DECOMPOSITION OF HIGH ORGANIC AND MOISTURE CONTENT MUNICIPAL SOLID WASTE IN BIOREACTOR LANDFILLS

By

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A thesis

presented to Ryerson University

in partial fulfillment of the requirements for the degree of

Master of Applied Science in the program of Civil Engineering

Toronto, Ontario, Canada, 2005

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Decomposition of High Organic and Moisture Content Municipal Solid Waste in Bioreactor Landfills

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Abstract

With the increase in waste recycling, municipal solid waste (MSW) with high organic and moisture contents are found in various landfills worldwide. If this kind of waste were put into anaerobic condition directly, the pH values will decrease sharply, which will seriously affect the biodegradation of the wastes. This study is aimed to investigate the decomposition of this type of MSW in aerobic condition before the anaerobic biodegradation. In the study, the effects of air addition and biosolids addition on the biodegradation of MSW with high organic and moisture contents were examined. Moreover, the flushing technology was compared with the leachate recirculation technology.

Six simulated bioreactor landfills were set up. After about 100 days' operation, it was observed that (1) the mass reduction rate in the aerobic-anaerobic bioreactor was approximately five times of that in the anaerobic bioreactor, the leachate quality was much better than that in the anaerobic bioreactor based on the final COD, BOD_5, TS, and NH_3 concentrations. (2) biosolids have strong buffering effects and the addition of biosolids accelerated the anaerobic biodegradation progress to a great extent.

Therefore, it was concluded that initially degrading MSW under aerobic condition before anaerobic degradation with biosolids addition is the optimum strategy for the decomposition of MSW with high organic and moisture contents.

Acknowledgements

I would like to express my appreciation to Dr. Mostafa Warith who gave me continuous guidance in the course of the thesis; Dr. Grace K. Luk who provided her lab for my long-term experimental analysis; Mr. Maqsoon Memoon, Mr. Youdong Wang, Mrs. Yongliang Xiao, Mr. Robin Luong, Mr. Dan Peneff, Mr. Nidal Jaalouk, and Mr. Domenic Valle who gave me help in the progress of the thesis experiment.

I am also grateful to NSERC for funding this research and the School of Graduate Studies for awarding me with Ryerson University Graduate scholarship.

Finally, I thank my parents, Mr. Xitai Jin and Mrs. Xiuying Feng,for their ongoing encouragement and support.

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BDOF	Biodegradable Organic Fraction
BMP	Biochemical Methane Potential
BOD_5	5-day Biochemical Oxygen Demand
COD	Chemical Oxygen Demand
LFG	Landfill Gas
MSW	Municipal Solid Waste
NH_3-N	Ammonia Nitrogen
TS	Total Solids
VFAs	Volatile Fatty Acids

1. INTRODUCTION

1.1 Generation and Management of Municipal Solid Waste

Municipal Solid Waste (MSW) include residential, commercial, and non-hazardous industrial wastes but exclude combustion ash, hazardous waste, sludge, and industrial process wastes. However, many of these other wastes are often deposited in the same landfills that receive MSW (Hilger and Barlaz, 2002). Therefore, many researchers use "refuse" instead of MSW for solid waste.

The generation of MSW has become an increasingly important worldwide issue over the last decade, because of the escalating growth in municipal populations, and the concomitant increase in waste production per capita. In 1997, 217,000,000 tons of MSW was generated in the U.S.A. alone, 8,000,000 tons more than 1996 (USEPA 1999). In 1994, Canadians disposed of 715 kg of solid waste per capita per year, totaling approximately 22,000,000 tons nationwide (Takata, 2002).

The increase in solid waste generation has promoted the development of the integrated management of MSW that is accomplished by recycling, composting, incineration, or landfilling of wastes. Among these methods, to date, the sanitary landfill is the predominant municipal solid waste disposal alternative because it is less expensive with less air pollution problems comparing with combustion, and there is a limit to the types of waste that can be recycled or composted (Hilger and Barlaz, 2002). Despite an increase in recycling, composting, and incineration, approximately 55% by weight of the MSW generated in the United States in 1997 was deposited in sanitary landfills (USEPA, 1999).

1.2 Development of Sanitary Landfilling

n the past, a landfill often represented little more than an open hole or mash where refuse vas dumped. The refuse was often not covered properly, sometimes it was burned for volume eduction, and there was little effort to control storm water runoff and downward migration of vater that had come into contact with the refuse (Barlaz, 1997). With the implementation of ncreasingly stringent regulations, landfills have become highly engineered facilities with ophisticated containment systems, environmental monitoring, and improved operational ractices.

As a generality, a typical dry landfill has an impermeable bottom liner, the wastes are elivered to the landfill, spread out, compacted and covered at the end of the day with a thin ayer of soil, until a planned depth is reached, then the waste is covered with an impermeable ap. The environmental barriers such as landfill liners and covers exclude moisture that is ssential to waste biodegradation. Consequently, wastes are contained in a "dry tomb" and emain intact for long periods of time ranging from 30 to 200 years, possibly in excess of the ife of the landfill barriers and covers. Liner failure could happen in conventional dry andfill sometime in future, which can cause serious groundwater and surface water ontamination (Warith, 2003).

Nowadays, siting new landfills has been very difficult and costly not only because landfills an threaten the environment, but also because the public opposition, this often called the NIMBY, or not in my back yard, syndrome. Therefore, the condition appeals to investigators o make efforts to make landfills more economically sound and environmentally friendly (Stessel and Murphy, 1992).

Today, the "bioreactor landfill" is one idea that has gained significant attention. A bioreactor landfill is a sanitary landfill that uses enhanced microbiological processes to transform and stabilize the readily and moderately decomposable organic waste constituents within 5 to 10 years of bioreactor process implementation. The bioreactor landfill significantly increases the extent of organic waste decomposition, conversion rates and process effectiveness over what would otherwise occur within the landfill (Pacey et al., 1999). The "bioreactor landfill" provides control and process optimization, primarily through the addition of leachate or other liquid amendments, the addition of sewage sludge or other amendments, temperature control, and nutrient supplementation (Reinhart et al., 2002). Beyond that, bioreactor landfill operation may involve the addition of air. Based on waste biodegradation mechanisms, different kinds of "bioreactor landfills" including anaerobic bioreactors, aerobic bioreactors, and aerobic-anaerobic (hybrid) bioreactors have been constructed and operated worldwide. According to the survey conducted by the Solid Waste Association of North America (SWANA) in 1997, there were over 130 leachate recirculating landfills in USA (Gou and Guzzone, 1997; Reihart et al., 2002).

Generally, there are four advantages for employing bioreactor landfill technology comparing to conventional dry landfills: (1) contain and treat leachate, (2) rapidly recover air space, (3) accelerate waste stabilization and avoid long-term monitoring and maintenance and delay siting of a new landfill, and (4) make more potential benefits from increased methane generation in anaerobic bioreactor landfill. For aerobic bioreactor landfills, there are three additional advantages: (1) significant increase in the biodegradation rate of the MSW over anaerobic processes, (2) a reduction in the volume of leachate, and (3) significantly reduced methane generation and "anaerobic" odors. However, Costs for continuous supply of air are excessively high for municipal solid waste treatment (Hanashima, 1999).

1.3 Composition of MSW

MSW composition can vary substantially with location and time depending on many factors, including socio-economic and climatic conditions, waste collection and disposal methods, sampling, and sorting procedures (El-Fadel and Khoury, 2000). The composition of buried MSW influences the biodegradation processes in the landfill ecosystem, which then affect not only landfill gas (LFG) production and composition but also leachate quality and quantity. The biggest environmental problem associated with landfilling practice is the generation of leachate and gas. Leachate is generated primarily as a result of precipitation falling on an active landfill surface, although other contributors to leachate generation include groundwater inflow, surface water runoff, moisture from emplaced waste, and biological decomposition (Reinhart and Townsend, 1998). Anaerobic degradation of organic matters in landfills leads to the generation of LFG containing methane (CH_4), carbon dioxide (CO_2), and trace concentrations of a range of vapors and gases. Methane is a very active greenhouse gas. Globally, landfills are the fourth largest anthropogenic source of methane emissions, releasing 22 to 36 million metric tons of methane annually (Doorn and Barlaz, 1995).

Many studies have examined the chemical composition of MSW (Barlaz et al., 1989a; Ham et al., 1993; Rhew and Barlaz, 1995; Eleazer et al., 1997; Ress et al., 1998). Table 1-1 presents typical organic composition of MSW.

Cellulose and hemicellulose represent the major degradable components of MSW. In contrast, lignin is essentially recalcitrant under methanogenic conditions; poly lignin is mineralized to

CO_2 and CH_4 in anoxic sediments at slow but environmentally significant rates (Colberg, 1988). It was reported that cellulose plus hemicellulose fraction of MSW accounts for 91% of its methane potential (Barlaz et al., 1990). Proteins and soluble sugars are other biodegradable organic materials that are present in smaller concentrations (Hilger and barlaz, 2002).

Table 1-1 Organic Composition of Residential Refuse (adapted from Hilger and Barlaz, 2002)

Source	% [dry wt]			
	Cellulose	Hemicellulose	Lignin	Volatile solids
Barlaz et al., 1989a*	51.2	11.9	15.2	78.6
Eleazer et al., 1997	28.8	9.0	23.1	75.2
Rhew and Barlaz, 1995	38.5	8.7	28.0	Not measured
Ress et al., 1998	48.2	10.6	14.5	71.4
Barlaz, unpublished data	36.7	6.7	13.6	Not measured

*: The following additional analyses were performed on this sample: protein, 4.2%; soluble sugars, 0.35%; starch, 0.6%; and pectin, <3%.

MSW recycle programs significantly contributed to change the composition of MSW, as well as the methane production from landfilled MSW. The actual methane yield of MSW decreased by 10% between the base case with no recycling (64.9 L wet kg $^{-1}$) and a case in which 31% of MSW is recycled (58.6 L wet kg $^{-1}$) (Eleazer et al., 1997).

1.4 Objective of This Study

With the increase in waste recycling and diversion, most dry waste (including paper, paperboard, bottles, cans as well as white goods) were diverted from the MSW stream which in turn increased the percentage of wet waste stream in the waste disposed in landfills, which included food wastes, yard trimmings, low-grade papers and inert residuals. In the wet waste stream, organic materials generally have high percentage, that could reach approximately

30% to 90% of the total mass of the MSW, and moisture content is also very high. These ypes of MSW can be found recently in several cities around the developed world, such as Toronto, and developing countries, such as the City of Istanbul (San and Onay 2001). If this kind of wastes were put into anaerobic condition directly, the pH values will decrease sharply, which will seriously affect the degradation of the wastes.

This study aims to examine the effect of initially degrading MSW with high organic and moisture contents under aerobic conditions prior to the anaerobic phase. Air addition was utilized to accelerate the degradation of the easily biodegradable organic materials as well as o create optimum environment for the downstream anaerobic degradation in bioreactor andfills. The objective of this study is threefold:

1. Examine the performance benefits of aerobic – anaerobic bioreactor landfills for treating high organic and moisture content wastes comparing to anaerobic bioreactor landfills.
2. Examine the performance benefits of biosolids addition for the decomposition of MSW with high organic and moisture contents in aerobic – anaerobic bioreactor landfills.
3. Compare the performance of the closed-loop leachate recirculation bioreactor landfills with single –pass bioreactor landfills.

Bioreactor landfills provide an innovative approach to control, monitor, and optimize the waste stabilization processes inside landfills rather than contain the wastes like “dry tombs”. Actually, they are more like organic solid waste digesters. In the newly developed aerobic or semi-aerobic bioreactor landfills, air is injected into the waste matrices, to increase the speed and extent of the organic waste degradation. Furthermore, offensive anaerobic odors can be removed. The degradation mechanism of the aerobic bioreactor landfills is similar to that of composting. In comparison, bioreactors have more benefits than composting: (1) land requirement would be less, (2) degradation would be enhanced because bioreactors have lining and capping, thus, airflow would be constant compared to periodic turning in composting (Stessel and Murphy, 1992).

It is hoped the this study finding will be applicable either in developed countries or in developing countries where MSW contains high organic and moisture contents in most cases.

1.5 Hypothesis

Active aeration will accelerate the degradation rate of MSW with high organic and moisture content, and decrease the time to stabilization.

2. LITERATURE REVIEW

2.1 The Anaerobic Degradation of MSW in Bioreactor Landfills

2.1.1 The Anaerobic Decomposition Process in Bioreactor Landfill Ecosystem

The technologies of enhancing biodegradation of organic waste in bioreactor landfills can be possibly developed upon understanding the basic biochemical processes that occur in such ecosystem. Numerous studies have been carried out on the anaerobic biodegradation process in the landfills. Many researchers (Reinhart and Al-Yousfi, 1996; Christensen and Kjeldsen, 1989; Barlaz et al., 1989a) have characterized the stabilization of wastes in terms of an idealized sequence of phases between the burial of fresh MSW and well-decomposed waste. Some investigations have suggested that the stabilization of waste proceeds in five sequential and distinct phase (Pohland and Harper, 1986; Reinhart and Townsend, 1998). The rate and characteristics of produced leachate and biogas vary from one phase to another, and reflect the microbially mediated processes taking place inside the landfill (Reinhart and Al-Yousfi, 1996). Major bacterial groups involved in this decomposition process include hydrolytic bacteria, fermentative bacteria, acetogenic bacteria, methanogenic bacteria and sulphate-reducing bacteria. The phases experienced by degrading wastes are described as following.

Phase I: Initial Adjustment Phase

In the aerobic phase, both oxygen and nitrate are consumed, with soluble sugars serving as the carbon source for microbial activity. The quantity of oxygen available is fairly low, depending on the degree to which the waste is compacted. All of the trophic bacteria groups required for MSW methanogenesis are present in fresh MSW (cellulolytics, acetogens, and methanogens), though there is little change in their populations (Barlaz et al., 1989a). In addition, this initial phase is associated with initial placement of solid waste and accumulation of moisture within landfills. An acclimation period (or initial lag time) is observed until sufficient moisture develops and supports an active microbial community (Reinhart and Al-Yousfi, 1996).

Phase II: Transition Phase

With the depletion of oxygen trapped within a landfill, a transformation from an aerobic to anaerobic environment occurs, and the facultative anaerobic microorganisms become active. The electron acceptors shift from oxygen to nitrates and sulfates (Reinhart and Al-Yousfi, 1996; Reinhart and Townsend, 1998). The hydrolytic and fermentative microorganisms hydrolyze polymers such as carbohydrates, fats, and proteins. The initial products of polymer hydrolysis are soluble sugars, amino acids, long-chain carboxylic acids, and glycerol (Barlaz et al., 1990). By the end of this phase, measurable concentrations of COD and volatile organic acids can be detected in the leachate (Reinhart and Townsend, 1998). In addition, the ammonia can be detected due to the hydrolysis and fermentation of protein compounds.

Phase III: Acid Formation Phase

During the first stage of this phase, the intermediates produced from Phase II, such as sugars, amino acids, long-chain carboxylic acids, and glycerol, are further fermented into short-chain carboxylic acids, carbon dioxide, and hydrogen. Acetate and alcohols are also formed. During the second stage of this phase, the obligate proton-reducing acetogens become active. They oxidize the fermentation products of the first stage to acetate, carbon dioxide, and hydrogen. The conversion of short-chain carboxylic acids to acetate is only thermodynamically favorable at very low hydrogen concentration. The thermodynamic favorability of reactions recognized as potentially operative during landfill stabilization is presented in Table 2-1 (Pohland and Kim, 2000). In nearly all cases, the role of hydrogen (H_2) is apparent and has led not only to the suggestion that H_2 will regulate reaction opportunity and pathway, but the relative predominance of process intermediates as well. However, there is a hydrogen-scavenging population, i.e., methanogens in an active anaerobic ecosystem. If fermentative and methanogenic activities are not balanced, intermediates will accumulate and may percolate from the landfill as leachate (Barlaz et al., 1990). Therefore, intermediate volatile fatty acids (VFAs) at high concentrations and a decrease in pH accompanied by metal species mobilization are often observed before the onset of MSW methanogenesis. The viable biomass growth associated with the acid formers bacteria, and rapid consumption of substrate and nutrients are the predominant features of this phase (Reinhart and Al-Yousfi, 1996).

Table 2-1 Representative Redox Half-Reactions during Waste Stabilization in Landfill Bioreactor (Adapted from Pohland and Kim, 2000)

Oxidation (electron donating reactions)[1]		ΔG^0 (KJ)
Caproate → Propionate	$CH_3(CH_2)_4COO^- + 2H_2O \rightarrow 2CH_3CH_2COO^- + H^+ + 2.5H_2$	+ 48.3
Caproate → Acetate	$CH_3(CH_2)_4COO^- + 4H_2O \rightarrow 3CH_3COO^- + H^+ + 4H_2 + 2H$	+ 96.7
Caproate → Butyrate+ Acetate	$CH_3(CH_2)_4COO^- + 2H_2O \rightarrow CH_3(CH_2)_2COO^- + CH_3COO^- + H^+ + 2.5H_2$	+ 48.4
Propionate → Acetate	$CH_3CH_2COO^- + 3H_2O \rightarrow CH_3COO^- + HCO_3^- + H^+ + 3H_2$	+ 76.1
Butyrate → Acetate	$CH_3(CH_2)_2COO^- + 2H_2O \rightarrow 2CH_3COO^- + H^+ + 2H_2$	+ 48.1
Ethanol → Acetate	$CH_3CH_2OH + H_2O \rightarrow CH_3COO^- + H^+ + 2H_2$	+ 9.6
Lactate → Acetate	$CH_3CHOH\ COO^- + 2H_2O \rightarrow CH_3COO^- + HCO_3^- + H^+ + 2H_2$	-4.2
Acetate → Methane	$CH_3COO^- + H_2O \rightarrow HCO_3^- + CH_4$	-31.0

[1] pH7, 1atm, 1 kg mol $^{-1}$ activity, 25°C

Phase IV: Methane Fermentation Phase

During phase IV, both methanogens and sulphate-reducing bacteria are involved in the anaerobic degradation. The hydrophilic methanogenic bacteria transform hydrogen and carbon dioxide into methane, and the acetophilic methanogenic bacteria transform acetic acid into methane and carbon dioxide. The rate of methane production increases rapidly to some maximum value. Methane gas constitutes approximately 50-60% (by volume) of gas composition (Barlaz, et al., 1990; Warith and Sharma, 1998). The pH value is increased, and consequently heavy metals are removed by precipitation. The organic matter present in the leachate declines, which causes the BOD and COD to fall.

In the mean time, sulphate-reducing bacteria convert hydrogen, acetic acid and higher VFAs into carbon dioxide and hydrogen sulphide. This group of bacteria competes with the methanogenic bacteria to transform the hydrogen and organic carbon. Based on their findings, Gurijala and Suflita (1993) indicated that methanogenesis might be limited to an unknown degree by the availability of sulfate. Fairweather and Barlaz (1998) reported that the presence of sulfate decreased methane yields, but sulfate reduction and methane production can occur concurrently during MSW decomposition and methanogenesis is the dominant electron sink process even in the presence of excess sulfate.

Phase V: Maturation Phase

During phase V, the easily biodegradable organic matter is stabilized, and nutrients and available substrate become limiting. Gas production drops dramatically and leachate strength stays steady at much lower concentrations. Reappearance of oxygen and oxidized species may be observed slowly (Reinhart and Al-Yousfi, 1996; Reinhart and Townsend, 1998). Concurrently, there is an increase in the rate of cellulose plus hemicellulose hydrolysis. The low level biodegradable matter gradually humifies (formation of complex molecules such as humic acid and filvic acid).

MSW degradation time span ranges from 30 to 100 years in traditional landfill ecosystem. However, with leachate recirculation, the temporal domain of the acid formation and methane fermentation phases is compressed, and accelerated stabilization of the readily degradable waste fractions typically leads to either accumulation and retention of more aggressive leachate during acid formation phase, or higher gas production/recovery potential and more stable leachate during subsequent methane fermentation phase than is encountered at conventional landfills (Pohland and Kim, 1999).

This idealized waste degradation sequence assumes that the waste is homogeneous and of constant age. A realistic landfill occupying waste cells with highly variable age and composition may yield a somewhat different picture (Barlaz et al., 1989b). In a large-scale landfill where waste is placed over a lengthy period of time, the waste stabilization phases tend to overlap and the leachate and gas characteristics reflect this phenomenon.

2.1.2 Governing Abiotic Factors for Anaerobic Degradation

Moisture Content

Moisture content is a critical factor affecting the rate and extent of organic waste decomposition. The benefits of increased water content in a landfill include limiting oxygen transport from the atmosphere, facilitating exchange of substrate, nutrients, buffer, and dilution of inhibitors and spreading of microorganisms within the landfill. The stimulatory effect of moisture content on anaerobic populations has been proved by numerous studies. Jones et al. (1983) characterized refuse samples from a sanitary landfill as a function of depth below the surface. The total anaerobic population as well as the populations of proteolytic, amylolytic, and cellulolytic bacteria increased near the water table, suggesting a stimulatory effect of moisture content. Protease and amylase activity increased sharply in the water table, which is consistent with the differences in enzyme activity between wet and dry refuse measured under laboratory conditions (Jones and Grainger, 1983; Barlaz, et al., 1990). The strong effect of moisture content was also seen in the correlations of total mass loss and moisture content according to the research results performed in full-scale landfills (Baldwin et al., 1998).

pH

At neutral pH, the bacteria responsible for MSW decomposition are most active. The optimal pH for refuse methanogenesis is 6.4 to 7.2 (Chugh et al., 1998). As discussed in the former section, the role of hydrogen is crucial, and the methanogen is hydrogen-scavenger. In low pH conditions, the activity of methanogenic bacteria is low. As a result, their conversion of hydrogen and acetic acid decreases. This causes the hydrogen pressure to build up, and at elevated pressures, acetogenic bacteria cannot convert volatile fatty acids, particularly butyric and propionic acid. The accumulation of these acids further lowers the pH within the landfill, and eventually stops methane production (Warith and Sharma, 1998). Therefore, the addition of buffering materials during bioreactor landfill operation is a critical strategy to maintain appropriate pH as well as balance relations between the various bacterial groups. The pH effect on the waste degradation is illustrated by the full-scale landfill studies in which a higher pH is correlated with more decomposed refuse reflected by the relationship between cellulose plus hemicellulose to lignin and pH (Ham et al., 1993; Mehta et al., 2002).

Nutrients

In landfill ecosystem, the anaerobic degradation of wastes particularly need such nutrients as nitrogen and phosphorous besides organic matters. The anaerobic ecosystem requires much less nitrogen and phosphorous than the aerobic system which assimilates much substrate into new cells. The optimal ratio between organic matter (expressed as COD), nitrogen and phosphorous is 100:0.44:0.08 (Christensen and Kjeldsen, 1989). In general, the well-mixed waste landfill will not be limited by nitrogen and phosphorous. Sometimes, the heterogeneity of landfill may limit the nutrients' availability to microorganism. Other micronutrients, e.g. sulphur, calcium, magnesium, potassium, iron, zinc, copper, cobalt, molybdanate and selenium, are found to be present in most landfills.

Temperature

Many studies have proved microbiological degradation rate increases along with temperature increase. The van't Hoff-Arrhenius equation (Metcalf and Eddy, 1991) is one of the most used equations that formulate the relationship between degradation rate and temperature as following:

[1] $k_t = k_{20} * \theta^{(T-20)}$

where: k_t = degradation rate constant at a particular temperature; k_{20} = degradation rate constant at 20°C = 0.23; θ = constant of 1.056 for temperatures between 20 and 30°C; and T = temperature for which k is desired.

The investigation done by Baldwin et al. (1998) tested this relationship as well. Blakey et al. (1997) documented that the role of temperature may be an important factor offering the potential means of manipulating the methane content of LFG. Rees (1980) observed that the optimum temperature for methane production from domestic refuse in a conventional anaerobic digester is about 40°C. Hartz et al. (1982) found that 41°C was the optimum for the generation of methane on a short-term basis, and methane generation would cease somewhere between 48 and 55°C. Mata-Alvares and Martina-Verdure (1986) reported the optimum temperature is 34 °C to 38 °C. In addition, it was documented that the rate of methane generation increased significantly (up to 100 times) when the temperature was raised from 20 to 30 and 40°C in laboratory simulations (Christensen and Kjeldsen, 1989).

Inhibitors

The anaerobic ecosystem is considered to be rather sensitive to inhibitors. Researchers have reported many inhibitors of anaerobic degradation, e.g. oxygen, carbon dioxide, hydrogen, proton activity, salt ions, sulphide, heavy metals, and specific organic compounds (Christensen and Kjeldsen, 1989). Cations such as sodium, potassium, calcium, magnesium and ammonium have been observed to stimulate anaerobic decomposition at low concentration while inhibit it at high concentrations. High sulphate concentration can inhibit methane generation.

It has been speculated that CO_2 acts as an inhibitor through the raising of the redox potential (Hansson, 1982), or the impairment of the methanogen cell membrane function by increasing its fluidity through CO_2 dissolving in the cell membranes of methanogens (Senior & Kasali, 1990). Additionally, it is possible that CO_2 acts as an end-product inhibitor during acetate and propionate degradation.

2.1.3 Technologies of Enhancing Degradation in Anaerobic Bioreactor Landfill

As discussed above, the principal and governing factors in the anaerobic degradation are very clear. How to manipulate these factors to accelerate the waste stabilization rate and get benefits from landfill is what numerous researchers have been attempting during the past over 30 years. Many technologies have been examined and applied in full-scale practices.

The stabilization means that the environmental performance measurement parameters (LFG composition, generation rate and leachate constituent concentrations) remain at steady levels, and should not increase in the event of any partial containment system failures beyond 5 to 10 years of bioreactor process implementation (Pacey et al., 1999). Therefore, the stabilization of waste is quantified by leachate quality, gas composition and production, landfill settlement. The effects of the following technologies are evaluated according to these aspects.

Leachate Recirculation and Moisture Control

Previous experiences and researches have indicated that moisture content is a critical factor in enhancing waste decomposition in bioreactor landfills. Moreover, some studies indicated not only moisture content but also moisture movement could affect waste stabilization. Therefore, moisture control (including moisture content and movement) is the essential for landfill operation. Leachate recirculation has been demonstrated to be a superior management strategy for moisture control. The study of leachate recirculation in landfills has attracted numerous researchers since mid 1970s (Pohland, 1975a, b, 1980; Mata-Alvarez and Martinez-Viturtia, 1986; Barlaz et al., 1990; Reinhart and Al-Yousfi, 1996; Reinhart, 1996; Pohland and Kim, 1999; San and Onay, 2001; Mehta et al., 2002). Through leachate recirculation, liquid movement distributes the inocula, minimizes local shortages of nutrients, provides better contact between insoluble substrates, soluble nutrients, and the microorganisms, dilutes potential toxins, and transfers heat. As a result, microbial activities are increased. The advantages of leachate recirculation include: providing in-situ leachate treatment instead of off-site treatment, thus saving costs; enhancing waste settlement, thus decreasing the risk of damage to the final cover and permitting recovery of valuable landfill air space; increasing gas generation rate which make energy recovery more favorable; accelerating waste decomposition, thus shortening the post closure monitoring period and reducing the overall landfill operation cost. Laboratory, pilot scale and full-scale studies have tested all these advantages.

Poland (1975 a, b) conducted studies on accelerating solid waste stabilization and leachate treatment by leachate recycle in simulated landfills. He concluded that the capture and recirculation of leachate through a simulated landfill can promote a more rapid development of an active anaerobic bacterial population of methane former, increase the rate and predictability of stabilization of readily available organic pollutants, dramatically decrease the time required for stabilization, and reduce the potential for environmental impairment.

Between 1993 and 1996 two pilot scale test cells were constructed at Yolo County, California, USA. Each cell has a surface area of about 930 m^2 and initial 12 m depth (Yolo Co., 1997; Mehta, et al., 2002). One cell was designated the "enhanced" cell in which supplemental liquid was added and leachate was recirculated. The "control" cell was constructed identically to the "enhanced" cell, however no liquid has been added. The two cells began operation in 1996. After about 3 years of operation, Mehta, et al. (2002) performed a comparison of the two test cells to evaluate the effects of leachate recirculation on refuse decomposition. After analyzing 44 samples from 33 distinct depth intervals and collecting the gas generation data and waste settlement data, Mehta, et al. (2002) arrived to the conclusion that the leachate recirculation has the potential to enhance settlement, methane production, and solids decomposition at field-scale. Refuse was excavated in three borings from the enhanced cell and two borings from the control cell. The moisture content analysis shows that the average moisture content in these samples range from 34 to 38 % in the enhanced cell, while the average moisture content in the control cell ranged from 14.6 to 19.2%. These data illustrate that leachate recirculation increase moisture content in enhanced cell. Leachate recirculation increased both methane production (63.1 versus 27.9 L CH_4 wet kg^{-1} over 1231 days) and waste settlement (15.5% versus 3% of the waste thickness).

During record period, the total volume of leachate recycled in the enhanced cells is equivalent to 570 L metric ton $^{-1}$. This volume should increase the refuse moisture content in the enhanced cell to 46%. However, only 2 of 33 collected samples reached this value. This illustrates that the liquid likely flow through the preferential flow paths in the waste. Therefore, the design of the system used for the distribution of recycled leachate is a critical factor for achieving good moisture management in bioreactor landfill.

Townsend et al. (1996) also presented the effects of leachate recycling on landfill stabilization at an existing lined landfill in North-Central Florida during the period from 1989 through July 1993. Leachate was recirculated to the landfill by means of an infiltration pond. The area of the landfill east of the ponds was left untreated to serve as the control area. The results indicated that leachate recycling significantly increased moisture content of the landfilled waste, and maintained conditions suitable for biological stabilization. The results of the settlement analysis illustrate the greatest subsidence occurred in the area close to the infiltration pond at 1.01 m (5.65% volume reduction), and the least subsidence was measured in the area farthest from the leachate recycle ponds at 0.69 m (3.82% volume reduction). The original average biochemical methane potential (BMP) from biodegradable organic fraction (BDOF) samples in the recycling area was 0.273 m^3 CH_4 kg^{-1} volatile solids (VS), and decreased to 0.196 m^3 CH_4 kg^{-1}VS. In contrast, the original average BMP from BDOF samples in control area is 0.297 m^3 CH_4 kg^{-1}VS, and only decreased to 0.281 m^3 CH_4 kg $^{-1}$VS.

In essence, the landfill itself can be used as a controlled anaerobic treatment system much analogous to an anaerobic trickling filter (Pohland, 1975a; Tittlebaum, 1982). Leachate recirculation can also supply effective in-situ treatment for landfill leachate. Even where recycled leachates are more concentrated than single-pass leachates, they are treated primarily inside the landfill, utilizing its storage and biodegradation capacity as an effective bioreactor (Reinhart and Al-Yousfi, 1996). The in-situ treatment effect can be illustrated by comparing the leachate data of the Trail Road Landfill (Warith, 2002) with the leachate data of the Tre Monti site (Frascari et al., 2004).

The Trail Road landfill Phase 3 in Ottawa, Canada started operation in 1991. The generated leachate was pumped into infiltration lagoons, which were constructed using on-site stockpiled clay for containment dykes. The infiltration lagoons were relocated periodically to ensure even distribution of the moisture and to accommodate the landfilling of the solid waste. The ratio of the BOD/COD decreased from about 0.9 to 0.4 over a period of eight years. Tre Monti is a 4.16-million-m^3 landfill located on a pliocenic clay declivity on the hills near Imola, in Northern Italy, and was built in 1989–1990. After a significant initial decrease, BOD remained relatively uniform from 1993 to 2001, ranging between 850 and 1700 mg l^{-1}, whereas COD, after a similar initial decrease, showed an overall tendency to increase in the 1994–2000 period. The BOD/COD ratio decreased from 0.50 to 0.18 from 1992 to 2001. Klink and Ham (1982) indicated that moisture content and movement are separate variables that affect landfill methane generation rates. They have observed that moisture movement through refuse bed enhanced the rates of methane production by 25 to 50%, compared to those rates during conditions of minimal moisture movement and at the same overall moisture content. Chugh et al. (1998) examined different leachate recirculation rates namely, 30%, 10%, 2% of the initial volume of waste in the reactors, and indicated that moisture movement significantly improve methane production rates.

Many studies agreed that leachate recirculation with pH control further enhanced landfill stabilization and treatment efficiency, and buffering the leachate prior to its recirculation is an important operation strategy for the maintenance of the desired pH values in the system (San and Onay, 2001). When Pohland and Kim (1999) examined the in-situ treatment of leachate and co-disposal of organic and inorganic hazardous wastes by using simulated landfill with leachate recirculation, they observed an early and rapid onset of acid formation with production of a strong and chemically aggressive leachate once field capacity was reached and excess moisture for leaching became available. However, the initial transformation pattern tended to persist until changes in leachate recirculation intervals occurred, neutralization and sludge seeding promoted the onset of active methane fermentation. The reason for this phenomena is that leachate recirculation, in some cases, can cause acid accumulation. In the acid formation phase, excess acids and hydrogen can make the thermodynamical favorable reactions (Table 2-1) reverse and shift the equilibrium to the left.

Buffering the recirculated leachate can reverse this phenomenon.

The effect of variable rates of leachate recirculation was examined by Al-Yousfi and Pohland (1998), where they employed a numerical model (PITTLEACH) to predict leachate quantity and quality, as well as biogas generation for both single-pass leaching and leachate recirculation. The results indicated that total volatile acids concentrations increased and the plateau section of concentration verses time relationship became more pronounced as the leachate recirculation rates increased. These profiles confirmed the effectiveness of landfills as in-situ bioreactors capable of treating and mineralizing high-strength leachate. The results indicated that higher leachate recirculation rates will cause higher methane gas generation and that there is a lag time needed for the methanogenic phase to prevail under leachate recycling operation. This confirmed that leachate recirculation can cause acidogenic conditions in landfills, and buffering with leachate recycling is very necessary to neutralize the acidic conditions.

Leachate recirculation reduces metal concentrations in leachate. The primary metal removal mechanisms appear to be sulphide and hydroxide precipitation and reaction with humic-like substance. Leachate recycling promotes neutral or above neutral leachate conditions as well as stimulates reducing conditions providing for the reduction of sulphate to sulphide (Gould et al. 1989). Additionally, moderate to high molecular weight humic-like substances are formed from waste organic matter with time. These substances tend to form strong complexes with heavy metals. However, over time, oxygen and water may enter the landfill creating conditions that may mobilize metals and flush remaining inorganic contaminants out of the landfill (Reinhart et al., 2002).

Some researchers have carried out some studies on the co-disposal potential of bioreactor andfills for organic and inorganic hazardous wastes with leachate containment and in-situ recirculation. Reinhart et al. (2002) documented that bioreactors would tend to optimize removal of hazardous organic contaminants by (1) stripping volatile organics by increased gas production, (2) optimizing conditions for biodegradation, and (3) stimulating immobilization of contaminants through humification. Sanin and Barlaz (1998) also confirmed these mechanisms. Pohland and kim (1999) reported that the effect of admixed loadings of inorganic and organic hazardous wastes on anaerobic degradation can be offset by managing their attenuation through leachate containment and regulated recirculation, the horizon of application of bioreactor landfills can be extended to co-disposal practices by implementing prospective design, construction and operational protocols consistent with simulated experiments' findings. The attenuation capacity of landfill bioreactors is equally effective for toxic organic compounds by employing leachate recirculation, and bioremediation with reductive dehalogenation is a prime example (Pohland et al. 1993; Pohland, 1995). Pagano et al. (1995) carried out a study to determine the reduction potential of PCB-contaminated sediments in anaerobic bioreactor systems with leachate recirculation. After 13 weeks of operation, the average total chlorine/biphenyl of the original Aroclor was reduced by 11% and 23%, respectively.

At landfills whereas leachate recalculation is practiced to enhance decomposition of readily degradable organic constituents, leachate ammonia nitrogen concentrations may accumulate to higher levels than during conventional single pass leaching, thereby requiring treatment prior to ultimate discharge (Pohland, 1995). Leachate recirculation could create an environment that promotes the rapid development of desired microbial populations of denitrifiers, nitrifiers, and methanogens. Onay and Pohland (1998 and 2001) reported nitrogen and sulfate attenuation in simulated landfill bioreactors. The experimental results indicated that both nitrogenous and sulfur compounds can be attenuated through autotrophic denitrification, and leachate nitrate concentrations of 750 mg/l reduced to less than 1 mg/l by denitrification to nitrogen gas. Promoting this process in landfill environment can result in the reduction of leachate ammonia and sulfate concentrations without any need for external leachate treatment. Furthermore, autotrophic denitrification can utilize sulfur compounds, prevent their accumulation in landfills and decrease their potential for inhibition of methanogenic bacteria by sulfate-reducing bacteria in competition for substrate. Therefore, it is recommended to modify landfill design by involving an aerobic zone associated with the leachate under-drain system, and an anoxic zone associated with a surface leachate distribution system below the final cap.

Leachate over recirculation can lead into saturation, ponding, and high level of acid conditions, particularly during early degradation phases (Reinhart and Townsend, 1998). The principal operational challenge is to manage leachate recirculation in such a manner that the excessive accumulation and retention of more aggressive leachate during the acid formation phase does not inhibit the onset and development of an active methane fermentation phase (Pohland and Kim, 2000). In order to maximize waste stabilization, leachate recirculation frequency must be carefully selected. Leachate application, with pH control, four times per week was reported (San and Onay, 2001) to effectively increase waste stabilization in terms of high gas yield and lower organic content in the leachate. It is extremely crucial, in full-scale leachate recirculation, that leachate is applied at a slow rate before the onset of methanogenic phase of waste biodegradation, and can be increased once LFG production reaches a reasonable flow rate (Reinhart and Townsend, 1998; Pohland, 1995).

Inocula Addition

Many researchers suggested adding inocula as a bioreactor management alternative. Municipal sewage sludge, animal manure, septic tank sludge and old MSW have been recommended as potential inocula. The addition of sludge to MSW has been reported to have both positive and negative effects in waste biodegradation. Anaerobically digested sewage sludge can serve as a seed to microorganisms as well as source of nitrogen, phosphorous, and other nutrients. Early studies by (Pohland, 1975a, b; Leckie et al., 1979; Buivid et al., 1981; Stegmann, 1983; Craft and Blakey, 1988) indicated that leachate recirculaion with pH control and sludge seeding enhanced biological stabilization of organic pollutants in the leachate and substantially increased biogas generation rates in span of few months rather than years. More recent laboratory study by Gulec et al. (2000) reported that in 10-liter laboratory-scale batch digesters filled with 2-year old MSW and sludge at ratios of 1:9, 1:6 and 1:4 (anaerobically digested sludge to waste on wet basis), pH of leachate ranged from 7.0 to 8.5 compared to sharp drop in pH levels to the acidic range in the control reactors (no sludge addition). This may be explained by the buffer capacity of sludge. Additional field practices of adding biosolids to waste by Blakey et al. (1997) and Viste (1997) indicated relative increase of biogas production and improvement of leachate quality.

On the other hand, Barlaz et al. (1987) observed carboxylic acid accumulations and decreases in pH associated with sludge addition to fresh MSW. The results of this study confirmed that sludge addition without buffer addition did not stimulate methane production. Moreover, it was suggested that sewage sludge addition to MSW might have a limiting effect on waste biodegradation if the anaerobic conditions are already established (Christensen and Kjeldsen, 1992).

Another alternative source of inocula is composted solid waste. Stegmann and Spendlin (1989) found that the addition of composted MSW to fresh MSW helps to initiate the methane phase relatively early. Furthermore, Suna Erses and Onay (2003) suggested that the utilization of external leachate recycled from old landfills having desired acclimated anaerobic microorganisms, low organic content and higher buffer capacity into a young landfill could be a promising leachate management strategy for faster waste stabilization. In the above study, old landfill leachate containing large number of methanogens served as inocula, and helped the onset of methanogenic conditions.

Particle Size

The use of MSW with a reduced particle size relative to unprocessed MSW provides a more homogenous waste. The well mixed shredded waste permits greater contact between the key refuse constituents required for methane production: moisture, substrate, and microorganisms (Barlaz et al., 1990). Waste shredding could lead to rapid oxygen utilization, increase rate of waste decomposition, and lead to early methane production (Ham and Booker, 1982; Otieno, 1989). Experimental results indicated that shredded MSW produces leachate with higher peak COD concentrations and slightly lower minimum pH levels than unprocessed MSW. However, too small particle sizes could cause rapid waste hydrolysis, and lead to a build-up of acidic end products, that will have a negative impact on methane production. MSW shredding to particle size in the range of 250 to 350 mm particle sizes produced 32% more methane after 90 days than MSW with 100 to 150 mm particle sizes, and 100-150 mm shredded MSW produced 16 times as much methane as a finely shredded MSW of less than 25 mm particle size (Buivid et al., 1981).

Temperature Control

As discussed above, the optimum higher temperatures will result in faster rates of gas production and refuse stabilization. The temperature attained by a landfill is determined by he balance between the rates of heat production and the rate of heat loss to the surrounding soil and atmosphere. The introduction of air and the consequential onset of aerobic activity contribute to rapidly increase temperature and have been found to stimulate methane production (Reinhart and Townsend, 1998; Rees, 1980). The phenomena was verified by full-scale tests (Mehta et al., 2002): temperatures in bioreactor cell with leachate recycle and cell without leachate recycle reached 50-55 °C in the top layer just after refuse burial. According to Mehta et al. (2002) observations, leachate recirculation accelerated the anaerobic reactions in landfills, and increased the temperatures inside the bioreactor landfill. It was reported that temperatures in the control cell without leachate recirculation stabilized at 25-32 °C, and temperatures in the enhanced cell with leachate recirculation increased with the initiation of leachate recirculation and ultimately stabilized at 35 °C in the bottom layer and 40 °C in the middle and top layers.

According to a full–scale investigation, Rees (1980) suggested that the method to maintaining temperatures of about 45 °C in an anaerobic landfill in a temperate climate is to allow water into the site from the bottom and maintain an insulating layer of about 4 m above the groundwater table in the landfill. Another potential method of temperature control is the heating of recirculated leachate such as used in Sweden's experimental "Energy Loaf", however the potential of this leachate heating needs further examination.

Lift Design

MSW is usually disposed of in 2 to 3 m lifts with or without daily cover. The depth of lifts, whether or not compacted, and with or without daily cover are important factors affecting the waste degradation. Early studies indicated that leachate COD concentration was a function of waste depth (Ham and Bookter, 1982), whereas COD of deeper cells (2.4 m) exhibited more than double the typical COD of the comparable shallow ones (1.2 m). Stegmann (1983) suggested that the first layer should be uncompacted, so readily degradable organics can decompose aerobically and are allowed to stabilize before addition of subsequent lifts. Reinhart et al. (2002) indicated that the increased MSW compaction not only reduces the waste ability to move moisture through the waste but also makes the waste achieve level of saturation with less moisture addition because both waste hydraulic conductivity and field capacity are inversely related to waste density. Moreover, compaction contributes to anisotropic conditions within the landfill that magnify lateral movement of moisture. Several bioreactors in Iowa, Wisconsin, and the UK have operated with little or no compaction (Viste, 1997; Blakey et al., 1997).

Field results confirmed that partially decomposed MSW has the ability to attenuate leachate (Ham and Bookter, 1982; Stegmann and Ehrig, 1982). The COD and BOD concentrations were reduced to 75% after leachate seeping through deeper lifts of MSW.

Applying of daily or intermediate cover of low permeability can lead to horizontal movement and the potential for leachate ponding or side seeps (Reinhart and Townsend, 1998). For example, Natale and Anderson (1985) reported saturated conditions and ponding at the Lycoming County site during periods when high volumes of leachate were recirculated in areas using clay and silty soils for daily cover. Therefore, many researchers suggested lift design without daily cover, or a cover should not be used immediately. However, in the actual bioreactor landfill operations, daily cover is used to improve the access to the landfill; reduce he amount of waste that can blow away; reduce the risk of disease; reduce odors; reduce the potential of landfill fire.

In order to minimize ponding and horizontal movement, Reinhart and Townsend (1998) suggested use of high permeability soils and/or alternative daily cover should be considered. Alternative daily cover materials include mulched or composted yard waste, foam, carpet, clay/cellulose additives, and geotextiles. The use of these alternative materials may result in andfill space and cost saving, increase of waste hydraulic conductivity within the landfill and extended life of the leachate drainage layers efficiency (Wiles and Hare, 1997). For example, he use of alternative daily cover in the form of green waste or tarps was successfully during he waste-filling phase of the Yolo County Central Landfill project (Yazdani et al., 2002).

Nutrients Addition

Nutrients required for waste degradation in landfills are generally met at least during early degradation phases. Sometimes, phosphorous may be limiting during later stages. Some studies found that the addition of nitrogen and phosphorous stimulated methane production or rapidly decreased BOD and COD concentrations in the leachate (Pohland, 1992; Warith, 2002). Moreover, some researchers (Stegmann, 1983; Leushner, 1989) observed that the addition of nutrients such as nitrogen and phosphorous, and buffer to the recycled leachate significantly shortened the initial phase of biodegradation, and methane generation commenced earlier. However, other studies found nutrient control had no significant effect on stabilization of the waste (Tittlebaum, 1982; Mata-Alvarez and Martinez-Verdure, 1986). Therefore, it is concluded that nutrient addition does not have sufficient advantages as other enhancement technologies.

2.2 The Aerobic Degradation of MSW in Bioreactor Landfills

Recently, increased interest has been focused on the introduction of oxygen to the landfill to create an aerobic bioreactor (Reinhart et al., 2002). In an aerobic environment, the indigenous, respiring microorganisms convert the biodegradable organic compounds in MSW to mostly carbon dioxide and water, instead of methane, with stabilized humus remaining. The aerobic degradation of MSW is mostly covered in the composting literature. Rich (1963) suggested the following stoichiometric equation for aerobic composting:

[2] $C_aH_bO_cN_d + 0.5(ny + 2s + r - c)O_2 = nC_wH_xO_yN_z + sCO_2 + rH_2O + (d - nz)NH_3$

Where: $r = 0.5[b-nx-3(d-nz)]$; $s = a-nw$

The terms $C_aH_bO_cN_d$ and $C_wH_xO_yN_z$ represent the compositions of feed substrate and final product, respectively.

Aerobic biodegradation processes have demonstrated that many of the organic compounds found in MSW can be degraded in significantly short time frames (as compared with anaerobic conditions) by the introduction of air and moisture in the proper proportions (Stessel and Murphy, 1992; Hudgins and Harper, 1999; Read et al., 2001). This leads to the idea of in-situ aerobic biodegradation of MSW in a landfill environment. The benefits of air injection are described as following:

Accelerating MSW Biodegradation

Aerobic biodegradation occurs at a much faster rate and to a greater extent than anaerobic conditions in a given time period (Stessel and Murphy, 1992). This conclusion was also testified by Hanashima (1999). Figure 2-1 shows the change in landfill types and its leachate BOD concentration with time.

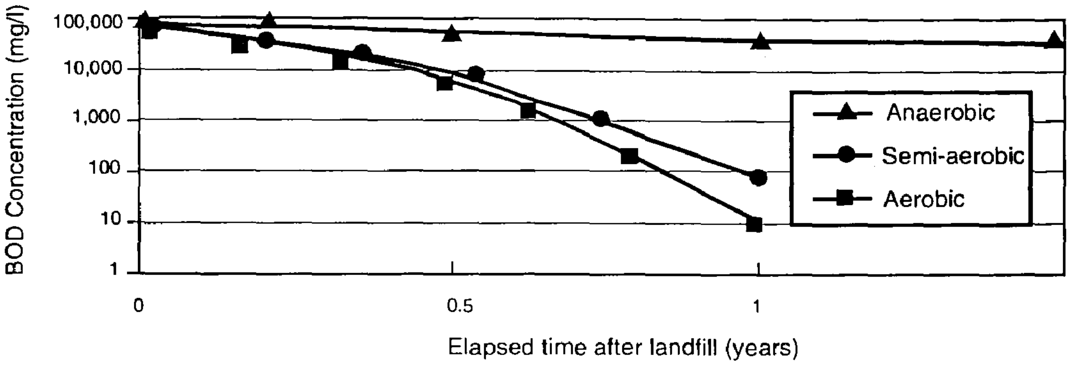


Figure 2-1 Changes in Landfill Types and Its Leachate BOD Concentration with Time (Adapted from Hanashima, 1999)

Reducing Leachate Volume

When the injected air passes the waste matrices, it is heated by the aerobically degraded materials, and it picks up moisture as well as dries the remaining materials. Therefore, the volume of leachate is reduced. This opinion is confirmed by two independent aerobic landfill demonstration projects in Columbia County landfill in Augusta, Georgia and Live Oak Landfill in North-Central Georgia. The leachate volume reduction rate in these two landfills s 86% and 50% separately (Hudgins and Harper, 1999).

Increasing Waste Settlement

Through lysimeter study, Stessel and Murphy (1992) concluded that aerobic degradation provided greatly enhanced degradation compared to the traditional anaerobic operation, the volatile carbon that was amenable to degradation degraded more quickly, and the waste matrix structurally weakened and promoting settlement. Many other investigators also confirmed this conclusion.

Inhibiting Methanogenesis

The inhibition of methanogenesis is based on the different degradation mechanism as discussed above. Although the reduction of methane generation sacrifices the landfill operation revenue, it is beneficial for global climate and environment.

Removing Offensive Anaerobic Odors and Ammonia

The aerobic degradation can reduce offensive smelling dramatically. Stessel and Murphy (1992) reported that the aeobic lysimeters smelled clean and earthy, like compost, while the anaerobic were somewhat rank. Anaerobic odors include a wide range of compounds, such as hydrogen sulfide, volatile fatty acids, ammonia, aromatic compounds and amines. Ammonia is the most common odor that can be formed aerobically as well as anaerobically. In an aerobic environment, bacteria can oxidize the ammonia nitrogen to nitrites and nitrates.

3. MATERIALS AND METHODS

3.1 Overview

This study was designed to examine the degradation potential of high organic and moisture content MSW in bioreactor landfills. Two types of synthetic MSW were utilized in the study: one consisted of synthetic MSW mixing with biosolids at the ratio of 4:1 (wet weight); the other one only consisted of synthetic MSW. Six bioreactors were constructed to simulate bioreactor landfills accepting either MSW with biosolids or MSW without biosolids.

Each bioreactor has a volume capacity of 220L and contains approximately 150L synthetic MSW (around 70kg in weight). The MSW compositions were broken down by shredding or cutting, weighed on the scale, and then mixed up thoroughly to get uniformity before being put into the bioreactors. The bioreactors were loaded up with the synthetic MSW that then were compacted to the densities reflecting typical landfill conditions. Tap water was added nto the bioreactor landfills, and the produced leachate was pumped back until the synthetic MSW matrices reached field capacity with respect to their moisture contents.

The six simulated bioreactor landfills have operated for 102 days in the laboratory. Five of hem operated in aerobic and anaerobic stages sequentially, each being supplied 0.25 $L S^{-1}$ of air continuously during the 18-day aerobic stage. The sixth one operated in constant anaerobic stage. Leachate recirculation and buffering technologies were applied to four of the bioreactors. The daily leachate recirculation rate for each bioreactor in anaerobic stage was 12 % of the matrix volume. Sodium hydroxide solution was used as buffering solution, and he amount added was based on the pH of the generated leachate from respective bioreactor, n order to keep the optimum pH range of 6.4 to 7.2 for anaerobic degradation. In comparison to four closed-loop leachate-recycling bioreactors, there were two single pass bioreactors that equipped with aerobic and anaerobic lagoons to treat leachate (refer to Figure 3-5). Based on their respective operational protocol, matrix components, and replicate number of the bioreactors, the six bioreactors were named as AMF, AMR1, AMR2, ASF, ASR, NMR (A: air; N: no air; M: MSW; S: biosolids; F: flushing; R: recirculation). The detailed information was summarized in Table 3-1.

Table 3-1 The Identification of Laboratory Bioreactor Landfills

ID	Matrix	Operational Protocol
AMF	MSW without Biosolids	Flushing; without leachate recirculation & buffering; aerobic and anaerobic stages.
AMR1	MSW without Biosolids	Leachate recirculation and buffering; aerobic and anaerobic stages.
AMR2	MSW without Biosolids	Leachate recirculation and buffering; aerobic and anaerobic stages.
ASF	MSW with Biosolids	Flushing; without leachate recirculation & buffering; aerobic and anaerobic stages.
ASR	MSW with Biosolids	Leachate recirculation and buffering; aerobic and anaerobic stages.
NMR	MSW without Biosolids	Leachate recirculation and buffering; constant anaerobic stage.

The analyses performed in this study included physical analyses of waste matrices (percent composition, moisture content, volume, density, settlement, and internal temperature) and parameters of leachate (pH, TS, BOD_5, COD, NH_3-N, Cl^{-1}, and metals). Through these analyses, the performance benefits of air injection were ascertained and characterized. In addition, the performances of bioreactors loaded with different synthetic MSW (MSW with biosolids Vs. MSW without biosolids) and bioreactors with different operational protocols (leachate recirculation mode Vs. flushing mode) were examined and compared.

3.2 Design and Construction of the Bioreactor Systems

3.2.1 Structural Components and Configuration of the Bioreactor Systems

The structural components and configuration the bioreactor systems were shown in Figure 3-1. Following is the detailed description of the design and construction of the systems.

Bioreactors

Six semi-transparent polyethylene cylindrical containers were used as the bioreactors. Each of them has diameter of 0.55 m, height of 0.92m, and total capacity of 220 L. A lid with the suitable size was used to cover each reactor. After the apparatuses inside each bioreactor were nstalled, the wastes were put inside, and the commissioning was finished, the seam between he body of the bioreactor and the lid has been sealed by silicone sealant in order to keep air ight and water tight. There is a ¾" port on the bottom and lid of each bioreactor respectively. Two fittings are installed in the ports. The bottom one is for discharging leachate, and the lid one is the pathway for the exhaust gas. A stainless steel sink screen is put onto each bottom fitting in order to prevent clogging or MSW loss. There is a 5cm thick layer of gravel lining the bottom of each bioreactor. The MSW was put on the gravel liner directly.

Laechate Collection Tanks

Each bioreactor was connected to one leachate collection tank with a ¾" x 1 ¼" PVC adaptor and suitable vinyl tubing. The capacity of each tank is 20 L. The function of the tank is to collect the leachate discharged from the bioreactor to prevent flooding inside the bioreactor. The generated acidic leachate during the anaerobic can be discharged very quickly, and never accumulated inside the MSW. There is a ½" PVC adjustable ball valve on the sidewall of each tank for sampling and discharging leachate to the leachate recirculation reservoir.

Leachate Recirculation Reservoirs and Pumps

Six transparent polyethylene tanks were used as leachate recirculation reservoirs. Each of them has a capacity of 25 L and graduations. The reservoir served as the "leachate transfer station" where the pH and volume of the recirculated leachate can be measured, and the acidic leachate can be buffered. The submersible pumps (Laguna Statuary Pump 3#) were used for pumping the leachate and makeup water back into the bioreactors.

Leachate Distribution Apparatus

The function of the leachate distribution apparatus is to distribute the recirculated leachate evenly on the top of the MSW matix, just like raining. The apparatus included a circle with a cross inside, and it is made by connecting a ¾" PVC cross, three ¾" PVC tees, and ¾" vinyl tubing. Sixteen holes with 3mm diameter were punched on the vinyl tubing. The apparatus was connected to the leachate recirculation pipe outside the bioreactor. A ¾" PVC adaptor was used for the pipe to pass the bioreactor wall. An adjustable PVC ball valve was installed on the recirculation pipe. When the recirculation was finished every time, the valve was closed to keep air tight of the bioreactor.

Air Distribution Pipes

Three PVC pipes were installed in each bioreactor to distribute the injected air evenly into the waste matrices. One third of each pipe near the bottom of the bioreactor was perforated. The air pipes were covered by plastic insect screen to prevent MSW getting inside the pipes and clog the pipes. The three pipes were connected together by PVC elbows and tubing, and become one "entity" which also served as a support for the water distribution apparatus. The injected air reached the "entity" through one air pipe that passed the bioreactor wall by one

PVC adaptor. An adjustable PVC ball valve was installed on the air pipe to adjust the airflow rate and shut down the air during the anaerobic stage.

Air Compressor

One air compressor (HUSKY, 2.75HP, 8 Gallons, 4.0 SCFM @ 40psi) was used to supply air o the six bioreactors. There was a standby air compressor (DeVILBISS ® Tradesman Model DAC-7118, DeVILBISS, Canada).

Temperature Sensor and Thermometer

A temperature sensor was put in the middle of the MSW matrix in each bioreactor. Wires connected the sensor with the thermometer outside the bioreactor. The thermometer (EXTECH 421307 thermometer) can read out the internal temperature of the MSW matrix.

Exhaust Gas Discharge System

The exhaust Gas pipe on the lid of each bioreactor was connected together by tubing and tees, and the exhaust gases were discharged outdoors through one ¾" vinyl tubing.

3.2.2 Leachate Recirculation Mode

The leachate recirculation system includes leachate collection tank, valves, leachate recirculation reservoir, submersible pump, tubing, leachate distribution apparatus. It operated n a closed-loop mode. The pathway of the leachate is presented in Figure 3-2. The leachate movement direction is shown by arrows in the figure.

3.2.3 Air Injection and Exhaust Mode

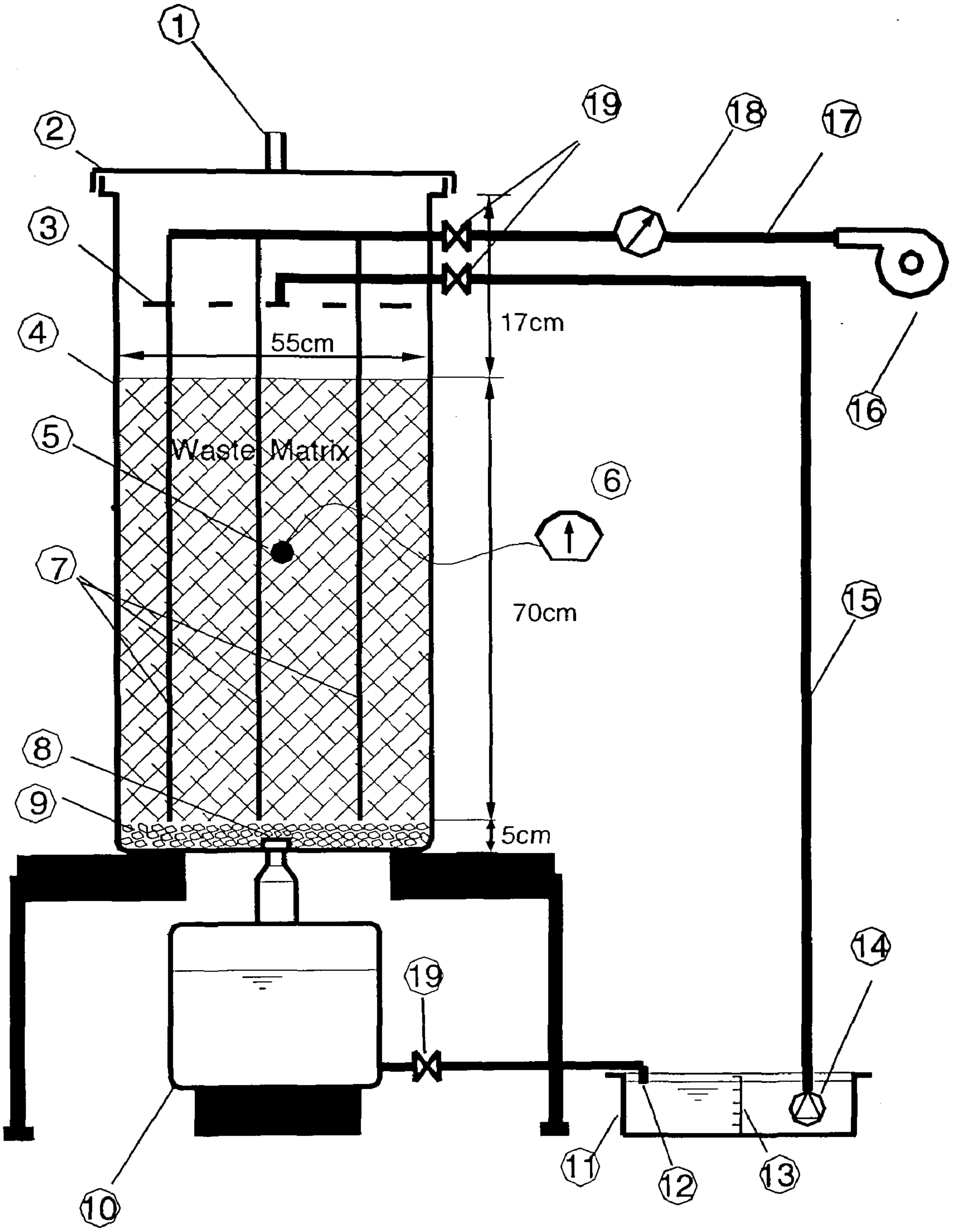
The air injection and exhaust system includes air compressor, air supply pipe, gas flow meter, valve, air distribution pipes, gas exhaust port, and gas exhaust pipe. The movement direction of the injected air is presented in Figure 3-3 as the arrows show. The connection of the air compressor and the five bioreactors is shown in Figure3-4.

3.2.4 The Operation Mode of Single Pass Bioreactors

In the single pass bioreactors, tap water was added into the waste matrix. The generated leachate was transported to the downstream treatment units: aerobic lagoon and anaerobic lagoon. The operation mode was displayed in Figure 3-5.

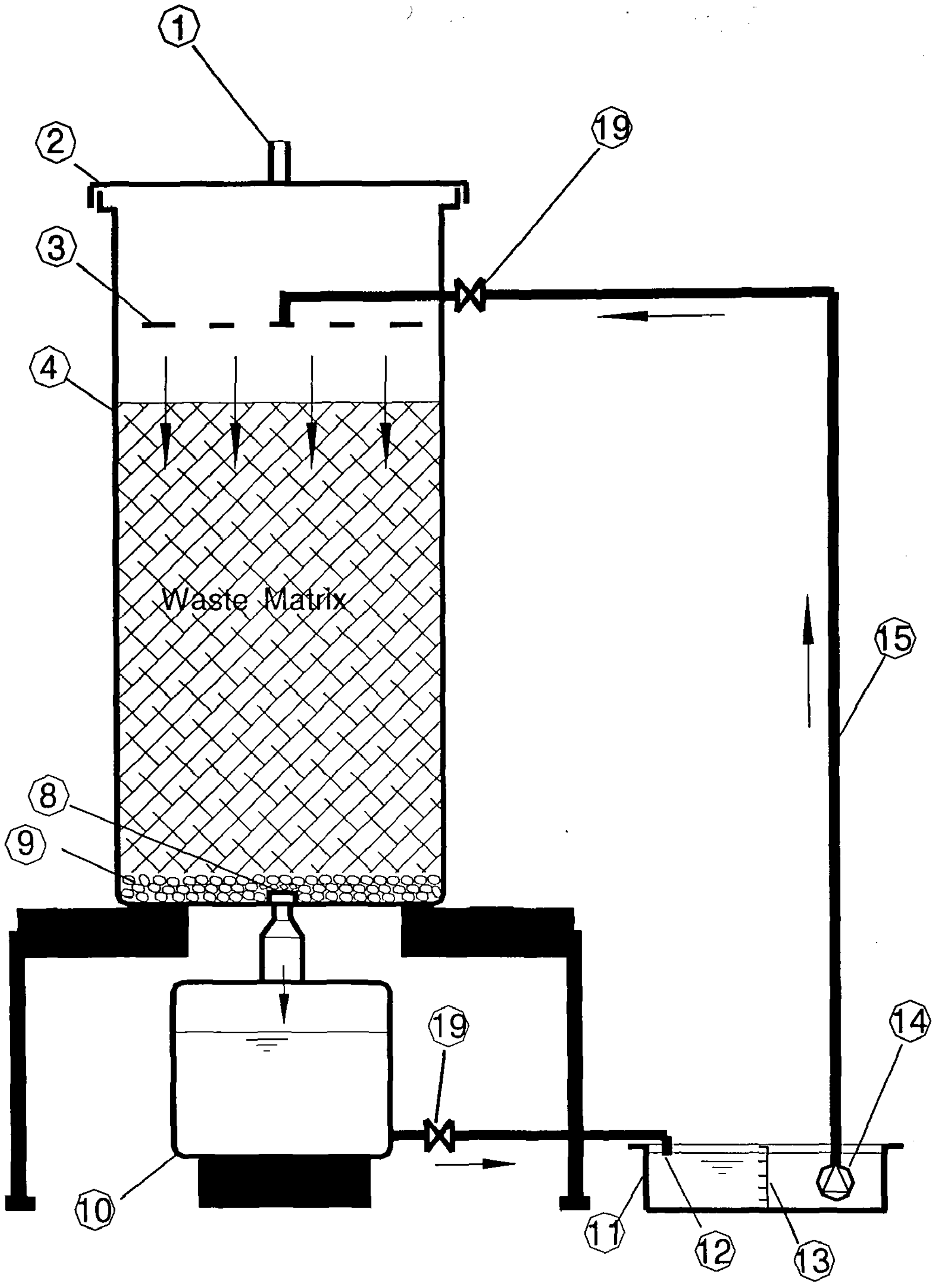
3.2.5 Bioreactor System Maintenance

The joints and seams were checked often during the study to make sure they were airtight and watertight. Spare parts, leachate tanks and reservoir, and stand-by air compressor and pump were in stock.



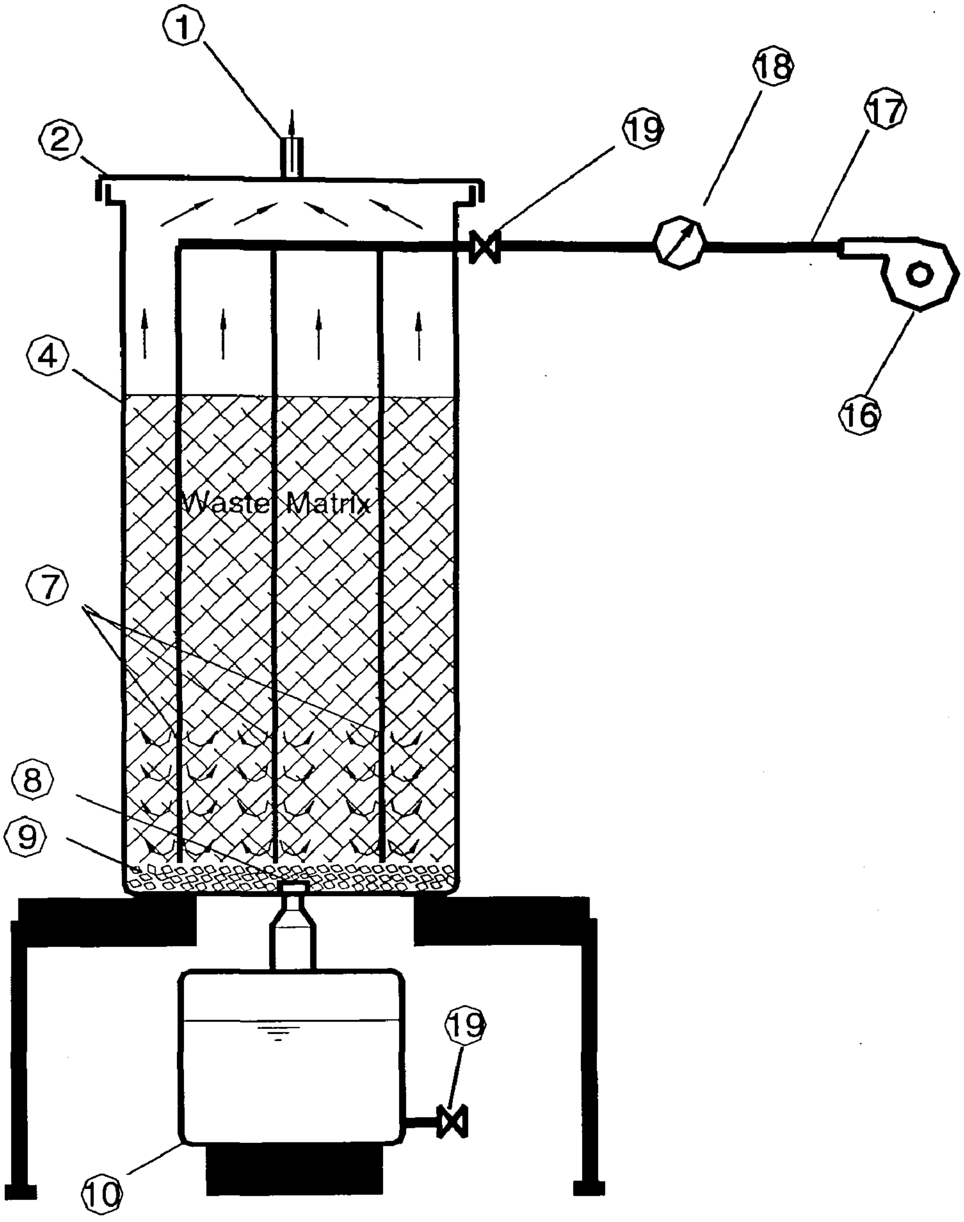
1	Gas Exhaust Port
2	Lid of Bioreactor
3	Leachate distribution Apparatus
4	Body of Bioreactor
5	Temperature Sensor
6	Thermometer
7	Air Distribution Pipes
8	Screen
9	Gravel
10	Leachate Collection Tank
11	Leachate Recirculation Reservoir
12	Leachate Sampling Port
13	Graduation
14	Leachate Recirculation Pump
15	Leachate Recirculation Pipe
16	Air Compressor
17	Air Supply Pipe
18	Gas Flow Meter
19	Ajustable Valve

Figure 3-1 Dimensions, Components, and the Configuration of the Simulated Bioreactor Landfill



1	Gas Exhaust Port	11	Leachate Recirculation Reservoir
2	Lid of Bioreactor	12	Leachate Sampling Port
3	Leachate distribution Apparatus	13	Graduation
4	Body of Bioreactor	14	Leachate Recirculation Pump
8	Screen	15	Leachate Recirculation Pipe
9	Gravel	19	Adjustable Valve
10	Leachate Collection Tank		

Figure 3-2. The Operation Mode of Leachate Recirculation System



1	Gas Exhaust Port	10	Leachate Collection Tank
2	Lid of Bioreactor	16	Air Compressor
4	Body of Bioreactor	17	Air Supply Pipe
7	Air Distribution Pipes	18	Gas Flow Meter
8	Screen	19	Adjustable Valve
9	Gravel		

Figure 3-3 The Operation Mode of Air System

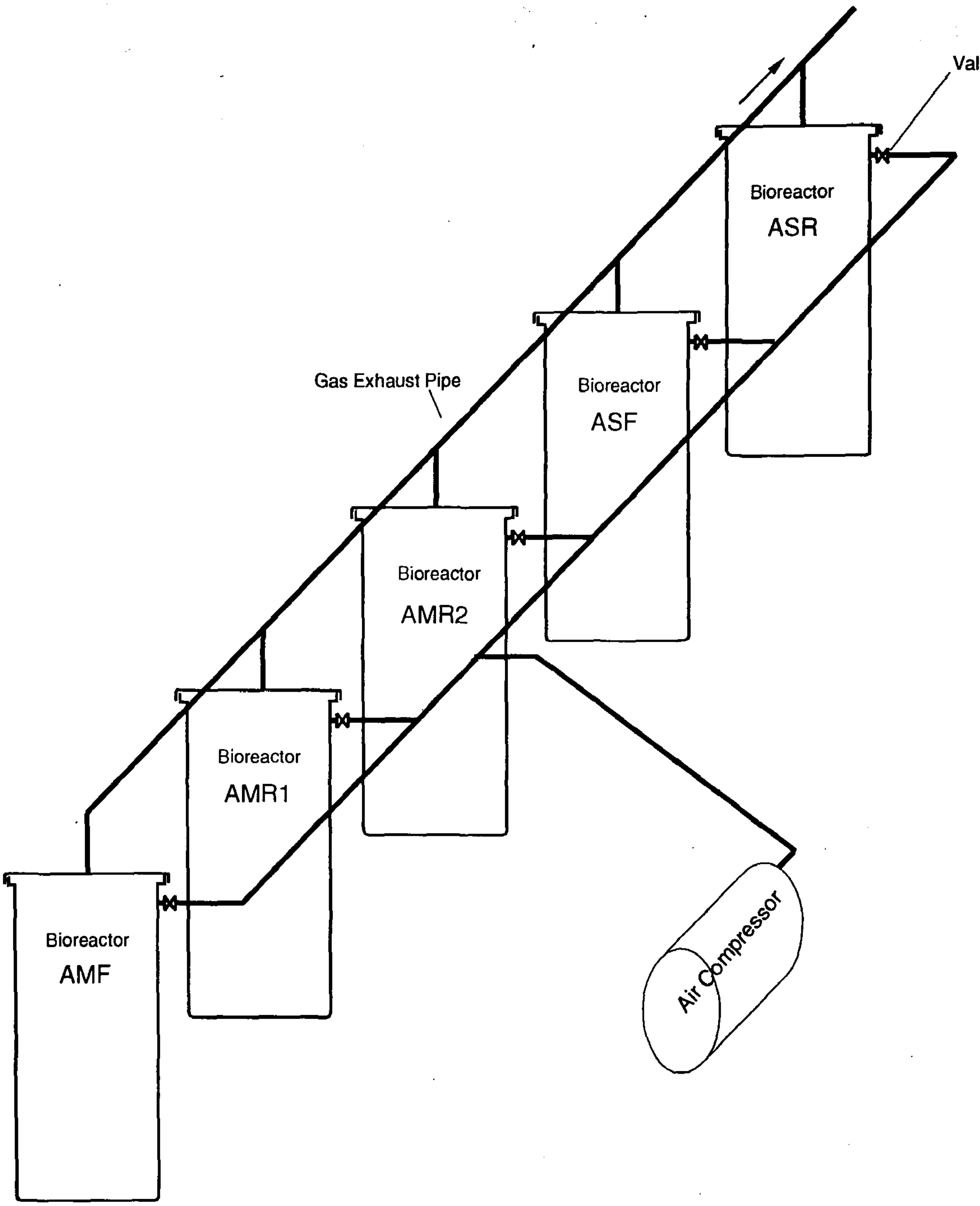
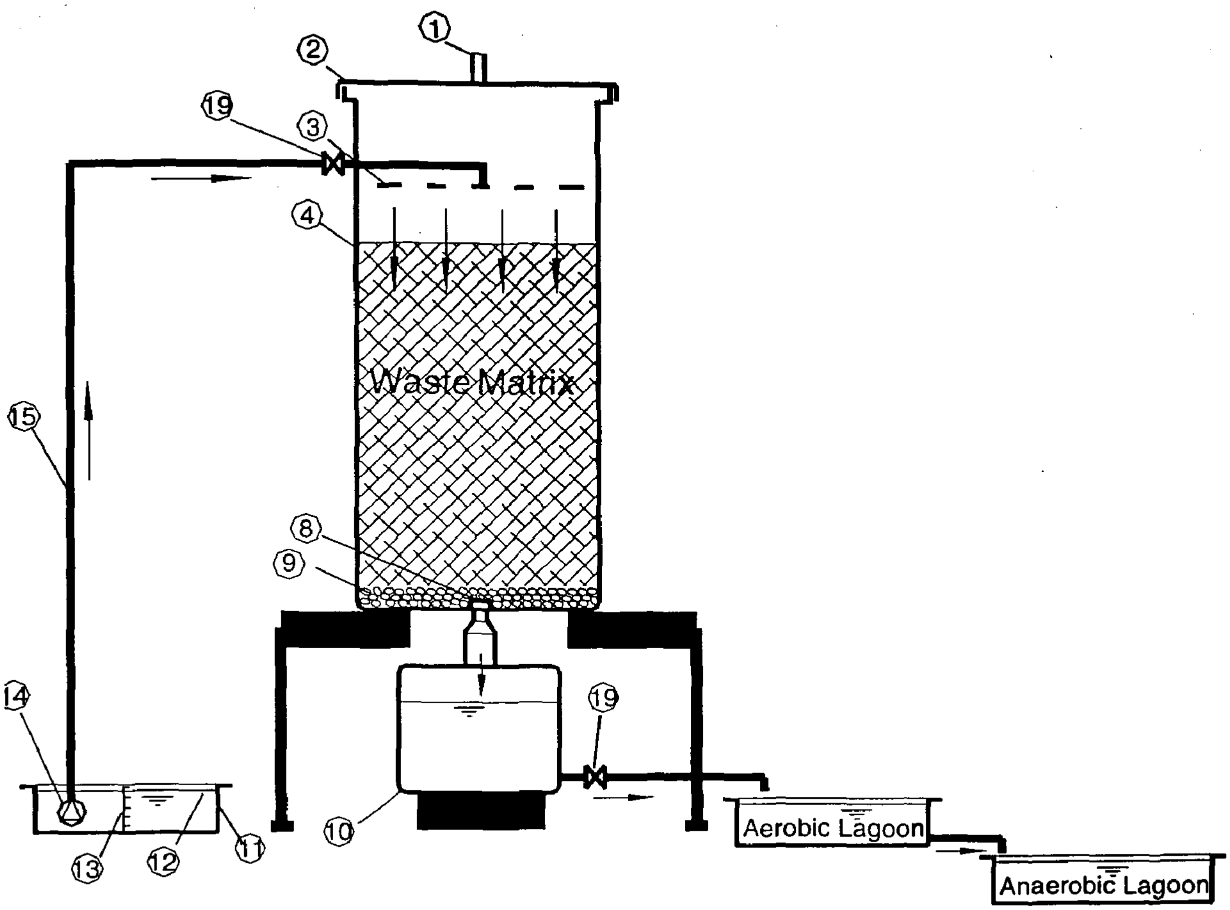


Figure 3-4 The Connection of the Air Compressor and the Five Bioreactors



1	Gas Exhaust Port	10	Leachate Collection Tank
2	Lid of Bioreactor	11	Leachate Recirculation Reservoir
3	Leachate distribution Apparatus	13	Graduation
4	Body of Bioreactor	14	Leachate Recirculation Pump
8	Screen	15	Leachate Recirculation Pipe
9	Gravel	19	Ajustable Valve

Figure 3-5 The Operation Mode of Single Pass Bioreactors

3.3 Preparation of Synthetic MSW and Loading of Bioreactors

According to the objective of the study as discussed in section 1.4, the synthetic MSW with high organic and moisture were prepared and loaded into the six bioreactors. There are two types of synthetic MSW made in this study: one with biosolids; the other one without biosolids. The bioreactors -- AMF, AMR1, AMR2, and NMR – were loaded with synthetic MSW without biosolids. The bioreactors – ASF and ASR -- were loaded with synthetic MSW with biosolids.

3.3.1 MSW Compositions

The synthetic MSW used in this study includes waste foods, waste vegetables, plastics, waste paper, textiles, and biosolids. The percentages of compositions were decided according to the purpose of this study. San and Onay (2001) used synthetic MSW according to the typical MSW Compositions in the city of Istanbul (Table 3-2). Hao (2004) used synthetic MSW according to the typical MSW Compositions in the city of Beijing (Table 3-3). The MSW compositions after recycling are presented in Table 3-4 (Source data refer to Appendix B). The biodegradable compositions were respectively 95%, 85% and 68% of the total MSW weight, belonging to high organic content MSW. Based on these data, the Synthetic MSW compositions and each component percentage in this study were designed (refer to Table 3-5).

3.3.2 Preparation of MSW

Waste foods and vegetables were taken from the nearby restaurants' kitchens. When the kitchen wastes were brought in, they were sorted, and only the waste foods and vegetables were used as the main organic components for the simulated bioreactor landfills. Waste foods nd vegetables were weighed separately.

he paper, including newsprint, computer printout, office paper, books, and packaging paper, ame from the offices at Ryerson University. All the paper was shredded.

ew garbage bags were used as plastic component in the synthetic MSW. In the field ioreactor landfills, the plastic bags channeled or blocked the recirculated leachate, as well as ncapsulated materials that would readily be biodegraded in the presence of air. Therefore, all e garbage bags in this study were cut to the size of 100-150 mm.

ld clothes were used as textile component in the study. Also, they were cut to the size of 00-150 mm.

iosolids were taken from Toronto Ashbridges Bay Treatment Plant . The biosolids were the ixture of primary sludge and secondary activated sludge that have undergone anaerobic igestion process. The properties of biosolids are presented in Appendix C.

able 3-2 Synthetic Solid Waste Compositions according to the MSW Compositions of tanbul (Adapted from San and Onay, 2001)

Composition	Percentage (%)
Food	76
Paper	12
Plastics	4
Textiles	4
Yard waste	3
Metal	1
Total	100

able 3-3 Synthetic Solid Waste Compositions according to the MSW Compositions of eijing (Adapted from Hao, 2004)

Composition	Percentage (dry weight %)
Food	55
Paper	25
Textiles	5
Plastics and rubber	15
Total	100

Table 3-4 MSW Compositions after Recycling in Toronto

Composition	Percentage (dry weight %)
Paper fibres	24
Plastics	9
Metals	3
Glass	3
Household special wastes	1
Compostables	44
Others	16
Total	100

Table 3-5 Synthetic MSW Compositions in the Study

Composition	AMF		AMR1		AMR2		ASF		ASR		NMR	
	kg	%	kg	%	kg	%	kg	%	kg	%	kg	%
Foods	29	42	31	43	30	43	26	35	26	35	30	43
Vegetables	23	33	22	32	23	32	19	26	19	26	23	32
Paper	8	12	9	12	8	12	7	10	7	10	8	12
Plastics	7	10	6	9	6	8	4	6	4	6	6	9
Textiles	3	4	3	4	3	4	2	3	2	3	3	4
Biosolids	0	0	0	0	0	0	15	20	15	20	0	0
Total Wet Weight (kg)	70	100	71	100	70	100	74	100	74	100	70	100
Initial Moisture Content (%)		58		63		63		67		67		58
Total Dry Weight (kg)	29	100	26	100	26	100	24	100	24	100	30	100

3.3.3 Loading of the Simulated Bioreactor Landfills

After all the MSW components were broken down to the suitable size, they were thoroughly mixed up and sampled for measuring the initial moisture content of the synthetic MSW. The synthetic MSW was loaded into each bioreactor and compacted. And then, about 39 L tap water was added into each bioreactor. In the first three days, the leachate was recirculated daily and the volume of the leachate was measured until the MSW in each bioreactor was brought to field capacity.

he moisture content is an essential factor that dramatically affects the waste degradation. On veight basis, moisture content is described as the weight of the water divided either by dry or vet waste weight. On a volumetric basis, moisture content is expressed as the volume of vater divided by the volume of wet waste. Generally, the field capacity is used to haracterize the moisture content of the MSW matrix. The field capacity is the concept to ualify the internal storage of a landfill, or the moisture content at which the maximum mount of water is held (through capillary forces) against gravity. The addition of more noisture will result in continuous leachate drainage (Reinhart and Townsend, 1998). When he amount of water in the leachate collection tank was approximately equal to the amount of eachate recirculated the previous day, the MSW inside the bioreactors was considered to each the field capacity.

he range of the field capacity is wide as expected since it is a function of the waste omposition, density and porosity, particle sizes, waste overburden, waste age (Yuen et al., 001; Reinhart and Townsend, 1998).

he characters of the loaded synthetic MSW in each bioreactor are summarized in the Table -6.

able 3-6 Physical Characters of Synthetic MSW

	AMF	AMR1	AMR2	ASF	ASR	NMR
nitial Volume (L)	150	157	160	152	152	173
nitial Height (m)	0.63	0.66	0.67	0.64	0.64	0.73
nitial Field Capacity (%)	65.1	69.6	70.4	70.6	69.9	67.0
nitial Density (kg m^{-3})	563	546	541	545	532	519
inal Density (kg m^{-3})	608	623	574	564	559	667

lote: "initial" means at the beginning of the study, "final" means at the end of the study.

.4 Operation and Sampling Protocol of the Simulated Bioreactor Landfills

uring the 102-day study, all the six bioreactors operated according to the proposed protocol based on the purpose of this study. All parameters of the operation were selected from the previous investigators' studies in the literature. Waste samples were collected before and after the operation of the bioreactors. Leachate samples were collected following a timeline.

3.4.1 Rate of Airflow

As described above, five bioreactors--AMF, AMR1, AMR2, ASF, and ASR—operated in two stages. In the 18-day aerobic stage, air was injected into the MSW matrices continuously to keep an aerobic condition for accelerating the organic waste decomposition. Stessel and Murphy (1992) indicated that there would be no maximum level of airflow; rather, one sought the lowest level at which accelerated degradation was obtained so as to minimize operating costs.

Haug (1993) provided the general formula – $C_{64}H_{104}O_{37}N$ – for the total organic fraction of MSW. Based on the formula, assume the organics are totally degraded, then Eq. [2] can be expressed as:

$$[3] \qquad \underset{(1478)}{C_{64}H_{104}O_{37}N} + \underset{(2264)}{70.75\ O_2} \longrightarrow 64\ CO_2 + 50.5\ H_2O + NH_3$$

The stoichiometric oxygen demand can be determined as 1.53 g O_2 g^{-1} waste (dry). Taking 2 as a safety factor, and converting to air volume 11.1 m^3 air kg^{-1} MSW (dry) (25 ^0C and at 1 Atm pressure) should be supplied. Assume all the organic portion was oxidized in the 18-day aerobic stage, 0.617 m^3 air d^{-1} kg^{-1} MSW (dry) needs to be supplied. Anderson (1990) also presented that a suitable aeration rate is considered by many operators to be between 0.6 to 1.8 m^3 air d^{-1} kg^{-1} volatile solids during the thermophilic stage with progressive decrease during the cooling down and maturation stages for composting.

hus, based on the dry weight of the wastes in each bioreactor (refer to Table 3-4), about 0.25 S^{-1} of air should be supplied to each bioreactor continuously. Five bioreactors operated at ıe same time, thus, 1.25 L S^{-1} of air should be supplied by the air compressor.

.4.2 Leachate Recirculation Rate

/ater is essential for both aerobic decomposition and anaerobic decomposition. As discussed ı section 2.1.3, moisture content and movement are separate variables. The function of :achate recirculation is to achieve optimum moisture content and movement inside waste ıatrices. The optimum leachate recirculation rate is different for aerobic and anaerobic egradation. The recirculation rate of this study was determined according to the previous :searchers' studies.

ı Hudgins and Harper's study (1999), they kept waste mass moisture contents above 60% in wo aerobic landfills. They also indicated that waste temperatures increased while moisture :vels decreased, in some cases to below 40%; as such, leachate flow and air delivery rates hould be adjusted based on the field data to keep the waste mass adequately moisturized and erated. Stessel and Murphy (1992) concluded that waste moisture levels had to be ıaintained at 75% for the optimum aerobic degradation. The field capacities in this study are ı the range of 60% - 70%. To keep the field capacity in the bioreactors, leachate was ecirculated daily. At the beginning of the study, the volume of leachate recirculated is pproximately equal to 15% of the waste matrix volume in one bioreactor. The volume of eachate generated decreased with time in the aerobic stage because of evaporation and legradation consumption. When the volume of leachate decreased to below 5 L approximately 4% of the waste matrix in one bioreactor), tap water was added to keep the minimum volume – 5 L.

Many researchers conducted studies on the effects of different leachate recirculation rates on anaerobic degradation. Chugh et al (1998) utilized three daily recirculation rates – 2%, 10%, 30% of initial volume of waste matrices. They concluded that the rate and extent of waste decomposition improved with the increase in moisture flow. However, the results of Sponza and Agdag's study (2004) contrasted the previous study. Their recirculation rates are 9 L d^{-1} (13% of the reactor volume) and 21 L d^{-1} (30% of the reactor volume) respectively. After 220 days of anaerobic incubation, they observed that the pH, COD, VFAs concentrations, methane gas productions and methane percentages in 9 L d^{-1} bioreactor were better than in 21 L d^{-1} bioreactor. They indicated that high recirculation volumes might deplete the buffering capacity and remove the activity of methanogens. Warith (2002) used 15% of the total volume of solid waste in each waste cell, and recirculated three times a week. Based on these previous studies, daily recirculation rate was determined as 12 % of the total waste matrix volume in each bioreactor for the anaerobic stage in this study.

3.4.3 Buffering Amount

Sodium Hydroxide solution was utilized as buffer solution. As discussed insection 2.1.2, the optimal pH for refuse methanogenesis is 6.4 to 7.2. At each time, the amount of buffering solution added into the leachate was determined according to the pH value of the leachate in order to keep it in the range of 6.4 to 7.2.

3.4.4 Operation and Analysis Timeline

The operation and analysis timeline of the bioreactors are summarized in Table 3-7. The duration of the aerobic stage was designed according to the literature and the economic onsideration. The duration of the anaerobic stage was determined according to the bservation of the experimental progress. When the parameters of the leachate kept constant ecrease over two weeks, the experiment was terminated.

able 3-7 The Operation and Analysis Timeline of Bioreactors

Date	03/09/04	06/09/04	23/09/04	16/12/04	17/12/04
Day Number	0	1	18	102	103
Aerobic Stage [1]		■	■		
Anaerobic Stage [1]				■	
Constant Anaeobic Stage [2]		■	■	■	
Physical MSW Analysis	Untreated MSW				Treated MSW
Leachate Recirculation		■	■	■	
Leachate Sampling		■	■	■	
Settlement Measurement		■	■	■	

ote: [1] represents bioreactors AMF, AMR1, AMR2, ASF, ASR; [2] represents NMR.

.5 Analytical Methods

.5.1 Methods for the Physical Analyses of MSW

he physical analyses of MSW were performed in this study in order to characterize the ature of the waste samples and get the bulk data of their decomposition as a whole. The arameters measured in the study consisted of percent composition, moisture content, volume, ensity, settlement, and internal temperature.

ercent Composition by Mass

he overall concept of MSW suitability for biodegradation can be obtained from its percent ompositions. Synthetic MSW was used in the study. The measurement of individual MSW omponent was carried out with reference to ASTM method D5321-92 (ASTM 2002). Scale vas used for the measurement. Each composition was weighed to the proposed weight for each bioreactor.

Moisture Content and Field Capacity

Moisture contents of matrices inside bioreactors were determined by ASTM Method 2216-98 (ASTM 2001). Approximately 1000g of specimen from each bioreactor was used for the measurement in order to avoid the interference of MSW heterogeneity. The initial moisture content was measured right after the MSW components were prepared and mixed up. The initial field capacity can be derived from the initial moisture content of MSW and the amount of tap water added into the MSW matrices for saturating the MSW. The moisture content was measured again at the end of the study to get the final field capacity for comparison.

Settlement of the MSW Over Time

After the MSW is disposed of in landfills, the thickness of waste layers will decrease with time. The Waste settlement analysis is very important because it can influence: (1) making projections of the remaining site life or remaining time before operations need to move to a new lined area; (2) the design of landfills' components, such as cover and liner systems; (3) post-closure development of landfills. The rate of landfill settlement depends primarily on the waste composition, operational practices and factors affecting biodegradation of the landfill waste, particularly moisture content (El-Fadel, 1998).

In this study, semi-transparent polyethylene containers were used as bioreactors, thus, the thickness of waste layers were easily measured outside the bioreactors. The settlement rates were expressed as the percent data of the decreased thickness divided by the initial waste thickness.

The volume and wet density of the waste were also easily derived from the measurement of he waste matrix thickness and mass as above.

nternal Temperature

Mercury thermometers were used to record the ambient air temperature. Temperature sensors and an EXTECH 421307 thermometer were used for monitoring the internal temperature to ensure that the MSW did not heat up to a dangerous level.

3.5.2 Methods for the Chemical Analysis of Leachate

Leachate samples of 150 ml from each bioreactor were collected throughout the study: three imes per week in the first 32 days; two times per week from day 32 to day 53; once a week from day 53 to the last day. The leachate parameters -- pH, TSS, TS, BOD_5, COD, NH_3-N, Cl^{-1}, and metals – were measured as:

pH

pH meter was used to measure the pH values of leachate in the study. Because leachate pH values changed along the time, it is very important to carry out the measurement right after sampling. Commercial standard solutions (including pH 4.0, 7.0, and 11.0) were used for calibrating the pH meter.

Total Suspended Solids (TSS) and Total Solids (TS)

The measurement of Total Suspended Solids (TSS) followed standard method 2540D (APHA et al., 1998). 1.5 um glass fiber filter and 10ml sample size were selected for this study. The measurement of Total Solids (TS) followed standard method 2540B (APHA et al., 1998). A sample size of 10ml was selected for this study. The apparatuses used for these two parameters included porcelain dishes, aluminum weighing dishes, Fisher Scientific Isotemp Oven (Model 630G, Fisher Scientific ltd.), OHAUS ® Precision Standard balance (GENEQ

Inc.), desiccators, vacuum pump, filtration apparatus.

Chemical Oxygen Demand (COD)

The COD of leachate was analyzed using standard method 5220D (APHA et al., 1998). A SPECTRONIC 20D spectrophotometer was used for the measurement, operating at 600nm. Each sample was analyzed in duplicate.

5-day Biochemical Oxygen Demand (BOD_5)

BOD_5 test followed standard method 5210B (APHA et al., 1998). In this study, samples were incubated in a Fisher Scientific Isotemp Incubator (Model 637D, Fisher Scientific Ltd.), and dissolved oxygen was measured by a YSI model 51B DO Meter (YSI Inc.). After each leachate sample was measured COD value, the BOD_5 value can be predicted, and then two dilution factors were selected for each sample according to Tchobanoglous (2003). Polyseed was used as seed source, each 300 ml BOD bottle was added 5 ml prepared polyseed solution.

Ammonia Nitrogen (NH_3-N)

Standard method 4500-NH_3 F (APHA et al., 1998) was used for NH_3-N measurement. A SPECTRONIC 20D spectrophotometer was used for the measurement, operating at 640nm. Each sample was analyzed in duplicate.

Chloride (Cl^{-1})

Standard method 4500- Cl^{-1} (APHA et al., 1998) was used for Cl^{-1} measurement.

Metals

The concentrations of two kinds of metals, Copper (Cu) and Zinc (Zn), were measured five times along the study: at the beginning of the study, at the end of the aerobic stage, at the eginning of the anaerobic stage, the pH lowest point in the anaerobic stage, at the end of the udy. The measurements were performed according to Standard method 3111C (APHA et al., 998). Atomic Absorption Spectrophotometer (AANALYST 800, PERKW ELMER) was sed.

.5.3 Statistical Analysis

ll the data obtained were calculated, analyzed, and plotted trend lines with Microsoft ® xcel ™ 2002.

4. EXPERIMENTAL RESULTS AND DISCUSSIONS

In this study, there are three comparison groups: (1) AMR Vs. NMR; (2) AMR Vs. ASR; and (3) AMR Vs. AMF and ASR Vs. ASF (the bioreactors have been identified in Table 3-1). The following results and discussions are divided into three parts, and each is related to one specific group. The physical variations of waste matrices and chemical variations of leachate in the bioreactors are presented in this section. Through the analyses and the comparison of the experimental data, the effects of air addition and biosolids addition on the biodegradation of MSW with high organic and moisture contents in bioreactor landfills are discussed. Moreover, the flushing technology is compared with the leachate recirculation technology. In the following sections, AMR represents the average values of the data from replicate bioreactors AMR1 and AMR2, and the complete data are presented in Appendix B.

4.1 Experimental Results and Discussions – AMR Vs. NMR

In order to exam the benefits of initially biodegrading MSW with high organic and moisture contents under aerobic condition before anaerobic degradation comparing with constant anaerobic degradation, the performance of bioreactor AMR is compared with that of bioreactor NMR in this section. Both bioreactors were loaded with synthetic MSW, operating with leachate recirculation and leachate buffering technologies. The only difference between them was that AMR operated in two sequential stages (aerobic stage and anaerobic stage) and NMR operated under constant anaerobic condition (refer to Tables 3-7).

4.1.1 Physical Variations – AMR Vs. NMR

Visible Changes in MSW Composition and Odors after Bioreactor Treatment

Most food and vegetable wastes were decomposed after approximately 100 days in bioreactor

AMR. There were more food and vegetable residuals in NMR than in AMR. There was no obvious degradation of the newsprints in NMR while the obvious volume reduction and biodegradation of the newsprints were noted in AMR. Textiles experienced no changes in both bioreactors. At the end of the study, the vinegar and alcohol odors were very strong in both bioreactors.

Changes in Mass, Settlement, Density, and Field Capacity

Due to the biodegradation of the MSW, both AMR and NMR experienced waste weight eduction after about 100 days bioreactor treatment. In bioreactor AMR, the wet waste weight educed from 86.2 kg to 73.8 kg and the reduction rate was 14.4%. In bioreactor NMR, the wet waste weight reduced from 89.9 kg to 87.1 kg and the reduction rate was 3.1%. The wet weight reduction rate in AMR was about five times of that in NMR.

As shown in Eqs [2] and [3], organic materials can beconverted to water, carbon dioxide and ammonia in the aerobic environment. The produced water can be vaporized due the heat generated in the aerobic reactions. As discussed in section 2.1.1, in anaerobic condition, the organic wastes, such as cellulose and hemicellulose, protein, and fats, can be converted to gases, such as methane, carbon dioxide, ammonia, and hydrogen, and VFAs. The generation of gases and volatile products can cause the loss of the waste weight in the bioreactors. In other words, the waste weight reduction rate reflected the biodegradation rate in the bioreactors. AMR achieved about 5 times wet weight reduction rate of NMR. This indicated he biodegradation rate in the aerobic-anaerobic bioreactor AMR was faster than that in the constant anaerobic bioreactor NMR.

The settlement rates in AMR and NMR with time are illustrated in Figure 4-1 and the complete data are presented in Appendix B. The overall settlement rates for AMR and NMR were 22% and 25%, respectively. Although the weight reduction rate in NMR was only one fifth of that in AMR, the settlement rate was higher in NMR than in AMR. The reason may be the lower initial waste density in NMR (refer to Table 4-1).

Although both the waste volume and waste weight decreased with time, densities in both bioreactors experienced increases due to a greater reduction in waste volume proportional to the waste weight (refer to Table 4-1). There was little change in field capacity in both bioreactors.

Table 4-1 Changes in Mass, Density, Field Capacity, and Settlement Rates in AMR and NMR

	AMR	NMR
Initial Wet Waste Weight (kg)	86.2	89.9
Final Wet Waste weight (kg)	73.8	87.1
Wet Weight Reduction (kg)	-12.4	-2.8
Wet Weight reduction Rate (%)	-14.4%	-3.1%
Initial Dry Waste Weight (kg)	25.8	29.7
Final Dry Waste Weight (kg)	21.9	28.6
Dry Weight Reduction (kg)	-3.9	-1.1
Dry Weight reduction Rate (%)	-15.1%	-3.7%
Initial Density (kg m^{-3})	543.8	518.6
Final Density (kg m^{-3})	598.5	666.6
Initial Field Capacity (W/W)	70.0%	67.0%
Final Field Capacity (W/W)	70.3%	67.1%
Final Settlement Rate (%)	22%	25%

Note: "Initial" means at the beginning of the study; "Final" means at the end of the study; Field capacity = the weight of water inside the waste divided by the total wet weight of the waste inside each bioreactor; Settlement rate = thickness reduction of the waste matrix divided by the initial thickness of the waste matrix inside each bioreactor.

4.1.2 The Variations of Leachate Chemical Parameters – AMR Vs. NMR

Variations in pH

The leachate pH variation trends in bioreactors AMR and NMR are displayed in Figure 4-2.

The buffer amount and buffering timeline are presented in Table 4-2. In about 100 days study period, both bioreactors utilized about the same amount of buffer. The leachate pH values ncreased from initially 4.40 to finally 7.0 in AMR while increased from initially 4.1 to finally 5.8 in NMR. As illustrated in Figure 4-2, the pH values in AMR were higher than in NMR at every sampling point in the study period. In the initial 18 days bioreactor operation aerobic stage in AMR), NMR utilized more than 10 times amount of buffer than AMR. Although the leachate pH values increased rapidly in both bioreactors and reached 5.5 and 6.3 for NMR and AMR, respectively, the final pH value in AMR was higher than in NMR. The neutralizing effect of air was obvious. From day 19, AMR was switched to anaerobic stage. From day 19 to day 28, there were no air addition and buffering in AMR. The pH values start to drop from 6.3 to 5.7 in AMR.

n order to keep the methanogenesis favorite pH range of 6.4 – 7.2 in the anaerobic stage in AMR, buffering technology was applied. When the leachate pH value dropped below 6.4, buffer solution was added to keep the leachate pH values between 6.4 and 7.2. After day 88, here were no pH values below 6.4, so the buffering addition was terminated. NMR followed AMR in the added buffer amount and buffering termination time from day 19 to the end of he study. However, after buffering was terminated on day 88 in NMR, its pH values slightly decreased from 6.2 to 5.8.

The anaerobically biodegradation of organics generated a large quantity of VFAs (refer to ection 2.1.1). The accumulation of VFAs can cause low pH values in bioreactors, as well as everse the anaerobic reactions in Table 2-1. As observed in the study, in AMR, after the aerobic stage was terminated and there was no buffering from day 19 to day 28, the pH values dropped to the acidic side, as well as in NMR, after the buffering was stopped on day 88, the pH values decreased quickly. The added buffer neutralized the produced VFAs and accelerated the reactions to occur in the direction as shown in Table 2-1. As a result, the anaerobic degradation was accelerated. Therefore, the buffering addition is an effective strategy in the operation of the anaerobic bioreactors.

The observations in this study also indicated that air has neutralizing effects on pH. The air brought the leachate pH value in AMR to a higher level than in NMR in the initial 18-day period although AMR only used less than one tenth buffer amount in this period. As Equations [2] and [3] presented, the aerobic biodegradation avoided the generation of VFAs comparing with the anaerobic degradation, thus, avoid the drop of pH.

After day 88, no more buffer was needed for AMR. The VFAs generation and consumption have achieved a balance. The ecosystem inside the AMR waste matrix maybe entered methanogenesis predominance phase -- *Phase IV* (refer to section 2.1.1). In contrast, after the buffering was stopped in NMR, the pH values dropped in the following study period. This result indicated that the production of VFAs exceeded the consumption of VFAs in NMR, and the anaerobic biodegradation progress in NMR lagged behind that in AMR.

The observations led to the conclusion that initially degrading high organic and moisture content MSW in aerobic stage created an optimum pH environment for the following anaerobic stage and saved the buffer amount, at the same time, accelerated the decomposition of the waste.

Table 4-2 Buffer Amounts and Buffering Timeline in AMR and NMR

Day Number		1-5	6	7-27	28-88	88-102	Total (g)
AMR	Buffer Amount (g)	0	20	0	707.6	0	727.6
	Time line		■		■		

Day Number		1-8	9-28	29-88	89-102	Total (g)
NMR	Buffer Amount (g)	256	0	475	0	731
	Timeline	■		■		

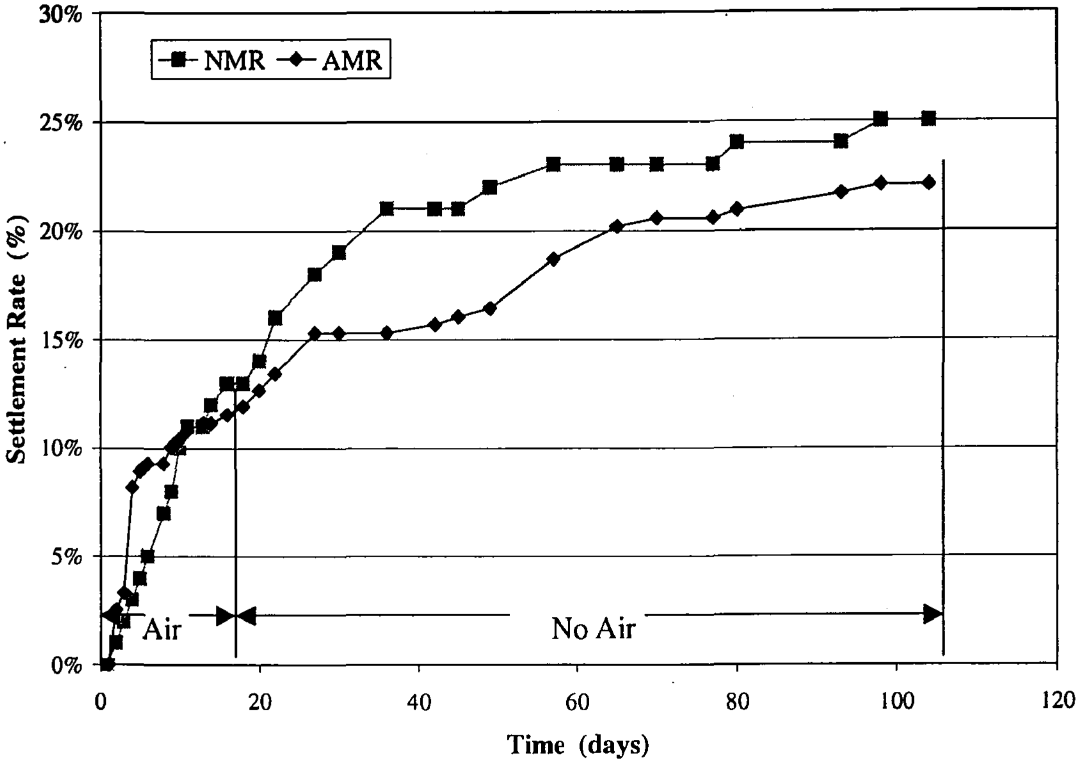


Figure 4-1 Waste Settlement Rates in AMR and NMR

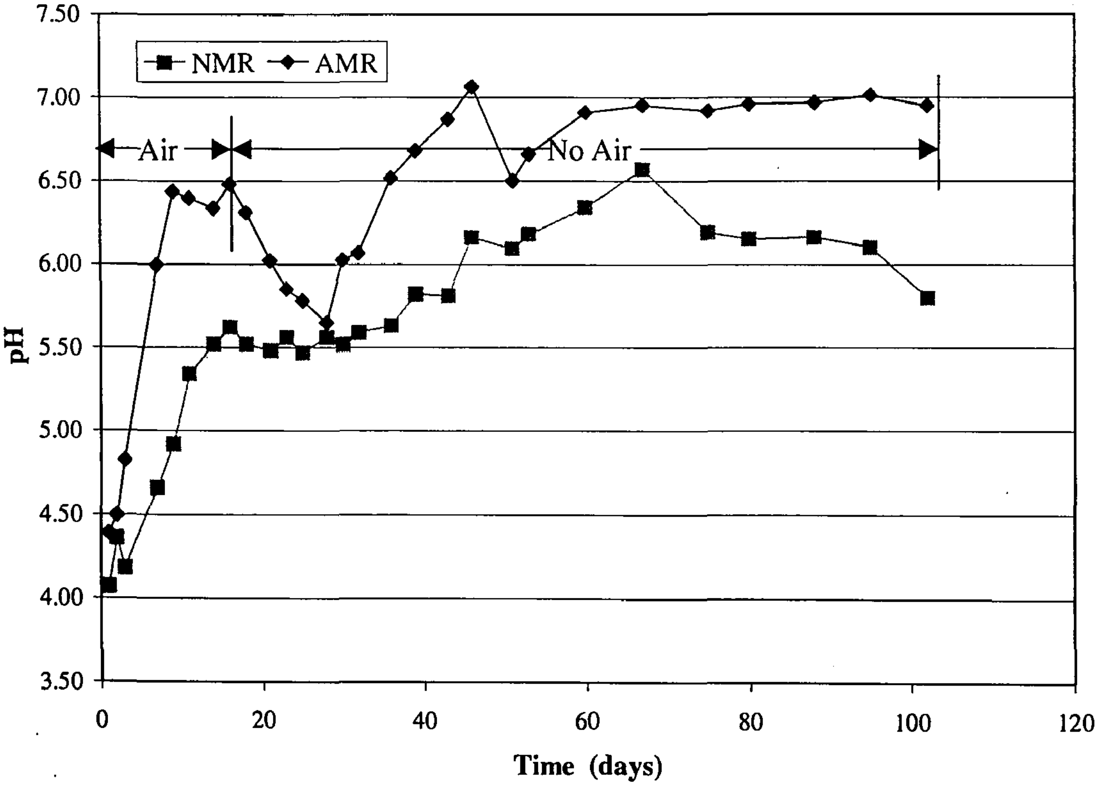


Figure 4-2 Variations in pH in AMR and NMR

Variations in COD

Figure 4-3 displays the COD variation trends in bioreactors AMR and NMR and Table 4-3 summarizes the representative COD concentrations. The initial COD concentrations in both bioreactors were approximately same (37,600 and 33,100 mg/l for AMR and NMR, respectively). The peak COD in NMR (62,300 mg/l) was higher than the peak COD in AMR (45,100 mg/l) by 38.1% of the AMR peak COD. At the end of the study, the final COD in NMR (44,200 mg/l) was higher than the final COD in AMR (31,700 mg/l) by 39.1% of the AMR final COD. Furthermore, the final COD in NMR was approximately the same as the peak COD in AMR.

n the first 18 days of the study, AMR operated in the aerobic stage. The COD concentration decreased sharply in AMR and reached the bottom concentration – 15,800 mg/l on day 18. At he same time, NMR operated in anaerobic condition. The leachate COD in NMR remained high concentration (59,500 mg/l) on day 18, when the aerobic stage terminated in AMR. The COD concentration reflected the leachate quality in the bioreactors. The above observations indicated that the leachate quality in AMR was much better than that in NMR after about 100 days bioreactor treatment. The other operation protocols were same for both bioreactors except AMR utilized air addition in the initial 18 days operation. Therefore, it was concluded that initially degrading MSW with high organic and moisture contents under aerobic condition improved the final leachate quality comparing with the constant anaerobic degrading in the study.

Table 4-3 Representative COD Concentrations in AMR and NMR and According Reduction Rates

	AMR	NMR
Initial COD (mg/l)	37,600	33,100
Fist Peak COD (mg/l)	45,100	62,300
Aerobic Stage End COD (mg/l)	15,800	59,500 [1]
COD Reduction Rate in Aerobic Stage [2]	65.0%	N.A.
Final COD in the study (mg/l)	31,700	44,200
Overall COD Reduction Rate in the Study [3]	29.7%	29.1%

[1]: COD concentration on day 18; [2]: the reduction rate is calculated based on the first peak concentration and the aerobic stage end concentration; [3]: the reduction rate is calculated based on the first peak concentration and the final concentration in the study.

Variations in BOD_5

Although there were larger fluctuations than COD curves, the BOD_5 variations followed the similar trends in AMR and NMR (refer to Figure 4-4). The representative BOD_5 concentrations were presented in Table 4-4. The initial BOD_5 concentrations in both bioreactors were approximately equal (18,800 and 19,400 mg/l for AMR and NMR, respectively). The peak BOD_5 in AMR and NMR were 23,500 and 30,800 mg/l, respectively, and the peak BOD_5 in NMR was higher than the peak BOD_5 in AMR by 31% of the peak BOD_5 in AMR. At the end of the study, the BOD_5 in NMR was almost double the BOD_5 in AMR, and were 21,200 and 10,700 mg/l, respectively. The overall reduction rates from the peak BOD_5 were 54.5% and 31.3%, respectively.

The BOD_5 data further confirmed that the final leachate quality in AMR was much better that that in NMR at the end of the study, and the initially degrading the MSW with high organic and moisture contents before the anaerobic degradation is an effective strategy for bioreactor landfills which accept this kind of MSW.

Table 4-4 Representative BOD_5 Concentrations in AMR and NMR and According Reduction Rates

	AMR	**NMR**
Initial BOD_5 (mg/l)	18,800	19,400
Fist Peak BOD_5 (mg/l)	23,500	30,800
Aerobic Stage End BOD_5 (mg/l)	6,100	30,800 [1]
BOD_5 Reduction Rate in Aerobic Stage [2]	74.0%	N.A.
Final BOD_5 at the end of the study (mg/l) [3]	10,700	21,200
Overall BOD_5 Reduction in the Study	54.5%	31.2%

[1]: BOD_5 concentration on day 18; [2]: the reduction rate is calculated based on the first peak concentration and the aerobic stage end concentration; [3]: the reduction rate is calculated based on the first peak concentration and the final concentration in the study.

Variations in BOD_5/COD Ratio

The BOD_5/COD ratio followed a decrease trend in both bioreactors AMR and NMR (refer to Table 4-5). In AMR, it decreased from initially 0.5 to finally 0.34, while, in NMR, it decreased from initially 0.58 to finally 0.48. The decrease in the BOD_5/COD ratio indicated he decrease in the percentage of the biodegradable compounds in the leachate.

Table 4-5 Representative BOD_5/COD Ratios in AMR and NMR

	AMR	NMR
Initial BOD_5/COD ratio	0.50	0.58
BOD_5/COD ratio at the end of aerobic stage	0.39	0.52
BOD_5/COD ratio at the end of the study	0.34	0.48

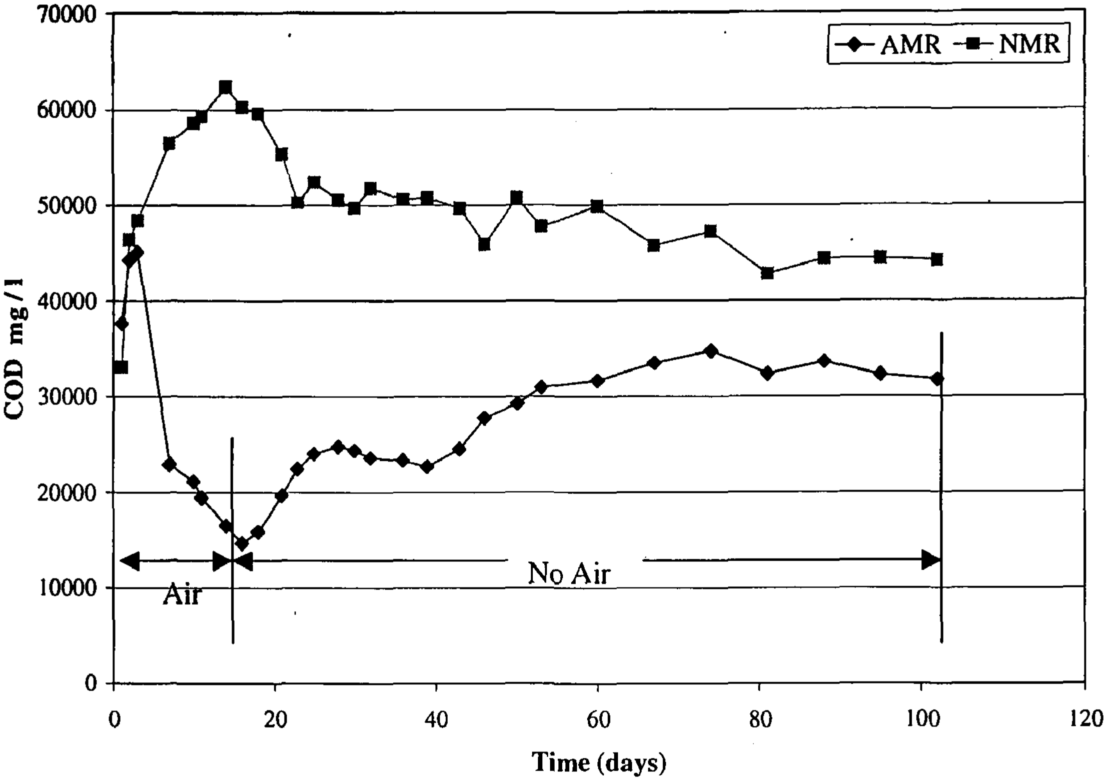


Figure 4-3 Variations in COD in AMR and NMR

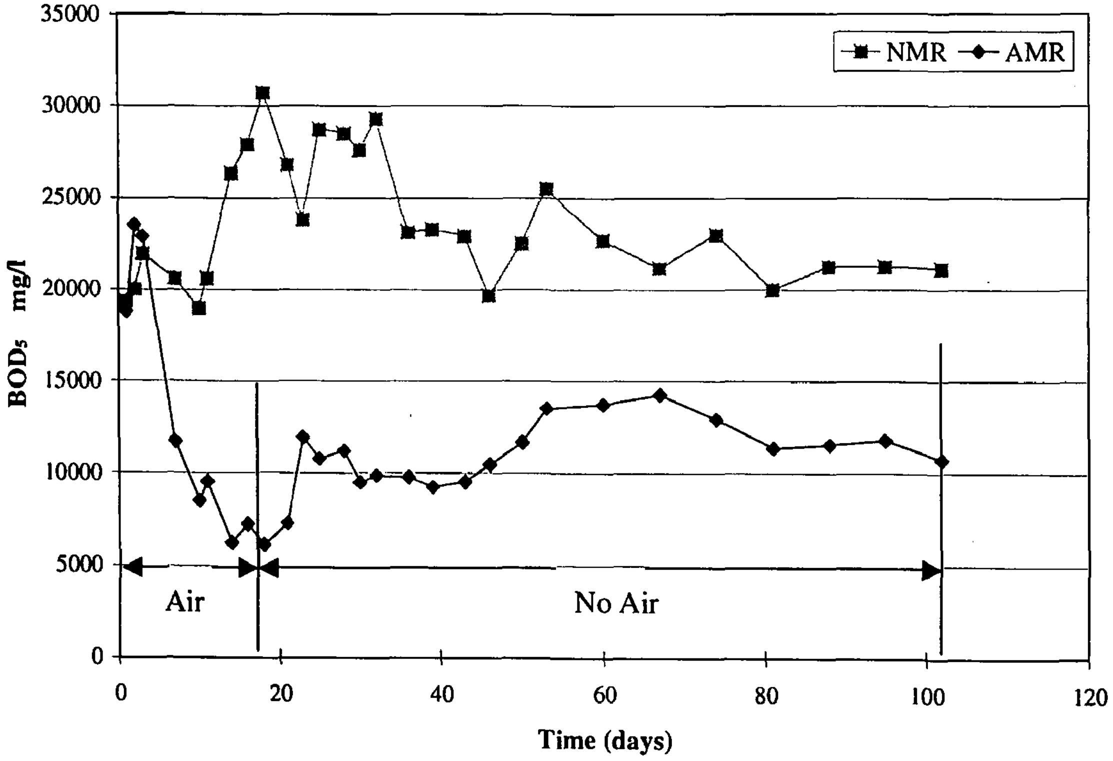


Figure 4-4 Variations in BOD_5 in AMR and NMR

ariations in NH_3-N

igure 4-5 displays the NH_3-N variation trends in both bioreactors AMR and NMR. Both ioreactors were loaded with the same kind of synthetic MSW and operated in the losed-loop leachate recirculation mode. In both bioreactors, the NH_3-N concentration ncreased progressively – from 24 to 550 mg/l in AMR while from 56 to 1300 mg/l in NMR. he initial and final NH_3-N concentrations in NMR were both double those in AMR.)bviously, the NH_3-N values in NMR were on a much higher level than in AMR. This result nay be attribute to the air addition in AMR because NH_3-N can be utilized as a substrate for erobic bacteria, and be converted to nitrite through nitrification. This result may also be ttributed to the inherent heterogeneity of the MSW. The NH_3-N was found at high oncentrations in leachate as a result of decomposition of the organic materials containing itrogen such as protein and amino acids. Although the overall weight of the waste foods and egetables inside each bioreactor was approximately same, the percentage of the nitrogenous rganic materials may be different.

ariations in TS

igure 4-6 shows the leachate TS variation trends in bioreactors AMR and NMR. The ariation trends were roughly the same as the COD and BOD_5 variation trends. The initial OD in AMR was approximately equal to the initial COD in NMR, as well as the initial OD_5 in AMR was approximately equal to the initial BOD_5 in NMR. However, the initial TS oncentration in NMR was even double that in AMR (17,250 and 32,500 mg/l for AMR and MR, respectively). This indicated that the leachate in NMR might contain more ndegradable solids. Table 4-6 summarized the representative TS concentrations in bioreactors AMR and NMR.

Table 4-6 Representative TS Concentrations in AMR and NMR and According Reduction Rates

	AMR	NMR
Initial TS (mg/l)	17,250	32,500
First Peak TS (mg/l)	44,650	60,300
Aerobic Stage End TS (mg/l)	13,050	32,700 [1]
TS Reduction Rate in Aerobic Stage	70.8% [2]	N.A.
Final TS at the end of the study (mg/l)	27,650	32,900
Overall TS Reduction Rate in the Study [3]	38.1%	45.4%

[1]: TS concentration on day 18; [2]: the reduction rate is calculated based on the first peak concentration and the aerobic stage end concentration; [3]: the reduction rate is calculated based on the first peak concentration and the final concentration in the study.

Variations in Metals and Chloride

Table 4-7 presented the concentrations of chloride and metals (copper and zinc) in leachate samples from AMR and NMR. The zinc concentration decreased with time in both bioreactors. The chloride concentration decreased with time in bioreactor NMR while in AMR the chloride concentrations increased with time in the first 30 days, and then dropped to approximately the initial concentrations at the end of the study. There were very small changes in copper concentration in both bioreactors.

Table 4-7 Metals and Chloride Concentrations in AMR and NMR

Day NO.	AMR				NMR			
	pH	Cu	Zn	Cl^-	pH	Cu	Zn	Cl^-
1	4.39	0.430	2.218	1.29	4.39	0.425	2.318	1.97
16	6.48	0.445	0.136	1.74	6.48	0.530	0.398	1.47
21	6.02	0.195	0.109	1.96	6.02	0.500	0.320	1.67
28	5.78	0.42	0.139	1.99	5.65	0.555	0.370	1.55
102	6.95	0.200	0.105	1.38	6.95	0.500	0.108	1.36

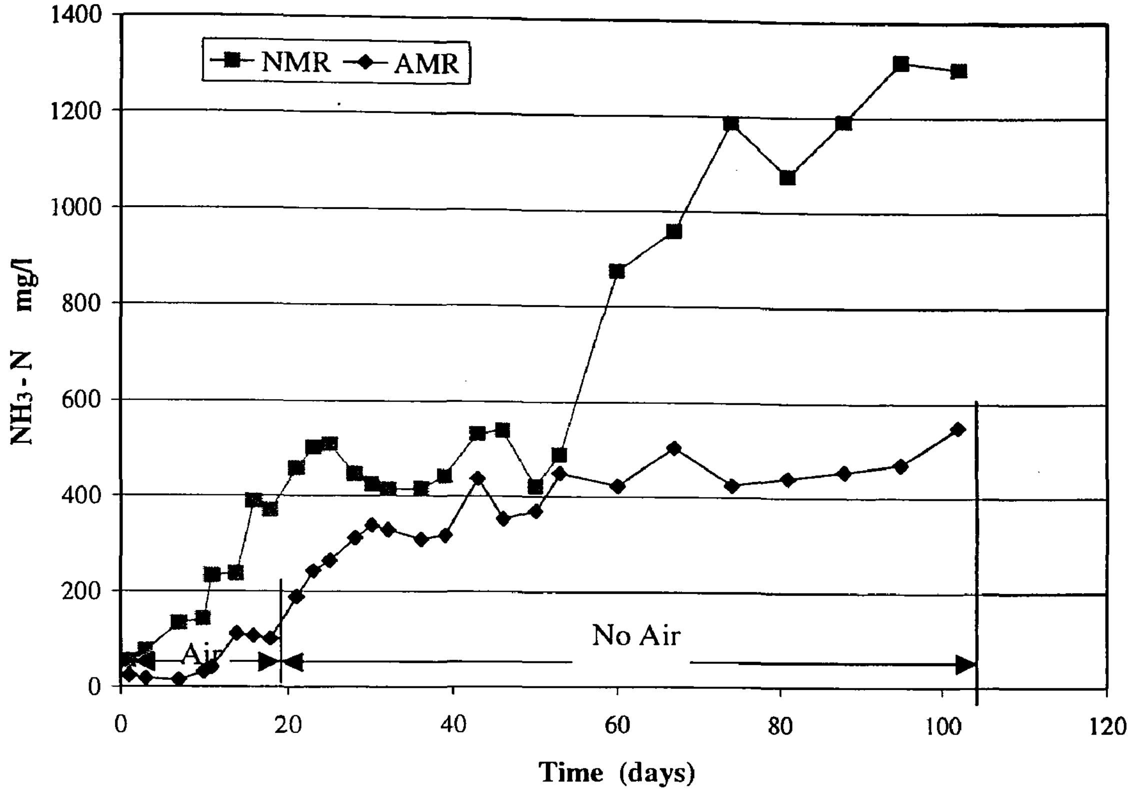


Figure 4-5 Variations in NH_3-N in AMR and NMR

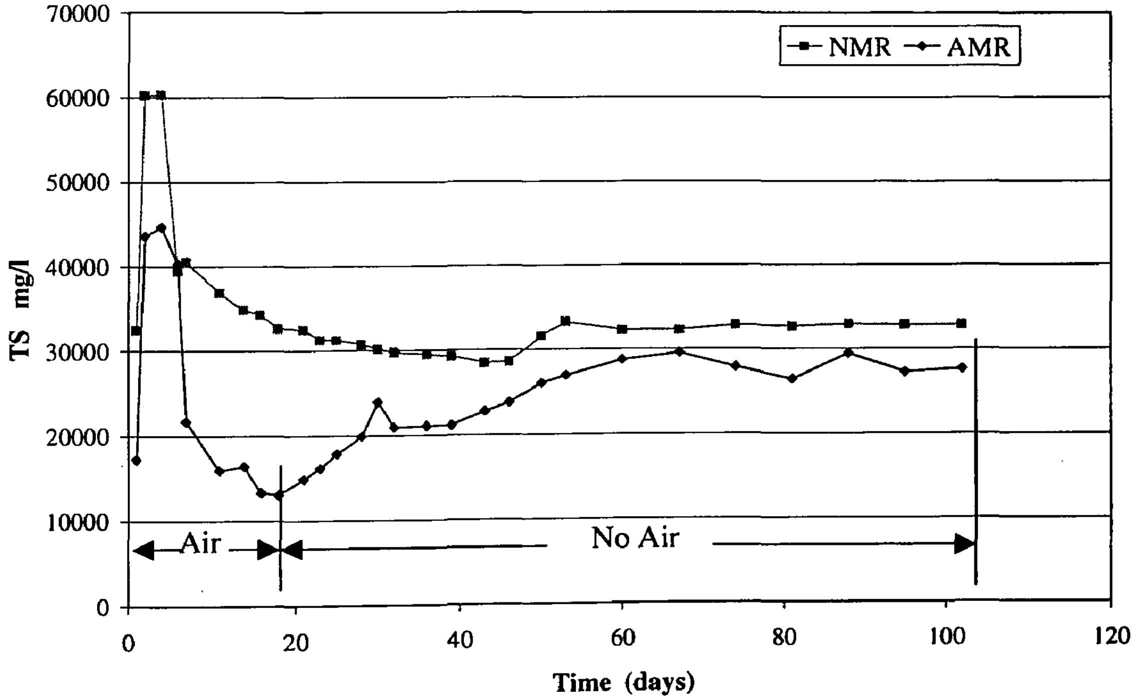


Figure 4-6 Variations in TS in AMR and ASR

4.1.3 Summary of Experimental Results and Discussions – AMR Vs. NMR

Based on the comparison of the performances of AMR and NMR, the initially degrading MSW under aerobic condition before the anaerobic degradation had positive effects on the decomposition of MSW with high organic and moisture contents.

Buffering can neutralize the generated VFAs in the anaerobic biodegradation, thus, be effective for accelerating the anaerobic biodegradation progress. The two bioreactors approximately utilized the same amount of buffer in the study. But, after about 100 days operation, the leachate pH values in AMR have been in the methanogenesis favorite range of 6.4 – 7.2 without buffering while the pH values still decreased after the buffering was stopped in NMR. This indicated that NMR already lagged behind AMR in the anaerobic degradation progress. In AMR, the initial aerobic degradation avoided the generation of large quantities of VFAs, thus, was beneficial for building up the optimum pH environment for the following anaerobic degradation and also saved buffer.

After about 100 days bioreactor treatment, the mass reduction rate in NMR was only one fifth of that in AMR. The initial COD concentration in AMR was approximately equal to that in NMR, as well as the initial BOD_5 concentration in AMR was approximately equal to that in NMR. However, the final BOD concentration in NMR was even double that in AMR (21,200 and 10,700 mg/l for NMR and AMR, respectively), while the final COD concentration in NMR was higher than that in AMR by 39.1% of the final COD in AMR (44,100 and 31,700 mg/l for NMR and AMR, respectively). The final TS concentrations were 32,900 and 27,650 mg/l for NMR and AMR, respectively. Therefore, it was concluded that the air addition accelerated the biodegradation of the organics as well as improved the final leachate quality.

he NH$_3$-N concentrations increased progressively with time in both bioreactors. The ecrease of NH$_3$-N was not observed in both bioreactors due to the time constraint of the tudy. The final NH$_3$-N concentrations were 550 and 1,300 mg/l for AMR and NMR, espectively. The lower NH$_3$-N in AMR may be attributed to the air addition, or may be ttributed to the inherent heterogeneity of the MSW.

he concentration of zinc decreased with time in both bioreactors. The concentration of hloride decreased with time in NMR while increased first and then decreased in AMR. here was no obvious variation trend in copper concentration.

.2 Experimental Results and Discussions – AMR Vs. ASR

he AMR bioreactors were loaded with synthetic MSW, operating with air addition, leachate ecirculation and leachate buffering technologies. The ASR bioreactor was loaded with ynthetic MSW and biosolids (at the wet weight ratio of 4:1), which also operated with air ddition, leachate recirculation and buffering technologies. The only difference between MR and ASR was that the waste in ASR consisted of biosolids, and the waste in AMR did ot consist of biosolids.

.2.1 Physical Variations – AMR Vs. ASR

isible Changes in MSW Composition and Odors after Bioreactor Treatment

ost food and vegetable wastes were decomposed after about 100 days in both bioreactors MR and ASR. There were only small amount of food and vegetable residuals, such as egetable husks, rinds, bones, and shrimp shells. The wet newsprints scattered in the plastics, hough obvious physical and chemical degradation occurred. Textiles changed very little after he treatment, and can still be identified as sweaters, bathing suits, etc. In both bioreactors, the anaerobic odors were very strong at the end of the study, however, the vinegar and alcohol odors were stronger in AMR, and the ammonia odor was stronger in ASR.

Changes in Mass, Settlement, Density, and Field Capacity

After about 100 days' operation, the wet waste weight in AMR reduced from 86.2 kg to 73.8 kg, as well as reduced from 80.9 kg to 64.4 kg in ASR due to aerobic and anaerobic degradation. The wet weight reduction rates for AMR and ASR were 14.4% and 20.4%, respectively (refer to Table 4-8). As discussed in section 4.1.1, the decomposition of MSW generated gases and intermediate volatile products, thus, caused the reduction of the waste weight, which in turn reflected the biodegradation speed in the bioreactors. Therefore, the above observation indicated that the overall biodegradation rate in ASR was faster than that in AMR.

The settlement trends are presented in Figure 4-7. The overall settlement rates for AMR and ASR were 22% and 24%, respectively. The settlement in ASR was faster than that in AMR. The settlement rate is governed by the biodegradation rate and other factors such as waste composition, density, and operational practices. The higher overall settlement rate in ASR may result from the greater extent of biodegradation.

With the decreases in waste volume and mass, densities in both bioreactors experienced increases due to a greater reduction in waste volume proportional to the waste mass (refer to Table 4-8). There was little change in field capacity in both bioreactors.

Table 4-8 Changes in Mass, Density, Field Capacity, and Settlement Rates in AMR and ASR

	AMR	ASR
Initial Wet Waste Weight (kg)	86.2	80.9
Final Wet Waste weight (kg)	73.8	64.4
Wet Weight Reduction (kg)	-12.4	-16.5
Wet Weight reduction Rate (%)	-14.4%	-20.4%
Initial Dry Waste Weight (kg)	25.8	24.4
Final Dry Waste Weight (kg)	21.9	18.9
Dry Weight Reduction (kg)	-3.9	-5.5
Dry Weight reduction Rate (%)	-14.9%	-22.5%
Initial Density (kg m^{-3})	543.8	532.3
Final Density (kg m^{-3})	598.5	558.9
Initial Field Capacity (W/W)	70.0%	69.9%
Final Field Capacity (W/W)	70.3%	70.6%
Final Settlement Rate (%)	22%	24%

Note: "Initial" means at the beginning of the study; "Final" means at the end of the study; Field capacity = the weight of water inside the waste divided by the total wet weight of the waste inside each bioreactor; Settlement rate = Thickness reduction of the waste matrix divided by the initial thickness of the waste matrix inside each bioreactor.

4.2.2 The Variations of Leachate Chemical Parameters – AMR Vs. ASR

Variations in pH

Figure 4-8 displays the trends of pH variations in leachate samples from AMR and ASR. The nitial pH values in AMR and ASR were 4.4 and 4.5, respectively. Continuous increases in pH were observed for both bioreactors during the 18-day aerobic stage, and reached 6.3 and 6.7, respectively on day 18, where the aerobic stage terminated. In day 6, AMR was buffered once, no buffering for ASR in aerobic stage. However, the pH level in ASR was higher than n AMR in this stage.

From day 19, both bioreactors were switched to anaerobic stage. From day 19 to day 28, here were no air addition and buffering. Meantime, the pH values started to drop from 6.7 to 5.7 in ASR, and from 6.3 to 5.7 in AMR. This observation coincided with the previous studies. In the anaerobic environment, the organics were hydrolyzed into VFAs, hence, caused the leachate pH to decrease. In contrast, the aerobic degradation broke down organics into carbon dioxide, ammonia and water, avoiding the generation of VFAs. Therefore, the pH increased in the aerobic stage and decreased in the initial anaerobic stage.

Previous studies have shown that methanogenesis is favored at a pH between 6.4 and 7.2. In order to achieve optimal pH for refuse methanogenesis, buffering leachate technology was applied from day 28 for both AMR and ASR. Once the leachate pH dropped below 6.4, buffering solution was added during the daily recirculation. Once the pH values of leachate were continuously in the range of 6.4 to 7.2, buffering was stopped. Buffering was stopped 30 days earlier in ASR than in AMR. The buffering stop point indicated that there was a balance between the VFAs generation and consumption, and the methanogenesis may became predominant (refer to section 2.1.1). This observation indicated ASR was earlier than AMR in the anaerobic degradation progress. This may attribute to the biosolids in ASR which can serve as seeds in the anaerobic degradation and accelerated the progress.

ASR only used half amount buffer of what AMR used (refer to Table 4-9). The pH level in ASR was higher than in AMR either in the aerobic stage or in the anaerobic stage. The only reason that caused the result is that ASR contained biosolids. This observation indicated that the biosolids had strong buffering effect. The biosolids have undergone anaerobic digestion that may cause the accumulation of ammonia. The buffering effect of biosolids may be attributed to the high ammonia concentration.

ble 4-9 Buffer Amounts and Buffering Timeline in AMR and ASR

ay Number		1-5	6	7-27	28-88	88-102	Total (g)
MR	Buffer Amount (g)	0	20	0	707.6	0	727.6
	Time line		█		█		

ay Number		1-27	28-59	59-80	81-83	83-102	Total (g)
SR	Buffer Amount (g)	0	310	0	8	0	318
	Timeline		█		█		

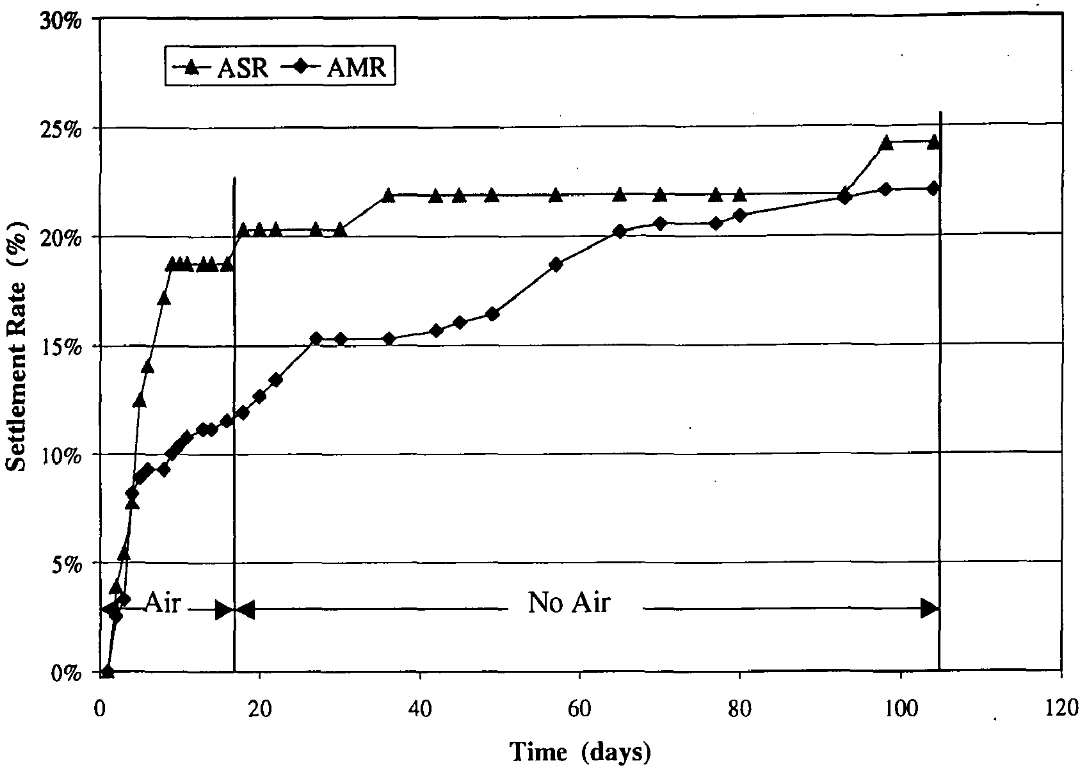


Figure 4-7 Waste Settlement Rates in AMR and ASR

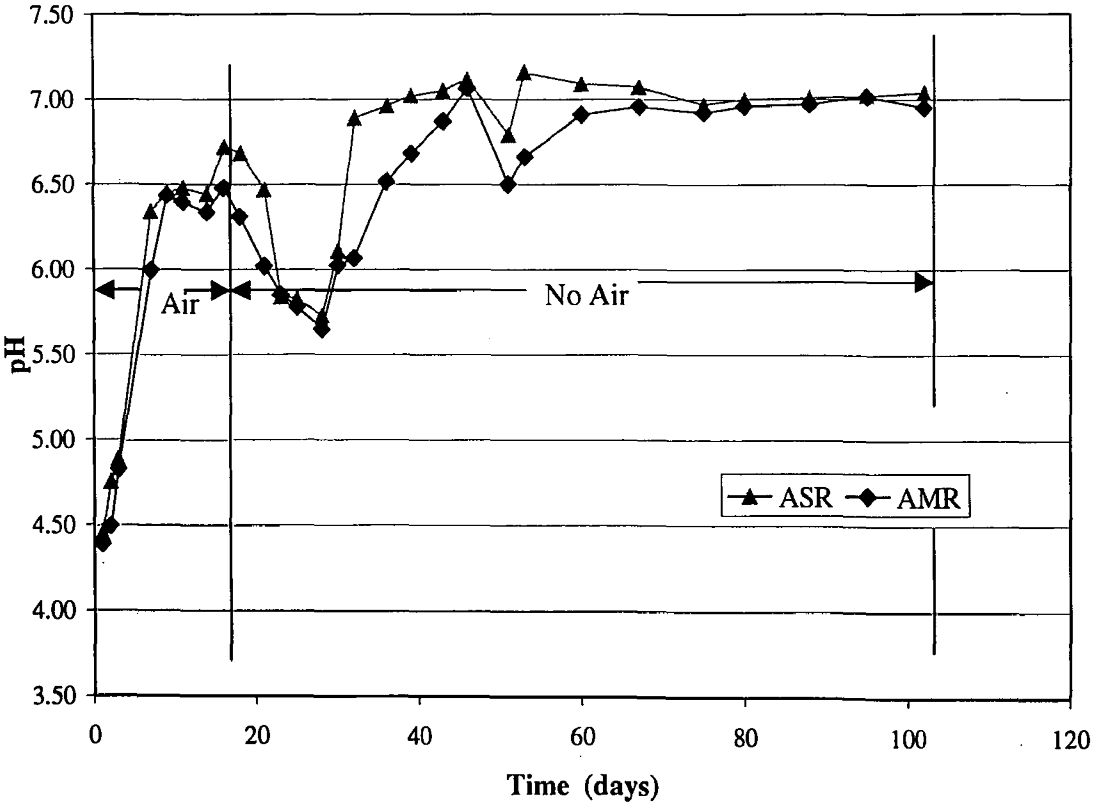


Figure 4-8 Variations in pH in AMR and ASR

ariations in COD

he trends of leachate COD variations in AMR and ASR are illustrated in Figure 4-9. The OD concentrations in AMR and ASR increased to first peak (45,100 and 36,500 mg/l, espectively) from the initial concentrations (37,600 and 30,000 mg/l, respectively) during the irst 3-day period. Following this initial period, sharp COD decreases in both bioreactors ere noted. The COD concentrations reached the bottom values (15,800 and 20,000 mg/l for MR and ASR, respectively) on day 18, when the aerobic stage terminated. The COD eduction rates from the first peak were 64.9% and 45.2% for AMR and ASR, respectively in he aerobic stage (refer to Table 4-10).

he COD concentrations in both bioreactors started to increase again right after the aerobic tage was switched to anaerobic stage. During day 30-39, makeup tap water was added into oth bioreactors in order to reach the optimum recirculation rate. This caused the COD oncentrations to fluctuate. The COD in ASR reached the second peak on day 60, while in MR on day 88. After the second peak, constant COD decreases were noted in both ioreactors. The overall COD reduction rates from the first peak in the study were 29.6% and 1.9% for AMR and ASR, respectively. (refer Table 4-10).

he COD reduction rate in aerobic stage was higher in AMR than in ASR. This observation ndicated that MSW without biosolids were more suitable for aerobic biodegradation than ISW with biosolids. It was noted that biosolids had higher density than MSW, and were very iscous when the bioreactors were loaded. The biosolids may encapsulate the organics, and lock the contact between the air and the organics. Such that, the COD reduction rate in ASR as much less than that in AMR in the aerobic stage. Another possibility may be that MSW without biosolids contained more biodegradable compositions.

In contrast, the overall COD reduction rate was higher in ASR than in AMR. The sharp increase of COD reduction rate in anaerobic stage caused the overall greater reduction rate in ASR. This observation may also result from the biosolids in ASR that served as seeds in the anaerobic stage. The biosolids addition accelerated the anaerobic biodegradation so that the COD reduction rate in anaerobic stage was much greater in ASR than in AMR. This result coincided with the previous studies (Blakey et al., 1997, Viste, 1997, and Gulec et al. 2000).

Table 4-10 Representative COD Concentrations in AMR and ASR and According Reduction Rates

	AMR	ASR
Initial COD (mg/l)	37,600	30,000
Fist Peak COD (mg/l)	45,100	36,500
Aerobic Stage End COD (mg/l)	15,800	20,000
COD Reduction Rate in Aerobic Stage (%) [1]	65.0%	45.2%
Anaerobic Stage End COD (mg/l)	31,700	21,200
Overall COD Reduction in the Study (%) [2]	29.7%	41.9%
COD Constant Drop Point in Anaerobic Stage	Day 88	Day 60
Buffering End Point	Day 88	Day 59

[1]: the reduction rate is calculated based on the first peak concentration and the aerobic stage end concentration; [2]: the reduction rate is calculated based on the first peak concentration and the final concentration in the study.

Variations in BOD_5

Figure 4-10 shows the BOD_5 variation trends in AMR and ASR. Although there were more fluctuations in BOD_5 data, the trends coincided with the COD trends very well. The initial BOD_5 concentrations in AMR and ASR were 18,800 and 14,200 mg/l, respectively. The first peak (23,500 and 18,500 mg/l for AMR and ASR, respectively) appeared around the third day of the study. The BOD_5 concentrations dropped to 10,700 and 7,700 mg/l at the end of the study. There were also two concentration peaks in the period of the study for each bioreactor.

he BOD_5 reduction rates from the first peak were 74.0% and 44.3% for AMR and ASR in he aerobic stage. In opposition to the aerobic stage, the overall BOD_5 reduction rates from he first peak were 54.5% and 58.4% for AMR and ASR, respectively (refer to Table 4-11). hese observations further confirmed the conclusions based on the COD data: MSW without iosolids were degraded faster in aerobic condition, while MSW with biosolids were egraded faster in anaerobic condition.

able 4-11 Representative BOD_5 Concentrations in AMR and ASR and According eduction Rates

	AMR	ASR
Initial BOD_5 (mg/l)	18,800	14,200
Fist Peak BOD_5 (mg/l)	23,500	18,500
Aerobic Stage End BOD_5 (mg/l)	6,100	10,300
BOD_5 Reduction Rate in Aerobic Stage [1]	74.0%	44.3%
Anaerobic Stage End BOD_5 (mg/l)	10,700	7,700
Overall BOD_5 Reduction Rate in the Study [2]	54.5%	58.4%
BOD_5 Constant Drop Point in Anaerobic Stage	67	53

: the reduction rate is calculated based on the first peak concentration and the aerobic stage nd concentration; [2]: the reduction rate is calculated based on the first peak concentration and he final concentration in the study.

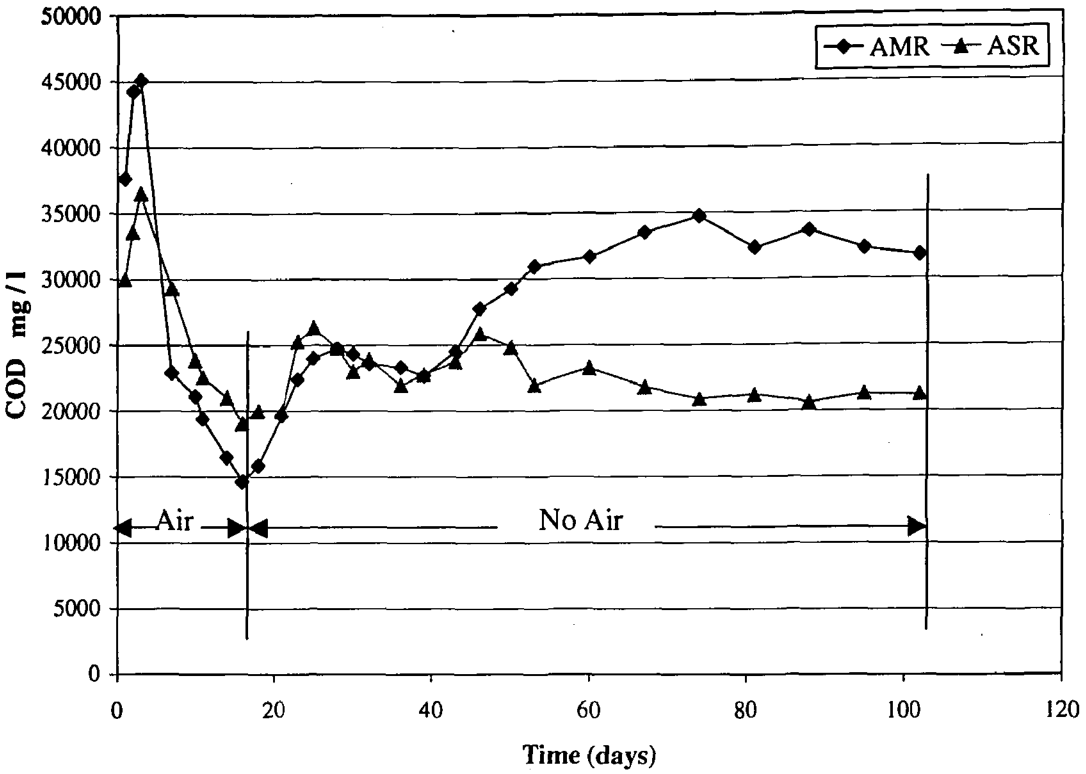


Figure 4-9 Variations in COD in AMR and ASR

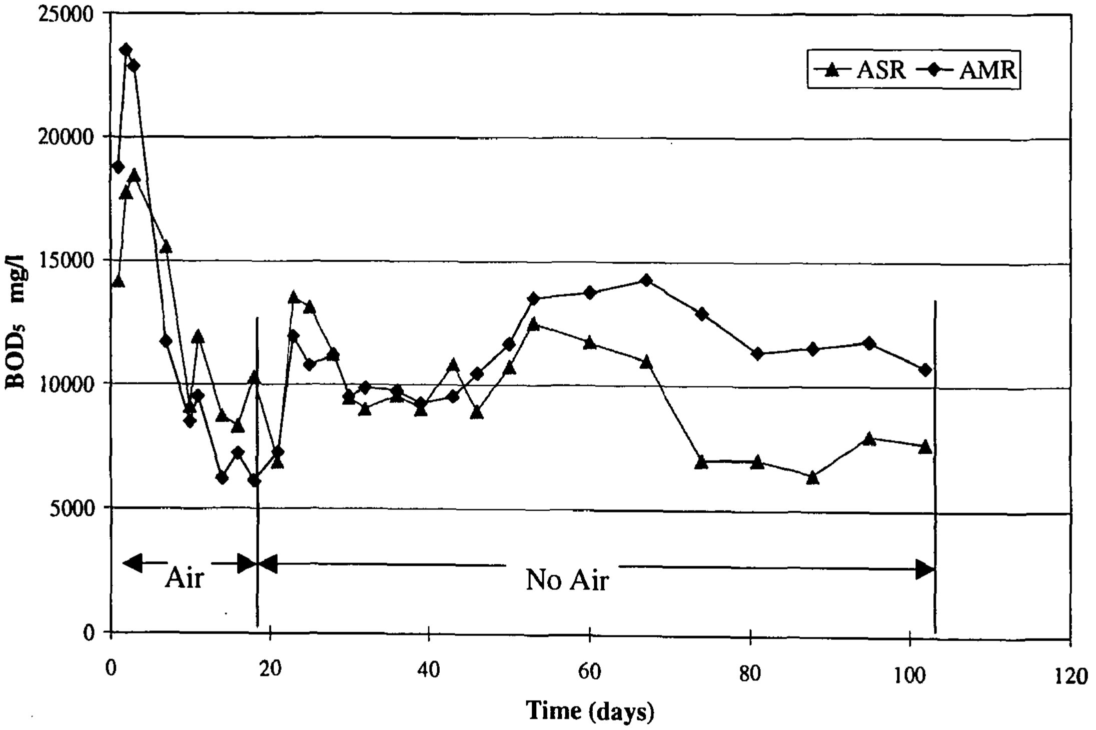


Figure 4-10 Variations in BOD_5 in AMR and ASR

ariations in BOD$_5$/COD Ratio

he BOD$_5$/COD ratios decreaed with time in both bioreactors – from 0.50 to 0.34 in AMR nd from 0.47 to 0.36 in ASR (refer to Table 4-12). The changes in BOD$_5$/COD ratio ndicated the variation in the percentage of biodegradable compounds in the leachate. Due to he degradation of the organics in MSW, the BOD$_5$/COD ratio would decrease. A ratio of .4-0.8 implies a highly biodegradable leachate (Warith, 2002). The leachate from both ioreactors initially belonged to this kind of leachate.

able 4-12 Representative BOD$_5$/COD Ratios in AMR and ASR

	AMR	ASR
Initial BOD$_5$/COD ratio	0.50	0.47
BOD$_5$/COD ratio at the end of aerobic stage	0.39	0.44
BOD$_5$/COD ratio at the end of the study	0.34	0.36

ariations in NH$_3$-N

he initial concentrations of NH$_3$-N in AMR and ASR were 24 and 340 mg/l, respectively. As lustrated in Figure 4-11, the NH$_3$-N concentration in AMR increased progressively and eached 550 mg/l at the end of the study. Although the NH$_3$-N concentration in ASR followed n increasing trend, there were great fluctuations. The final concentration in ASR was 1,020 ng/l.

he NH$_3$-N concentration values in ASR and AMR were not in the same order of magnitude. t was also noted that the NH$_3$-N concentrations in bioreactors with biosolids were much igher than bioreactors without biosolids (refer to appendix A). This observation indicated hat biosolids, which have undergone anaerobic digestion, introduced a large quantity of mmonia into the leachate.

n the aerobic environment, NH$_3$-N can be utilized as a substrate for aerobic bacteria, and be converted to nitrite through nitrification. Therefore, the aerobic stage is beneficial for the removal of high concentration ammonia in MSW with biosolids.

In the anaerobic stage, the NH_3-N concentrations increased more sharply because the nitrogenous organics were continuously converted into NH_3, and the leachate recirculation reintroduced NH_3 into the system. Some researchers (Sponza and Agdag, 2004, Ozturk, 1991) observed that the ammonium concentration started to decrease after a period of anaerobic incubation, since the ammonia was consumed by the anaerobic bacteria to develop their cellular components. This phenomenon was not observed in this study probably due to the time constraints.

Variations in TS

The leachate TS in AMR and ASR followed the same variation trends as COD and BOD_5 (Figure 4-12). Table 4-13 summarized the representative TS concentrations in AMR and ASR. The TS reduction rates from first peak concentrations in aerobic stage were 70.8% and 61.8% for AMR and ASR, respectively. In comparison, the overall reduction rates in the study were 38.1% and 48%, respectively. Combining with the COD and BOD_5 data, this observation further confirmed that MSW without biosolids can be degraded faster than MSW with biosolids in aerobic environment while MSW with biosolids can be degraded faster than the MSW without biosolids in anaerobic condition.

able 4-13 Representative TS Concentrations in AMR and ASR and According leduction Rates

	AMR	ASR
nitial TS (mg/l)	17,250	12,500
irst Peak TS (mg/l)	44,650	34,600
erobic Stage End TS (mg/l)	13,050	13,200
S Reduction Rate in Aerobic Stage (%) [1]	70.8%	61.8%
naerobic Stage End TS (mg/l)	27,650	18,000
verall TS Reduction Rate in the Study (%) [2]	38.1%	48.0%

: the reduction rate is calculated based on the first peak concentration and the aerobic stage nd concentration; [2]: the reduction rate is calculated based on the first peak concentration and e final concentration in the study.

ariations in Metals and Chloride

able 4-14 shows the concentrations of chloride and metals (copper and zinc) in leachate mples from AMR and ASR. The concentrations of metals decreased with time. There was o evident relationship between the variations of metals concentrations and the pH variations. he chloride concentrations increased with time in the first 30 days, and then dropped to pproximately the initial concentrations.

able 4-14 Metals and Chloride Concentrations in AMR and ASR

Day NO.	AMR				ASR			
	pH	Cu	Zn	Cl^-	pH	Cu	Zn	Cl^-
1	4.39	0.430	2.218	1.29	4.45	0.670	0.830	1.37
16	6.48	0.445	0.136	1.74	6.72	0.455	0.094	1.40
21	6.02	0.195	0.109	1.96	6.47	0.175	0.080	1.60
28	5.78	0.42	0.139	1.99	5.83	0.350	0.112	1.70
102	6.95	0.200	0.105	1.38	7.04	0.266	0.075	1.14

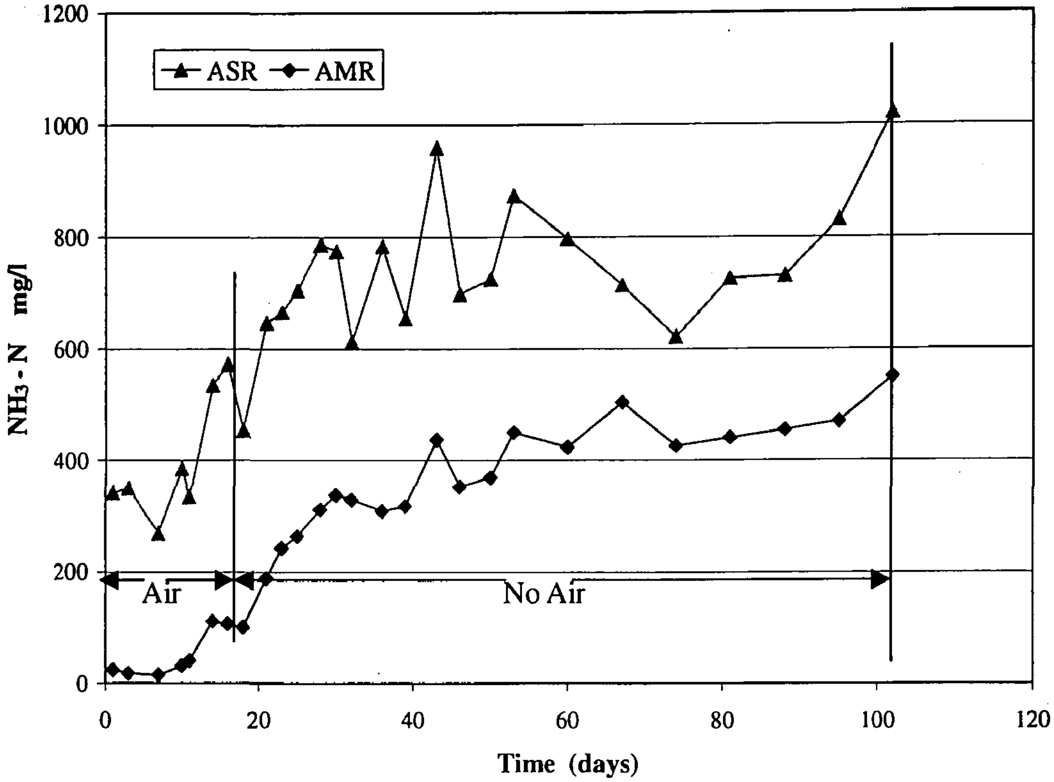


Figure 4-11 Variations in NH_3-N in AMR and ASR

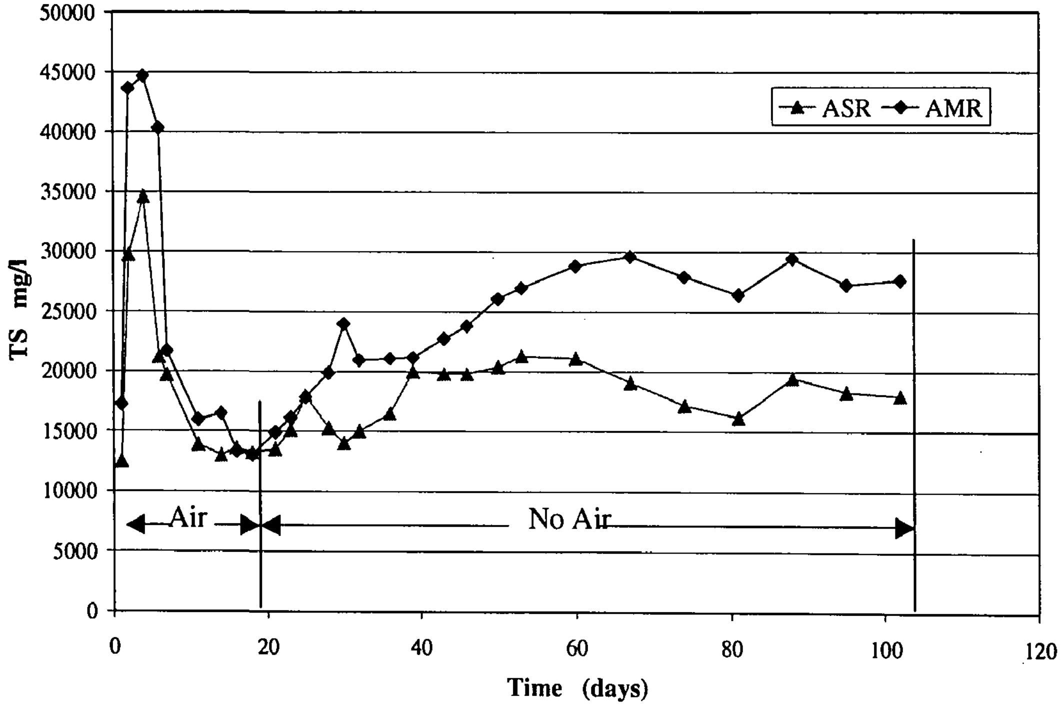


Figure 4-12 Variations in TS in AMR and ASR

2.3 Summary of Experimental Results and Discussions – AMR Vs. ASR

e experimental results in this section indicated that biosolids addition had positive effects the decomposition of MSW with high organic and moisture contents in aerobic-anaerobic oreactor landfills.

e leachate ammonia concentration in both AMR and ASR increased progressively in the dy, and no ammonia decrease was observed in both bioreactors probably due to the time nstraint. The ammonia concentration in ASR was not in the same order of magnitude as in MR. Combining with the other bioreactors' data, the leachate ammonia concentration in oreactors containing biosolids was much higher than in those without biosolids. This servation can be attribute to the added biosolids because they have undergone anaerobic gestion.

SR only used half amount of buffer as AMR, but the leachate pH values were higher in SR than in AMR at every sampling point in the study. Both bioreactors had the same eration protocols except ASR containing biosolids. Therefore, it was concluded that osolids have strong buffering effects that may result from the high ammonia content in the osolids.

he waste weight reduction rates in AMR and ASR were about 15% and 21%, respectively. his indicated that the overall biodegradation rate in ASR was faster than in AMR. This nclusion was also confirmed by the overall reduction rates of COD, BOD$_5$ and TS ncentrations.

contrast, the reduction rates of COD, BOD$_5$ and TS concentrations were higher in AMR an in ASR in the aerobic stage. It can be derived that the reduction rates of these parameters were higher in ASR than in AMR in the anaerobic stage. It can be concluded that MSW without biosolids were degraded faster in the aerobic environment and MSW with biosolids were degraded faster in the anaerobic environment. Biosolids served as seeds in the anaerobic degradation and accelerated the anaerobic degradation progress. This can also be verified by the earlier buffering stop time in the anaerobic stage.

4.3 Experimental Results and Discussions – AMR Vs. AMF and ASR Vs. ASF

This section presents the experimental results of bioreactors AMR, AMF, ASR and ASF, and focuses on the discussion about the effects of single-pass operation and closed-loop leachate recirculation operation on the decomposition of MSW with high organic and moisture contents. There are two comparison groups: AMR Vs. AMF and

ASR Vs. ASF. Bioreactors AMR and AMF were loaded with MSW without biosolids. AMR operated in leachate recirculation mode with buffering and AMF operated in single-pass mode without buffering. Bioreactors ASR Vs. ASF were loaded with MSW with biosolids. ASR operated in leachate recirculation mode with buffering and ASF operated in single-pass mode without buffering.

4.3.1 Physical Variations – AMR Vs. AMF and ASR Vs. ASF

After about 100 days bioreactor treatment, the weight of the wastes in each of the four bioreactors decreased to a different extent. The wet weight reduction rates in AMR, AMF, ASR and ASF were 14.4%, 21.1%, 20.4% and 24%, respectively. The weight reduction rates in single-pass bioreactors AMF and ASF were obviously higher than those in the leachate recirculation bioreactors AMR and ASR.

Figures 4-13 and 4-14 illustrate the waste settlement rates in the four bioreactors. The overall ttlement rates in AMR, AMF, ASR and ASF were 22%, 27%, 24% and 27%, respectively. he settlement characters in single-pass bioreactors were better than those in the leachate circulation bioreactors. The density and field capacity changes in the four bioreactors are resented in Table 4-15.

able 4-15 Changes in Mass, Density, Field Capacity, and Settlement Rates in AMR, MF, ASR and ASF

	AMR	**AMF**	**ASR**	**ASF**
nitial Wet Waste Weight (kg)	86.2	84.2	80.9	82.8
inal Wet Waste weight (kg)	73.8	66.4	64.4	62.9
Vet Weight Reduction (kg)	-12.4	-17.8	-16.5	-19.9
Vet Weight reduction Rate (%)	-14.4%	-21.1%	-20.4%	-24%
nitial Dry Waste Weight (kg)	25.8	29.3	24.4	24.4
inal Dry Waste Weight (kg)	21.9	22.3	18.9	18.5
ry Weight Reduction (kg)	-3.9	-7.0	-5.5	-5.9
ry Weight reduction Rate (%)	-14.9%	-23.9%	-22.5%	-24.2%
nitial Density (kg m $^{-3}$)	543.8	562.7	532.3	544.8
inal Density (kg m $^{-3}$)	598.5	607.5	558.9	563.7
nitial Field Capacity (W/W)	70.0%	65.1%	69.9%	70.6%
Final Field Capacity (W/W)	70.3%	66.4%	70.6%	70.6%
Final Settlement Rate (%)	22%	27%	24%	27%

ote: "Initial" means at the beginning of the study; "Final" means at the end of the study; ield capacity = the weight of water inside the waste divided by the total wet weight of the aste inside each bioreactor; Settlement rate = thickness reduction of the waste matrix divided y the initial thickness of the waste matrix inside each bioreactor.

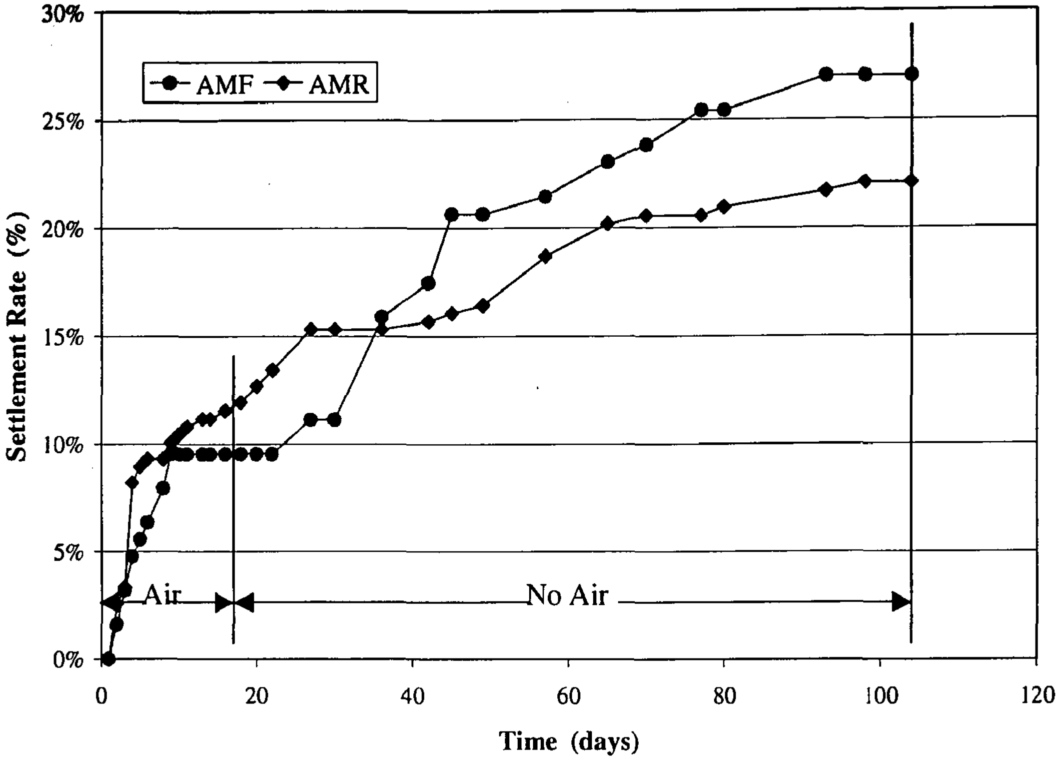


Figure 4-13 Waste Settlement Rates in AMR and AMF

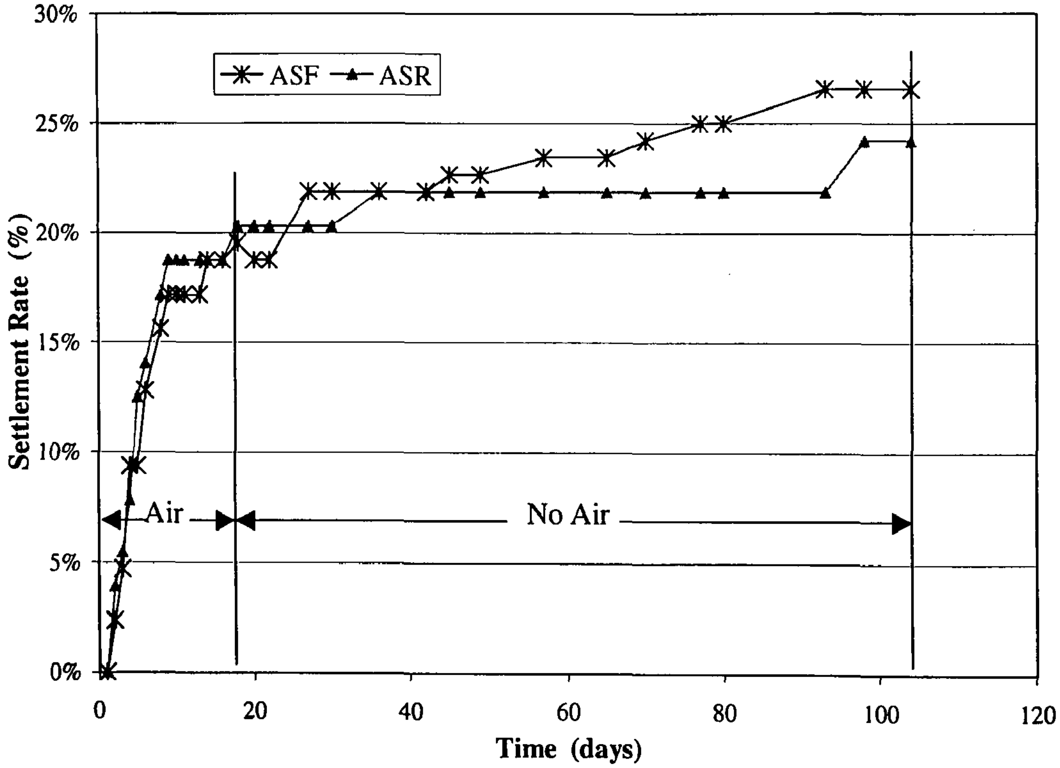


Figure 4-14 Waste Settlement Rates in ASR and ASF

4.3.2 The Variations of Leachate Chemical Parameters – AMR Vs. AMF and ASR Vs. ASF

Variations in pH

The initial pH values in AMR, AMF, ASR and ASF were 4.40, 4.60, 4.50 and 4.90, espectively. All the four bioreactors operated with air addition in the first 18 days. The air rought the pH values to about the neutral values in the four bioreactors (refer to Table 4-16). As illustrated in Figures 4-15 and 4-16, the pH curves of the single-pass bioreactors AMF and ASF were above those of the leachate recirculation bioreactors AMR and ASR. This may esult from the different operation modes because the added tap water flushed away the enerated VFAs. In the anaerobic stage, the leachate recirculation bioreactors operated with eachate buffering and the single bioreactors operated without leachate buffering. Therefore, he pH values in leachate recirculation bioreactors kept the increase trend and were around e neutral value, while the pH values in single-pass bioreactor kept decrease trend and were way from the neutral value.

able 4-16 Representative pH Values in AMR, AMF, ASR and ASF

	AMR	AMF	ASR	ASF
Initial pH	4.40	4.60	4.50	4.90
pH at the end of aerobic stage	6.30	6.50	6.70	7.60
pH at the end of the study	7.00	6.30	7.00	6.90

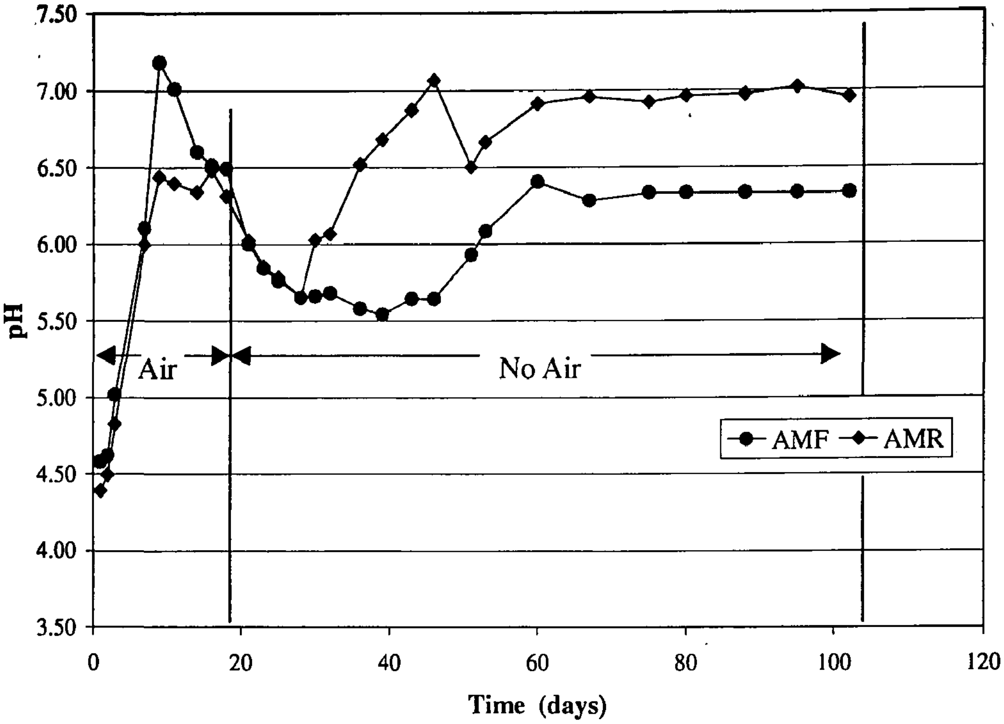


Figure 4-15 Variations in pH in AMR and AMF

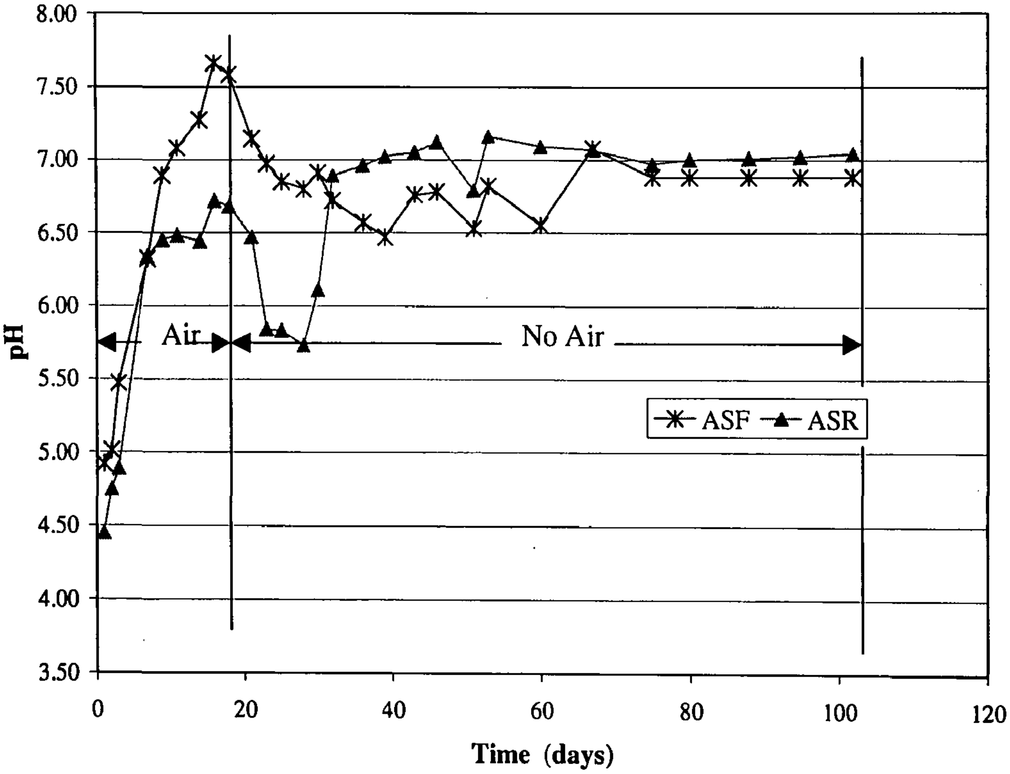


Figure 4-16 Variations in pH in ASR and ASF

ariations in COD, BOD$_5$ and TS

able 4-17 summarized the representative COD, BOD5 and TS concentrations in AMR, AMF, SR and ASF, and Figures 4-17 to 4-22 presented the variation curves of these parameters in e comparison groups.

the comparison group: AMR and AMF, the initial and peak COD, BOD$_5$ and TS ncentrations in the single-pass bioreactor AMF were approximately equal to the initial and ak COD, BOD$_5$ and TS concentrations in the leachate recirculation bioreactor AMR. owever, the final COD in AMF was about one forth of that in AMR, the final BOD$_5$ in AMF as about one third of that in AMR, and the final TS in AMF was only about one tenth of that AMR. These data indicated that the final leachate quality was much better in the ngle-pass bioreactor AMF than in the leachate recirculation bioreactor AMR.

the comparison group: ASR and ASF, the COD, BOD$_5$ and TS reduction rates in the ngle-pass bioreactor ASF were all higher than those in the leachate recirculation bioreactor SR (refer to Table 4-17).

herefore, it was concluded that the single-pass operation mode was more effective for nproving the final leachate quality in the bioreactors. In the operation of single-pass ioreactors, tap water was constantly added into the bioreactors to simulate the rainfall. The dded water diluted the leachate in the bioreactors and flushed away the pollutants and ansported to the downstream treatment units.

Table 4-17 Representative COD, BOD_5 and TS concentrations in AMR, AMF, ASR and ASF

	AMR	AMF	ASR	ASF
Initial COD (mg/l)	37,600	38,000	30,000	25,500
Peak COD (mg/l)	45,100	38,000	36,500	32,800
Final COD at the end of the study (mg/l)	31,700	8,300	21,200	16,000
COD Reduction Rate (%)	29.7%	78.2%	41.9%	51.2%
Initial BOD_5 (mg/l)	18,800	20,000	14,200	11,000
Peak BOD_5 (mg/l)	23,500	25,200	18,500	16,200
Final BOD_5 at the end of the study (mg/l)	10,700	4,000	7,700	5,600
BOD_5 Reduction Rate (%)	54.5%	84.1%	58.4%	65.4%
Initial TS (mg/l)	17,300	17,000	12,500	14,400
Peak TS (mg/l)	44,700	36,500	34,600	30,000
Final TS at the end of the study (mg/l)	27,700	2,500	18,000	6,600
TS Reduction Rate (%)	38.0%	93.2%	48.0%	78%

Note: "Initial" means at the beginning of the study; "Final" means at the end of the study; all the reduction rates were calculated based on the peak concentrations and the final concentrations.

Variations in NH_3-N

In comparison group: AMR and AMF, the initial NH_3-N concentrations were 24 and 25 mg/l, respectively. As illustrated in Figure 4-23, in AMR, the NH_3-N concentration increased progressively and reached the highest concentration 550 mg/l at the end of the study. In AMF, the NH_3-N concentration increased to the peak (500 mg/l) on day 36 and then sharply decreased. The final NH_3-N in AMF was about half of that in AMR.

In comparison group: ASR and ASF, the initial NH_3-N concentrations were 330 and 340 mg/l, respectively. In ASR, the NH_3-N variation trend was the same as in AMR. The final NH_3-N concentration was 1,020 mg/l. In ASF, the NH_3-N variation trend was the same as in AMF. It jumped to peak (2,890 mg/l) on day 30 and then sharply decreased (refer to Figure 4-24). The final NH_3-N in ASF was also about half of that in ASR.

The lower final NH_3-N concentrations in the single-pass bioreactors can also be attributed to the flushing effects.

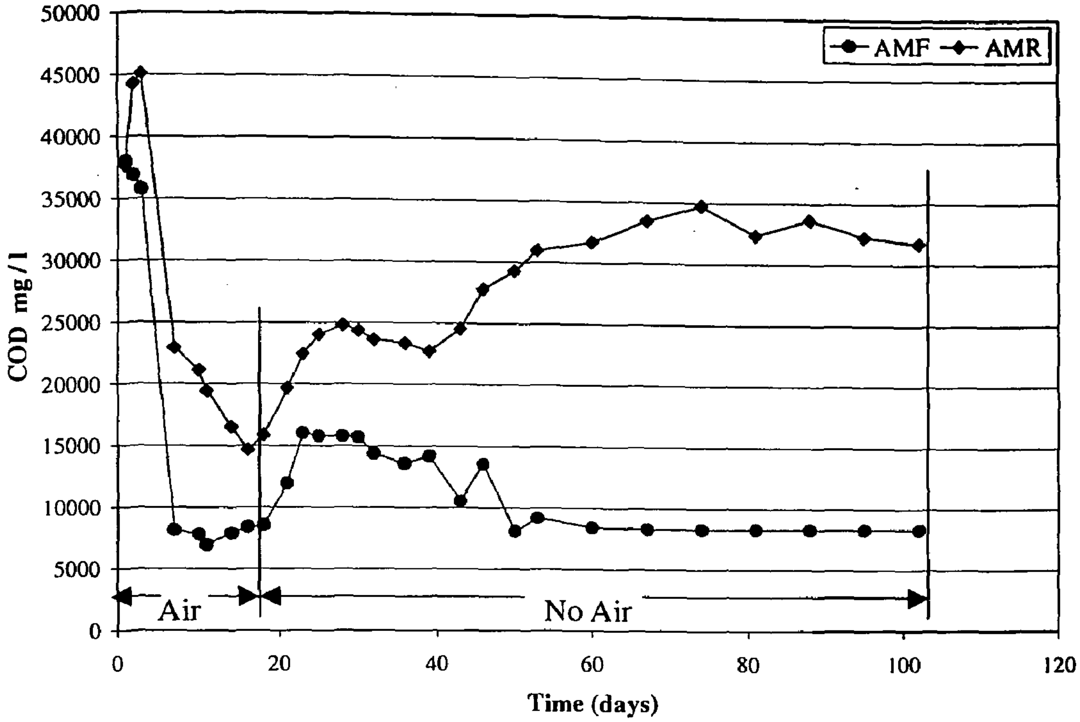


Figure 4-17 Variations in COD in AMR and AMF

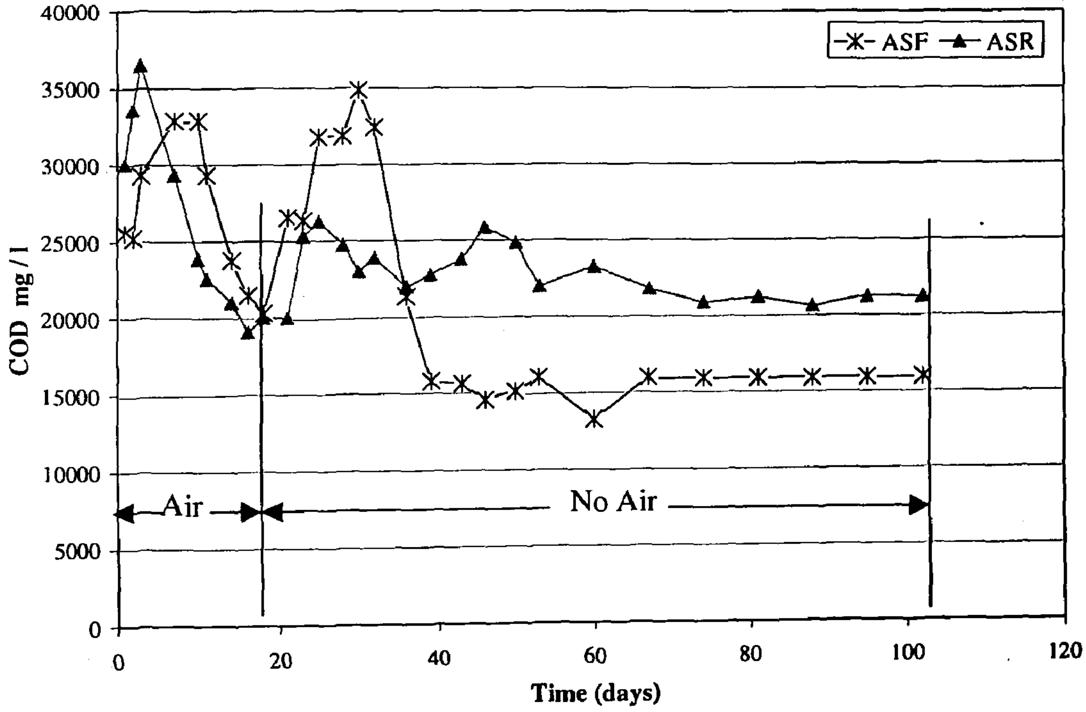


Figure 4-18 Variations in COD in ASR and ASF

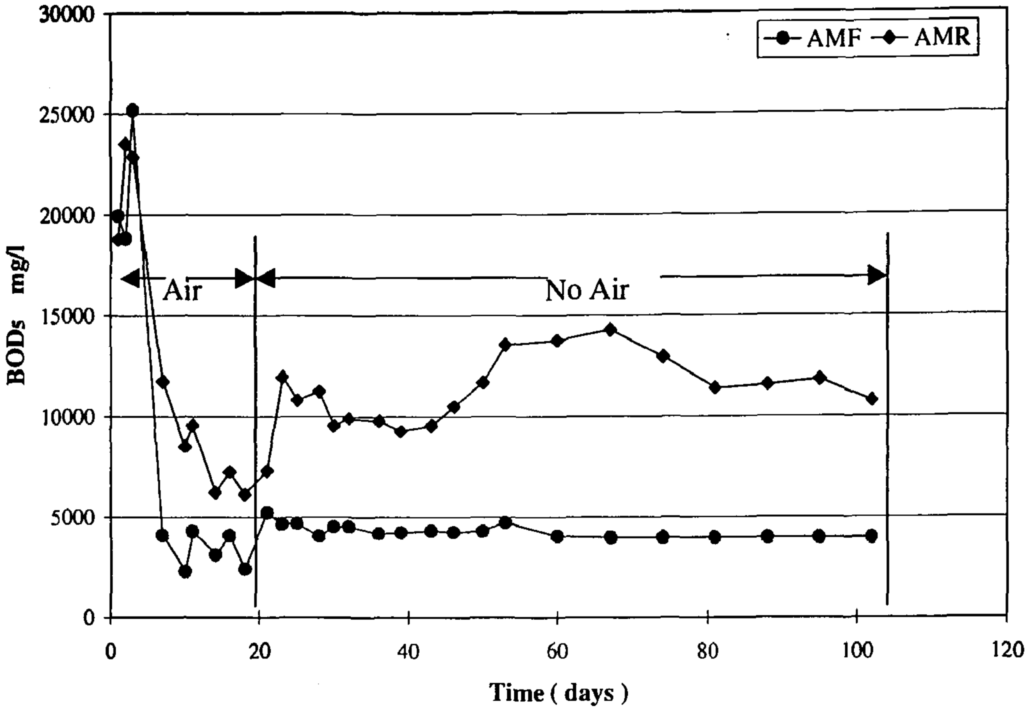


Figure 4-19 Variations in BOD$_5$ in AMR and AMF

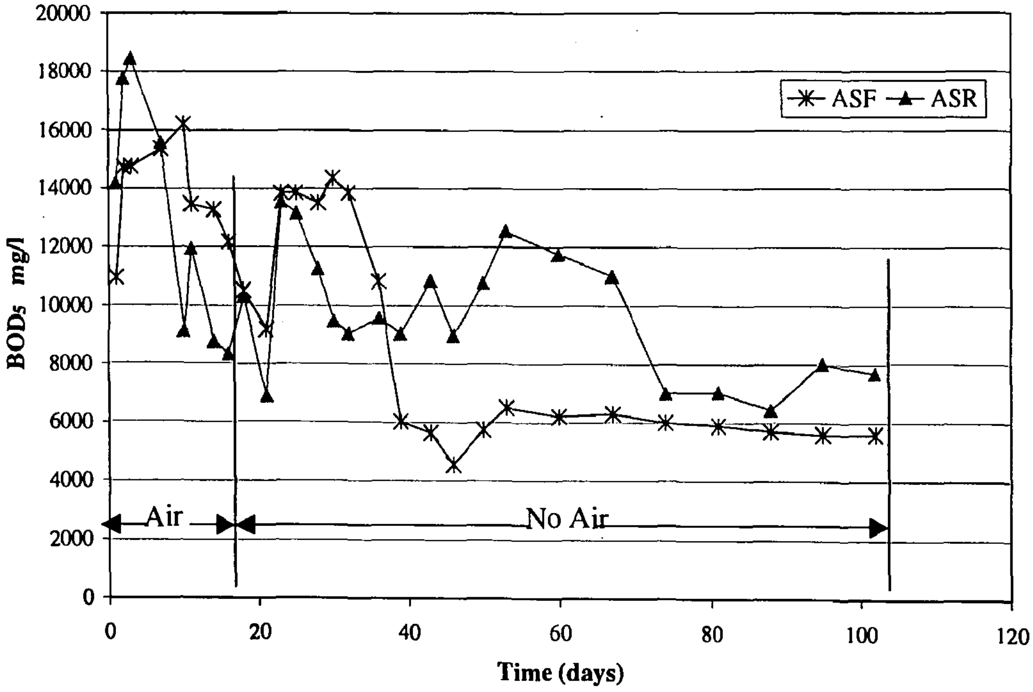


Figure 4-20 Variations in BOD$_5$ in ASR and ASF

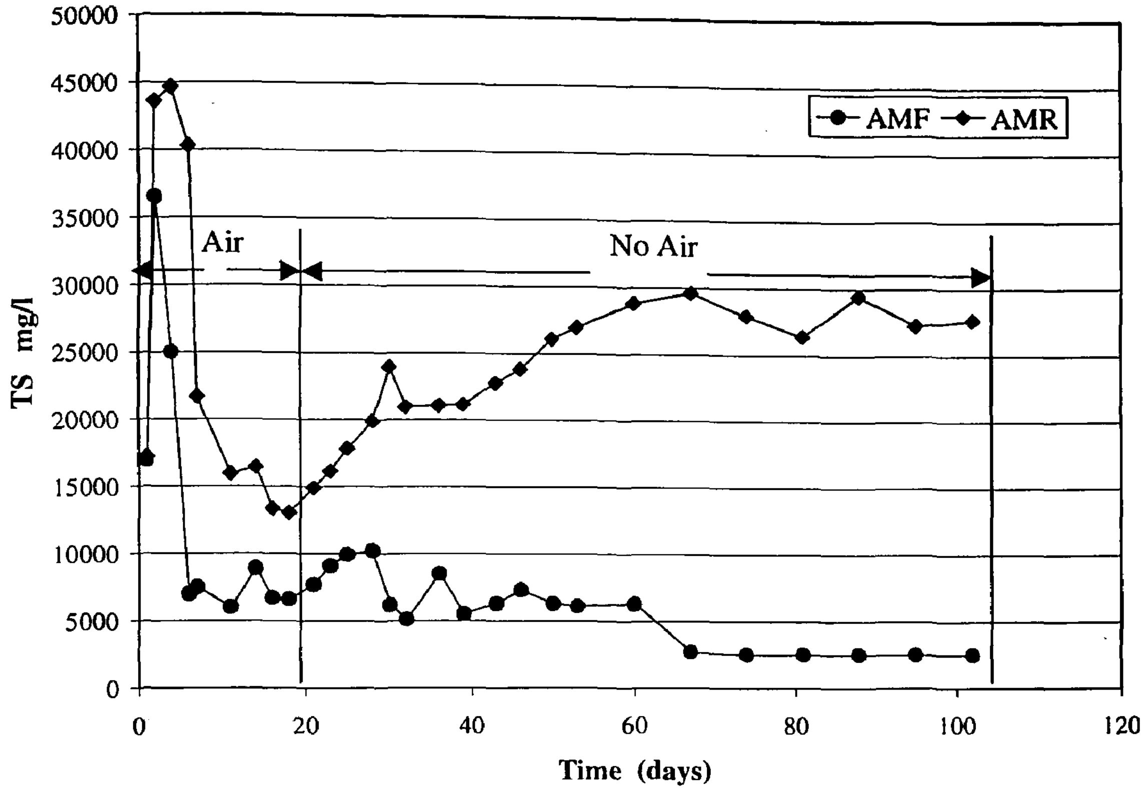


Figure 4-21 Variations in TS in AMR and AMF

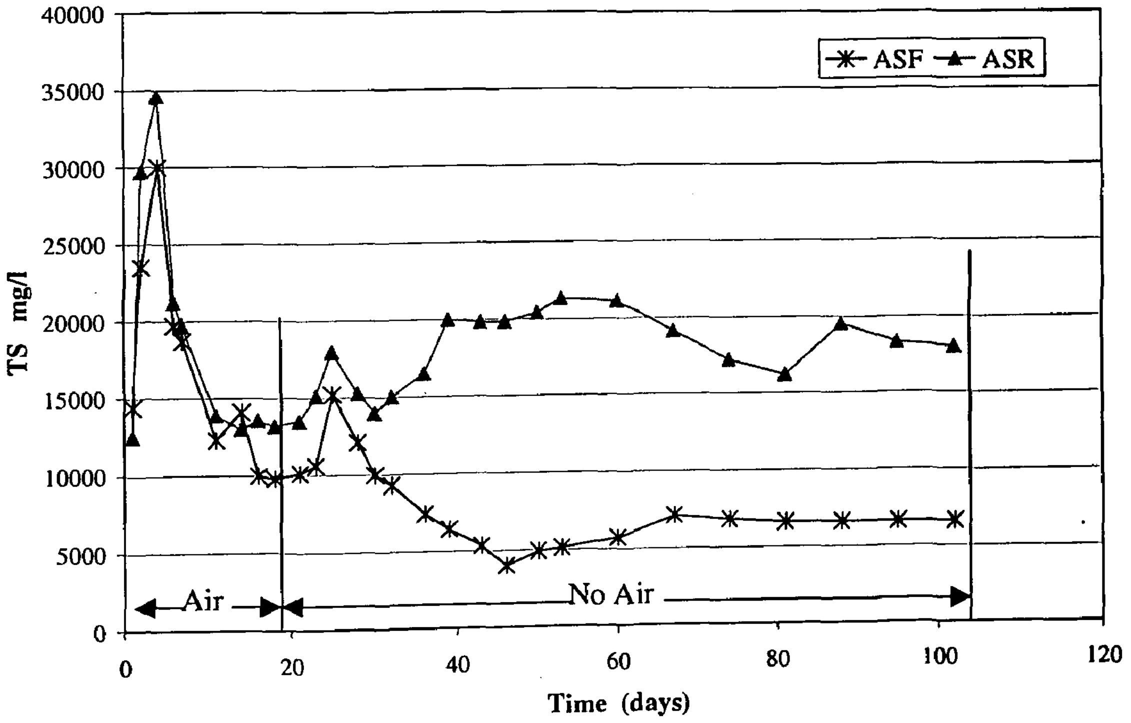


Figure 4-22 Variations in TS in ASR and ASF

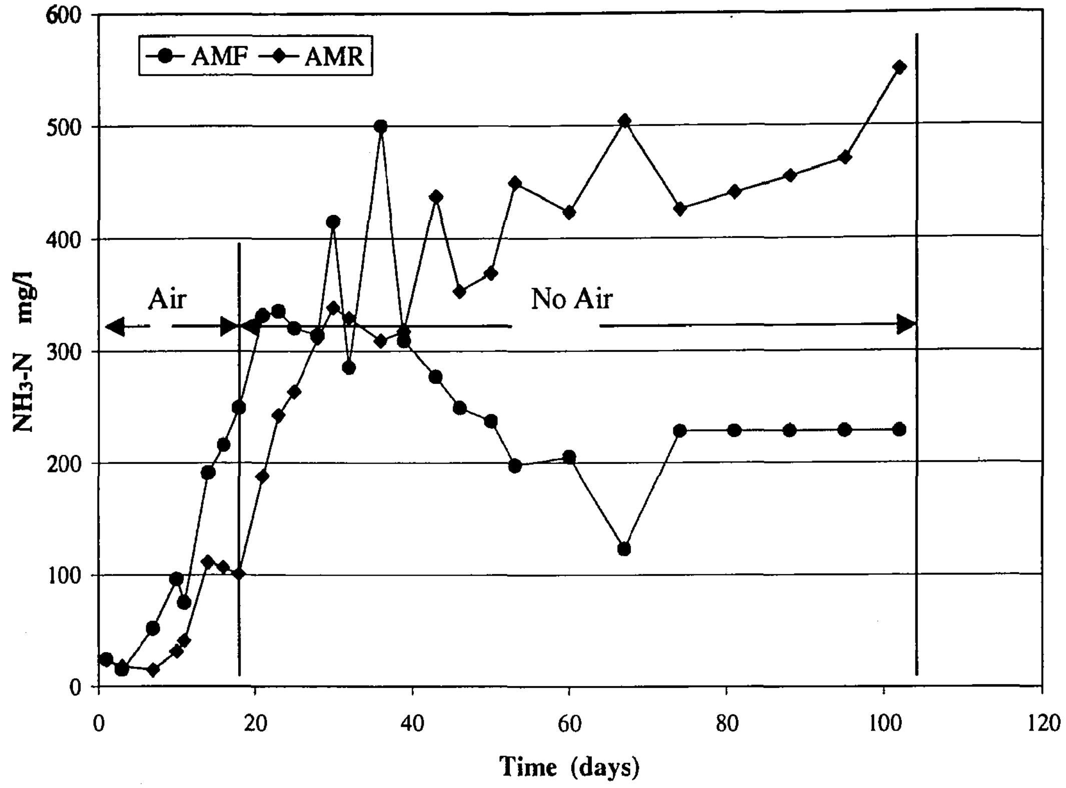


Figure 4-23 Variations in NH$_3$-N in AMR and AMF

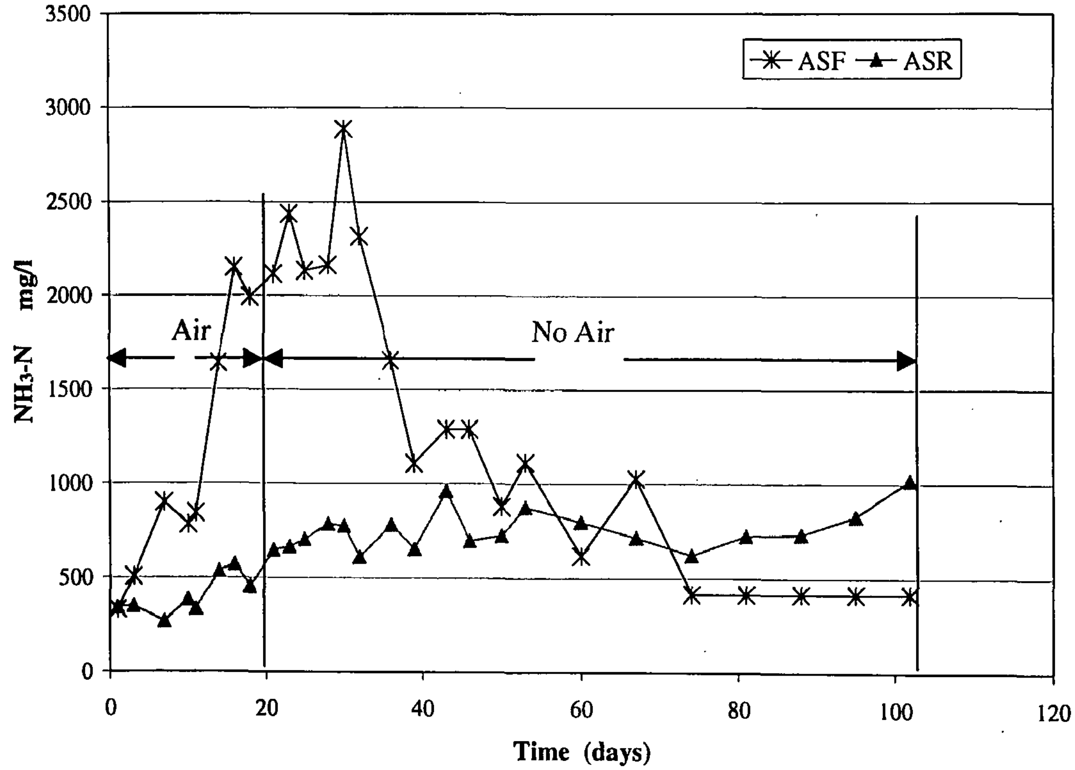


Figure 4-24 Variations in NH$_3$-N in ASR and ASF

3.3 Summary of Experimental Results and Discussions – AMR Vs. AMF and ASR Vs. SF

he waste weight reduction rates and settlement rates of the waste matrices in single-pass ioreactors were obviously higher than the leachate recirculation bioreactors. The final achate quality was much better in the single-pass bioreactors than in the leachate ecirculation bioreactors, especially for the comparison group AMR Vs. AMF. The final COD AMF was about one forth of that in AMR, the final BOD_5 in AMF was about one third of at in AMR, the final TS in AMF was only about one tenth of that in AMR, and the final H_3-N in AMF was only half of that in AMR.

ased on these data, the single-pass operation mode was more effective for removing ollutants from the bioreactor landfills. But, it transported the pollutants to the downstream reatment units. From the economic view of point, this operation mode is not attractive.

5. CONCLUSIONS AND RECOMMENDATIONS

5.1 Conclusions

With the increase in waste recycling and diversion, MSW with high organic and moisture contents (80% -90% of the total weight of the MSW) can be found in various landfills worldwide. This study aimed to decompose this type of MSW in aerobic – anaerobic bioreactor landfills. In the study, the effects of air addition and biosolids addition on the biodegradation of MSW with high organic and moisture contents were examined. Moreover, the flushing technology was compared with the leachate recirculation technology.

Six bioreactor landfills have been set up in the lab. Based on their respective operational protocol, matrix components, and replicate number of the bioreactors, the six bioreactors were identified as AMF, AMR1, AMR2, ASF, ASR, and NMR (A: air; N: no air; M: MSW; S: biosolids; F: flushing; R: recirculation). Based on the analyses of the experimental results, conclusions were as following:

- The positive effects of air addition were concluded based on the performance comparison between AMR and NMR. AMR operated in two sequential stages (aerobic and anaerobic stages) while NMR operated in the constant anaerobic stage. The two bioreactors approximately utilized the same amount of buffer in the study. But, after about 100 days operation, the leachate pH values in AMR have been in the methanogenesis favorite range of 6.4 – 7.2 without buffering while the pH values still decreased after the buffering was stopped in NMR, as well as the pH values in AMR were higher than in NMR at every sampling point in the study period. In the anaerobic environment, organic materials were hydrolyzed to VFAs that can cause acidic environment in the waste ecosystem. The low pH values even can reverse the anaerobic biodegradation. In the aerobic environment, organics were broke down into carbon dioxide, ammonia and water, at the same time, avoiding the generation of VFAs. Hence, it was concluded that initially degrading MSW with high organic and moisture content under aerobic environment was effective for building up the optimum pH environment for the following anaerobic degradation. Meanwhile, this observation indicated that NMR lagged behind AMR in the anaerobic progress. In other words, the air addition in the aerobic stage accelerated the downstream anaerobic biodegradation.

After about 100 days bioreactor treatment, the mass reduction rate in NMR was only one fifth of that in AMR. The initial COD concentration in AMR was approximately equal to that in NMR, as well as the initial BOD_5 concentration in AMR was approximately equal to that in NMR. However, the final BOD_5 concentration in NMR was even double that in AMR (21,200 and 10,700 mg/l for NMR and AMR, respectively), while the final COD concentration in NMR was higher than that in AMR by 39.1% of the final COD in AMR (44,100 and 31,700 mg/l for NMR and AMR, respectively). The final TS concentrations were 32,900 and 27,650 mg/l for NMR and AMR, respectively. Therefore, it was concluded that the air addition accelerated the biodegradation of the organics as well as improved the final leachate quality.

The NH_3-N concentrations increased progressively with time in both bioreactors. The final NH_3-N concentrations were 550 and 1,300 mg/l for AMR and NMR, respectively. The lower NH_3-N in AMR may be attributed to the air addition, or may be attributed to the inherent heterogeneity of the MSW.

- The positive effects of biosolids addition were concluded based on the performance comparison between AMR and ASR. The leachate ammonia concentrations in ASR was higher than in AMR by one order of magnitude. This resulted from the added biosolids because they have undergone anaerobic digestion. ASR only used half amount of buffer as AMR, but the leachate pH values were higher in ASR than in AMR at every sampling point in the study. Both bioreactors had the same operation protocols except ASR containing biosolids. Therefore, it was concluded biosolids have strong buffering effects that may result from the high ammonia content in the biosolids.

  The waste weight reduction rates in AMR and ASR were about 15% and 21%, respectively. This indicated that the overall biodegradation rate in ASR was faster than in AMR. This conclusion was also confirmed by the overall reduction rates of COD, BOD_5 and TS. In contrast, the reduction rates of COD, BOD_5 and TS were higher in AMR than in ASR in the aerobic stage. Hence, it can be derived that the reduction rates of these parameters were higher in ASR than in AMR in the anaerobic stage. It can be concluded that MSW without biosolids were degraded faster in the aerobic environment and MSW with biosolids were degraded faster in the anaerobic environment. Biosolids served as seeds in the anaerobic degradation and accelerated the anaerobic degradation progress. This can also be testified by the earlier buffering stop time in the anaerobic stage.

- Based on the experimental data of waste weight reduction rates, COD, BOD_5, TS, and NH_3-N, the single-pass operation mode was more effective for removing pollutants from the bioreactor landfills. But, it transported the pollutants to the downstream treatment units. From the economic view of point, this operation mode is not attractive. Initially degrading MSW under aerobic condition before anaerobic degradation with biosolids addition is the optimum strategy for the decomposition of MSW with high organic and moisture contents.

.2 Recommendations

s concluded above, air addition was beneficial for building up the optimum pH environment r the downstream anaerobic stage and accelerated the decomposition of the MSW with high rganic and moisture contents. However, the air supply is costly in the field bioreactor ndfills. From the economic view of point, the optimal period of aerobic stage needs further vestigation.

heoretically, in the aerobic environment, ammonia can be utilized as a substrate for aerobic acteria, and be converted to nitrite through nitrification. Therefore, the aerobic stage is eneficial for the removal of high concentration ammonia in MSW with biosolids. It needs rther investigation to conclude the extent of ammonia removal in the aerobic stage with the ISW containing the given percentage of nitrogenous organics.

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PPENDICES

ppendix A – Experimental Data

esults of Leachate pH Analysis

itial Data **Unit:**
H

Date	Day No.	AMF	ASF	ASR	NMR	AMR1	AMR2	AMR
5-Sep-04	1	4.6	4.9	4.5	4.1	4.3	4.5	4.4
7-Sep-04	2	4.6	5.0	4.8	4.4	4.4	4.6	4.5
8-Sep-04	3	5.0	5.5	4.9	4.2	4.9	4.8	4.8
2-Sep-04	7	6.1	6.3	6.3	4.7	6.0	6.0	6.0
4-Sep-04	9	7.2	6.9	6.5	4.9	6.5	6.4	6.4
6-Sep-04	11	7.0	7.1	6.5	5.3	6.4	6.4	6.4
9-Sep-04	14	6.6	7.3	6.4	5.5	6.2	6.5	6.3
1-Sep-04	16	6.5	7.7	6.7	5.6	6.4	6.6	6.5
3-Sep-04	18	6.5	7.6	6.7	5.5	6.2	6.4	6.3
6-Sep-04	21	6.0	7.1	6.5	5.5	5.9	6.2	6.0
8-Sep-04	23	5.8	7.0	5.8	5.6	5.8	5.9	5.9
0-Sep-04	25	5.8	6.9	5.8	5.5	5.7	5.8	5.8
3-Oct-04	28	5.7	6.8	5.7	5.6	5.6	5.7	5.7
5-Oct-04	30	5.7	6.9	6.1	5.5	6.0	6.1	6.0
7-Oct-04	32	5.7	6.7	6.9	5.6	6.0	6.2	6.1
1-Oct-04	36	5.6	6.6	7.0	5.6	6.5	6.6	6.5
4-Oct-04	39	5.5	6.5	7.0	5.8	6.7	6.6	6.7
8-Oct-04	43	5.6	6.8	7.1	5.8	6.9	6.8	6.9
1-Oct-04	46	5.6	6.8	7.1	6.2	7.1	7.1	7.1
6-Oct-04	51	5.9	6.5	6.8	6.1	6.6	6.5	6.5
8-Oct-04	53	6.1	6.8	7.2	6.2	6.7	6.6	6.7
4-Nov-04	60	6.4	6.6	7.1	6.3	6.9	6.9	6.9
1-Nov-04	67	6.3	7.1	7.1	6.6	7.0	7.0	7.0
9-Nov-04	75	6.3	6.9	7.0	6.2	6.9	7.0	6.9
4-Nov-04	80	6.3	6.9	7.0	6.2	7.0	7.0	7.0
2-Dec-04	88	6.3	6.9	7.0	6.2	7.0	7.0	7.0
9-Dec-04	95	6.3	6.9	7.0	6.1	7.0	7.0	7.0
6-Dec-04	102	6.3	6.9	7.0	5.8	6.9	7.0	7.0

Results of Waste Matrix Settlement

Initial Data **Unit:**
cm

	AMF			ASF			ASR			NMR			AMR1			AMR2			AMR
IH	63.0			64.0			64.0			73.0			66.0			67.4			
Day No.	H	S	R %	H	S	R %	H	S	R %	H	S	R %	H	S	R %	H	S	R %	R %
1	63	0	0%	64	0	0%	64	0	0%	73	0	0%	66	0	0%	67	0	0%	0%
2	62	1	2%	63	2	2%	62	3	4%	72	1	1%	64	2	3%	66	1	2%	3%
3	61	2	3%	61	3	5%	61	4	5%	72	2	2%	63	3	5%	66	1	2%	3%
4	60	3	5%	58	6	9%	59	5	8%	71	2	3%	60	6	9%	63	5	7%	8%
5	60	4	6%	58	6	9%	56	8	13%	70	3	4%	60	6	9%	62	6	9%	9%
6	59	4	6%	56	8	13%	55	9	14%	70	4	5%	60	6	9%	61	6	9%	9%
8	58	5	8%	54	10	16%	53	11	17%	68	5	7%	60	6	9%	61	6	9%	9%
9	57	6	10%	53	11	17%	52	12	19%	67	6	8%	60	6	9%	60	7	11%	10%
10	57	6	10%	53	11	17%	52	12	19%	66	7	10%	60	7	10%	60	7	11%	10%
11	57	6	10%	53	11	17%	52	12	19%	65	8	11%	60	7	10%	60	8	12%	11%
13	57	6	10%	53	11	17%	52	12	19%	65	8	11%	60	7	10%	59	8	12%	11%
14	57	6	10%	52	12	19%	52	12	19%	65	9	12%	60	7	10%	59	8	12%	11%
16	57	6	10%	52	12	19%	52	12	19%	64	10	13%	59	7	11%	59	8	12%	12%
18	57	6	10%	52	13	20%	51	13	20%	64	10	13%	59	8	11%	59	8	12%	12%
20	57	6	10%	52	12	19%	51	13	20%	63	10	14%	58	8	12%	59	9	13%	13%
22	57	6	10%	52	12	19%	51	13	20%	62	12	16%	57	9	14%	59	9	13%	13%
27	56	7	11%	50	14	22%	51	13	20%	60	14	18%	56	11	16%	58	10	15%	15%
30	56	7	11%	50	14	22%	51	13	20%	59	14	19%	56	11	16%	58	10	15%	15%
36	53	10	16%	50	14	22%	50	14	22%	58	15	21%	56	11	16%	58	10	15%	15%
42	52	11	17%	50	14	22%	50	14	22%	58	15	21%	56	11	16%	57	10	15%	16%
45	50	13	21%	50	15	23%	50	14	22%	58	15	21%	56	11	16%	57	11	16%	16%
49	50	13	21%	50	15	23%	50	14	22%	57	16	22%	56	11	16%	56	11	17%	16%
57	50	14	21%	49	15	23%	50	14	22%	56	17	23%	53	13	20%	56	12	18%	19%
65	49	15	23%	49	15	23%	50	14	22%	56	17	23%	53	14	20%	54	13	20%	20%
70	48	15	24%	49	16	24%	50	14	22%	56	17	23%	52	14	21%	54	13	20%	21%
77	47	16	25%	48	16	25%	50	14	22%	56	17	23%	52	14	21%	54	13	20%	21%
80	47	16	25%	48	16	25%	50	14	22%	56	18	24%	52	15	22%	54	13	20%	21%
93	46	17	27%	47	17	27%	50	14	22%	56	18	24%	51	16	23%	54	13	20%	22%
98	46	17	27%	47	17	27%	49	16	24%	55	18	25%	50	16	24%	54	13	20%	22%
104	46	17	27%	47	17	27%	49	16	24%	55	18	25%	50	16	24%	54	13	20%	22%

Note: IH: matrix initial height; H: matrix height with time; S: matrix settlement with time; R: matrix reduction rate from the initial height.

Results of Leachate COD Analysis

nitial Data

Unit:
ıg/l

Date	Day No.	AMF	ASF	ASR	NMR	AMR1	AMR2	AMR
6-Sep-04	1	37992	25539	30000	33142	35753	39510	37632
7-Sep-04	2	36867	25204	33560	46390	41050	47439	44245
8-Sep-04	3	35753	29301	36494	48346	41050	49097	45074
12-Sep-04	7	8102	32838	29301	56458	21255	24536	22896
15-Sep-04	10	7795	32838	23872	58549	20525	21701	21113
16-Sep-04	11	6882	29301	22555	59251	17509	21306	19408
19-Sep-04	14	7855	23755	21000	62319	17408	15567	16488
21-Sep-04	16	8348	21466	19088	60201	14623	14623	14623
23-Sep-04	18	8515	20291	20001	59499	16113	15567	15840
26-Sep-04	21	11909	26533	20022	55323	20395	18918	19657
28-Sep-04	23	16010	26323	25289	50215	23874	20960	22417
30-Sep-04	25	15794	31812	26323	52332	25701	22300	24001
3-Oct-04	28	15741	31931	24743	50485	23907	25591	24749
5-Oct-04	30	15701	34893	21874	49647	23907	24743	24325
7-Oct-04	32	14344	32414	23907	51755	23701	23495	23598
11-Oct-04	36	13546	21361	21966	50572	23650	23000	23325
14-Oct-04	39	14166	15842	22782	50739	22577	22782	22680
18-Oct-04	43	10508	15698	23770	49610	24395	24605	24500
21-Oct-04	46	13524	14637	25882	45887	26098	29443	27771
25-Oct-04	50	8054	15226	24843	50700	27428	31079	29254
28-Oct-04	53	9136	16115	21974	47700	27428	34488	30958
4-Nov-04	60	8380	13288	23253	49828	28346	34963	31655
11-Nov-04	67	8274	15980	21778	45725	32144	34788	33466
18-Nov-04	74	8274	15980	20876	47158	34602	34850	34726
25-Nov-04	81	8274	15980	21223	42753	32380	32283	32332
2-Dec-04	88	8274	15980	20600	44354	34650	32600	33625
9-Dec-04	95	8274	15980	21250	44354	33039	31444	32242
16-Dec-04	102	8274	15980	21215	44154	32549	30902	31726

Results of Leachate BOD$_5$ Analysis

Initial Data

Units: mg/l

Date	Day No.	AMF	ASF	ASR	NMR	AMR1	AMR2	AMR
6-Sep-04	1	19950	10950	14160	19350	16200	21360	18780
7-Sep-04	2	18825	14700	17760	19995	22050	24960	23505
8-Sep-04	3	25200	14760	18450	21960	20760	24960	22860
12-Sep-04	7	4080	15343	15557	20600	11280	12180	11730
15-Sep-04	10	2280	16200	9128	18997	9342	7680	8511
16-Sep-04	11	4280	13450	11950	20598	8040	11040	9540
19-Sep-04	14	3117	13251	8751	26327	6390	6057	6224
21-Sep-04	16	4071	12167	8333	27900	6107	8375	7241
23-Sep-04	18	2400	10500	10312	30750	6000	6250	6125
26-Sep-04	21	5203	9159	6909	26787	6009	8559	7284
28-Sep-04	23	4640	13839	13539	23802	13908	10000	11954
30-Sep-04	25	4657	13832	13149	28725	12285	9285	10785
3-Oct-04	28	4029	13500	11250	28500	11325	11100	11213
5-Oct-04	30	4487	14332	9458	27600	9708	9332	9520
7-Oct-04	32	4507	13830	9025	29256	9650	10093	9872
11-Oct-04	36	4159	10813	9573	23100	9675	9879	9777
14-Oct-04	39	4208	6021	9025	23250	9128	9380	9254
18-Oct-04	43	4300	5649	10833	22916	9720	9345	9533
21-Oct-04	46	4224	4570	8957	19666	9432	11535	10484
25-Oct-04	50	4287	5750	10750	22500	10429	12905	11667
28-Oct-04	53	4714	6500	12535	25500	13392	13660	13526
4-Nov-04	60	4035	6200	11750	22698	13750	13750	13750
11-Nov-04	67	3950	6300	10999	21165	15082	13451	14267
18-Nov-04	74	3950	6000	7000	22998	12750	13125	12938
25-Nov-04	81	3950	5900	7042	20000	11214	11535	11375
2-Dec-04	88	3950	5700	6428	21250	11500	11600	11550
9-Dec-04	95	3950	5600	7999	21250	11856	11749	11803
16-Dec-04	102	3950	5600	7678	21150	10892	10583	10738

esults of COD / BOD_5 Ratios Analysis

nitial Data

Units:

Date	Day No.	AMF	ASF	ASR	NMR	AMR1	AMR2	AMR
-Sep-04	1	0.53	0.43	0.47	0.58	0.45	0.54	0.50
7-Sep-04	2	0.51	0.58	0.53	0.43	0.54	0.53	0.53
-Sep-04	3	0.70	0.50	0.51	0.45	0.51	0.51	0.51
2-Sep-04	7	0.50	0.47	0.53	0.36	0.53	0.50	0.51
5-Sep-04	10	0.29	0.49	0.38	0.32	0.46	0.35	0.40
6-Sep-04	11	0.62	0.46	0.53	0.35	0.46	0.52	0.49
9-Sep-04	14	0.40	0.56	0.42	0.42	0.37	0.39	0.38
1-Sep-04	16	0.49	0.57	0.44	0.46	0.42	0.57	0.50
3-Sep-04	18	0.28	0.52	0.52	0.52	0.37	0.40	0.39
6-Sep-04	21	0.44	0.35	0.35	0.48	0.29	0.45	0.37
8-Sep-04	23	0.29	0.53	0.54	0.47	0.58	0.48	0.53
0-Sep-04	25	0.29	0.43	0.50	0.55	0.48	0.42	0.45
3-Oct-04	28	0.26	0.42	0.45	0.56	0.47	0.43	0.45
5-Oct-04	30	0.29	0.41	0.43	0.56	0.41	0.38	0.39
7-Oct-04	32	0.31	0.43	0.38	0.57	0.41	0.43	0.42
1-Oct-04	36	0.31	0.51	0.44	0.46	0.41	0.43	0.42
4-Oct-04	39	0.30	0.38	0.40	0.46	0.40	0.41	0.41
8-Oct-04	43	0.41	0.36	0.46	0.46	0.40	0.38	0.39
1-Oct-04	46	0.31	0.31	0.35	0.43	0.36	0.39	0.38
5-Oct-04	50	0.53	0.38	0.43	0.44	0.38	0.42	0.40
8-Oct-04	53	0.52	0.40	0.57	0.53	0.49	0.40	0.44
-Nov-04	60	0.48	0.47	0.51	0.46	0.49	0.39	0.43
1-Nov-04	67	0.48	0.39	0.51	0.46	0.47	0.39	0.43
8-Nov-04	74	0.48	0.38	0.34	0.49	0.37	0.38	0.37
5-Nov-04	81	0.48	0.37	0.33	0.47	0.35	0.36	0.35
2-Dec-04	88	0.48	0.36	0.31	0.48	0.33	0.36	0.34
-Dec-04	95	0.48	0.35	0.38	0.48	0.36	0.37	0.37
6-Dec-04	102	0.48	0.35	0.36	0.48	0.33	0.34	0.34

Results of Leachate NH_3 –N Analysis

Initial Data

Units: mg/l

Date	Day No.	AMF	ASF	ASR	NMR	AMR1	AMR2	AMR
6-Sep-04	1	25	331	342	56	22	26	24
8-Sep-04	3	15	505	351	76	11	25	18
12-Sep-04	7	52	899	269	134	13	17	15
15-Sep-04	10	96	782	385	142	33	30	32
16-Sep-04	11	75	842	335	233	45	37	41
19-Sep-04	14	191	1641	535	237	133	89	111
21-Sep-04	16	216	2148	572	390	120	93	107
23-Sep-04	18	249	1990	454	370	123	78	101
26-Sep-04	21	331	2113	646	458	215	161	188
28-Sep-04	23	335	2433	665	501	268	217	243
30-Sep-04	25	320	2131	704	508	295	232	264
3-Oct-04	28	314	2160	786	446	335	288	312
5-Oct-04	30	415	2886	774	425	330	346	338
7-Oct-04	32	285	2313	612	414	278	380	329
11-Oct-04	36	500	1654	783	416	326	292	309
14-Oct-04	39	309	1107	653	441	335	300	318
18-Oct-04	43	277	1289	959	531	408	466	437
21-Oct-04	46	249	1289	697	539	273	432	353
25-Oct-04	50	237	878	724	421	336	402	369
28-Oct-04	53	198	1111	873	488	367	532	450
4-Nov-04	60	205	616	796	874	371	475	423
11-Nov-04	67	123	1027	713	959	443	565	504
18-Nov-04	74	228	416	621	1187	433	418	426
25-Nov-04	81	228	416	725	1076	428	453	441
2-Dec-04	88	228	416	730	1192	421	488	455
9-Dec-04	95	228	416	830	1317	440	500	470
16-Dec-04	102	228	416	1021	1302	566	533	550

esults of Leachate TS Analysis

itial **Data**

nits: mg/l

Date	Day No.	AMF	ASF	ASR	NMR	AMR1	AMR2	AMR
6-Sep-04	1	17000	14400	12500	32500	16000	18500	17250
7-Sep-04	2	36500	23500	29700	60200	39100	48100	43600
9-Sep-04	4	25000	30000	34600	60300	39600	49700	44650
1-Sep-04	6	7000	19700	21200	39500	41700	38900	40300
2-Sep-04	7	7500	18700	19700	40500	19800	23600	21700
6-Sep-04	11	6000	12300	13900	36900	15000	16900	15950
9-Sep-04	14	8900	14100	13000	34900	17400	15500	16450
1-Sep-04	16	6700	10000	13600	34300	12900	13800	13350
3-Sep-04	18	6600	9800	13200	32700	12500	13600	13050
6-Sep-04	21	7700	10100	13500	32500	14800	15000	14900
8-Sep-04	23	9100	10600	15100	31300	15800	16500	16150
0-Sep-04	25	9900	15200	18000	31300	15700	20000	17850
3-Oct-04	28	10200	12100	15300	30800	17800	22000	19900
5-Oct-04	30	6200	10000	14000	30200	23100	24800	23950
7-Oct-04	32	5100	9300	15000	29800	20100	21800	20950
1-Oct-04	36	8500	7400	16500	29600	20150	22000	21075
4-Oct-04	39	5500	6400	20000	29300	20300	22000	21150
8-Oct-04	43	6300	5300	19800	28500	20900	24600	22750
1-Oct-04	46	7300	4000	19800	28600	22200	25400	23800
5-Oct-04	50	6300	4900	20400	31600	24700	27400	26050
8-Oct-04	53	6100	5100	21300	33300	24700	29200	26950
4-Nov-04	60	6300	5700	21100	32300	26600	31000	28800
1-Nov-04	67	2700	7100	19100	32300	27200	32000	29600
8-Nov-04	74	2500	6800	17200	32900	26600	29200	27900
5-Nov-04	81	2500	6600	16200	32700	26600	26200	26400
2-Dec-04	88	2500	6600	19500	33000	29600	29300	29450
9-Dec-04	95	2500	6600	18300	32900	27500	27000	27250
6-Dec-04	102	2500	6600	18000	32900	28800	26500	27650

Results of Leachate Metals and Chloride Analysis

Initial Data Units:
mg/l

Cu

Date	Day No.	AMF	ASF	ASR	NMR	AMR1	AMR2	AMR
06-Sep-04	1	0.400	0.880	0.670	0.425	0.320	0.540	0.430
21-Sep-04	16	0.715	0.370	0.455	0.530	0.520	0.370	0.445
26-Sep-04	21	0.340	0.225	0.175	0.500	0.180	0.210	0.195
30-Sep-04	25	0.340	0.375	0.350	0.555	0.410	0.430	0.420
16-Dec-04	102	0.340	0.375	0.266	0.500	0.173	0.226	0.200

Zn

Date	Day No.	AMF	ASF	ASR	NMR	AMR1	AMR2	AMR
06-Sep-04	1	1.900	1.182	0.830	2.318	1.955	2.480	2.218
21-Sep-04	16	0.064	0.130	0.094	0.398	0.114	0.158	0.136
26-Sep-04	21	0.054	0.112	0.080	0.320	0.102	0.116	0.109
30-Sep-04	25	0.052	0.118	0.112	0.370	0.120	0.158	0.139
16-Dec-04	102	0.050	0.090	0.075	0.108	0.100	0.110	0.105

Cl^-

Date	Day No.	AMF	ASF	ASR	NMR	AMR1	AMR2	AMR
04-Sep-04	1	1.52	0.85	1.37	1.97	1.22	1.35	1.29
21-Sep-04	16	0.77	1.05	1.40	1.47	1.52	1.95	1.74
26-Sep-04	21	1.00	1.25	1.60	1.67	1.67	2.25	1.96
30-Sep-04	25	0.95	0.92	1.70	1.55	1.70	2.27	1.99
16-Dec-04	102	0.28	0.28	1.14	1.36	1.32	1.44	1.38

ppendix B – Data Adapted from Waste Composition Study 2000/2001 – City of oronto

Waste Audit Results Table

page 1 of 2

Single Family and Multi Family Combined Results – Fall and Winter 6 week survey **Number of households:160**

Waste collection streams		BB	Blue Box	Organics	Garbage	Generation			Recovery
Waste sort categories and descriptions		Y	Net Weight (kg)	Net Weight (kg)	Net Weight (kg)	Total Net Weight (kg)	Percent of Total (%)	kilograms per hhld per year	Blue Box Rate (%)
1. PAPER FIBRES									
Newspaper	ONP, inserts	Y	1624		489.19	2113.10	16.13	114	77%
Magazines & paperpacks	OMG, catalogues, soft covers	Y	274		203.23	477.05	3.64	26	57%
Phone Books	OTB	Y	45		16.51	61.85	0.47	3	73%
Cardboard	OCC	Y	270		255.03	524.37	4.00	28	51%
Boxboard/Rolls	OBB	Y	107		234.54	341.46	2.61	18	31%
Mixed Papers	junk mail, fine household papers	Y	192		478.53	670.68	5.12	36	29%
Molded Pulp	egg cartons, drink trays	Y	6		21.00	27.26	0.21	1	23%
Books	hard covered	Y	7		12.97	19.46	0.15	1	33%
Kraft Paper	paper bags	Y	9		45.72	55.07	0.42	3	17%
Spiral Wound	frozen juice, pringles type packaging		1		18.50	19.74	0.15	1	6%
Tissue/Toweling	tissues, napkins, paper towels		7		395.26	402.51	3.07	22	2%
Other Paper	multi-layered, waxed, wrapping, fast food		19		66.96	86.28	0.67	5	22%
Gable Top Cartons	milk, juice		4		44.73	48.76	0.37	3	8%
Aseptic Containers	tetra type packaging		1		12.10	12.70	0.10	1	5%
Sub-total Paper Fibres			**2,566**	**0**	**2,296**	**4,862**	**37**	**263**	**53%**
2. PLASTICS									
PETE Soft Drink	# 1 soft drink	Y	20		17	37	0	2	54%
LCBO containers	alcoholic beverage containers	Y	10		2	12	0	1	83%
PETE Other	water, juice, food, dish soap, trays		33		29	62	1	3	53%
HDPE bottles	# 2	Y	45		39	84	1	5	53%
PVC	# 3, bottles, packaging		5		9	14	0	1	39%
LDPE & PP Bottles	# 4 and # 5, squeezable		3		14	17	0	1	18%
PS	# 6, trays, cups, packaging		4		94	99	1	5	4%
Recyclable Film	shopping bags, milk pouches,		7		182	189	2	10	4%
Non-Recyclable Film	garbage bags, chip bags, shrink wrap		3		289	292	2	16	1%
Wide Mouth Tubs & Lids	# 2, 4, 5 & 6		7		58	65	0	4	11%
Other Containers	# 7, trays, bottles, unmarked plastics		5		39	44	0	2	12%
Other Plastics	non-pkg, garden hose, VCR tape, toys		10		227	236	2	13	4%
Sub-total Plastics			**152**	**0**	**999**	**1,151**	**9**	**62**	**13%**
3. METALS									
Aluminum Cans	food & beverage cans	Y	27		22	49	0	3	55%
Aluminum Foil Trays	pie plates, etc	Y	4		27	31	0	2	12%
Steel Cans	food & beverage cans	Y	111		92	202	2	11	55%
Aerosol Cans	empty		4		10	14	0	1	30%
Paint Cans	empty		0		5	5	0	0	3%
Other Metal	scrap metal, other containers, bikes		5		144	149	1	8	3%
Sub-total Metals			**151**	**0**	**300**	**450**	**3**	**24**	**33%**

Waste Audit Results Table

Single Family and Multi Family Combined Results - Fall and Winter

Waste collection streams		BB	Blue Box	Organics	Garbage	Generation			Blue Box
Waste sort categories and descriptions		Y	Net Weight (kg)	Net Weight (kg)	Net Weight (kg)	Total Net Weight (kg)	Percent of Total (%)	Kg per hhld per year	Recovery Rate (%)
4. GLASS									
LCBO Clear	clear glass alcoholic beverages	Y	99		56	154	1	8	64%
LCBO Coloured	coloured glass alcoholic beverages	Y	159		39	195	2	11	80%
Clear	food & beverage containers	Y	167		111	278	2	15	60%
Coloured	food & beverage containers	Y	42		17	59	0	4	74%
Other Glass	lightbulbs, window glass, cups, ceramics		15		73	88	1	5	17%
Sub-total Glass			488	0	296	783	6	42	62%
5. HOUSEHOLD SPECIAL WASTES									
Batteries	all types		0		3	3	0	0	
Paint	mostly half to full containers		0		39	39	1	2	
Motor Oil	used oil, filters		0		1	1	0	0	
Flammables	starter fluid, solvents		0		3	3	0	0	
Other HSW	sharps, drugs, acids, antifreeze		0		29	29	0	1	
Sub-total HSW			0	0	75	75	1	4	
6. COMPOSTABLES									
Vegetable Food, Animal Food Waste	vegetable and fruit peelings, meats, fats, oils		2		3,522	3,524	27	191	
Grass, Woody Yard, Other Yard Waste	grass clippings, brush, branches, wood chips, leaves, soil, garden wastes		0		407	407	3	22	
Animal waste	feces, animal litter and bedding		0		320	320	2	17	
Wood ashes	fireplaces & wood stoves		0		0	0	0	0	
Sub-total Compostables			2	0	4,249	4,251	32	238	
7. OTHER WASTE MATERIALS									
Textiles	clothing, shoes		0		276	277	2	15	
Building Renovations	drywall, lumber, carpeting		0		565	565	4	31	
White Goods	large appliances	Y	0		13	13	0	1	
Sanitary Products	diapers, napkins		0		362	362	3	20	
Electronics/Appliances	small appliances, computers, radios		0		105	105	1	6	
Rubber	tires, hoses, tubing		0		21	21	0	1	
Furniture	sofas, chairs, cabinets		0		91	91	1	5	
Other	materials not classified elsewhere		0		87	87	1	5	
Sub-total Other Waste Materials			0	0	1,524	1,525	12	85	
Total weight in kilograms			3,359	0	9,739	13,098		709	
Total percentages by waste type			26%	0%	74%	100%			

Notes:

Yardwaste cannot be assumed to be generated at the same rate (or at all) for multi family as for single family. Do not use the generation per unit amount noted here.

pendix C – Properties of Biosolids

ronto Works & Emergency Services Department: Wastewater Quality Laboratory Services

hbridge’s Bay Treatment Plant Biosolids Demonstration Project: Laboratory Report – Year

01

tal solids in sludge = 2.0%

latile total solids = 43.0%

mmonia nitrogen (as N) in sludge = 777 mg/l

itrate nitrogen (as N) in sludge = 0.1 mg/l

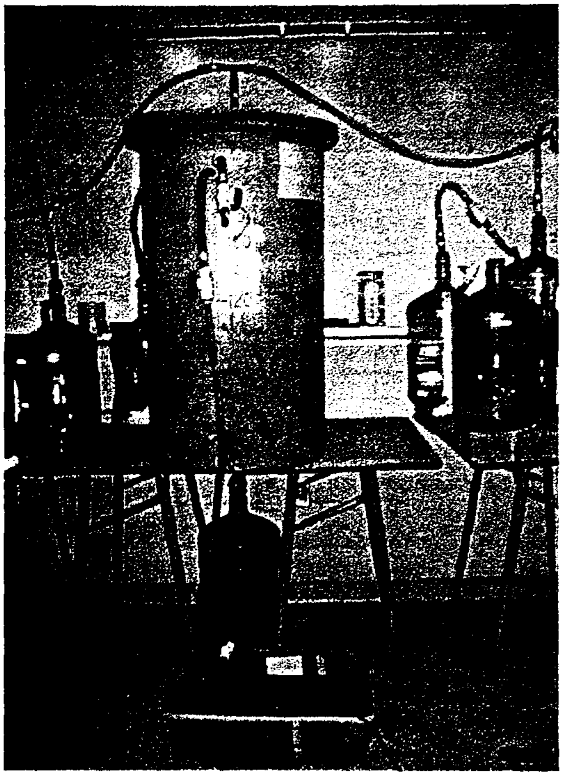
otal solids in cake = 28.4%

olatile total solids = 36.2%

otal phosphorus (as P) in cake = 15,600 mg/l

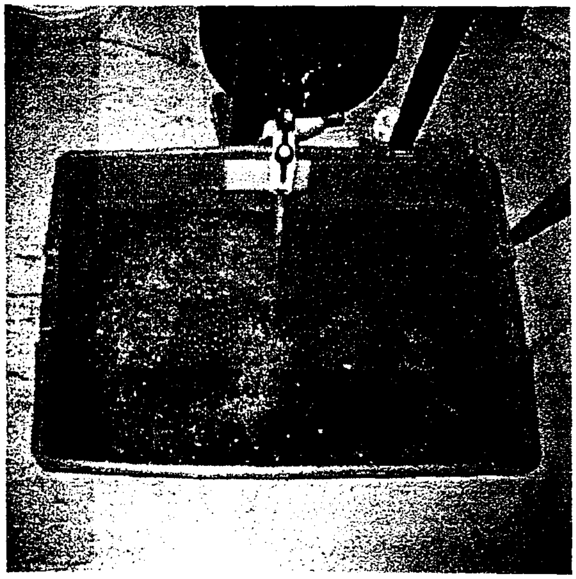
KN (as N) in cake = 39,600 mg/l

Appendix D – Supplementary Photographs

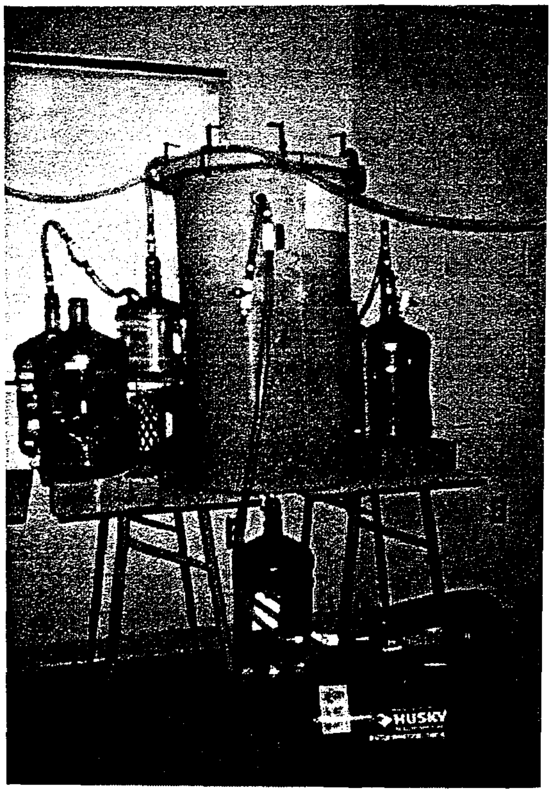


(a) Leachate Recirculation

(b) Waste Foods and Vegetables



(c) Leachate Discharge



(d) Air Injection