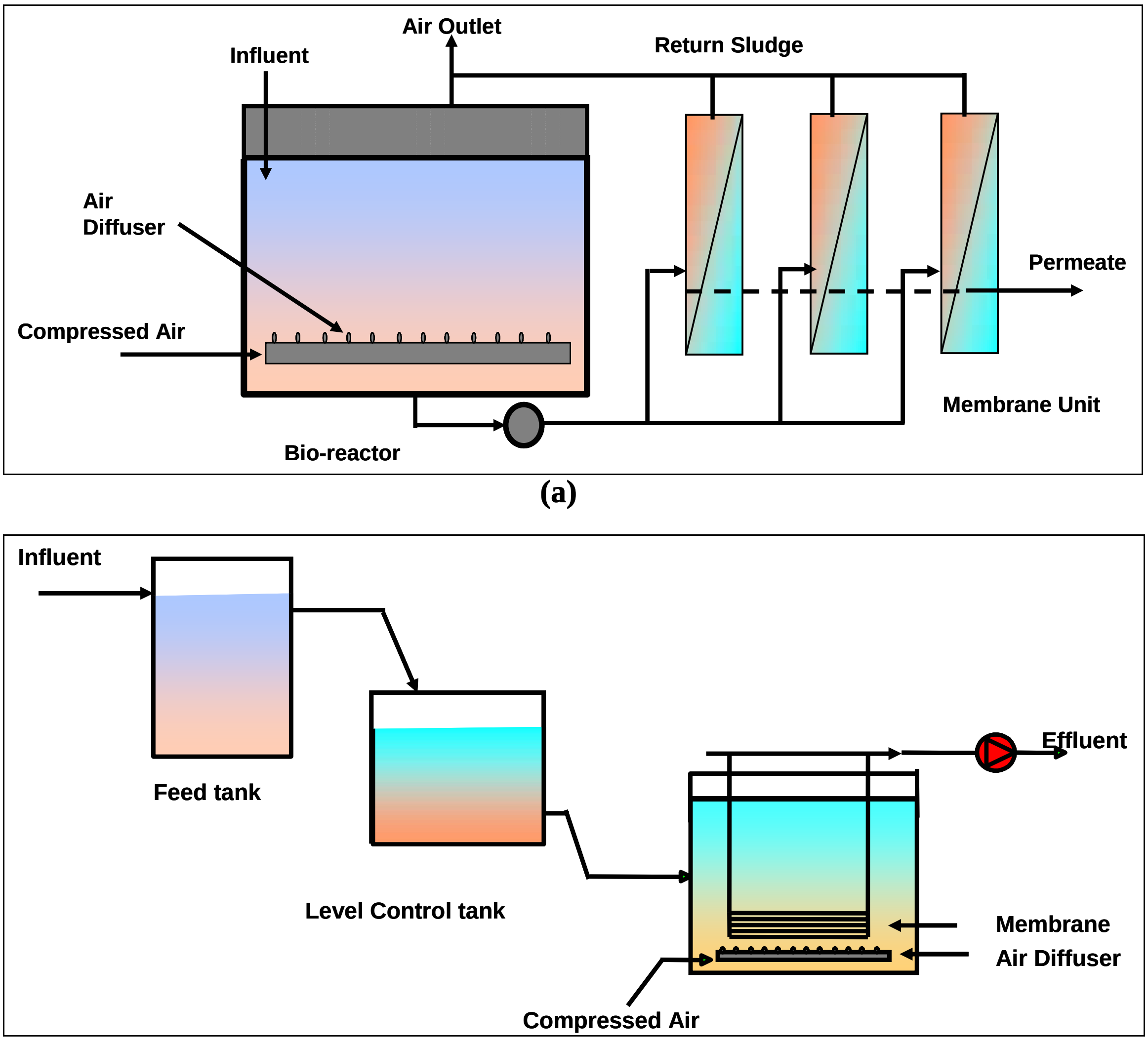
4.8.1 Membrane bioreactors (MBR)

The combination of membranes to biological processes for treatment has led to the emergence of membrane bioreactors for the separation and retention of solids, for bubble-less aeration within the bioreactor; and for extraction of priority organic pollutants from industrial contaminated water (Stephenson *et al*., 2000).



Source: Gierlich and Kollbach, 1998

Figure 4.3 Typical biological treatment scheme for leachate treatment in European countries



(b)

Figure 4.4 Schematic diagrams of (a) External recirculation MBR, and (b) Submerged MBR systems

The membrane unit can be configured external to or immersed in the bioreactor (Figure 4.4). Bressi and Favari (1997) conducted studies on MBR system consisting of activated sludge process coupled with external hollow fiber ceramic mircofiltration unit. When the performance of the MBR was evaluated, removal efficiencies were found to be 78 - 94% for young leachate with COD > 10,000 mg/L, 60 - 65% for intermediate leachate with COD ranging from 5,000 – 7,000 mg/L and 23 - 46% in case of stabilized leachate with COD < 2,500 mg/L. However, MBR alone was not able to treat the pollutants to meet the effluent discharge limit as it was unable to reduce chlorides, sulphates, ammonia-nitrogen and refractory organic compounds.

Reverse osmosis was used for large-scale leachate treatment in Germany. The RO system has a capacity of 36 m^3/h and has been in operation since 1989 with a change of membrane in 1997 (Peters, 1997). Operational pressures ranged from 36 - 60 bars

depending on feed water characteristics with a permeate flux of 15 L/m^{2}h. The COD of the leachate was around 1,797 mg/L with ammonium concentration of 366 mg/L. The effluent had a COD of less than 15 mg/L with elimination more than 99%. Peters (1997) also reported that a combination of reverse osmosis and nanofiltration was more effective treatment than reverse osmosis alone.

When a review of MBR systems for leachate treatment was done in different parts of Europe, it was found that MBR systems were effective in pollutant reduction. Table 4.12 summaries the performance of MBR systems in different parts of Europe.
The primary disadvantages of membrane bioreactors include capital costs for the membranes and operating costs associated with routine membrane cleaning. Another major disadvantage of RO and the membrane processes is membrane fouling, more specifically biofouling. Clogged pores result in reduced trans-membrane fluxes, increased operating pressures and deterioration of the membrane. If the initiation of biofouling is eliminated, the costs associated with cleaning the membranes could be dramatically reduced.

Table 4.12 Performance of MBR systems in leachate treatment (Wehrle Environmental Company case studies)

Place/Year	Volume (m^3)	Membrane surface area (m^2)	Flow rate (m^3/d)	COD		Ammonia	
				Input (mg/L)	Output (mg/L)	Input (mg/L)	Output (mg/L)
Germany, 1994	90	UF - 33 NF - 150	60	300	120	1200	
Germany, 1994*	252	NF -120	230	5000	< 400	1600	< 50
Spain, 2001	130	30	50	3600	< 160	1000	< 60
Germany, 1999*	98	84	65-120	1500	400	900	100
Spain, 1998	420	NF - 300 UF - 100	190	5000	1500	2000	60
Germany, 2000*	648	372	720	2000	200	600	5
Germany, 1999	680	240	290	5000	1200	2200	< 10
Germany, 1992	63			1500	200	500	50
Spain, 1996	600	288	550	4000	1500	2000	50
Germany, 1996*	114	UF - 60 NF - 100	70	3000	< 200	1250	< 150
France, 2001*	235	24	14.4	1950	< 300	800	< 150
Germany, 2000*			65	1100	< 400		

*With activated carbon

Membrane bioreactors can further be used to pre-treat leachate more successfully than SBR processes, prior to its disposal into the sewers. However, membrane systems are susceptible to shock loading of ammonia that may affect the biomass. However due to the presence of membrane, a complete retention of solids is still possible to maintain.

4.8.2 Ultrafiltration and biologically activated carbon (UF-BAC)

Pirbazi *et al*., (1996) used ultrafiltration coupled membrane technology with powdered activated carbon to treat landfill leachate collected from two different sources. This process scheme includes adsorption, biodegradation and filtration to develop a highly efficient process for leachate treatment. Further experiments were also conducted on

pre-treatment of leachate by coagulation/flocculation and oxidation to develop an effective treatment scheme.

In the process, biodegradation of leachate occurs in the activated carbon and an external ultrafiltration unit separates the effluent. While operating the system, the flux deterioration of the UF module was controlled by maintaining a turbulent flow regime within the membrane module and by applying powdered activated carbon (PAC). Three main functions of PAC were to depolarize the membrane surface, to act as a filter aid by forming an incompressible particulate layer of high fluid permeability and to reduce the thickness of the biofilm layer on the membrane. The pre-treatment included coagulation/flocculation by ferric chloride, lime and alum for the leachate obtained from the first source and oxidation of ferrous iron in the leachate obtained from the second source. The treatment scheme was able to attain a removal efficiency of greater than 90% and pre-treatment reduced suspended solids, which in turn reduced membrane fouling. The addition of PAC was found to enhance the membrane flux.

4.8.3 Denitrification-nitrification and UF

Luning and Notenboom (1997) evaluated the application of membrane bioreactors on a full-scale leachate treatment plant. Initially an RO unit was used in the leachate treatment and a combination of treated wastewater and leachate led to decrease in the RO performance. Therefore, an aerobic MBR was adopted. The treatment scheme consisted of separate nitrification-denitrification reactors followed by an external UF membrane.

The operational conditions of the system are summarized in table 4.13. The study indicated that the treatment of weak leachate by conventional aerobic methods was inadequate due to a loss of biomass along with activated sludge. The application of MBR would be advantageous in such cases. Operation at higher pressure aided in the degradation of long-chain components in the treatment process.

Keenan *et al*. (1984) studied a full-scale leachate treatment process, which included chemical/physical treatment by chlorination and biological treatment. The chemical/physical pre-treatment incorporated equalization, precipitation, sedimentation of precipitate, air stripping at elevated pH for ammonia removal, neutralization and nutrient supplementation. Chemical precipitation by lime was necessary for the removal of heavy metals and a portion of the organic matter. Air stripping of ammonia was done to remove excess ammonia. Sulphuric and phosphoric acids were added to reduce the pH of the leachate prior to biological treatment and supplementary phosphorus was provided as the phosphorus present could have been depleted during lime precipitation.

4.8.4 Coagulation-activated sludge and chlorination-ammonia stripping

Biological treatment was conducted in two aeration tanks and clarifiers, which could be operated in series or parallel. The effluent was chlorinated prior to discharge into a river. The study revealed that phosphorus deficiency and excess nitrogen reduced the efficiency of activated sludge process. Hence, ammonia stripping and phosphorus supplements were necessary to maintain a high performance of the treatment plant.

Table 4.13 Operational conditions and performance of leachate treatment plant			
Operation	**Parameters**	**Unit**	**Value**
Process conditions	Temperature	°C	30
	Pressure	Bar (g)	3
	Additional C-source		Methanol
	pH		6.8
Operational conditions	Membrane flux	L/m^2.h	> 110
	Energy consumption (membrane unit)	kWh/m^3	8
	Membrane life expectancy	years	5 - 7
	Cleaning interval	years	4
Results	N removal	%	> 98
	NH_4-N removal	%	> 99
	NO_3-N removal	%	> 80
	COD removal	%	> 93
	BOD removal	%	> 99

Source: Luning and Notenboom, 1997

4.8.5 Microfiltration and reverse osmosis

A two-stage precipitation/microfiltration and reverse osmosis process for the treatment of landfill leachate was investigated (Krug and McDougall, 1988). In this process scheme, use of caustic soda, lime, lime with PAC and soda ash for chemical precipitation was also evaluated. The samples were filtered by MF prior to RO treatment. The study revealed that the choice of chemical for precipitation played a major role on the membrane performance. A flux reduction could be noticed with the use of caustic ash since the solids formed have a greater tendency of adhering to and fouling the membrane. Lime addition and MF were found to be effective in removing suspended solids, metals and hardness from raw leachate and subsequently, salts could be removed by RO.

4.9 Development of Landfill Leachate Treatment Sequence

A single treatment technology is not efficient for leachate due to the complexity involved in treating it having a varied composition and characteristics. Leachate treatment entails the integration of several processes. The coupling of units for the development of treatment sequences should be modular to allow maximum flexibility in order to vary the order of arrangement and the addition/removal of unit operations. This effectively creates different treatment lines and thus can be better adapted to the changing qualitative conditions of the leachate (Qasim and Chiang, 1994; Bressi and Favali, 1997).

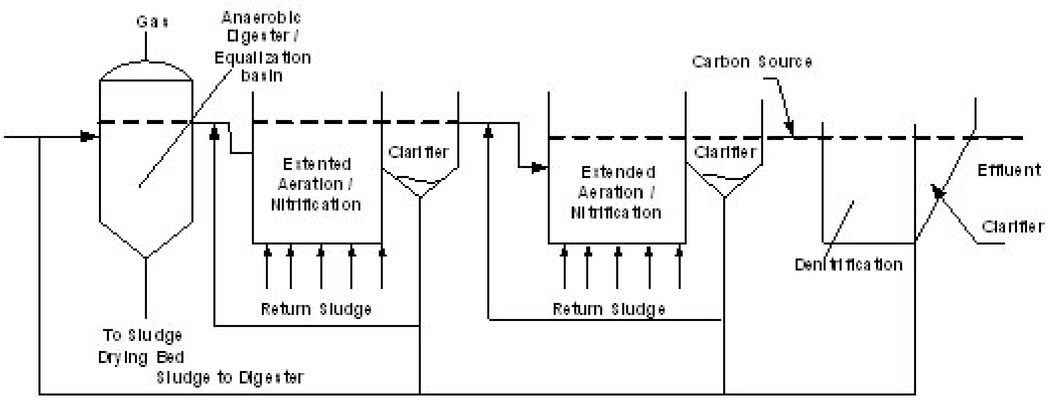
The discharge of leachate directly into receiving water sources enforces stringent measures to meet mandatory discharge limits and conditions. In selecting a treatment sequence, it is necessary to review and consider the development of sequences in the past. Different treatment sequences which can be used for leachate are described below:

4.9.1 Two-stage suspended growth reactors

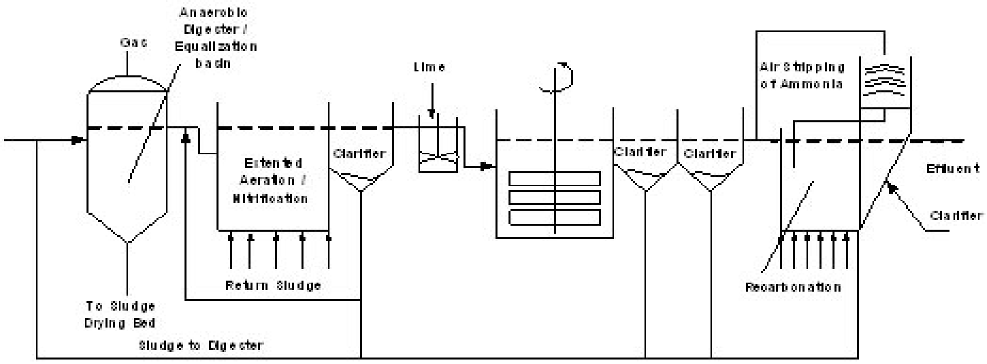
A single-stage activated sludge process is incapable of removing residual refractory organics, which remain after stabilization of biodegradable organics. The existence of a

second stage process can maintain organisms acclimatized to refractory organic compounds and for further degradation.

A treatment sequence comprising of an anaerobic reactor followed by aerobic process where biodegradable refractory organics and nitrogen compounds are simultaneously removed could be effective in treating a young and middle aged leachate. PAC could also be added to enhance organic removals. Finally, a denitrification reactor can be introduced with an addition of external carbon source. This treatment sequence is represented in figure 4.5.



(a)



Source: Qasim and Chiang, 1994

(b)

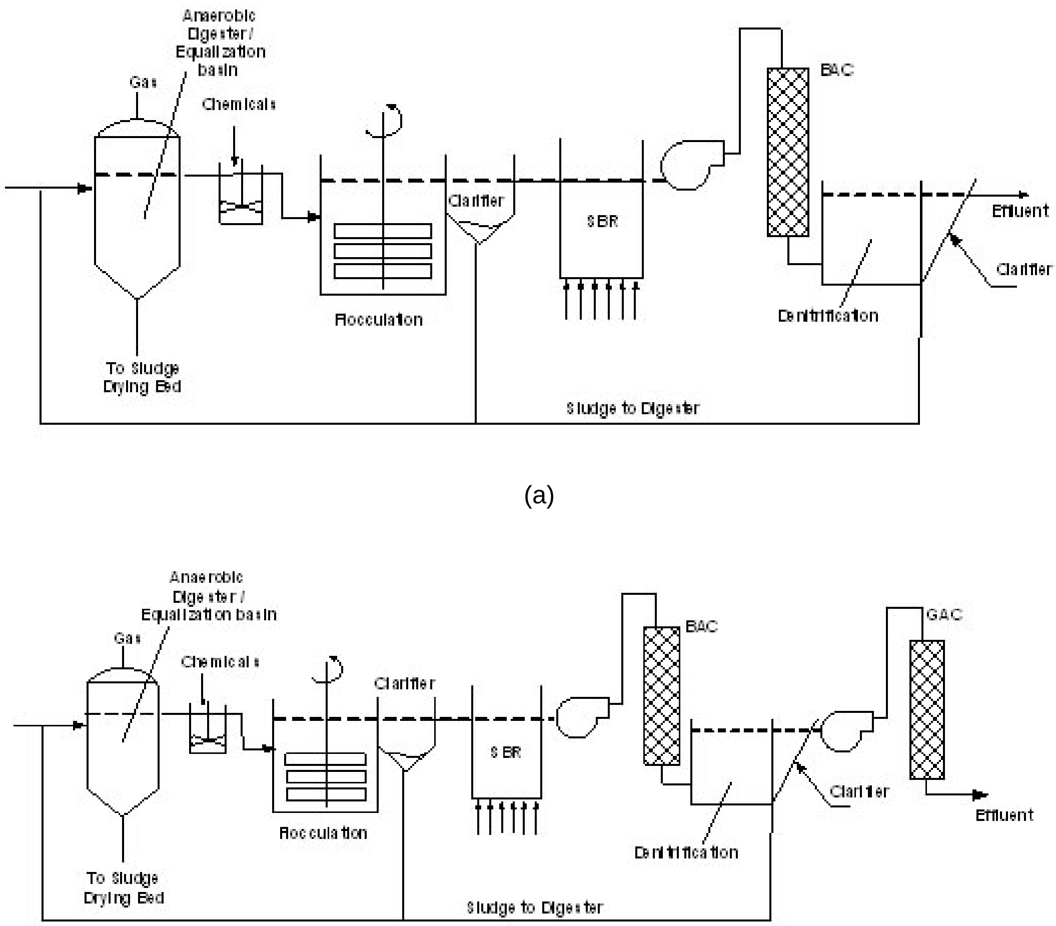
Figure 4.5 Two-stage suspended growth reactors (a) young to medium age landfill (b) old landfill

As the landfill age increases, the leachate characteristics changes and treatment sequence could be subsequently changed. The second aerobic reactor could be converted into an ammonia-stripping column to compensate the increase in ammonia and incomplete nitrification in the first aerobic zone. The denitrification zone can be used for re-carbonation and pH adjustment of the effluent from the ammonia stripping phase.

The process could be effective in removing biodegradable and refractory organics, heavy metals, nitrogen and phosphorus. However, this scheme would not be efficient in removal of dissolved solids.

4.9.2 Sequencing batch reactor (SBR) - biological activated carbon (BAC) filter and denitrification process

Treatment process with pre-treatment using an anaerobic reactor followed by coagulation-flocculation and subsequently by SBR-BAC filter and denitrification process (Figure 4.6) could be successful in treating young and middle aged leachate. Equalization would take place in the anaerobic filter and heavy metal precipitation in the coagulation-flocculation stage. The SBR would enrich the microbial population with desired metabolic capabilities, stability and settling characteristics in which both the biodegradable and refractory organics could be removed.



Source: Qasim and Chiang, 1994

(b)

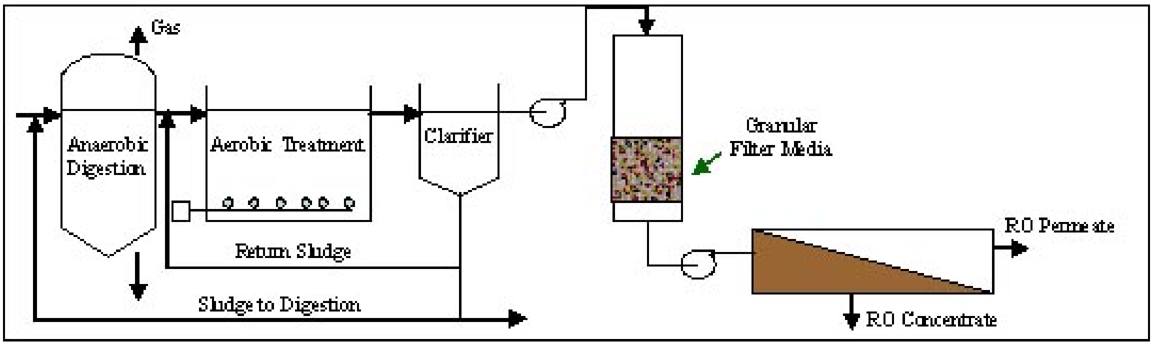
Figure 4.6 SBR-BAC and denitrification process for (a) young to medium age leachate (b) old leachate

The biological activated carbon would help in creating a biofilm to metabolize refractory organics and initiate nitrification of the residual ammonia. The nitrate nitrogen would be

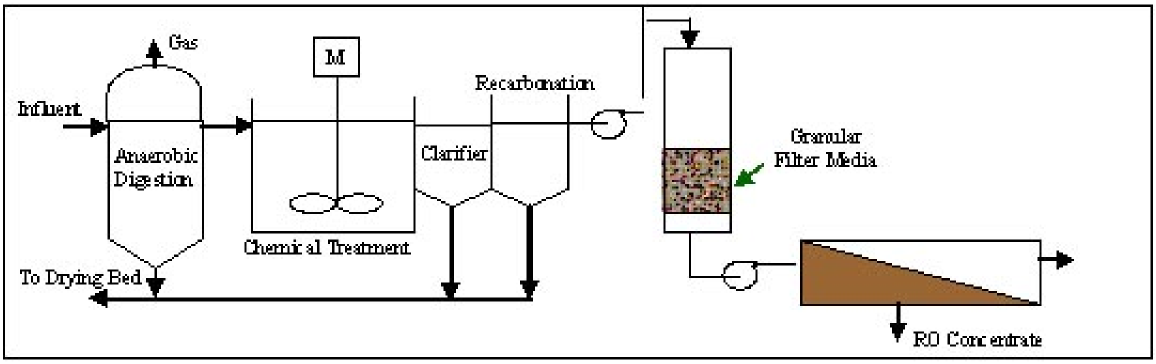
removed in the denitrification tank. With an increase in the landfill age, the facility could be upgraded by the addition of granular activated carbon columns.

4.9.3 Biological treatment and reverse osmosis

A treatment sequence (Figure 4.7) that is capable of removing mineralized material should include anaerobic digestion, suspended growth biological waste treatment, partial softening, filtration and reverse osmosis. The anaerobic digester stabilizes the waste while the aeration system degrades the biological matter. The effluent is polished in a gravity filter and dematerialized in RO unit, thus achieving an effluent devoid of dissolved salts and low in organics. With increase in age, the biological treatment could be replaced by coagulation- precipitation process followed by re-carbonation, filtration and RO.



(a)



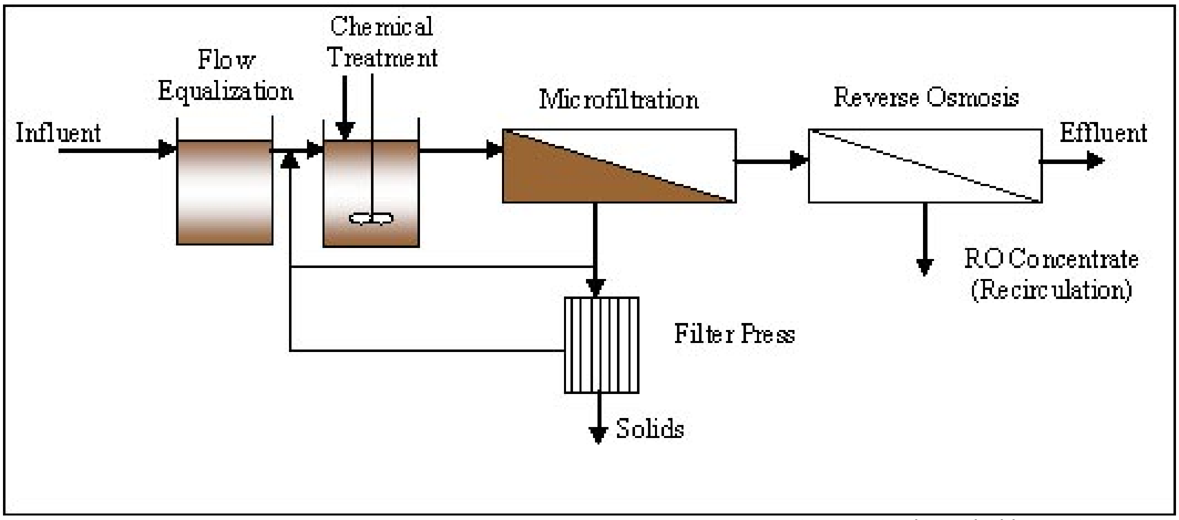
(b) *Source: Qasim and Chiang, 1994*

Figure 4.7 Leachate treatment with biological treatment and reverse osmosis (a) young to medium age leachate (b) old leachate

4.9.4 Microfiltration/reverse osmosis

Incorporation of a multiple membrane system by the combination of microfiltration and reverse osmosis could also lead to a well-developed treatment sequence (Figure 4.8). The process is suitable for leachates of all ages. The two-stage process incorporates precipitation and MF for the removal of toxic metals and suspended solids, and RO for the concentration of residual organics. The first step of precipitation and microfiltration provides a simple pre-treatment for the RO unit and thus produces a high quality effluent free of solids and dissolved organics. However, similar to other membrane

processes, the system is susceptible to fouling. Hence, development of antifouling strategies and reduction in biofouling needs to be evaluated.



Source: Qasim and Chiang, 1994

Figure 4.8 MF and RO process for leachate treatment

4.9.5 MBR-UV/ozone-reverse osmosis

Bressi and Favali (1997) evaluated various treatment schemes to develop a modular treatment system with flexibility, which is required to treat varied composition and characteristics of landfill leachate over the entire lifetime of a landfill. The basic technologies selected in the study were:

- Membrane bioreactor for biological treatment by aerobic oxidation and nitrification;
- UV/ozone for increasing biodegradability and for partial oxidation; and
- Reverse osmosis treatment for the elimination of dissolved solids and reducing of organic load.

By combining these technologies in various configurations, the removal efficiencies were evaluated and ultimately the best sequence was recommended for young, intermediate and stabilized leachate. The choice of ozone as a pre-treatment for stabilized leachate and as a post-treatment for young leachate, increases the biodegradability and aids in the partial removal of organic residuals after the MBR process respectively. UV and ozone also offer the advantage of breaking down and partially oxidizing low degradable molecules.

Although the treatment process performed reasonably well for leachate from different landfill ages, energy consumption and economics of the process need to be considered. From the review of the process schemes, Bressi and Favali (1997) suggested that the membrane bioreactors should be supported by UV/ozone for the oxidation of refractory compounds. Ideally, the UV/ozone process should be located after the MBR process. As a post or polishing treatment, they are also effective in degrading color contributing humic compounds. However, it was found that the UV/ozone process was more efficient when placed after MBR process in young leachate and before MBR process in the old leachate.

From the various treatment schemes evaluated, MBR followed by RO proved to be the most promising treatment sequence. Removal efficiencies of 96 - 99% could be obtained and the effluent could be directly discharged into the environment. The RO process, unlike the MBR, is purely a concentration process, which effectively reduces the pollutants discharged. From a landfill management point of view, this proves to be advantageous. This process, unlike the MF-RO proposed by Qasim and Chiang (1994), incorporates the activated sludge and microfiltration unit operations in a single-step MBR process thus eliminating the clarifier present in a conventional process. Furthermore, the biological pre-treatment of the leachate ensured a better quality permeate from the RO unit and prolonged the life span of the RO unit by reducing fouling effects and treatment costs. By incorporating RO as a post-treatment to MBR process, UV/ozone is not required.

4.9.6 Ammonia stripping, MBR and ozonation

The high ammonium concentration present in the leachate poses toxicity to the biological processes thus affecting the degradation process. Due to the toxicity, removal of ammonia is necessary for leachate treatment. As the treatment efficiency by nitrification-denitrification is considered poor with BOD/TKN <2.5, BOD/NH_3 <4 and COD/TKN <5 (Tchobanoglous and Burton, 1991), ammonia stripping could be an effective option. Ammonia removal by air stripping can reduce ammonia concentration from 2,000 - 200 mg/L. Ammonia stripping as pre-treatment also has an advantage of reducing refractory compounds, thereby reducing the COD by precipitation when the pH is adjusted.

If pre-treatment of ammonia stripping fails, this would lead to shock loading caused by ammonia in the biological system, making it difficult for the floc to settle down. This problem can be solved by the adoption of membrane system to replace the clarifier in the activated sludge process since membranes can retain total solids until the sludge recovers from the shock loading of ammonia. In this regard, membrane filtration has proven to be a justifiable solution in terms of economic feasibility and affordability in most cases, even when the overall costs for the purification are compared with other approaches for leachate treatment (Peters, 1997). By coupling a membrane with the activated sludge reactor, an MBR emerges as a logical treatment option.

In the MBR systems, maintaining a lower MLSS is advantageous since the lower the sludge produced, the greater is the effectiveness of aeration. This approach can effectively further reduce the aeration requirements and a smaller SRT and HRT can reduce the reactor volume, thus decreasing the capital cost.

It is commonly known that young leachate has a high biodegradability and can most effectively be treated by biological processes while old leachate is less susceptible to biological treatment. Therefore, the focus on increasing the biodegradability of older leachate should also be emphasized. A possible approach to increase leachate biodegradability is to ozonate either the mixed liquor or effluent streams from the membrane bioreactor. The objective is to maintain a constant MLSS as well as breakdown of high molecular weight refractory compounds into simpler molecules, which increases the nutrients within the reactor. Thus, this increases the overall

biodegradability within the bioreactor and improves the effluent quality to create a zero-discharge system. It also eliminates the capital and operational costs associated with the treatment of sludge in its disposal facility prior to discharge. Therefore, a sequence with ammonia stripping as pre-treatment followed by MBR and finally with ozonation as post-treatment could be effective in leachate treatment.

Chapter 5

CASE STUDIES OF LEACHATE TREATMENT PLANTS

To select an effective leachate treatment train, it is necessary to understand the sequence of treatment applied in various landfills in different parts of the world. High variability in leachate volume and strength with time makes its treatment challenging. To arrive at an appropriate option, a review of the experiences obtained from different case studies would be valuable. A gist of various case studies is given below.

5.1 Hempsted Landfill, Gloucester, UK

Hempsted landfill was in operation for 30 years and is located close to the River Severn which has a large salmon and eel population. Shallow waste deposits contained significant depths of leachate close to ground level. Leachate management was done in two stages. Weak leachate was pretreated by the addition of hydrogen peroxide to remove soluble sulphides, followed by vigorous aeration, mixing and stripping to remove methane prior to discharge to the sewer system for treatment by the municipality. This option has a capacity of 605 m^3/d which reduced the volume of leachate from the landfill.

The second stage was designed to treat high-strength leachate with contaminants primarily of ammonia-nitrogen, organic compounds and metals. This was accomplished in a sequence batch reactor with a capacity of 280 m^3/d which performed satisfactorily even at a low temperature of 10°C. Table 5.1 presents the detailed treatment performance of the twin SBR plant and compares the results for effluent quality with the consented limits for discharge into the river.

Table 5.1 Performance data and consent limits for SBR plant at Hempsted landfill

Parameters (mg/L except pH)	**Untreated leachate**	**Effluent SBR-1**	**Effluent SBR-2**	**Compliance sample**	**Consent limits**
Suspended solids	-	-	-	21	100
pH	6.9	7.8	7.0	7.7	-
COD	229	194	229	-	-
BOD_5	50	18	19	7.5	50
NH_4^+-N	168	0.8	<0.3	1.08	25
Nitrate-N	<0.3	232	254	247	-
Nitrite-N	<0.1	<0.1	<0.1	-	-
Chloride	356	571	612	575	-
Iron	17.8	3.8	5.3	-	-
Zinc	0.10	0.14	0.20	0.0967	0.250

Source: Robinson, 1999

The plant was able to achieve a high COD, BOD and NH_4^+-N removal efficiency. The metal concentration in the effluent was within the standards, though nitrate concentration of 247 mg/L was obtained. It is evident that absence of a denitrification

reactor resulted in a high nitrate concentration. Thus, it could be said that an additional denitrification reactor would be required for the effluent quality to comply with the standards.

5.2 Trecatti Landfill, Merthyr Tydfil, South Wales, UK

Trecatti landfill is a ten million cubic meter lined landfill located in an open coal mine. The treatment plant comprises of a pair of identical, modified sequencing batch reactor (SBR) units. The concentric concrete tanks have roof covers for heat retention. Originally, during 1995 a single SBR was commissioned and fitted with mechanical and electrical equipments and controls. The leachate treatment plant was designed to withstand adverse weather conditions - temperatures below freezing point.

The unit treated up to 6,000 m^3 of leachate each month before the second treatment tank was commissioned in September 1998. The plant had treated a total of more than 90,000 m^3 of strong methanogenic leachate over a period of 30 months consistently to high standards. Table 5.2 represents leachate and the effluent quality. Complete nitrification and removal of NH_4^+-N could be maintained despite frequent adverse weather - snowdrifts up to several feet deep over the surface of the treatment plant.

Two covered sequential batch reactors had a treatment volume ranging from 100 - 180 m^3/d. The treated effluent was discharged into the sewers with the following final effluent qualities, COD 29 mg/L, BOD 3 mg/L and NH_4^+-N 0.5 g/L. The nitrate level was as high as 616 mg/L due to the absence of a denitrification process.

Table 5.2 Performance of Trecatti landfill

Parameter (mg/L except pH)	Leachate	Effluent
pH	7.1	7.6
COD	~1000	299
BOD_5	210	3
TOC	299	115
NH_4^+-N	541	0.5
Nitrate-N	<0.1	616
Nitrite-N	0.6	<0.1
Chloride	1070	991
Iron	10.4	<0.6
Zinc	0.53	0.12

Source: Robinson, 1999

Operation of the Trecatti plant is fully automated - controlled by a computer system that maintains detailed records of all plant operating features (dissolved oxygen, pH, temperature, volume treated, etc). Fail-safe systems have been incorporated throughout, and pH-values in the treatment reactors are controlled automatically to maintain within a preset range.

5.3 Buckden South Landfill, Eastern England, UK

The Buckden South Landfill has been closed since mid-1994. As the leachate could pose adverse effects on the fish population when discharged into a river nearby, extensive treatment was required. The leachate also contained contaminants typical of methanogenic conditions including NH_4^+-N at values in excess of 400 mg/L with a required effluent discharge standard of 10 mg/L. The treatment plant design had to ensure that these effluent quality standards including fish toxicity criteria could be met at all times. The plant was commissioned during late 1994 and has been operating reliably and consistently for more than eight years.

The treatment system was developed to treat methanogenic leachate with a capability of removing pesticides present in it. As high quality effluent discharge was required, biological treatment could not alone suffice. Hence, a treatment sequence comprising of twin sequencing batch reactors with engineered reed bed was designed. Each of the SBR had heat efficiency and was capable of treating a maximum volume of 200 m^3/d leachate. The engineered and planted reed bed with a capacity of 2,000 m^2 was used as a polishing treatment of the effluent. An ozonation plant with dosing up to 500 g/h (150 mg/L at maximum treatment rates) was used to break down the residual pesticides into smaller organic molecules and these were discharged into a 500 m^2 reed bed for complete degradation prior to discharge into the river (Robinson, 1999).

The second SBR was used as a storage tank, which could be converted into a denitrification tank in the later stages. The influent characteristics, effluent discharge and discharge standards are given in table 5.3

Table 5.3 Influent characteristics and effluent discharge requirements for Buckden leachate treatment plant

Parameter (mg/L)	Mean influent concentration	Final effluent	Effluent discharge requirement
COD	1093	307	500
BOD_5	284	15	20
Ammonia-N	285	0.3	10
Chloride	1455	1500	2000
Iron	19.6	1.8	2.5

Source: Robinson et al, 2002

The treatment process scheme was able to meet the effluent discharge limits though COD was not within the limits. Ozone was found to be ineffective in removing COD, despite of an increase in the BOD. The elevated COD was not detrimental to the receiving wastewater body and hence, the COD discharge range was acceptable. Pesticides could be effectively removed from the leachate.

5.4 Borde-Matin Landfill, France

The Borde-Martin landfill uses the natural contours of a valley for step-wise filling of the waste. The waste is pre-sorted and glass and paper are removed for recycling. The remaining waste is screened and landfilled. The leachate percolates through the valley

walls and reaches the treatment facility located on the other side of the valley. The treatment plant has been in operation since 1998 and was designed to mimic natural treatment phenomena at an accelerated rate. Leachate is channeled into a lagoon with treatment taking place in three different stages: nitrification-denitrification, coagulation with lime and iron salts, and ozonation. Figure 5.1 shows the nitrification-denitrification facility in the foreground, the physical-chemical treatment in the centre and the ozonation facility at the background.

Nitrification-denitrification was done twice by specific bacterial population to oxidize high ammonium concentrations into nitrogen gas. At the end of the first stage, ammonia levels could meet the discharge requirements although the organic fraction remained untreated. This was accomplished by coagulation with lime and salts, which reduced 50% of the COD. After coagulation, ozonation reduced the residual organic fraction as well as removed color prior to discharge into a river. The treatment scheme was able to achieve 92% carbon removal and 98% nitrogen removal.

The treatment plant also incorporates a biogas facility capable of transforming the generated biogas into energy. The expected energy output has been estimated at 50, 000,000 kWh/year.

Figure 5.1 Aerial view of the leachate treatment facility in Borde-Matin Landfill, France

5.5 Bryn Posteg Leachate Treatment Plant, UK

Bryn Posteg landfill site (Mid Wales, UK) is an engineered landfill site constructed on a former lead mine. The site receives about 150 tonnes of waste each day and the area has an average annual rainfall of 1200mm. High water quality of the Severn River and the nature of other rivers nearby make the treatment and local discharge impossible. The effluent from the treatment plant is discharged into a small sewage treatment plant for further treatment prior to discharge into the river.

The rainfall rate is an important factor affecting the leachate quality. The leachate treatment plant had been commissioned in 1983 consisting of a large HDPE-lined lagoon with a long mean retention period of a maximum capacity of treating 150 m^3/day. High-efficiency floating aerators provided the required aeration. Detailed operational monitoring was done from 1983 to 1986. It was found that the unit costs were inevitably greater when less leachate was treated but a number of modifications significantly improved the energy efficiency at low rates. The plant has performed for several years even in severe weather conditions. Table 5.4 gives the performance of leachate treatment plant during the winter season 30 months since the operation of the plant. A total of 26,275 m^3 with a mean rate of 29 m^3/d was treated during this period.

Table 5.4 Bryn Posteg landfill leachate treatment plant during winter

Parameter (mg/L except pH)	**Leachate**		**Overall removal (%)**
	Influent	**Effluent**	
pH	6.3	7.7	-
COD	9750	210	97.2
BOD_5	7000	37	99.5
NH_4^+-N	175	0.9	92.8
Organic N	27	11	59.3
SS	160	4.5	78.8
VSS	110	20	81.8
Na	904	808	10.6
Cl^-	1522	1300	14.6
Fe	295	9.9	98.7
Zn	11.5	0.5	95.5

Source: Robinson et al, 1992

The treatment plant could achieve excellent removal of organic content, and ammonia could be maintained even when the COD and ammonia levels in the leachate increased to 23,000 mg/L and 600 mg/L respectively. The results showed that around 97% of COD and 99% of BOD could be removed. The COD was found to be higher than that indicated through BOD due to the long retention time used in the process. Excellent removal of iron and zinc were also maintained.

5.6 Compton Bassett Leachate Treatment Plant, UK

The Compton Bassett landfill site has been in operation since 1985. Its leachate treatment plant was designed to treat leachate from an old, closed district council waste disposal site at Sands Farm and for a large adjacent containment site at Compton Bassett. Leachate is being treated in an aerated lagoon SBR plant at rates up to 100 m^3/d, primarily to nitrify high concentrations of NH_4^+-N. Effluent is being discharged via a small village sewage works into a tiny stream.

This landfill had a complicated leachate management as the discharge to be treated was generated from several sources including an old landfill containing well-stabilized leachate (BOD=1000 mg/L, NH_4^+-N=700 mg/L). As the treated effluent had to be discharged into a small stream, strict standard of ammonia concentration of 7.5 mg/L required to be maintained. Because of these constraints, the process was later modified. Due to the scarcity in organic matter for the removal of NH_4^+-N, a liquid waste stream

from a local jam factory containing a high COD of 330,000 mg/L was directed into the lagoon to maintain optimum ratios of BOD:N ratio. The plant maintained a high effluent quality with 10 mg/L BOD, 268 mg/L COD, 3 mg/L NH_4^+-N, 8 mg/L total nitrogen and 38 mg/L SS (Robinson *et al.*, 1992).

5.7 Pirai Sanitary Landfill Leachate Treatment, Brazil

The Pirai sanitary landfill is located in Rio de Janeiro State, Brazil. It covers an area of 3,500 m^2. After a year of operation, a leachate treatment plant consisting of biological filter followed by a reed bed was established for discharges into the lagoon. The lagoon had a sufficient capacity to accumulate 500 m^3 of leachate. When the overall evaluation of treatment plant was done, it was found that the performance of the treatment plant was efficient. Table 5.5 summarizes the efficiency of the treatment plant.

Table 5.5 Performance of the treatment plant

Parameter (mg/L)	Leachate	
	Influent	**Effluent**
Total alkalinity	1,746	100
NH_4^+-N	64.16	0.60
TOC	58.3	-
BOD_5	244	4
COD	554	79
TKN	403	2.95
TDS	4,460	140

Source: Ferreira, 2001

The leachate quality was similar to the young leachate due to dilution effect caused by rainfall. The creek nearby was also evaluated to find out any significant environmental impact of the landfill.

Chapter 6

FUTURE RESEARCH DIRECTIONS IN LEACHATETREATMENT

Leachate treatment is an integral part of the sustainable solid waste management practices. However, the design, construction and operation of leachate treatment facilities have not been standardized due to uncertainties in leachate production and characteristics. This uncertainty is due to the factors that effect the generation of leachate and its composition from a landfill. There are several techniques used for landfill leachate treatment, which have been discussed in the previous two chapters. The future directions in the treatment of the leachate are based on the findings and techniques described in the following sections. To design an effective treatment system, it is necessary to control the generation of leachate in the first place before opting for a treatment technology. Based on the controlled leachate generation, treatment scheme can be furnished.

6.1 Pre-sorting and Pre-treatment of Waste

The solid waste disposed into the landfill has a great influence on the leachate generation and its characteristics. Through pre-sorting and pre-treatment of the waste dumped into the landfill (either by pre-selection or by rigorous sorting of the waste constituents), it is possible to manage the rate of generation and transfer of the waste constituents into the liquid medium. This option would be beneficial for leachate treatment.

Pre-sorting not only helps to recover recyclable or reusable waste and separation of hazardous waste, but is also useful in reducing the pollutant load into the landfill, thus making the leachate treatment less complicated. Pre-treatment would facilitate resource recovery in the form of recoverable materials, energy and compost and the reduced volume to exhibit consistent leachate characteristics that would make the process of designing a leachate treatment sequence easier as well as cost-effective. Enhanced leaching of the waste constituents by flushing could also be an innovative pre-treatment technique for effective leachate treatment. It can therefore be suggested that pre-sorting and pre-treatment of solid waste could aid in leachate treatment.

6.2 Natural Treatment Methods in Combination with Conventional Treatment Processes

Leachate re-circulation and reed beds have been established on full-scale leachate treatment plants and are found to be effective when coupled with conventional biological processes. Since the landfill effectively acts as an anaerobic digester, organic loading can be reduced while ammonia removal can be accomplished in a conventional nitrification reactor.

Lin *et al* (2003) used vetiver grass (*Vetiveria zizanioides*) with different substrates (coal refuse, fly ash, cinder, soil and gravel) to treat leachate and found that the removal of COD, NH_4^+-N, NO_3^--N and TKN removal was from 33-73, 46-74, 72-94 and 46-73%, respectively. They found that coal was a good substrate for treating landfill leachate with high NH_4^+-N, TP and TSP removal, and denitrification increased with the addition of saw dust. It was found that without the grass, the performance of the wetland was

poorer. Heavey (2003) studied the use of peat as a treatment medium for landfill leachate in Ireland and concluded that it could be treated successfully to achieve almost 100% removal of BOD and ammonia. This method could be affordable for developing countries where finance is a major constraint and in closed landfills, which do not need constant monitoring.

Leachate irrigation is another treatment technique which would help to increase moisture content and induce enhanced degradation of the waste during the prolonged dry season in the tropical countries. It would reduce the pollution load, as further natural degradation would take place within the landfill. It will also help in recirculation of nutrients into the landfill for microbial growth to maintain a balance in their population within the landfill. Such simple treatment would reduce the need for variation in young and old leachate treatment plants and management practices (Kuruparan *et al*, 2003).

The combination of natural treatment methods offers the advantages of reducing the volume of leachate and subsequent treatment costs as well as enables on-site leachate treatment instead of leachate disposal into the sewers. It can be further suggested that leachate with high ammonia content should undergo nitrification process with a conventional treatment plant prior to re-circulation into the landfill. This alternative is in comparison with conventional two stage nitrification-denitrification process.

6.3 Analysis of Biodegradable and Other Leachate Fractions

Among the various treatment techniques, in comparison with the physico-chemical treatment process, biological treatment process is cost-effective. In order to verify the applicability of the biological treatment process, it is necessary to have a deep understanding about the biodegradability of the landfill leachate at different stages. According to Kylefors (1997), there is a lack of knowledge on COD contributing constituents present in the leachate and environmental impacts of the residual COD. It was suggested that acetic acid, which is the dominant part of organic fraction in acidogenic leachate, may also contribute to the COD.

Therefore, in combination with the conventional closed reflux COD analysis as in Standard Methods for Wastewater Treatment (APHA, 1995); volatile fatty acid analysis, BOD and COD fractionation by either gel chromatography or membrane filtration should be performed. The combined results would aid in establishing the biodegradable organic fraction as well as the COD fractions based on their molecular weights. This molecular weight distribution can be related to the fractions of COD which form a part of the easily degradable or non-degradable organic constituents. Further, this would also signify the effectiveness of the treatment process in terms of COD fractions.

Analysis of the various functional groups in the leachate will also aid in choosing the appropriate treatment process. The various compounds present in the landfill leachate may include alkaline compounds, phenolic compounds, acidic compounds such as polyaromatic compounds, chlorinated aromatic compounds, phenolic compounds, aromatic compounds, heavy metals, etc. The functional groups may include asters, aldehydes, ketones, alcohols, acid group, carboxylic groups, etc. Presence of heavy metals may also affect the performance of the biological leachate treatment process.

Thus, analysis and determination of various leachate components can help in better understanding of suitable leachate treatment process.

6.4 Leachate Toxicity

Though various treatment techniques and sequences have been investigated to a great extent, little importance has been given on the toxicity of the treated leachate. The effect of different treatment techniques on the toxicity of the leachate needs to be evaluated in order to determine the effectiveness of the treatment process and the safety in effluent discharge. Toxicity studies are important as the treatment process may also lead to refractory intermediate compounds, which may or may not be toxic then disposed into the environment. Thus, the future direction in determining the leachate treatment should also include the safety of the treatment process.

6.5 Development and Modification of Treatment Sequences

A treatment sequence selected to treat leachate of a particular age should offer the flexibility in adopting other unit operations or replacing some unit operations in favor of newer technologies. This is one of the most important and necessary criteria in developing a treatment scheme for high organic, high ammonia leachate and leachate with variable characteristics at different time period. This should also be economically and technological feasible with a long-term sustainability.

Future developments in this aspect could be achieved by evaluating the schemes for young and old leachate as well as exploring options for coupling membrane technologies with various physical-chemical or other biological processes to treat leachate of variable characteristics. In addition, by creating a modular design, the modular expansion of any or all of the unit operations would be easy and financially affordable to accommodate increased flow and load conditions due to climatic variations or other external factors.

6.6 Membrane Treatment Techniques

From the review of treatment sequences, it could be suggested as given by various authors (Qasim and Chiang, 1994; Bressi and Favari, 1997) that a biological process coupled with MF to form the MBR followed by an RO (post-treatment) could be successful in treating leachate of different ages. It is therefore recommended that this scheme should be further studied. Moreover, RO is also gaining popularity in various sectors including desalination and thus becoming economically viable option. However, a thorough characterization of the sludge and leachate should be performed and techniques should be developed to control biofouling/fouling of the membranes to render this solution conclusive.

Thus, it is evident that the focus of future research should be directed towards membrane techniques, fouling control measures and improving the design and operation of MBR processes.

6.7 Residual Treatment

A review of current and past treatment sequences for landfill leachate indicates that a combination of MBR and RO appears to be the most suitable alternative for treating leachate of variable quality - young, intermediate and stabilized. Though MBR is an

advanced treatment technology, total elimination of pollutants is impossible. Further, the RO unit is merely a physical separator. This implies that excess sludge disposal and concentrated waste from the RO unit would still require further treatment before disposal. Thus, the problem in leachate management and disposal is not fully addressed. However, by advanced oxidation of sludge, a zero-discharge system can be created by returning the oxidized sludge into the biological treatment process. This would increase the biodegradability of the process by reducing humic and other bio-refractory compounds into easily degradable entities, possibly increase the removal efficiencies of organic and color components and eliminating the need for sludge wastage. This would also reduce the overall effluent discharged from the process. Studies done by Yeom *et al.* (2001) indicated that ozonated sludge when recycled in the activated sludge process had an effect in increasing the biodegradability of the sludge and reducing excess sludge production.

Thus, biological treatment combined with advanced oxidation technology can be effective in leachate treatment. Among the various biological treatment processes, membrane treatment systems are efficient options. Oxidized sludge can also be used as nutrient or an internal carbon source for denitrification process.

6.8 Conclusion

The future research strategies for landfill leachate treatment could focus mainly on the following four systems besides modifications of conventional aerobic biological systems like sequencing batch reactors and anaerobic systems such as upflow anaerobic sludge blanket reactors, etc. The systems mentioned below could be promising in terms of innovative and effective leachate treatment.

1. Landfill irrigation system that could naturally reduce the leachate pollutant load as the re-circulated leachate would react in the bioreactor enhancing the degradation process.
2. Natural treatment systems using wetlands and different species of grass would facilitate an economical system where space does not become a constraint. Monitoring of such systems and financial inputs would be significantly low.
3. Investigations on natural treatment processes with use of different substrates like peat, coal, sawdust, clinker, soil etc. would facilitate the removal of pollutants from the leachate. This can be further studied in the light of its low cost and availability of substrate.
4. Membrane bioreactor can be used for the treatment of leachate where space and time become a major constraint as the system can be compact. Further, the use of different types of membrane processes has been proved successful and also enables the reuse of the effluent for secondary purposes.

References

1. Aberling, U. and Seyfried, C.F. (1992). Anaerobic-Aerobic Treatment of High-Strength Ammonium Wastewater - Nitrogen Removal via Nitrite, Water Science and Technology, 26 (5-6): 1007-1015.
2. Agamuthu, P. (1999). Characterization of Municipal Solid Waste and Leachate from Selected Landfills in Malaysia, Malaysian Journal of Science, 18: 99-103.
3. Ahn, K.H., Yeom, I.T., Park, K.Y., Maeng, S.K., Lee, Y., Song, K.G and Hwang, J.H. (2001). Reduction of Sludge by Ozone Treatment and Production of Carbon for Denitrification, Asian Water Quality Conference, September 12 to15, Fukuoka.
4. Amokrane, A., Comel, C. and Vernon, J. (1997). Landfill Leachates Pre-treatment by Coagulation-Flocculation, Water Research, 31(11):2775-2782.
5. APHA-AWWA-WPCF (1995). Standard Methods for the Examination of Water and Wastewater.19th edition.
6. Avezzu, F., Bissolotti, G. and Collivignarelli, C. (1992). Combination of Wet Oxidation and Activated Sludge Treatment. Landfill of Waste: Leachate. Ed. Christensen, Cossu, and Stegmann. Elsevier Science Publishers Ltd. England: ISBN: 1-85-166733-4: 333-352.
7. Bae, J-H, Kim, S-K. and Chang, H-S. (1997). Treatment of Landfill Leachates: Ammonia Removal via Nitrification and Denitrification and Further COD Reduction via Fenton's Treatment Followed by Activated Sludge, Water Science and Technology, 36 (12): 341-348.
8. Barlaz, M.A., Ham, R.K. and Schaefer, D. M. (1989). Mass balance analysis of decomposed refuse in laboratory scale lysimeter, Journal of Environmental Engineering, ASCE 115: 1088-1102.
9. Bjorkman, V.B. and Mavinic, D.S. (1977). Physico-chemical Treatment of a High Strength Leachate. Proceedings 32nd Annual Industrial Waste Conference, Purdue University, Lafayeete, Indiana: 189-195.
10. Blakey, N.C., Cossu, R., Maris, P.J. and Mosey, F.E. (1992). Anaerobic Lagoons and UASB Reactors: Laboratory Experiments. Landfill of Waste: Leachate. Ed. Christensen, Cossu, and Stegmann. Elsevier Science Publishers Ltd. England: ISBN: 1-85-166733-4: 245-263.
11. Boyle, W.C. and Ham, R.K. (1974). Biological Treatability of Landfill Leachate. Journal of Water Pollution Control Federation, 46(5): 860.
12. Bressi, G. and Favali, G. (1997). Uses of MBR (Membrane Bioreactor) in Leachate Treatment, Proceedings Sardinia 1997, Sixth International Symposium, S. Margherita Pula, Cagliari, Italy: 13 to 17 October.
13. Bull, P.S., Evans, J.V., Wechsler, R.M. and Cleland, K.J. (1983). Biological Technology of the Treatment of Leachate from Sanitary Landfills. Water Research, 17(11): 1473-1481.
14. Cameron, R.D. and Koch, F.A. (1980). Trace Metals and Anaerobic Digestion of Leachate, Journal of Water Pollution Control Federation, 52(2): 282.
15. Chang, L. (1993). Investigating of landfilling of MSW pretreated with different processes. Proceedings of Sardinia 1993, Fourth International Landfill Symposium. Vol. 2 Calgari, Italy: 2049-2056.
16. Cheung, K.C., Chu, L.M. and Wong, M.H. (1997). Ammonia Stripping as a Pre-treatment for Landfill Leachate, Water, Air and Soil Pollution, 94: 209-221.

17. Chian E.S.K. and DeWalle, F.B. (1976). Sanitary Landfill Leachates and Their Treatment, Journal of the Environmental Engineering Division, Proceedings of the American Society of Civil Engineers, 102: 411-431.
18. Chian, E.S.K. (1977). Stability of Organic Matter in Landfill Leachates, Water Research, 11(2): 225-232
19. Chian, E.S.K. and DeWalle, F.B. (1977). Treatment of High Strength Acidic Wastewater with a Completely Mixed Anaerobic Filter. Water Research, 11: 295-304.
20. Clabaugh, M. M. (2001). Nitrification of landfill leachate by biofilm columns, Virginia Polytechnic Institute and State University, Blacksburg, Virginia: MSc Thesis.
21. Cook, E.N. and Foree, E.G. (1974). Aerobic Bio-stabilization of Sanitary Landfill Leachate, Journal of Water Pollution Control Federation, 46(2): 380-392.
22. Cossu, R., Serra, R. and Muntoni, A. (1992). Physico-chemical Treatment of Leachate. Landfill of Waste: Leachate, Ed. Christensen, Cossu, and Stegmann. Elsevier Science Publishers Ltd. England: ISBN: 1-85-166733-4: 265-304.
23. Dollerer, J. and Wilderer, P.A. (1996). Biological Treatment of Leachates from Hazardous Waste Landfills using SBBR Technology. Water Science and Technology, 34(7-8): 437-444.
24. Doyle, J., Watts, S., Solley, D. and Keller, J. (2001). Exceptionally high-rate nitrification in sequencing batch reactors treating high ammonia landfill leachate, Water Science and Technology, 43 (3): 315-322.
25. Dzombak, D.A., Langnese, K.M., Spengel, D.B. and Luthy, R.G. (1990). Comparison of Activated Sludge and RBC Treatment of Leachate from a Solid Waste Landfill. Proceeding of 1990 WPCF National Specialty Conference on water quality Management of Landfills, Chicago, July 15 -18: 4-39 to 4-58.
26. Ehrig, H.J. (1998). Water and Element Balances of Landfills, Lecture Notes in Earth Sciences, (ed. P. Baccini), Springer-Verlag, Berlin.
27. Fairweather, R.J. and Barlaz, M.A. (1988). Hydrogen sulphide production during decomposition of landfill inputs, Journal of Environmental Engineering, ASCE 124: 353-361.
28. Farquhar, G.J. (1989). Leachate: Production and Characteristics, Canadian Journal of Civil Engineering, 16: 317-325.
29. Ferreira, J.A., Ritter, E., Rosso, T.C.A, Giordano, G. and Campos, J.C. (2001) Treatment of Landfill Leachate using Biological Filter and Reed Bed system, Sardinia, Proceedings of Eight International Waste Management and Landfill Symposium, (II): 241-247.
30. Fettig, J., Stapel, H., Steinert, C. and Geiger, M. (1996). Treatment of landfill leachate by preozonation and adsorption in activated carbon columns. Water Science and Technology, 34 (9): 33–40.
31. Fisher, M. and Fell, C. (1999). Ammonia Removal from High Strength Leachate Using a Sequencing Batch Reactor. Waste Management: 25-26.
32. Garcia, H., Rico, J.L. and Garcia, P.A. (1996). Comparison of Anaerobic Treatment of Leachate from an Urban-Solid-Waste Landfill at Ambient Temperature and at 35^{o}C, Bioresource Technology, 58: 273-277.
33. Gaudy, A.F., A.F. Rozich and S. Garniewski.(1986). Treatability Study of High Strength Landfill Leachate, Proceeding of the 41st Industrial Waste Conference, Purdue University, West Lafayette, Indiana: 627-638.

34. Gierlich, H. and Kollbach, J. (1998). Treating Landfill Leachate in European Countries, Pollution Engineering International, Autumn 1998.
35. Graham, D.W. (1981). Biological-Chemical Treatment of Landfill Leachate, Department of Civil Engineering, University of British Columbia, Vancouver.
36. Gurijala, K.R. and Suflita, J.A. (1993). Environmental factors influencing methanogenesis from refuse in landfill samples, Environmental Science and Technology, 27: 1176-1181.
37. Harmsen, J. (1983). Identification of Organic Compounds in Leachate from a Waste Tip, Water Research, 17 (6): 699-705.
38. Henderson, J.P., Besler, D.A., Atwater, J.A. and Mavinic, D. (1997). Treatment of Methanogenic Landfill Leachate to Remove Ammonia Using a Rotating Biological Contactor (RBC) and a Sequencing Batch Reactor (SBR). Environmental Technology, 18(7): 687-698.
39. Henry J. G., Prasad, D., Sidhwa, R. and Hilgerdenaar, M. (1982). Treatment of Landfill Leachate by Anaerobic Filter, Water Pollution Research Journal, Canada, 17: 45-56.
40. Henry, J.G., Prasad, D. and Young, H. (1987). Removal of Organics from Leachates by Anaerobic Filter. Water Research, 21(11): 1395-1399.
41. Ho, S. Boyle, W.C. and Ham, R.K. (1974). Chemical treatment of Leachate from Sanitary Landfills, Journal of Water Pollution Control Federation, 46(7): 1776-1791.
42. Horan, N.J., Gohar, H. and Hill, B. (1997). Application of a Granular Activated Carbon-Biological Fluidized bed for the Treatment of Landfill Leachate containing high concentrations of Ammonia, Water Science and Technology: 36(2-3): 369-375.
43. Hosomi, M., Matsusige, K., Inamori, Y., Sudo, R., Yamada, K. and Yoshino, Z. (1989). Sequencing Batch Reactor Activated Sludge Processes for the Treatment of Municipal Landfill Leachate: Removal of Nitrogen and Refractory Organic Compounds, Water Science and Technology, 21: 1651-1654.
44. Hu, T. (1989). Treatment of Vermicelli Wastewater by an Acid-tolerant, Starch Degrading Yeast. Biological Waste, 28(3): 163-174.
45. Illies, P. (1999). Biological Nitrification and Denitrification of High Ammonia Landfill Leachate Using Pre and Post Treatment Denitrification Systems and Methanol as Supplementary Source of Organic Carbon, University of British Columbia: MSc Thesis.
46. Imai, A., Iwami, N., Matsushige, K., Inamori, Y. and Sudo, R. (1993). Removal of Refractory Organics and Nitrogen from Landfill Leachate by the Microorganism-Attached Activated Carbon Fluidized Bed Process, Water Research, 27(1): 143-145.
47. Imai, A., Onuma, K., Inamori, Y. and Sudo, R. (1995). Biodegradation and Adsorption in Refractory Leachate Treatment by the Biological Activated Carbon Fluidized Bed Process. Water Research, 29(2): 687-694.
48. Jans, J.M., Schroeff, A.V.D and Jaap, A. (1992). Combination of UASB Pre-treatment and Reverse Osmosis. Landfill of Waste: Leachate. Ed. Christensen, Cossu, and Stegmann. Elsevier Science Publishers Ltd. England: ISBN: 1-85-166733-4: 313-321.
49. Joseph, K., Nagendran, R. and Palanivelu, K. (2002). Open Dumps to Sustainable Landfills, Centre for Environmental Studies, Anna University, Chennai, India.
50. Keenan, J.D., Steiner, R. L. and Fungaroli, A.A. (1984). Landfill Leachate Treatment, WCPF, 56(1): 33-39.

51. Kennedy, K. J., Hamoda, M. F. and Guiot, S. G. (1988). Anaerobic Treatment of Leachate using Fixed Film and Sludge Bed Systems, J.WPCF, 60: 1675-1683.
52. Kennedy, K. and Lentz, E. (2000). Treatment of Landfill Leachate Using Sequencing Batch and Continuous Flow Upflow Anaerobic Sludge Blanket (UASB) Reactors. Water Research, 34(14): 3640-3656.
53. Kettunen, R. H., Hoilijoki, T. H. and Rintala, J. A. (1996). Anaerobic and Sequential Anaerobic-Aerobic Treatment of Municipal Landfill Leachate at Low Temperature, Bioresearch Technology, 58(1):, 31-40.
54. Knox, K. (1985). Leachate Treatment with Nitrification of Ammonia, Water Research, 19 (7): 895-904.
55. Krug, T.A. and McDougall, S. (1988). Preliminary Assessment of Microfiltration/ Reverse Osmosis Process for the Treatment of Landfill Leachate, Proceedings of the Environment Ontario Environment Research Technology Transfer Conference, 28-29 November 1998, Toronto, Canada.
56. Kylefors, K. (1997). Landfill Leachate Management-Short and Long Term Perspectives, Lulea University of Technology, 1997:16, ISSN 1402-1757.
57. Leckie, J.O., Pacey, J.G. and Halvadakis, C. (1975). Acceleration Refuse Stabilization through Controlled Moisture Application, Paper Presented in the Second Annual National Environmental Engineering Research Development and Design, Gainsville, Florida.
58. Leckie, J.O., Pacey, J.G. and Halvadakis, C. (1979). Landfill Management with Moisture Control. Journal Environmental Engineering Division, ASCE, 105(EE2): 337-355.
59. Leikam, K. and Stegmann, R. (1999). Influence of mechanical biological pretreatment of municipal solid waste on landfill behavior, Waste Management and Research, 17: 424-429.
60. Lema, J.M., Mendez, R. and Blazquez, R. (1988). Characteristics of Landfill Leachates and Alternatives for their Treatment: A Review, Water, Air and Soil Pollution, 40: 223-250.
61. Luning, L. and Notenboom, G. (1997). The Membrane Bioreactor, Advanced Leachate Treatment, Proceedings Sardinia 1997, Sixth International Symposium, S. Margherita Pula, Cagliari, Italy: 13 to 17 October.
62. Martin, G.M.A., Auzmenti, A.I. and Olozaga, C.P. (1995). Landfill leachate: Variation in quality and quantity, Proceedings of Sardinia 95th International Landfill Symposium, CISA, Cagliari (Italy): 345-354.
63. Mavinic, D.S. (1998). Leachate Quality: Effects on Treatability. Proceedings of the International training Seminar: Management and Treatment of MSW Landfill Leachate, Venice, Italy, 2-4 December 1998, CISA, Sanitary Environmental Engineering Centre.
64. Mendez, R., Lema, J.M., Blazquez, R., Pan, M. and Forjan, C. (1989). Characterization, Digestibility and anaerobic Treatment of Leachates from Old and Young Landfills, Water Science and Technology, 21: 145-155.
65. Nakagawa, S., Tanaka, T., Ito S. and Katsu, Y. (1999). Decomposition of Dioxins in a Landfill Leachate by Advanced Oxidation Processes, Journal of the Japan Society of Waste Management Experts, 10 (2).
66. National Research Institute (NRI) China (2003). National Engineering Research Center for Urban Pollution Control Tongji University, Shanghai, China

67. NRI India (2003). Centre for Environment Studies, Anna University, Chennai – 600 025, India
68. NRI Sri Lanka (2003). Department of Agricultural Engineering, Faculty of Agriculture, University of Peradeniya, Kandy, Sri Lanka.
69. NRI Thailand (2003). Department of Environmental Engineering, Kasetsart University, Bangkok, Thailand
70. Nedwell, D. and Reynolds, P. (1996). Treatment of Landfill Leachate by Methanogenic and Sulphate-Reducing Digestion. Water Research, 30(1): 21-28.
71. Peters, T. (1997). Treatment of Landfill Leachate by Reverse Osmosis, Proceedings Sardinia 1997, Sixth International Landfill Symposium, S. Margherita di Pula, Cagliari, Italy: 13-17 October 1997.
72. Pirbazi, M., Ravindran, V., Badriyha, B.N. and Kim, S.H. (1996). Hybrid Membrane Filtration Process for Leachate Treatment, Water Research, 30 (11): 2691-2706.
73. Pohland, F.G. and Harper, S.R. (1985). Critical Reviews and Summary of Leachate and Gas Production from Landfills, EPA Report Number 600/2-86/073.
74. Pohland, F.G. and Kim, J.C. (1999). In Situ Anaerobic Treatment of Leachate in Landfill Bioreactors, Water Science and Technology, 40 (8): 203 -210.
75. Pohland, F.G. (1972). Landfill Stabilization with Leachate Recycle, Interim Progress Report, Grant EP 00658-01, Solid Waste Research Division, U.S.EPA, Cincinnati, Ohio.
76. Pohland, F.G. (1975). Sanitary Landfill Stabilization with Leachate Recycle and Residual Treatment, EPA 600/2-75-043, U.S. EPA, Cincinnati, Ohio.
77. Pohland, F.G., Stratakis, M., Cross, W.H. and Tyahla, S.F. (1990). Controlled Landfill Management of Municipal Solid Waste and Hazardous Wastes, Proceedings of 1990 WPCF National Specialty Conference on Water Quality Management of Landfills, Chicago, Ill., July 15-18, 3-16 - 3.32.
78. Pollution Control Department (2000). Research and Development of Landfill Leachate Treatment, Ministry of Science, Technology and Environment. Bangkok, Thailand, ISBN: 9-74-787938-7.
79. Qasim, S.R. and Chiang, W. (1994). Sanitary Landfill Leachate-Generation, Control and Treatment. Technomic Publishing Co., Inc. Pennsylvania, USA: ISBN 1-56-676129-8
80. Ragle, N. Kissel, J.C., Ongerth, J.E. and De Walle, F.B. (1995). Composition and Variability of Leachate from Recent and Aged Areas within a Municipal Landfill, Water Environment Research, 67(2): 238-243.
81. Raina, S. and Malvinic, D.S. (1985). Comparison of Fill and Draw Reactors for Treatment of Landfill Leachate. Water Pollution Research Journal, of Canada, 20(2): 12-28.
82. Ranaweera, R.M.R.P. and Tränkler, J. (2001). Pre-Treatment Prior Final Disposal - A Case Study For Thailand, Proceedings Sardinia 2001, Eighth International Waste management and Landfill Symposium, Pula, Cagliari, Italy , Vol III pp.187-196
83. Reeves, T.G. (1972). Nitrogen Removal: A Literature Review, Journal WPCF, 44(10): 1895-1908.
84. Robinson, H.D. and Grantham, G. (1988). The Treatment of Landfill Leachates in On-site Aerated Lagoon Plants: Experience in Britain and Ireland, Water Research, 22(6): 733-747.

85. Robinson, H.D. and Luo, M.M.H. (1991). Characterization and Treatment of Landfill Leachates from Hong Kong Landfill Sites, J.IWEM. 5: 326-335.
86. Robinson, H.D. and Maris, P.J. (1985). Treatment of Leachate from Domestic Waste in Landfill sites, Journal Water Pollution Control Federation, 57(1): 30-38.
87. Robinson, H.D., Barr, M.J. and Last, S.D. (1992). Leachate Collection, Treatment and Disposal, Journal of Institute Water and Environmental Management, 6: 321-332.
88. Robinson, H.D. and Maris, P.J. (1983). The Treatment of Leachates from Domestic Wastes in Landfills – I: Aerobic Biological Treatment of Medium-Strength Leachate, Water Research 17(11): 1537-1548.
89. Robinson, H.D. and Maris, P.J. (1985). The treatment of leachates from domestic waste in landfill sites, Journal of Water Pollution Control Federation, 57 (1): 30-38.
90. Robinson, H.D., Walsh, T. and Carville, M. (2002). Advanced Leachate Treatment at Buckden Landfill, Huntington, UK, IWM Annual Conference, 2002. www.leachate.co.uk
91. Robinson, H. D. (1999) State-of-the-Art Landfill Leachate Treatment Systems in the UK and Ireland, Paper presented to the IWM Annual Conference.
92. Schuk, W.W. and James, S.C. (1986). Treatment of Landfill Leachate at Publicly Owned Treatment Works, Waste Management and Research, 4: 265-277.
93. Siegrist, H., Reithaar, S. and Lais, P (1998). Nitrogen loss in a nitrifying rotating contactor treating ammonium rich leachate without organic carbon, Water Science and Technology, 37 (4-5): 589–591.
94. Sivapornpun, V. (2000). Heavy Metals in Landfill Leachate: Options for Reduction and Treatment. AIT Thesis, EV-00-17, Thailand.
95. Steensen, M. (1997). Chemical Oxidation for the Treatment of Leachate-Process Comparison and Results from Full-Scale Plants. Water Science and Technology, 35(4): 249-256.
96. Stephenson, T., Judd, S., Jefferson, B. and Brindle, K. (2000). Membrane Bioreactors for Wastewater Treatment. IWA Publishing: ISBN 1-90-022207-8.
97. Strachan, L.J., Trois, C., Robinson, H.D. and Olufsen, J.S. (2000). Appropriate Biological Treatment of Landfill Leachate with full Nitrification and Denitrification. Conference Proceedings, WISA, 2000, Sun City, South Africa.
98. Suflita, J.M., Gerba, C.P., Ham, R.K., Palmisano, A.C., Rathje, W.L. and Robinson, J.A. (1992). The world's largest landfill: multidisciplinary investigation, Environmental Science Technology, 26:1486-1494.
99. Tatsi, A.A. and Zouboulis, A.I. (2002). A field investigation of the quantity and quality of leachate from a municipal solid waste landfill in a Mediterranean climate (Thessaloniki, Greece), Advances in Environmental Research, 6:207-219.
100. Tchobanoglous, G. and Burton, F.L. (1991). Wastewater Engineering: Treatment, Disposal and Reuse, 3rd edition, McGraw-Hill: ISBN: 0-07-100824-1
101. Tchobanoglous, G., Theisen, H. and Vigil, S. (1993) Integrated Solid Waste Management: Engineering Principles and Management Issues, 1993 International editions, McGraw-Hill.
102. Thurman, E.M. and Malcolm, R.L. (1981). Preparative Isolation of Aquatic Substances. Environmental Science and Technology, 15: 463-466.
103. Timur, H. and Ozturk, I. (1999). Anaerobic Sequencing Batch Reactor Treatment of Landfill Leachate, Water Research, 33(15): 3225-3230.

104. Turk, O. and Mavinic, D.S. (1989). Maintaining Nitrite Build-Up in a System Acclimatized to Free Ammonia, Water Research, 23 (11): 1383-1388.

105. Venkatadri, R. and Peters, R.W. (1993). Chemical Oxidation Technologies: Ultraviolet Light/Hydrogen peroxide, Fenton's Reagent and Titanium Dioxide-Assisted Photocatalysis. Hazardous Waste and Hazardous Materials, 10(2): 107-149.

106. Vicevic, G.P.., Top, P.J. and Laughlin, R.G.W. (1992). Aerobic and Anaerobic Fixed Film Biological Reactors. Landfill of Waste: Leachate. Ed. Christensen, Cossu, and Stegmann. Elsevier Science Publishers Ltd. England: ISBN: 1-85-166733-4: 229-243.

107. Visvanathan, C., Ben Aim, R. and Parameshwaran, K. (2000). Membrane Separation Bioreactors for Wastewater Treatment, Environmental Science and Technology, 30(1): 1-48.

108. Webber, W.J. and Smith, E.H. (1986). Removing Dissolved Organic Contaminants from Water. Environmental Science and Technology, 20(10): 970-979.

109. Wehrle Environmental, Industrial and Landfill Leachate Case Studies, Company Information, Website: www.wehrle-env.co.uk

110. Welander, U., Henrysson, T. and Welander, T. (1998). Biological Nitrogen Removal from Municipal Landfill Leachate in Pilot Scale Suspended Carrier Biofilm Process, Water Research, 32(5): 1564-1570.

111. Wong, P.T. and Mavinic, D.S. (1984). Treatment of Municipal Leachate under Multi- Variable Condition, Journal Water Pollution Control Federation, Research, Canada.

112. Wu, Y.C., Hao, O.J., Ou. K.C. and Scholze, R.J. (1988). Treatment of Leachate from a Solid Waste Landfill Site Using a Two-Stage Anaerobic Filter. Biotechnology and Bioengineering, 31: 257-266.

113. Li, X., Yan, J., Chai, Y., Ni, M. and Cen, K. (2002), Development of municipal solid waste incineration technologies, Better Air Quality in Asian and Rim Cities, 16-18 Dec, 2002, Hong Kong.

114. Yalmaz, G. and Ozturk, I. (2001). Biological ammonia removal from anaerobically pre-treated landfill leachate in sequencing batch reactors (SBR), Water Science and Technology, 43 (3): 307-314

115. Yeom, I.T., Lee, K.R., Ahn, K.H., Lee, Y.H. and Lee, S.H. (2001). Effect of Ozone Treatment on the Biodegradability of Sludge from Municipal Wastewater Treatment Plants, Water Science and Technology, 46(4-5) in press and IWA Annual Congress, October 15 to 19, Berlin, Germany.

116. Zach, A., Binner, E. and Latif, M. (2000). Improvement of municipal solid waste quality for landfilling by means of mechanical-biological pretreatment, Waste Managemnt and Research, 18: 25-32.

117. Zaloum, R. and Abbott, M. (1997). Anaerobic Pretreatment Improves Single Sequencing Batch Reactor Treatment of Landfill Leachates. Water Science and Technology, 35(1): 207-214.

118. Zapf-Gilje, R. and D.S. Mavinic. (1981). Temperature effects on Biostabilization of Leachate, Journal of Environmental Engineering Division, ASCE, 107(EE4: 653-663

119. Zhao, Y., Liu, J., Huang, R. and Gu, G. (2000). Long-term monitoring and prediction for leachate concentrations in Shanghai Refuse Landfill, Water, Air, and Soil Pollution, 122:281–297.

Appendix

Landfill Leachate and Gas Constituent Concentration Ranges encountered in the Literature and their Relative Significance to the Degree of Landfill Stabilization					
	Phase of Biological Stabilization				
Leachate / Gas Constituent	**Transition Phase**	**Acid Formation Phase**	**Methane Fermentation Phase**	**Final Maturation Phase**	**Overall Range (All Phases)**
Biochemical Oxygen Demand (BOD_5) - mg/L	100 - 10,900 Influence of dilution and aerobic solubilization of waste organic	1000 - 57,700 Accumulation of biodegradable organic acids due to methanogenic lag	600 - 3400 Conversion of biodegradable organics to gaseous end products (CH_4 and CO_2)	4 - 120 Influence of high - molecular weight organic residuals (humics, fulvics)	4 - 57,700
Chemical Oxygen Demand (COD) - mg/L	480 - 18,000 Trending similar to BOD_5	1500 - 71,100 Trending similar to BOD_5	580 - 9760 Trending similar to BOD_5	31 - 900 Higher influence of residual organics than in BOD_5 assay.	31 - 71,700
Total Organic Carbon (TOC), mg/L	100 - 3000 Beginning to appear due to aerobic solubilization	500 - 27,700 Increasing rapidly; accumulation due to methanogenic lag	300 - 2230 Conversion of volatile acids to CH_4; decrease in aqueous carbon	70 - 260 Influence of higher molecular weight organics (humics, fulvics)	70 - 27,700
Total Volatile Acids (TVA), mg/L as acetic acid	100 - 3000 Just beginning to appear as a result of solubilization	3000 - 18,800 Solubilization of organic polymers to monomers; beta - oxidation to volatile acids	250 - 4000 Conversion to fatty acids; fermentation of acetic acid to methane	Essentially absent - methanogenic system under - saturation	0 - 18,800
BOD_5/COD Ratio	0.23 - 0.87 Increasing biodegradability of organics due to solubilization	0.4 - 0.8 High biodegradability	0.17 - 0.64 Decreasing biodegradability due to methanation	0.02 - 0.13 Low degree of biodegradability	0.02 - 0.87
COD/TOC Ratio	4.3 - 4.8 Low oxidation state or organics	2.1 - 3.4 Low to moderate oxidation state of organics	2.0 - 3.0 Moderate to high oxidation of organics	0.4 - 2.0	0.4 - 4.8
Total Kjeldahl Nitrogen (TKN), mg/L	180 - 860	14 - 1970 Low due to microbial assimilation of nitrogeneous compounds	25 - 82 Low due to microbial assimilation of nitrogenous compounds	7 - 490	7 - 1,970
Nitrate Nitrogen, mg/L	0.1 - 5.1 Increasing due to oxidation of NH_3	0.05 - 19 Decreasing due to reduction to NH_3 or N_2 gas	Absent: Complete conversion to NH_3 or N_2 gas	0.5 - 0.6	0 - 51
Ammonia Nitrogen , mg/L	120 - 125	2 - 1030 Increasing due to nitrate reduction and protein breakdown	6 - 430 Decreasing due to biological assimilation	6 - 430	2 - 1030

NH_3/TKN Ratio	0.1 - 0.9	0 - 0.98 Protein breakdown; biological assimilation	0.1 - 0.84	0.5 - 0.97	0.1
Total Phosphate (PO_4^{3-}—P)mg/L	0.6 - 1.7	0.2 - 120 Biological assimilation and metal complexation	0.7 - 14 Low due to biological assimilation	0.2 - 14	0.2 - 120
Total Alkalinity, mg/L as $CaCO_3$	200 - 2500	140 - 9650 Increasing due to volatile acid formation and bicarbonate dissolution	760 - 5050 Decreasing due to volatile acid removal	200 - 3520	140 - 9650
Total Solids (TS), mg/L	2450 - 2050	4120 - 55,300 Increasing due to solubilization or organics and mobilization of metals	2090 - 6,410	1460 - 4640	1460 - 55,300
pH	6.7	4.7 - 7.7 Low due to volatile acid accumulation	6.3 - 8.8 Increasing due to volatile acid removal and bicarbonate dissolution	7.1 - 8.8	4.7 - 8.8
Oxidation - Reduction Potential (ORP), mV	+40 to +80	+80 to –240 Decreasing due to the depletion of oxygen	- 70 to - 240	+97 to +163	- 240 to +16
Copper, mg/L	0.085 - 0.39	0.005 - 2.2	0.03 - 0.18 Decreasing – complexation (comp)	0.02 - 0.56	0.005 - 2.2
Iron, mg/L	68 - 312	90 - 2,200	115 - 336 Decreasing (comp)	4 - 20	4 - 2200
Lead, mg/l	0.001 - 0.004	0.01 - 1.44	0.01 - 0.1 Decreasing (comp)	0.01 - 0.1	0.001 - 1.44
Magnesium, mg/L	66 - 96	3.1 - 1140	81 - 505 Decreasing (comp)	81 - 190	3 - 1140
Manganese, mg/L	0.6	0.6 - 41	0.6 Decreasing (comp)	0.6	0.6 - 41
Nickel, mg/L	0.02 - 1.55	0.03 - 79	0.01 - 1.0 Decreasing (comp)	0.07	0.02 - 79
Potassium, mg/L	35 - 2300	35 - 2000	35 - 2300	35 - 2300	35 - 2300
Sodium, mg/L	20 - 7600				20 - 7600
Zinc, mg/L	0.06 - 21	0.65 - 220	0.4 - 6.0	0.4	0.06 - 220
Total Coliform, CFU/100 ml	10^0 - 10^5	10^0 - 10^5	Essentially absent	Absent	0 - 10^5
Fecal Coliform, CFU/100 ml	10^0 - 10^5	10^0 - 10^5	Essentially absent	Absent	0 - 10^5
Fecal Streptococci, CFU/100 ml	10^0 - 10^6	10^0 - 10^6	Essentially absent	Absent	0 - 10^5
Viruses, PFU/100 ml	- -	Essentially absent	Essentially absent	Essentially absent	Absent
Conductivity, μmhos/cm	2450 - 3310	1600 - 17,100 Increasing due to mobilization of metals	2900 - 7700 Decreasing due to metals complexation with sulfides	1400 - 4500	1400 - 17,100
Chloride (Cl^-), mg/L	30 - 5000 Biologically stable: good indicator of washout	30 - 5000 Stable: good hydraulic tracer	30 - 5000 Stable: good hydraulic tracer	30 - 5000 Stable: good hydraulic tracer	30 - 5000

Sulfate (SO_4^{2-}), mg/L	10 - 458 Increasing due to aerobic oxidation	10 - 3240 Increasing initially due to aerobic solubilization then decreasing as anaerobic conditions are established	Absent Complete conversion to sulfides	5 - 40 Reappearing due to aerobic oxidation	-
Sulfide (S^{2-}), mg/L	Essentially absent	0 - 818 Begins to appear and increases due to sulfate reduction under anaerobic condition	0.9 Low due to heavy metal precipitation	Absent	-
Cadmiun, mg/L	190 - 490	70 - 3900	76 - 490 Decreasing due to complexation and precipitation	76 - 254	70 - 3900
Chromium, mg/L	0.023 - 0.28	0.06 - 18	0.05 Decreasing due to complexation, precipitation with sulfides	0.05	0.02 - 18
Methane, %	Essentially absent (aerobic metabolism)	Very low (<1%); Transition to anaerobic metabolism	30 - 60 Suitable for energy recovery	0 - <10 Decreasing due to substrate limitation and reversion to aerobic metabolism	0 - 60
Carbon Dioxide, %	0 - 10 Product of aerobic decomposition of organic	10 - 30 Increasing due to waste decomposition	30 - 60 Decreasing to <5% as methanogenesis Increase	<40 Aerobic metabolism	0 - 60
Nitrogen Gas, %	70 - 80 Influence of trapped air	60 - 80 Decreasing due to dilution with CO_2	<20 Artefact of trapped air; dentrification	>20 Aerobic metabolism	<20 - 80
Oxygen, %	20 Influence of trapped air	0 - 5 Decreasing due to aerobic utilization; shift towards anaerobic metabolism	0 - 5 Disappearing as methanogenesis increases	>5 Increasing due to introduction of air	0 - 20
Hydrogen, %	Essentially absent in the presence of oxygen	0 - 2 Begins to appear as oxygen is depleted; accumulates until methanogenesis occurs	<0.1 Maintained at low levels by methanogenesis; difficult to measure	0 - 2 Essentially absent	-

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The Asian Institute of Technology (AIT) is an international graduate institution of higher learning in Bangkok,Thailand. Its mission is to develop highly qualified and committed professionals who will play leading roles in the sustainable development of the region and its integration into the global economy. Its vision is to become a leading and a unique regional and cultural institution of higher lerning, offering state of the art education, research and training in technology, management and societal development.

The Environmental Engineering and Management Field of Study of AIT and the National Engineering Research Center for Urban Pollution Control ,Tongji University, China have jointly carried out this study on Sustainable Solid Waste and Landfill Management (SWLM) in Asia under ARRPET with the support of the Swedish International Development Cooperation Agency (Sida). The study has culuminated into this publication State of the Art Review Landfill Leachate Treatment. The review is aimed at providing the readers - environmental managers and researchers an in-depth knowledge about the characteristics of leachates, its variations and treatment techniques with various case studies from Europe. The future directions in leachate management are given to highlight the research and developments in the subject.

For details regarding publications of the sustainable solid waste landfill projects, Please visit: http://www.arrept.ait.ac.th/swlf/index.htm or contact visu@ait.ac.th