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Comparative Study of Biodegradation of Municipal Solid Waste in Simulated Aerobic and Anaerobic Bioreactor Landfills

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COMPARATIVE STUDY OF BIODEGRADATION OF MUNICIPAL SOLID WASTE IN SIMULATED AEROBIC AND ANAEROBIC BIOREACTOR LANDFIILS

by

Septa Rendra

A thesis submitted to
the Faculty of Graduate and Postdoctoral Studies
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ABSTRACT

The biodegradation of municipal solid waste (MSW) was investigated in simulated bioreactor landfills under aerobic and anaerobic conditions. The bioreactors were operated to determine the amount of leachate recirculation and municipal wastewater sludge addition to optimize waste degradation. The leachate generated was recycled over 47 and 63 weeks for aerobic and anaerobic bioreactors, respectively. Leachate samples were collected on a weekly basis and analyzed for pH, biochemical oxygen demand (BOD), chemical oxygen demand (COD), total Kjeldahl nitrogen (TKN), ammonia nitrogen (NH_3-N), total phosporus, and metals. The temperature of the MSW in the bioreactors was measured on a daily basis. In addition, the generation of biogas was monitored in the anaerobic bioreactors during the operating period.

The leachate generated was recirculated at the rates of 285 to 855 mL/kg of MSW.d (5 to 15 L/wk) and sludge was added at the rates of 28.5 to 85.5 mL/kg of MSW.d (0.5 to 1.5 L/wk). Within 27 and 39 weeks enhanced MSW degradation in the aerobic and anaerobic bioreactors were observed at a leachate recirculation rate of 855 mL/kg of MSW.d and sludge addition rate of 85.5 mL/kg of MSW.d. During this period, the COD concentration in the leachate dropped from 38,000 mg/L for aerobic and 47,000 mg/L for anaerobic to approximately 1000 mg/L. This is an indication that the aerobic biodegradation is 1.5-fold faster compared to the biodegradation under anaerobic operation. A reduction in the leachate recirculation and sludge addition rate to 285 and 28.5 mL/kg of MSW.d respectively, increased the waste stabilization period up to 45 and 63 weeks for aerobic and anaerobic bioreactors, respectively.

The statistical empirical models based on two levels factorial design were used to describe the effects of leachate recirculation, sludge addition and their combination on biodegradation of the waste. For both aerobic and anaerobic bioreactors the values of

estimate parameter β_1 were higher compared to β_2 and β_{12}. This indicated that the effect of leachate recirculation was much stronger compared to the effect of sludge addition and their combination on biodegradation of the waste.

A statistic procedure, F-test and ANOVA-test were used to determine whether or not there is a significant difference between the aerobic and anaerobic biodegradation. The result of the paired F-test show that F calculation was 270.85 and F critical was 160 at a 95% confidence level. This confirms that there was a significant difference between the aerobic and anaerobic biodegradation. In addition, the ANOVA test show that effect of air flow addition on the MSW biodegradation was very significant. The results of these tests indicated that the addition of air affected positively the biological activities and consequently enhanced the MSW biodegradation process.

A fuzzy logic model, describing the dynamics of the biodegradation and stabilization process during the experiment, was developed to simulate the effect of leachate recirculation and sludge addition on MSW biodegradation in landfills. The model was based on the COD concentration of the leachate, temperature of the MSW in the bioreactors, and biogas production from anaerobic bioreactors. Subsequently, the model was evaluated by comparing the simulation with the experimental results. The model shows that the higher rate of leachate recirculation and sludge addition, the faster biodegradation of MSW. In addition, the model could be used to predict the rate of MSW biodegradation under various operating conditions.

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GLOSSARY

AER	Aerobic Bioreactor
ANA	Anaerobic Bioeactor
AIR	Air flow at constant rate
ANOVA	Analysis of Variance
BD	Below Detection limit
BOD	Biochemical Oxygen Demand at 5 days incubation
COD	Chemical Oxygen Demand
C/N	Carbon-to-Nitrogen ratio
CR	Medium of leachate recirculation
CS	Medium rate of sludge addition
DF	Degree of Freedom
FIE	Fuzzy Interference Engine
FIS	Fuzzy Inference System
FLC	Fuzzy Logic Controller
GHG	Greenhouse Gases
HR	High of leachate recirculation
HS	High of sludge addition
ICP	Inductively Coupled Plasma
IPCC	Intergovernmental Panel on Climate Change
LFG	Landfill gas
LR	Low rate of leachate recirculation
LS	Low rate of sludge addition
MC	Moisture Content

MSE	Mean Square Error
MSW	Municipal Solid Waste
SSE	Sum Square Error
TC	Total carbon
TKN	Total Kjeldahl Nitrogen
TP	Total Phosporus
TS	Total Solids
U.S. EPA	United States Environmental Protection Agency
VS	Volatile Solids
VOA	Volatile Organic Acids

Btu	British thermal unit
ppbv	parts per billion (by volume)
ppm	parts per million (by volume)
Tg eCO_2	teragrams of carbon dioxide equivalents
MC	Moisture Content
AER	Aerobic Bioreactor
ANA	Anaerobic Bioeactor
TC	Total carbon
TKN	Total Kjeldahl Nitrogen
TP	Total Phosporus
BOD	Biochemical Oxygen Demand at 5 days incubation
COD	Chemical Oxygen Demand
α	pProbability level of statistical test
LFG	Landfill gas
FIS	Fuzzy Inference System
FLC	Fuzzy Logic Controller
FIE	Fuzzy Interference Engine
η_i	,i^{th} rule base
IPPC	Intergovernmental Panel on Climate Change
ppbv	parts per billion (by volume)

ppm	parts per million (by volume)
GHG	Greenhouse Gases
Tg eCO_2	teragrams of carbon dioxide equivalents
Btu	British thermal unit
MSW	Municipal Solid Waste
VOA	Volatile Organic Acids
C/N	Carbon-to-Nitrogen ratio
$\hat{\sigma}^2$	estimate of pure pure error variance
β	parameter estimate
λ	sensitivity parameter for leachate recirculation
γ	sensitivity parameter for sludge addition
x_1	coded variable for leachate recirculation
x_2	coded variable for sludge addition
$\hat{Y}$	process response
H_0	null hypothesis
H_A	alternate hypothesis

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CHAPTER 1

INTRODUCTION

1.1 Background

A major problem associated with solid waste landfilling practice is the generation of leachate. Leachate is the result of water percolation through the solid waste matrix. Its volume depends mainly on rainfall and melted snow percolating into the solid waste as well as the original water content of the solid waste. The leachate has a high concentration of organic and inorganic pollutants, including heavy metals and toxic compounds (Ozkaya *et al.*, 2006; El-Fadel *et al.*, 2003; Tatsi and Zouboulis, 2002; Flyhammar *et al.*, 1998; Reinhart and Townsend, 1998). A major hazard associated with leachate is its potential to contaminate groundwater and surface water.

The production of landfill gas (LFG) poses another environmental problem. LFG is mainly composed of carbon dioxide and methane, these are two greenhouse gases (GHG). Methane usually accounts in the range of 50 to 60 percent by volume of the total gas production in a landfill (U.S. EPA, 1994, Tchobanoglous *et al.*, 1993). It is colorless, odorless, and highly combustible when mixed with sufficient amount of oxygen. Carbon dioxide on the other hand is also colorless and odorless but non combustible. However, high concentrations of carbon dioxide will tend to acidify groundwater via the production of carbonic acid. Often this is a good indicator that interaction of the groundwater with LFG is occurring and therefore affecting the cabonate-bicarbonate chemistry of the subsurface environment (Kraemer *et al.*, 1998).

In light of the above mentioned issues, it is necessary to address the environmental problems caused by the operation and post-closure of a landfill site. Leachate recirculation as well as LFG recovery and utilization are the methods commonly used to control these problems (Reinhart and Townsend, 1998; Reinhart and Al-Yousfi, 1996; Barlaz, *et al.*, 1992).

1.2 Statement of the Problem

1.2.1 Landfill Gas Contribution to Global Warming

Carbon dioxide, the most important GHG, is mainly produced from human activities. Reducing the emission of carbon dioxide could augment the contribution of another GHG, methane, which is second only to carbon dioxide in its contribution to global warming. Methane is also the main component of the natural gas and is a valuable energy source. Although a recent study by Intergovernmental Panel on Climate Change (IPCC, 2001) shows that the concentration of methane in the atmosphere was very small, 1750 ppb, compared to carbon dioxide, 370 ppm in 2000, it could turn out to be the major GHG in the future.

In 2004, Canada's total GHG emissions reached an estimated 758 teragrams of carbon dioxide equivalents (Tg eCO_2), and this increased 27 percent from 1990 to 2004 (Environment Canada, 2006). At the same time, total U.S. GHG emissions were 7,074.4 teragrams of carbon dioxide equivalents. Overall, total U.S. emissions have risen by 15.8 percent from 1990 to 2004 (U.S. EPA, 2006). In Canada, total GHG emissions from landfills were estimated to be 22 Tg eCO_2, accounting for about 3 percent of the national GHG emissions.

According to IPCC (2001) and Hofmann (2004) methane is more than 21 times as effective as carbon dioxide at trapping heat in the atmosphere. Over the last two hundred and fifty years (1750 to 2004), the concentration of methane in the atmosphere increased by 143 percent. Methane produced at landfills contributes approximately 3 to 4 percent to the annual global anthropogenic greenhouse gas emissions (IPCC, 2001). In Canada and the U.S.A., methane from landfill sector is responsible for approximately 23 and 25 percent of total methane emissions, respectively (Environment Canada, 2005; U.S. EPA, 2006).

It has become apparent that in order to find solutions addressing the impact of GHG on global warming, an engineering approach is required. Since methane is a source of energy as well as a GHG, the reduction of methane emissions from landfills could be an economically valuable option. Reduction in LFG emissions can be achieved by promoting energy recovery rather than the flaring of methane.

1.2.2 Landfill Gas as an Energy Source

Energy from LFG is a renewable energy source that can be utilized as an alternative energy in the future. The energy value of unprocessed LFG is about 500 Btu /ft^3 (US. EPA, 1994). This energy value is approximately half that of natural gas, primarily because only about half of LFG is methane. Higher energy value can be achieved by processing LFG, and can result in a heat value of approximately 1000 Btu /ft^3 (US. EPA, 1994). The recovered LFG can be used to run a generator that converts gas to electrical energy.

Energy recovery is being used at some landfill sites, particularly at large landfills, over 3 million ton of waste, where the magnitude of the operation and the potential life span of the project make energy recovery economically feasible. (US. EPA, 1994). However, the recovery and utilization of methane at a reasonable cost depends on the availability of a technology as well as the quality and volume of LFG, which is highly dependent on the nature and characteristic of the landfill (US. EPA, 1994).

The production of LFG can vary with time; thus, energy recovery and utilization systems are designed to run on less than the maximum amount of available LFG. Generally, landfills that are closed for fewer than 5 years are the best candidates for energy recovery since the ability of a landfill to generate gas decreases over time (US.

EPA, 1994). In addition, due to the high moisture of LFG and with the presence of trace corrosive gases, the collected gas should be pretreated prior to use.

1.2.3 Leachate Recirculation

Excess moisture collected at the bottom of a sanitary landfill as the leachate is usually transported off-site for treatment. An alternative to off-site treatment is to recycle this leachate back into the landfill, which enhances the opportunity for natural attenuation to perform the bulk of the biochemical transformations processes. The recirculated leachate increases the moisture within the waste accelerates the biodegradation of solid waste and reduces the time required for stabilization. It is desirable to minimize the time period in which waste biodegradation occurs in order to reduce gas emissions after the landfill is closed.

Full-scale applications of leachate recirculation in municipal solid waste landfills are very few. This is due to most of landfill were not designed as bioreactor landfills. Therefore, the actual operating data describing performance of current designs are also quite limited. Since leachate recirculation has not been applied in practice on a widespread basis, single pass leaching (no leachate recirculation) systems continue to prevail, even though data from various studies indicate that leachate recirculation strategies create a more rapid development of active microbial populations, higher reaction rates, and compressed time for stabilization (Pohland and Al-Yousfi, 1994). Leachate recirculation also extends the moisture retention time of the landfill. By recycling the leachate, trace nutrients, organic materials, and other substances are returned to the landfill for in-situ conversion. As a results, the organic strength of leachate concentrations increases at the beginning and eventually will decrease with time. In addition, leachate recirculation decreases both the total volume and the strength of leachate that requires ultimate treatment prior to or after landfill closure (Rendra *et al.* 2006, 2003; He *et al.*, 2005)). Other advantages of leachate recirculation include improved distribution of nutrients and enzymes, pH buffering, dilution of inhibitory compounds, recycling and distribution of methanogens, liquid storage, and evaporation opportunities (Reinhart, 1996).

An apparent disadvantage of leachate recirculation is the additional capital and operating/maintenance costs that are required for re-distribution systems, associated pumps, and appurtenances as well as operation difficulties. These costs however could be offset by a reduction of off-site transport and treatment requirements. Since the landfill will take less time to reach final maturation, the amount of time required for analysis and management of leachate and gas are both decreased.

1.3 Bioreactor Landfills

There are several interdependent factors that influence the rate of biodegradation in landfill, and those must be considered. The use of buffers and nutrients, the addition of sludge, the reduction of waste particle size, and moisture content management are factors that can be manipulated in order to enhance waste biodegradation. The direct end product of solid waste degradation is landfill gas production. Thus, a bioreactor landfill, which has the potential to rapidly stabilize the waste, will also have rapidly enhanced gas production. The operation of MSW landfills as bioreactors has been practiced to some extent at a number of landfill sites in Canada and the United States (Warith, 2002; Yazdani et al., 2002; Reinhart and Townsend, 1998). The level to which bioreactor landfill operations have been implemented, however, were commonly limited to some form of leachate recirculation. Thus, experience is minimal with regard to: (a) controlling or monitoring the treatment process occurring within the landfill, and (b) the impact of leachate recirculation has on the internal landfill system. In most cases, leachate recirculation is practiced as a novel approach for managing leachate without using the landfill as a treatment system.

Landfills utilizing leachate recirculation have not been specifically designed to operate as a leachate treatment systems. The full-scale implementation of bioreactor technology has not yet been established, and optimal design and operation methods for bioreactor landfill treatment systems have not been developed. The absence of operating models has limited the widespread use of this technology for solid waste management.

Investigations have been performed in order to study bioreactor landfill operations at the full-scale level beyond the simple operation of leachate recirculation systems. One of the first bioreactor landfill systems was operated at the Southwest Landfill in Alachua County, Florida. University of Florida researchers operated segments of the existing sanitary landfill as a bioreactor treatment system. The landfill however was not designed with leachate recirculation in mind. A variety of leachate recirculation methods were employed (Townsend *et al.*, 1995). Solid waste samples were collected from treated and untreated areas within the landfill, and the degree of waste treatment was measured. This research demonstrated that waste treatment was rapidly accelerated by leachate recirculation.

1.4 Scope and Objectives of the Study

Due to the uncertainty of solid waste characteristics, as well as the complex processes taking place within the landfill, it is difficult to assess the effect of separate factors influencing the degradation of solid waste in-situ. To simplify the problem, laboratory scale systems were used to facilitate the investigation of individual factors that affect waste biodegradation. In the laboratory-scale systems, the complexity and uncertainty in an actual landfill can be eliminated. In addition, laboratory-scale systems have the advantages of standardization, ease of control, and manipulation. However, these systems should be operated under conditions as close as possible to those in an actual landfill.

Presently, there is limited information in the literature regarding the use of aerobic landfill as a method of solid waste disposal. In the aerobic landfill, the addition of air, and thus oxygen, promotes the aerobic stabilization of the landfilled waste. This is the same process that occurs in a waste composting system. However, aerobic composting is accomplished within the landfill rather than in windrows or other methods in which composting is normally practiced. In addition, significant waste fractions such as lignin and ligneous materials, which are not degradable anaerobically can be degraded aerobically (Stessel and Murphy, 1992). It is apparent that the aerobic bioreactor landfill has some advantages over the anaerobic bioreactor landfill; however, there is no experimental data available to support this. Thus, it is necessary to know whether the aerobic bioreactor results in a more consistent and more effective approach than the anaerobic approach.

An evaluation of the use of aerobic and anaerobic processes for the rapid stabilization of MSW in simulated bioreactor landfills is the major focus of this research. In addition, this research also examined the effect of two factors, leachate recirculation and sludge addition, in order to optimize the MSW biodegradation. Another aspect of this research is to evaluate the use of anaerobic bioreactor landfills technology and compare them to the use of aerobic bioreactor landfills technology in term of the biodegradation rate.

In order to facilitate the prediction of bioreactor landfill performance under long term operation and its life expectancy, a fuzzy logic model simulation of leachate quality in terms of COD concentrations was carried out. In addition, statistical empirical model was used to evaluate the effect of leachate recirculation and sludge addition on the MSW biodegradation.

Specific objectives of this research are to:

1. Establish the optimal conditions for operating bioreactor landfills under aerobic and anaerobic conditions.
2. Evaluate the effect of leachate recirculation and sludge addition using statistical approach.
3. Compare the rate of municipal solid waste municipal solid waste biodegradation and process under aerobic and anaerobic conditions.
4. Predict the performance of bioreactor landfills with the aid of a fuzzy logic model.

1.5 Organization of the Thesis

This thesis is divided into six chapters. The first chapter provides an introduction and presents the objectives of the research. Chapter Two is a literature review that summarizes some of the biological principles involved in landfill and describes aerobic and anaerobic landfill. Chapter Three presents the experimental set-up and operation as well as the materials and analytical methods used in the experiments. Chapter Four presents analysis, discusses significant performance results, and develop statistical empirical model. Modeling biodegradation using Fuzzy Logic are presented and discussed in Chapter Five. Finally, the conclusions, recommendations, and contributions are presented in Chapter Six and followed by References and Appendices.

CHAPTER 2

LITERATURE REVIEW

2.1 Solid Waste Degradation Sequence

A consortium of bacteria will directly degrade solid waste disposed in a sanitary landfill. Degradation begins aerobically for some short period after the waste has been placed in a landfill until the oxygen is depleted. When this is completed, anaerobic degradation begins and goes on for a long period. Anaerobes are microorganisms that do not use oxygen for cellular respiration. In anaerobic respiration, microorganisms like bacteria derive energy from organic (carbon-containing) molecules by using an inorganic molecule other than oxygen as an electron acceptor, typically nitrate or sulphate.

As with other anaerobic processes, waste degradation in landfill involves several communities of bacteria such as hydrolytic bacteria, fermentative bacteria, acetogenic bacteria, methanogenic bacteria, and sulphate-reducing bacteria (Lay *et al.*, 1998b; Kim *et al.,* 1997; Christensen and Kjeldsen, 1989; Barlaz *et al.*, 1989a; Zaid *et al.*, 1986). These bacteria are responsible for the conversion of organic wastes into organic acids and ultimately into methane and carbon dioxide. Fermentative bacteria hydrolyze and ferment carbohydrates, proteins, and lipids to volatile organic acids. Acetogenic bacteria then convert these organic acids to acetate, CO_2, and H_2. These products are in turn converted to methane and carbon dioxide by a special group of bacteria known as methanogens. The changes in landfill gas as well as leachate quality reflect the progress of waste stabilization. The composition and rate of gas generated and leachate produced vary from one stage to another (San and Onay, 2001.; Katsiri *et al.*, 1999; Lay *et al.*, 1998a; Lay *et* *al.*, 1998b; Bae *et al.*, 1998; Barlaz, *et al.*, 1992; Christensen and Kjeldsen, 1989). Anaerobic conditions develop naturally in nearly all landfills without any intervention. There are five sequential and distinct phases in the degradation of solid waste. Figure 2.1 show the phase of wastes degradation.

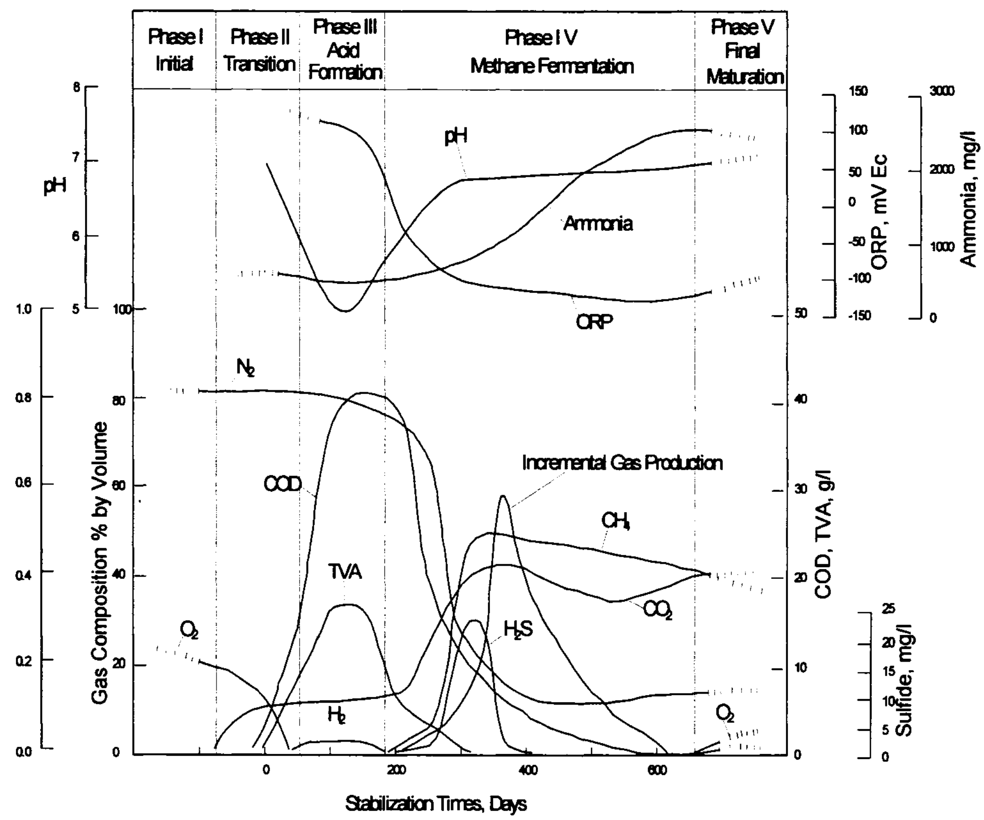

Figure 2.1 Stabilization patterns in landfill (Adapted from Pohland and Kim, 1999)

2.1.1 Phase I: Initial adjustment phase

This is a short aerobic decomposition phase that occurs immediately after placement of solid waste and accumulation of moisture within landfills. In this phase, easily degradable organic matter is oxidized and is converted into carbon dioxide in the presence of oxygen.

2.1.2 Phase II: Transition phase

In phase II, the field capacity is sometimes exceeded and absence of oxygen will support the development of facultative and anaerobic microorganisms. A great variety of commons types of facultative and anaerobic microorganisms mediate a wide variety of biochemical reactions. In this process, complex particulate matter is hydrolized into lower molecular weight dissolved compounds. Carbohydrates are degraded to simple sugars, proteins are converted amino acids, and lipids are converted to long chain fatty acids and glycerine. By the end of this phase, measurable concentration of volatile organic acids (VOA) and chemical oxygen demand (COD) in the leachate can be detected (Reinhart and Townsend, 1998; Tchobanoglous *et al.*, 1993).

2.1.3 Phase III: Acid formation phase

The first stage of acid formation is acidogenesis in which dissolved compounds produced in the hydrolysis stage are taken up in the cells of fermentative bacteria. In this stage, fermentative bacteria convert amino acids and simple sugars into acetic acid, volatile acids, hydrogen, carbon dioxide, ammonia, and hydrogen sulphide gas. In the absence of oxygen, nitrate and sulphate act as electron acceptors. In microbial conversion reactions nitrate is often reduced to nitrogen gas and sulphate is converted to hydrogen sulphide. The microorganisms involved in this conversion, described collectively as nonmethanogenic, consist of facultative and obligate anaerobic bacteria.

The acidic leachate with a pH of less than 6.5 may contain high concentrations of fatty acids, calcium, iron, heavy metals, and ammonia (Al-Yousfi, 1992; Christensen and Kjeldsen, 1989; Doedens and Cord-Landwehr, 1989). As the redox potential drops, the initial high concentration of sulphate may slowly reduce. Precipitation of sulphides of iron, manganese, and heavy metals, may also occur when sulphide is generated (Flyhammar *et al.*, 1998; Christensen and Kjeldsen, 1989). Generation of carbon dioxide and hydrogen will reduce the content of nitrogen in the gas phase.

The acidogenesis stage is followed by acetogenesis in which the products of acidogenesis (acetate, hydrogen, and carbon dioxide) are converted into the final products for methane production. These products will be converted to methane and carbon dioxide in the presence of methane-forming bacteria. In some cases, these microorganisms will slowly start to develop toward the end of the acetogenesis stage.

2.1.4 Phase IV: Methane formation phase

During phase IV, acetate, hydrogen, and carbon dioxide are consumed by a consortium of methanogenic bacteria and are converted into methane and carbon dioxide. The steady production of methane is characteristic of this phase and it constitutes approximately 50 to 60 % by volume of the LFG composition. The high rate of methane formation maintains low concentrations of volatile fatty acids and hydrogen. As a result, the pH within the landfill will rise to more neutral values in the range of 6.3 to 8 (Reinhart and Al-Yousfi, 1996; Al-Yousfi, 1992; Ehrig, 1989, Pohland and Harper, 1986). At this higher pH, very small amounts inorganic constituents can remain in solution. As a result, the concentration of heavy metals present in the leachate will be reduced by complexation and precipitation.

2.1.5 Phase V: Maturation phase

In this phase, the generation rate of LFG decreases significantly and leachate strength remains steady at low values. This is due to the fact that most of the biodegradable organic fractions and nutrients were removed in the previous phase. However, recalcitrant wastes are still being decomposed and produce humic subtances (Reinhart and Townsend, 1998). Due to the low methane generation rate, nitrogen and oxygen from the atmosphere diffuse into the landfill. Therefore, small amounts of nitrogen and oxygen can be found in the landfill gas during the maturation phase. (Warith and Sharma, 1998; Christensen and Kjeldsen 1989).

2.2 Factors Affecting Biodegradation

Important environmental factors that affect solid waste degradation are the presence of oxygen, temperature, pH, sulphate, the presence of essential nutrients, the absence of excessive concentrations of toxic compounds in the waste, and moisture (Rendra *et al.* 2006, 2003, Warith, 2002; Warith *et al.*, 1999; San and Onay, 2001; Kim *et al.*, 1997; Reinhart, 1996; Christensen and Kjeldsen, 1989; Barlaz *et al.*, 1990; Pohland, 1980). These abiotic factors develop the consortium of bacterial groups that participate in the degradation of solid waste. Brief descriptions of these factors are discussed below.

2.2.1 Oxygen

Methanogenic bacteria are the most sensitive to the presence of oxygen because these bacteria are obligate anaerob. From the atmosphere, a certain amount of air will always diffuse into and be trapped within the landfill. Vacuum conditions may occur in the landfill because of extensive gas recovery pumping. This may extend the aerobic zone in the top of the landfill and eventually preclude methane generation in these layers. However, aerobic bacteria in these layers will easily use the oxygen and limit the aerobic zone to less than 1 m of compacted waste (Christensen and Kjeldsen, 1989).

2.2.2 pH

The value and stability of pH in the landfill is extremely important because the methanogenic bacteria operate only within a pH range of 6 to 8 (Warith, *et al.*, 1999, Barlaz *et al.*, 1989b). Delolme *et al.* (1998) reported that the acetogenesis and methanogenesis phases of waste degradation were not reached at a pH range of 5 to 6. Acidogenic bacteria are significantly less sensitive to low or high pH values and hence acid fermentation will prevail over methanogenic fermentation. The transformation of hydrogen and acetic acids will decrease if the activity of methanogenic bacteria is low. Consequently, the hydrogen pressures build up and pH decreases. Due to this build-up, acetogenic bacteria cannot convert volatile fatty acids, particularly butyric and propionic acids to methane and carbon dioxide, which will further lower the pH within the landfill. Eventually, methane production reduces substantially.

2.2.3 Sulphate

Sulphate reducing bacteria cause major problems in the landfill stabilization process, especially when the solid waste contains significant amounts of sulphate. These bacteria always take place prior to methane production based on energy required for this reaction. Sulphate reducing bacteria are all obligate anaerobes. It was reported by Kim *et al.* (1997) that in the sulphate reducing reactor of laboratory landfill simulators, methane production was effectively suppressed. Other investigations have also shown the same results (Campbell, 1993; Stegman and Spendlind, 1989). The reduction of methane production by sulphate is not related to any toxic effect of sulphate on methane-forming bacteria but mainly due to substrate competition (Campbell, 1993).

2.2.4 Nutrients

In anaerobic digestion, nutrients such as nitrogen and phosphorus must be present in sufficient amounts. The maintenance of optimal compositions of organic carbon and nitrogen is critical for cell synthesis and metabolism in anaerobic landfill stabilization. For example, a high ratio of carbon-to-nitrogen (C/N) in the waste causes potential problems affecting microbial growth due to lack of nitrogen. In contrast, a low C/N ratio could result in problems with ammonia toxicity. The inhibitory effects of ammonia are caused by free ammonia and hence are increasing with increasing pH. In landfill environments the concentration of total ammonia become quite toxic if exceeds 3,000 mg/L (Christensen and Kjeldsen, 1989). Similar to the C/N ratio, carbon and phosphorus ratio must be optimized. The optimum ratio between carbon, nitrogen, and phosphorus ratio is listed as 100:0.44:0.08 (Stegman and Spendlind, 1989; Christensen and Kjeldsen, 1989).

Other micro nutrients such as sulphur, calcium, magnesium, potassium, iron, zinc, copper, cobalt, molybdate, and selenium, are also necessary for metabolism of anaerobic microorganisms. These nutrients can be found in most sanitary landfills. Generally, nitrogen and phosphorus will not be limiting in the waste matrix, but insufficient homogenization of the waste may result in nutrient-limited zones (Christensen and Kjeldsen, 1989).

2.2.5 Inhibitors

The inhibitory effects of oxygen, hydrogen, and sulphate in methane production have already been described. There are several other compounds that affect the anaerobic degradation process, even at a very low concentration, such as heavy metals and chloro-organic compounds. In addition, carbon dioxide, salt ions, and sulphide can potentially inhibit methanogenic bacteria. The degradation rate of acetate rapidly decreases when the partial pressures of carbon dioxide falls in the range of 0.2 to 1.0 atmospheres (Hansson and Molin, 1981; Cossu *et al.*, 1993). Several cations such as sodium, potassium, calcium, magnesium, and ammonium have been reported to stimulate anaerobic degradation at low concentrations but inhibit it at high concentrations (Warith *et al.*, 1999).

2.2.6 Temperature

The anaerobic waste degradation, like other biological processes, strongly depends on temperature. As temperature increases, the rate of reaction is much faster. With respect to the waste biodegradation process, the maximum temperature must be between 30 to 40^0C for the mesophilic range and 50 to 60^0C for the thermophilic range. It was also reported that in laboratory landfill reactors, the methane production rate increased significantly (up to 100 times) when the temperature was raised from 20 to 40^0C (Christensen and Kjeldsen, 1989).

The heat generated in a deep landfill with a moderate water flux may cause a temperature rise in the landfill. This is because the heat flux from the landfill to the surrounding environment is small due to insulating waste matrix (Rees, 1980). Landfill temperatures of 30-45^{0}C can be expected even in temperate climates. It should be noted that this range of temperatures might be found in some landfill sites, but may not be found in others.

2.2.7 Moisture Content (MC)

Barlaz *et al.* (1987) reported that a higher moisture content (55%) in shredded municipal solid waste leads to a more rapid accumulation of acid, hindering methane production. They also found that there was no methane production at moisture contents below 35 %. A laboratory investigation by Lay *et al.* (1998c) showed that a moisture content of 75% was observed to be necessary for initial rapid decomposition for the methane production. They used non-shredded organic fraction of municipal solid waste in simulated landfill bioreactor.

The advantage of increased water content in the landfill is that it limits oxygen transfer from the atmosphere. Other benefits include the facilitated exchange of substrate, nutrients, buffer, and dilution of inhibitors and spreading of microorganisms within the landfill. Moisture content can be increased through leachate recirculation.

2.3 Methods of Enhancing Degradation

Numerous landfill studies, both in laboratory and full-scale, have been conducted to investigate the methods for enhancement of solid waste degradation. As mentioned earlier, several factors influence biodegradation of waste in landfill. These factors can be manipulated to enhance biodegradation in a landfill. These manipulation techniques include moisture content management, the use of buffers, nutrients addition, reduction of waste particle size, and sludge addition. Moisture content can be most practically controlled through leachate recirculation. The use of buffer is for controlling the pH (6.5

to 7.5), and the use of shredded waste is to increase surface area and implies some mixing of the solid waste associated with the size reduction. Well-mixed, shredded solid wastes permit greater contact between components required for methane production such as moisture, substrate, and microorganisms. Nutrients are usually present in sufficient quantities in the waste, hence nutrients will only be added when the waste contains insufficient nutrient. The impacts of these parameters on enhancement of waste biodegradation are described below.

2.3.1 Leachate Recirculation

Some studies have been conducted to investigate the effects of leachate recirculation on biological waste stabilization, leachate quality, gas production, waste settlement, and other factors (Morisa *et al.*, 2003; San and Onay, 2001; Warith *et al.*, 1999; Pohland and Kim, 1999; Bae *et al*, 1998; Reinhart and Al-Yousfi, 1996; Townsend *et al.*, 1996; Pohland and Al-Yousfi, 1994; Stessel and Murphy, 1992; Barlaz *et al.*, 1992). The general conclusion of all these studies was that waste stabilization can be achieved by leachate recirculation.

Bae *et al.* (1998) operated three lysimeters to investigate the effects of leachate recirculation on methane production from solid waste. These lysimeters were designed as a control reactor (L-control), a recycle reactor (L-recycle), and a sludge reactor (L-sludge). Each lysimeter was air tight and was filled with 114 kg (wet weight) of shredded solid waste with a density of 0.7 ton/m^3. In the L-control, no leachate recycling was carried out, and only received simulated rainfall. A combination of simulated rainfall and leachate recirculation was applied to the other lysimeters. Sludge from a sewage treatment plant was added to the L-sludge lysimeters. All lysimeters were operated at 35 ^{0}C. The methane gas production rate observed in the lysimeters was used as a measure of the waste stabilization. It was found that gas production in the L-sludge lysimeter was the most significant of all lysimeters. Cumulative gas production from L-sludge reactor was 10-30 times greater than that from the other lysimeters. In addition, the test also indicated that the L-recycle reactor produced about two times more methane than L-control reactor, indicating that leachate recirculation has a positive effect on methane production. The L-sludge reactor, however, produced 1.3 times more methane than the L-recycle reactor.

The conclusions from above study were in agreement with previous reports, which indicated that leachate recirculation without nutrient addition is ineffective in enhancing methane production (Barlaz *et al.* 1989b; Doedens and Cord-Landwehr 1989). However, it was noted that leachate recirculation enhanced microbial activity by providing better contact between the microorganisms, substrates and nutrients.

Warith *et al.* (1999) operated three laboratory-scale landfills to investigate the effects of leachate recirculation on waste degradation. The recirculated leachate was supplemented with two types of material to enhance waste stabilization. Three PVC landfill models were used, namely; a control cell (C cell), a buffer and nutrients cell (B&N cell) and a sludge cell (S cell). Each cell had a diameter of 0.7 m and a height of 1.2 m, and were filled with municipal solid waste. The wastes were broken down to a diameter of about 75 mm prior to loading of the landfill model. Thereafter, the wastes were compacted to a density of 278 kg/m^3. The initial moisture content of the wastes were 9.8%, and then increased to field capacity (45%) by adding tap water to the waste daily. Leachate recirculation was applied for all cells. In the S cell, the amount of municipal sewage sludge added to the recirculated effluent was 5% of the total leachate volume. The S cell was also operated similar to the C cell and B&N cell. All of the cells were air tight and equipped with leachate collection and redistribution systems.

The experiments showed that the concentration of the COD in the leachate increase rapidly in the S cell. A COD peak concentration of about 65,000 mg/L was detected in the S cell after about 30 weeks, and the leachate from the B&N cell exhibited a COD concentration of about 52,000 mg/L. The COD concentration in the C cell was about 32,000 mg/L. After a period of about 65 weeks, the COD concentrations of about 17,000 mg/L, 21,000 mg/L, and 25,000 mg/L, were detected in the S cell, B&N cell, and C cell, respectively. These COD concentrations indicated that the degradation of the waste was achieved more rapidly in the S cell. While in the B&N cell, the degradation rate was lower than in the S cell, but it was higher compared to the C cell. This indicated that nutrient and buffer addition have a positive effect on the rate of biological degradation. Settlement measurement results indicated that the solid waste reduction in volume was in the range of 16-20.2% of the initial solid waste volume. The highest degree of settlement was found in the B&N cell (20.2%), while in the S and C waste settlement were at 18.6% and 16%, respectively.

2.3.2 Particle Size

The significance of the particle size is mainly due to the amount of surface area of the waste particles exposed to microbial interaction. The higher ratio of the surface area to volume, the higher microbial activity within the landfill. As a result, waste biodegradation will also accelerate.

Oteino (1989) reported that shredding waste with a moisture content of 44% and a density of 306 kg/m^3 increased the biodegradation rate, thereby producing better quality leachate in a shorter period of time. In contrast to the work of Oteino (1989), Tittlebaum (1982) worked with shredded refuse with a moisture content of 70% and a density of 240 kg/m^3. Tittlebaum concluded that the shredding of waste did not have any significant effect on waste degradation. This is further supported by the work done by Barlaz *et al.* (1990 and 1989b), they reported that refuse with a particle size ranging from 250 mm to 350 mm produced 32% more methane after 90 days, compared to the refuse with a particle size ranging from 100 mm to 150 mm. The reason for this result is that further stimulation by a reduced particle size may have caused a rapid rate of hydrolysis, resulting in accumulation of acidic end products and a lower pH. The acidic conditions would then limit the activity of microorganisms and consequently affect methane production and waste biodegradation rates.

2.3.3 Temperature

Potential effects of temperature are important because of the varying ranges found in actual landfills. The direct effect of temperature on microbial activity can be manipulated to optimize methane production. The optimum temperature for methane production in the mesophilic range has been reported to be in the range of 30 to 35 ^{0}C (Watson-Craik *et al.* 1994), with temperature above and below this causing inhibition of microbial activity. Furthermore, it was concluded that prolonged periods at these temperatures might result in unbalanced fermentation. It was suggested that a refuse emplacement strategy be employed to control temperature. Rees (1980), for example, suggested that low density refuse, diluted with wood, paper, or aerobically stabilized refuse, should be placed at the landfill base and allowed to decompose. Aerobic decomposition would then cause a temperature increase and heat at the subsequent layers of refuse will release. Heat loss from the site can be minimized by an insulating layer of refuse above the reactive zone.

Temperature in the landfill depends on the production of heat and addition from external heat such as solar energy and heat loss to the cover soil and to the atmosphere (Rees, 1980). Numerous studies were carried out to examine the effect of temperature on waste biodegradation in landfills. In addition, many researchers have investigated methods for controlling temperature for optimum waste biodegradation.

An experiment was conducted by Mehanta *et al.* (2002), on a bioreactor landfill model with applying leachate recirculation and without leachate recirculation. Both bioreactors with and without leachate recirculation reached temperature in the range of 50 to 55 °C in the top layer just after the solid waste was deposed and covered. It was noted that leachate recirculation caused the acceleration of the anaerobic reactions in landfills and increased the temperatures inside the bioreactor landfill as well. It was reported that the temperatures in the bioreactor with leachate recirculation stabilized at about 40 °C from the middle to the top layer of the waste matrix and it was about 35 °C at the bottom layer. While for the bioreactor without leachate recirculation, the temperature stabilized at 25-32 °C throughout the entire system.

2.3.4 Sludge Addition

Anaerobically digested sludge can serve as a seed of microorganisms as well as source nutrients such as nitrogen and phosphorus, and other micro nutrients. Barlaz et al. (1989a) concluded that all microorganisms required for methanogenesis were present in the fresh refuse. They also reported that acids accumulation and consequently pH decrease was due to an imbalance between acid fermentative and methanogenic activity. Once the pH decreases, there is little mechanism within a conventional landfill for refuse neutralization. The use of wastewater sludge balances the activities of the two types of bacteria involve in these processes and prevents the initial pH decrease. Rapid waste decomposition was observed in test lysimeters when sewage sludges were added (Barlaz *et al.*, 1990).

Old refuse is another potential inoculum that can be added to fresh refuse. Adding old, anaerobically degraded refuse can stimulate methanogenesis, and the waste acts as a dilutant against the accumulation of toxic compounds. The addition of old refuse has been proven to be a very effective enhancement technique (Barlaz *et al.*, 1990).

Doedens and Cord-Landwehr (1989) reported that the addition of sludge and composted refuse to shredded fresh refuse improved the conditions for methane production. The sludge was anaerobically digested but not dewatered, and the old refuse was composted for 1 to 2 months. The ratio of composted refuse to sludge was 7:1 on a dry weight basis. A positive effect resulting from this experiment was that methane production increased. Consistent with this work, Stegman and Spendlin (1989) found that methane gas production was highest when fresh refuse was mixed with composted refuse. It was also noted that the BOD concentration rapidly decreased when composted refuse was added.

2.3.5 Enzyme Addition

Lagervist and Chen (1993) evaluated the effect of enzyme addition under methanogenic and acidogenic conditions using a 0.1 m^3 landfill model. Cellulotic enzymes were added to fresh refuse through leachate recirculation and resulted in rapid degradation of waste. The conversion of volatile solids was approximately 40 to 50% for both acidogenic and methanogenic conditions.

Cellulotic enzyme was used because the major component of the degradable MSW is cellulose, and by manipulating the enzyme activity, it is possible to control and enhance the hydrolysis of cellulose. It was concluded that the increased acid production generated by enzyme addition during intense methane production was not sufficient to suppress the subsequent methanogenesis that occurred. This was explained by the nature and content of the waste and its non-homogeneous distribution. Enzyme addition during the decline of gas production did not cause an increase in the gas production rate (Lagervist and Chen 1993).

2.3.6 Nutrient Addition

As mentioned previously, nutrients play an important role in the biodegradation of solid waste in landfills. Some studies found that the addition of nutrients accelerated the methane generation or rapidly decrease COD and BOD concentrations in the leachate (Warith, 2002, Warith *et al.*,1999, and Pohland and Harper, 1986). Warith *et al.* (1999) tested the effect of nutrient addition on the degradation of MSW. A nitrogen content of 20% and phosphorus content of 20% from plant food were added into landfill model with the size of 0.7-m in diameter and a height of 1.2-m. The average density of the waste matrix was 278 kg/m^3. It was observed that the addition of supplemental nutrients rapidly increased and decreased of COD concentrations in the leachate in a shorter time compared to the control model that did not have any nutrients addition. The conclusions was addition of nutrients enhance microbial activities in the landfill model, resulting in the rapid biodegradation of solid waste.

2.3.7 Moisture Content (MC)

As discussed earlier, many studies have been conducted to investigate the effect of moisture content in the decomposition of solid waste. The general concluded that moisture content is one of the most critical variables in the decomposition of solid waste. If the moisture content is less than 50%, either methanogenesis would not prevail or the rate of methanogenesis would be very slow (Nyns, 1980). It was reported by Christensen and Kjeldsen (1989) that there is an exponential increase in gas production rates when moisture content is between 25% and 60%.

Rahtje (1989) dug seven landfills to investigate the material component inside the landfills, and found that newspaper dating back to 1952 was still fresh and readable. This showed that without adequate moisture content, solid waste biodegradation would not occur. In a heterogeneous system such as a landfill, the lack of opportunity for contact between microorganisms, substrates, and other necessary growth factors may limit biodegradation. As the refuse moisture content increases, the opportunity for contact increases, which should enhance microbial activity. Therefore, when field capacity is adjusted initially or if a continuous flow of water through the refuse is provided, decomposition accelerates.

High moisture content stimulates hydrolysis, but an accelerated rate of hydrolysis can be inhibitory. If the hydrolysis rate is high, or if a great quantity of soluble substrates are available, acid forming bacteria will grow faster than methanogenic bacteria. As a result, there will be an accumulation of acid, and pH will decrease, thus causing methane production to decrease.

At high densities, each particle of refuse is in closer contact with adjacent particles. Thus, for the same moisture content, there is more water in contact with each individual particle when the densities are higher. If moisture contact stimulates the rate of hydrolysis to inhibitory levels, then as the density increases, the optimum moisture content would be expected to decrease.

Warith *et al.* (1999) demonstrated that leachate recirculation was most stimulatory when municipal sewage sludge was added to the refuse. This indicated that sewage is an excellent source of microbial inoculum. On the other hand, leachate recirculation without sludge addition or neutralization stimulated the accumulation of carboxylic acids, which inhibit the rate of waste biodegradation. This result leads to the conclusion that leachate recirculation alone did not enhance the stabilization of the solid waste.

As presented above, the importance of moisture sufficiency is well confirmed and pronounced in the leachate recirculation strategy. However, the addition of buffer, nutrients, and source of inoculum, such as sewage sludge are important in enhancing solid waste biodegradation and stabilization.

2.4 Bioreactor Landfill

The following section provides an overview on laboratory, pilot and full-scale work conducted on bioreactor landfills with particular emphasis on leachate recirculation, sludge and nutrients addition as well as on landfill gas.

2.4.1 Background

A bioreactor landfill operates leachate recycle to accelerate transformation and degradation of organic waste. The increase in waste degradation and stabilization is accomplished through the addition of liquid, nutrients, and air to enhance microbial processes. Leachate recirculation is a key component of the bioreactor process and substantially lowers the cost of leachate management without having to change current liner and leachate head requirements, and eliminates the need for off-site management at municipal wastewater treatment plants.

This bioreactor concept differs from the conventional MSW landfill approach which is commonly uses the "dry tomb" concept. Consequently, wastes decomposition remaining active for long periods of time after post closure of landfill, as long as even for 30-50 years. Post closure landfill can create environmental problems, such as excess of LFG and leachate. As discussed earlier, LFG contribute to global warming. Pollution of groundwater and surface water could occur sometime in future when using conventional dry landfill.

A variety of operational approaches of bioreactor landfill have been investigated to find the optimum condition for MSW degradation. This approach could be divided into three different general types of bioreactor landfill configurations: aerobic bioreactors, anaerobic bioreactors, and hybrid bioreactors.

2.4.2 Aerobic Bioreactor

Theory of aerobic degradation of MSW is mostly covered in the composting literature. Aerobes are organisms that require oxygen for cellular respiration. In aerobic respiration, energy is derived from organic molecules in a process that consumes oxygen and produces new cell, carbon dioxide, water, and heat. In composting water and sewage sludge are commonly used as amendments. The practices of adding water through leachate recirculation however is unique to landfilling. In landfills aerobic activity is promoted through injection of air or oxygen into the waste matrix.

At present, there is limited information in the literature with regards to the use of aerobic landfill technology as a method of solid waste disposal (Barlaz *et al.*, 1992). In an aerobic landfill, the addition of air, and in turn oxygen, promotes the aerobic stabilization of the landfilled waste. The aerobic stabilization process occurring in a landfill is similar to that which occurs in a waste composting system, the main difference being aerobic composting is accomplished within the landfill rather than in windrows or other methods in which composting is normally practiced (Barlaz *et al.*, 1992). In an aerobic landfill, methane energy is sacrificed by the addition of air. On the other hand significant waste fractions such as lignin and ligneous materials, which are not degradable anaerobically are degraded aerobically (Stessel and Murphy, 1992)

Stessel and Murphy (1992) conducted aerobic and anaerobic lysimeter studies and reported that within 75 days a MSW settlement of 35% for the aerobic and 20% for the anaerobic were achieved. In addition, the concentration of COD in the leachate was also reported to be lower for aerobic as compared to anaerobic landfills. It was demonstrated that by recirculating leachate through the landfill waste matrix while at the same time adding oxygen, aqueous concentrations of organic compounds could also be reduced (Stessel and Murphy, 1992). The study demonstrated that aerobic degradation had a better performance than anaerobic, and thus aerobic landfill could have advantages in the recovery of valuable airspace, minimization of landfill gas (LFG) and other leachate management related problems. In aerobic bioreactor, the waste matrix serve as an efficient "trickling media", allowing an effective recycling of leachate and introduction of oxygen. This process continually provides the respiring microorganisms with moisture, nutrients, and oxygen needed to biodegrade the solid and aqueous organic compounds in the MSW.

Two pilot plant studies of aerobic landfills, at the Colombia County and Atlanta landfills, Georgia, USA, were investigated by Hudgins and Harper (1999). The results of these studies indicated that aerobic landfills can reduce the volume of methane ranged from 50 to 90% and minimize odour problems. In addition, the COD concentration in leachate was also reduced from 1,100 to 300 mg/L (73%) at the Colombia county landfill. Consistent with this work, Vitello (2001) observed that methane generation decreased by more than 98% at the pilot scale aerobic landfill in Williamson County, Tennessee, U.S.A. An experiment conducted by Doedens and Cord-Landwer (1989) at the Bornhausen landfill in Goslar, Germany, three test sites were setup with and without cover layer and recirculation. The waste was build into 4-m layer in six month. The test cell without cover layer combined with aeration through drainage system at the bottom revealed that reduction of BOD concentration of 90% was achieved. While in the test cell without aeration, the reduction of BOD concentrations was about 50%. They observed that aeration could accelerate waste biodegradation.

2.4.3 Anaerobic Bioreactor

Anaerobic bioreactor seeks to accelerate the degradation of waste by optimizing conditions for anaerobes. Anaerobic processes could be enhanced by manipulating environmental factors that affect solid waste biodegradation as previously described in this chapter. The most common manipulation is recycling of leachate into the landfill waste matrix.

A number of lab-scale, pilot and full scale studies of bioreactor landfills were conducted in North America and European countries. Comprehensive reviews of the technical literature on the landfill bioreactor concept can be found in a book by Reinhart and Townsend (1998) entitled Landfill Bioreactor Design & Operation. Some of the studies cited are as follows. A lab-scale study to evaluate the effects of leachate recirculation on solid waste biodegradation was conducted by Sponza and Agdag (2004) at the Dokuz Eylul University, Turkey. The simulated bioreactors were made from Plexiglas with the diameter of 30-cm and the height was 100-cm. Two bioreactors were operated with leachate recirculation (9 and 21 L/d) and the third was used as a control with no leachate recirculation. These bioreactors were operated anaerobically for about 220 days. After analyzing some parameters (volatile acids and methane) in the leachate generated from bioreactors and collecting the gas generation data, they observed that leachate recirculation can accelerate biodegradation of solid waste. However, the highest rate of leachate recirculation of 21 L/d caused in a decreased in methane generation. These researchers attributed the decreasing of methane generation to an accumulation of volatile acids at the high rate of leachate recirculation.

A full-scale landfill bioreactor project at the Yolo County Central Landfill, California, U.S.A. was reported by Yazdani *et al.* (2002). The construction of a 12-acre bioreactor that contains a 6-acre anaerobic bioreactor, a 3.5-acre anaerobic bioreactor, and a 2.5-acre aerobic bioreactor were used to study the performance of the biodegradation solid waste. The 3.5-acre anaerobic and the aerobic bioreactors had been filled with waste and the instrumentation, leachate injection, and gas collection systems have been installed. A total of 140,000 and 65,104 tons of waste were placed in the 6-

acre and 3.5-acre anaerobic bioreactors, respectively. While a total of 11,942 tons of waste were placed in the aerobic bioreactor. They reported that enhance methane/gas recovery at a rate 10-fold the rate normally seen in conventional landfill practice in the anaerobic bioreactors. This indicated that leachate recirculation enhance the biodegradation of MSW. The aerobic biodegradation of MSW was faster compared to anaerobic biodegradation. The reclaiming landfill space could improve the overall economics of aerobic operation by creating a sustainable operation.

CHAPTER 3

MATERIALS AND METHODS

In order to accomplish the objectives of this research project, two sets of laboratory scale landfill bioreactors were used to conduct the experiment. The first set consists of six aerobic simulated bioreactors and the second was six anaerobic simulated bioreactors. A total of 12 experimental runs were carried out at 10 different operating conditions. Materials and procedures used in this research are presented and discussed in the following sub sections.

3.1 Experimental Design

The objective of the experimental study was to optimize the performance of the MSW bioreactor landfill. In this study, the effect of leachate recirculation and sludge addition on the rate of biodegradation was examined to determine the optimal operating conditions in terms of cumulative methane production for anaerobic reactors and leachate quality, in terms of COD concentration, for both aerobic and anaerobic bioreactors.

3.1.1 Two Level Factorial Designs

A statistical empirical model was used to describe the effects of operating variables on the COD concentration of the leachate (process response, $\hat{Y}$). In this model, the data was fitted to the first order linear model using a factorial design. The model is expressed as follows:

$$\hat{Y} = \beta_0 + \beta_1 x_1 + \beta_2 x_2 + \beta_{12} x_1 x_2 \qquad (3.1)$$

where, $\hat{Y}$ is the expected value of the observed response, β_i is a model parameter and x_i is a coded operating variable. The model parameters were estimated using data collected according to a two levels (2^2) factorial experimental design including two center point replicates, from which reproducibility and lack of fit can be evaluated.

To assess the effects of leachate recirculation and sludge addition on the biodegradation rate of MSW, three scenarios providing different rates of leachate recirculation and sludge addition were examined. These scenarios were coded in a 2^2 factorial experimental design as shown in Table 3.1 (Box et al., 1978).

Table 3.1 Different scenarios at which operating variables were investigated in the factorial design

Type of operating variable	Levels		
	Low	Center	High
Rate of leachate recirculation (mL/day)	714	1428	2142
Rate of sludge addition (mL/day):	71.4	142.8	214.2
Coded	-1	0	+1

To simplify notations and subsequent calculations, x_1 and x_2 in (3.1) were dimensionless coded variables defined as follows:

x_1 = (rate of leachate recirculation-1428)/714

x_2 = (sludge addition-142.8)/71.4

Two runs of center point replicates and four possible combinations of operating conditions using the coding x_1 (leachate recirculation) and x_2 (sludge addition) were derived from Table 3.1. The two center point replicates were used as a control for the bioreactors operation. The following design strategy of four combinations of operating conditions and two center point replicates, as outlined in Table 3.2, was put forward for this study. A total of six 9-L aerobic and six 9-L anaerobic reactors were required to run the experiments.

Table 3.2 Operating conditions for leachate recirculation and sludge addition

Bioreactor's name	Rate of leachate recirculation (mL/kg of MSW.d)	Rate of sludge addition (mL/kg of MSW.d)
AER1 and ANA1	285	28.5
AER2 and ANA2	855	28.5
AER3 and ANA3	285	85.5
AER4 and ANA4	855	85.5
AER5 and ANA5	570	57
AER6 and ANA6	570	57

For the two center point replicates, the estimation of the pure error variance ($\hat{\sigma}^2$) was determined as follows:

$$\hat{\sigma}^2 = \sum_{u=1}^{2} \frac{(y_u - \bar{y})}{1}$$

where, y_u is the measured response value for the two center point replicates, and $\bar{y}$ is the sample mean of all measured response value at the two center point replicates.

Responses may have a high degree or low degree uncertainty which was indicated by the value of pure error variance. If the value of the pure error variance was small the variation in the reactor performance was also small. This means that the reactor has a low degree of uncertainty. In contrast, if the pure error variance was large then the variability of the reactor performance is expected to be high. Under this condition, the response has a high degree of uncertainty.

3.2 Composition of the MSW

To achieve a representative sample of MSW commonly disposed in a landfill, the MSW was collected from the curbside of the city of Ottawa. Samples were collected from curbside of restaurants, grocery stores, houses and apartments. Information and data on the physical composition of MSW obtained from the City of Ottawa was used as a guideline in this experiment. Since the objectives of the experiment is to compare the rate of MSW biodegradation of aerobic and anaerobic bioreactor landfills, only the biodegradable material of MSW used in the experiment. The physical and chemical properties of MSW collected are shown in Table 3.3. Separation of the individual components that make up a solid waste stream and their relative distribution was carried out in the laboratory and is shown in Table 3.4.

Table 3.3 Physical and chemical properties of MSW

Component	Percent by weight
MC	20.2
Total Solids	79.8
Volatile Solids	91.4
Total carbon	49.8
Total nitrogen	1.7
Total phosporus	0.8

Table 3.4 Composition of MSW used in simulated bioreactors

Component	Total weight of MSW (wet basis) (Kg)	(%)
Paper:		
Corrugated and Kraft	0.319	12.76
Used newspaper	0.174	6.96
Computer and high grade paper	0.099	3.96
Others	0.323	12.92
Sub total	0.915	36.60
Food:		
Fruits	0.120	4.80
Vegetables	0.137	5.48
Potatoes	0.102	4.08
Pasta and rice	0.085	3.40
Bread	0.085	3.40
Meat	0.085	3.40
Fish	0.051	2.04
Diary product	0.051	2.04
Coffee ground	0.017	0.68
Tea leaf	0.017	0.68
Others	0.154	6.16
Sub total	0.904	36.16
Yard trimmings:		
Grass	0.256	10.24
Leaves	0.077	3.08
Trimmings	0.348	13.92
Sub total	0.681	27.24
Total	2.50	100.00

3.3 Simulated Bioreactor Landfills Model

This experimental setup consisted of six simulated bioreactor landfill models made from Plexiglas with dimensions of 150-mm in diameter and 550-mm in height. The schematic representation of the aerobic bioreactor landfill is shown in Figure 3.1. All six bioreactors were operated with a leachate recirculation system, each equipped with an outlet port at the bottom for leachate collection, as well as an inlet port to distribute the simulated rainfall and the recycled leachate. The leachate was collected in tanks before recycling at the top of each reactor.

All bioreactors were sealed using construction-type sealant to avoid leaks. A PVC pipe with a diameter of 12.7-mm was used as an air distribution system and it was connected to the laboratory air supply system. The bottom part of the pipe was perforated and wrapped in a plastic mesh prior to inserting it into the waste matrix. Three thermocouple ports were inserted at three different heights in the solid waste matrix in order to monitor temperatures. In addition, a supporting media, consisting of a 50-mm thick layer of marbles (15-mm in diameter) was placed at the bottom of each reactor and used as a leachate drainage system as well as to prevent clogging of the leachate outlet at the bottom of the bioreactors. A perforated plate was placed at the top of the waste matrix to provide an equal distribution of simulated rain water and recycled leachate. To maintained temperatures, the bioreactors were insulated using fiberglass insulation material. Two layer of insulation were applied on the bioreactors,i.e., the 1st layer (thickness 50mm) was Pink Insulation Fiberglass^{TM} R-12 RSI-2.1, and the 2nd layer (thickness 10 mm) was polyurethane foam. A total of six additional simulated bioreactor landfills identical to an aerobic system with the exception of no air supply were operated to simulate anaerobic bioreactors.

3.4 Experimental Methods and Operations

3.4.1 Bioreactors Filling

The collected solid waste was shredded manually to approximately 50 to 100-mm in size and then mixed uniformly before being loaded into the bioreactors. A 100-mm layer of shredded solid waste was filled and compacted to attain an approximate density of 350 kg/m^3. This technique was repeated until the thickness of solid waste in each reactor reached a height of 400-mm. Total mass and volume of MSW in the bioreactors were 2.50 kg and 7-L, respectively.

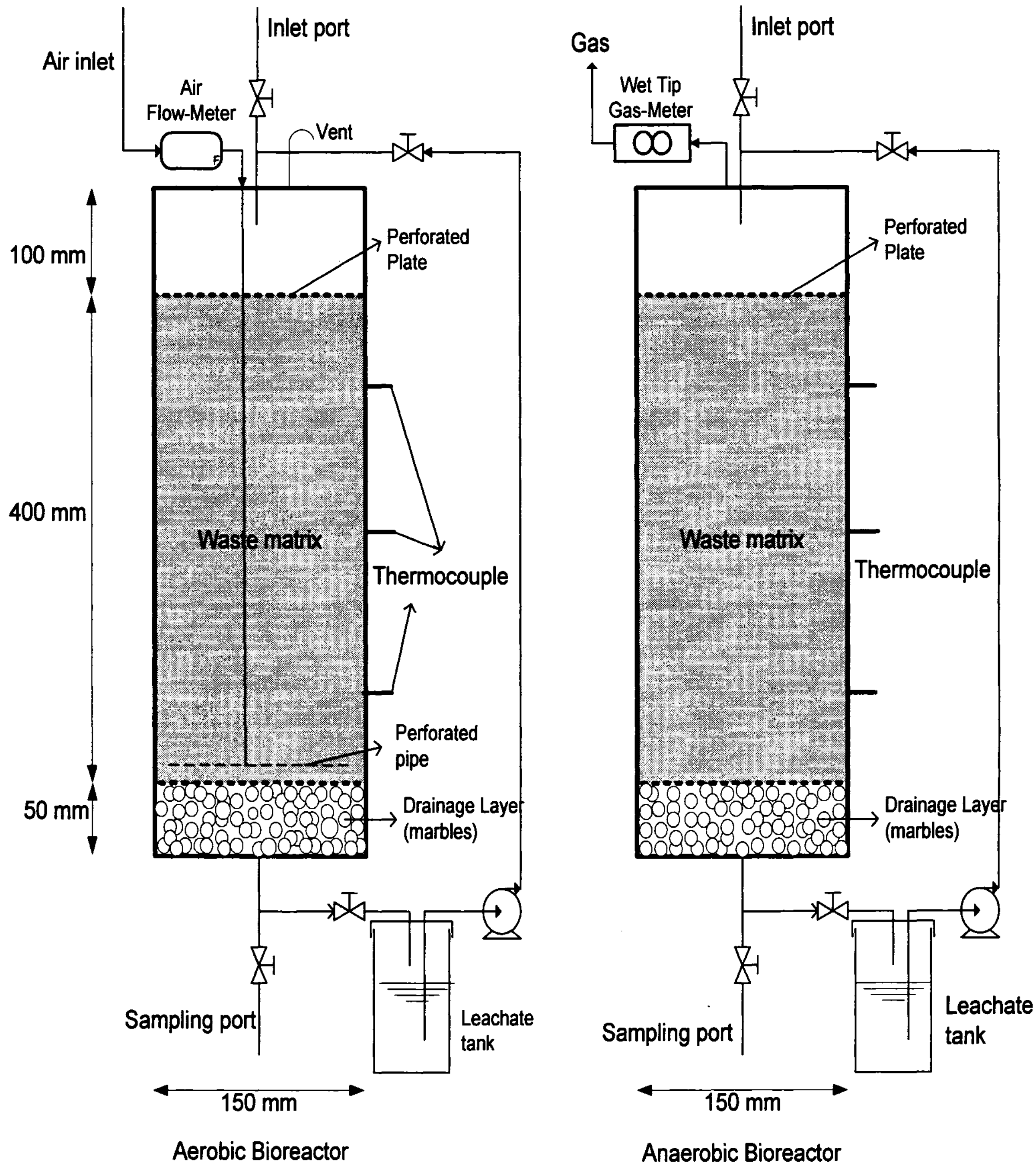

Figure 3.1 Configuration of simulated bioreactors

3.4.2 Leachate Recirculation

In this research, the initial moisture content of MSW in the bioreactor was 20.2%. According to Tchobanoglous *et al.* (1993), the initial moisture content of the MSW was typically between 15 to 25 percent. In order to attain the MSW field capacity, 2 liters of tap water were added per day to each of the bioreactors. After day 4, when leachate was generated, the simulated rainfall was reduced to 285 mL/d starting at day 8. The leachate generated was collected in a 15-L tank, then recycled daily using a pump at the rate of 714, 1428 and 2142 mL/d to the top of the bioreactors which were started on day 15. The use of these rates was selected to keep the waste matrix under unsaturated conditions based on moisture content removed and to examine the rate of the MSW degradation.

3.4.3 Buffer and Sludge Addition

In all twelve bioreactor landfills, the pH was controlled by adding a buffer solution of sodium bicarbonate ($NaHCO_3$) 1.0 N through leachate recirculation systems. Addition of the buffer solution was used to maintain the pH in the neutral range.

Anaerobically digested and thickened waste activated sludge from ROPEC Municipal Wastewater Treatment Plant in Ottawa were added to the bioreactors at the rates indicated in Table 3.2 through the leachate recirculation system. The physical and chemical properties and metal concentrations in the sludge are presented in Table 3.5. and Table 3.6.

Table 3.5 Physical and chemical properties of sludge

Component	Thickened waste activated sludge	Anaerobic digested sludge
Total Solids, %	4.2	3.3
Volatile Solids, % of TS	58.6	60.7
pH	7.1	7.4
Total carbon, % of TS	60.5	65.9
Total nitrogen, % of TS	2.1	2.5
Total phosporus, % of TS	1.3	1.8

Table 3.6 Concentration of metal in sludge

Parameter	Thickened waste activated sludge	Anaerobic digested sludge	Limit of Detection
B	1.0255	1.272	0.0069
Cd	0.1174	0.1691	0.0114
Cr	0.3023	0.388	0.0066
Cu	4.037	3.361	0.0048
Fe	32.44	42.25	0.0168
Mn	1.246	0.7398	0.0015
Ni	0.3626	0.3932	0.0177
Pb	0.3319	0.235	0.0723
Zn	4.5165	3.129	0.0093

Note: All concentrations are in ppm

3.4.4 Air Requirements

Air was supplied to the six bioreactors to maintain aerobic conditions during the experimental study. The air flowrate supplied needed to satisfy the oxygen demand exerted by the organic waste decomposition (stoichiometric demand). For this research study, the solid waste composition was approximated by the general formula $C_{64}H_{104}O_{37}N$ provided by Haug (1993). Using this composition, the oxygen demand can be determined as follows:

$$\underset{1478}{C_{64}H_{104}O_{37}N} + \underset{2264}{70.75\ O_2} \longrightarrow 64\ CO_2 + 50.5\ H_2O + NH_3$$

Based on the assumed solid waste composition, the oxygen demand was estimated as

$$O_2 = 2264/1478 = 1.53 \text{ g } O_2 \text{ /g waste (dry)}$$

This was the total quantity of oxygen which must be supplied to the aerobic reactors to meet the stoichiometric demand. However, in this research, a safety factor of 2 was applied to avoid anaerobic conditions within the reactors. Air contains 23% oxygen by weight and has a specific weight of 1.20 kg/m^3 at 25^0C and 1 atmosphere pressure. Thus, the volume of air required was determined as:

$$\text{Air} = (2)\ (100/23)\ (1.53 \text{ g } O_2 \text{ /g waste})/(1.20 \text{ kg/m}^3) = 11.1 \text{ m}^3 \text{ air/ kg of MSW}$$

Mass of the solid waste in each reactor was 2.5 kg with initial moisture contents of 20.2%. It was estimated that the decomposition will prevail during 120 days period. Thus, the air flowrate was calculated as:

$$\begin{aligned}\text{Air flowrate} &= (79.8\% \times 2.5 \text{ kg of waste})\ (11.1 \text{ m}^3 \text{ air/ kg waste})/120 \text{ days} \\ &= 0.185 \text{ m}^3 \text{ air/day} \\ &= 185 \text{ L air/day}\end{aligned}$$

3.5 Analytical methods

In order to record the progress of landfill stabilization within the reactors, a combination of parameters concerning solid waste, leachate, and gas were monitored and analyzed throughout this experiment. Analytical methods of these parameters are discussed in the following sub sections. To record the progress of landfill waste stabilization within the bioreactors, a combination of parameters concerning solid waste and leachate were monitored throughout the experimental study.

3.5.1 pH

The pH analysis of the leachate samples were determined by using the procedure 4500-H^+ B., as outlined in *Standard Methods for the Examination of Water and Wastewater, 20th Edition* (AWWA, APHA, WEF, 1998) on a weekly basis. The instrument used to analyze the pH was a Fisher Accumet Electrode pH meter (model 320, Fisher Scientific, San Diego, CA, USA).

3.5.2 Total Chemical Oxygen Demand (COD)

The total COD concentrations of the leachate samples were determined by using the Closed Reflux Titrimetric Method according to procedure 5220 C in the Standard Method (AWWA, APHA, WEF, 1998).

3.5.3 Biochemical Oxygen Demand (BOD_5)

The BOD concentrations of the leachate samples were determined according to procedure 5210 B in the Standard Method (AWWA, APHA, WEF, 1998) on a biweekly basis. All the BOD concentrations reported in this experiment were 5-day BOD concentrations.

3.5.4 Nitrogen

Total Kjeldahl nitrogen (TKN) concentrations of the leachate samples were determined according to procedure 4500-N_{org}C Semi-Micro Kjeldahl to Standard Method (AWWA, APHA, WEF, 1998).

Ammonia nitrogen (NH_3-N) concentrations of the leachate samples were determined on a biweekly basis by using the Closed Reflux Titrimetric Method according to procedure 5220 C in the Standard Method (AWWA, APHA, WEF, 1998).

Nitrate concentrations of the leachate samples were determined after 5 weeks of operation and then every 10 weeks. The procedure used in this determination was 4500-NO_3^- D Nitrate Electrode Method as outlined in Standard Method (AWWA, APHA, WEF, 1998).

3.5.5 Total Phosphorus

Total phosphorus (TP) concentrations of the leachate samples were determined according to procedure 4500-P B Persulfate Digestion in Standard Method (AWWA, APHA, WEF, 1998). The TP concentrations were analyzed after 5 weeks of bioreactor operation and then every 10 weeks.

3.5.6 Metals

Selected metals (B, Cd, Cr, Cu, Fe, Mn, Ni, Pb, Zn) were measured after 5 weeks of operation and then every 10 weeks. Leachate samples were filtered using Glass fiber filter (0.45 μm) and acidified to a pH of 2. Then the filtrate was analyzed by using a Thermo Jarrell ICP analyzer.

3.5.7 Temperature

The temperature of the waste in the bioreactors was measured on a biweekly basis using the "Barnant Thermocouple" type T model. Thermocouples were inserted into three ports of the bioreactors to monitor temperatures. These parameters were analyzed using procedures that are described in "Standard Method" (AWWA, APHA, WEF, 1998).

3.5.8 Solids

The solid waste was characterized prior to placement into reactors, and at the end of the experiment. The purpose of characterizing the solid waste was to show that characteristics of actual landfill materials are adequately simulated. A sample size of 2.5 kg with the physical composition described in Table 3.1 were grounded to 40 mesh and analyzed for the parameters: moisture content (MC), total solids (TS), volatile solids (VS), pH, total carbon (TC), total Kjeldahl nitrogen (TKN), and total phosphorus (TP).

Moisture content, TS, and pH were analyzed according to the method developed by McKeague (1978), whereas VS was analyzed by oven drying at 550 ^{0}C (AWWA-APHA-WEF,1998). TC, TKN and TP were analyzed according to Tiessen and Moir (1993), McGill and Figueiredo (1993), and O'Halloran (1993). These procedures are described in Soil Sampling and Methods of Analysis (Carter, 1993).

3.5.9 Measurement of Biogas

The volume of gas produced in the simulated bioreactor landfills was measured using a wet tip meter device from Wet Tip Inc. During the course of the experiment, the volume of biogas production were recorded. The number of tip readings from the wet tip meter were then converted into a rate of biogas generated using a calibration curve.

The composition of methane, carbon dioxide, and nitrogen produced by the bioreactors was determined using a Hewlett-Parckard (HP) 5710a gas chromatograph (GC) equipped with a thermal conductivity detector and a 3380A model integrator (Hewlett-Packard, Polo Alto, CA). Biogas sample volumes of 0.5 mL were manually injected into the port of the GC machine for analysis.

3.5.10 Settlement of MSW in bioreactors

Volume reduction of MSW was investigated in each of the twelve bioreactors. Settlement was used to measure the volume reduction of the compacted MSW in the bioreactors. The height of the compacted MSW in the bioreactors was measured to determine the settlement and was started from the beginning (0 day), 9, 15, 25, 35 weeks of operation of bioreactors and at the end of experiment (47 weeks) for aerobic and (0 day), 9, 15, 25, 35, 45, 55 and 63 weeks for anaerobic. The distance from the top of the drainage layer (marbles) to the top of the MSW matrix initially was 40 cm. This distance was the original height of the compacted MSW.

3.5.11 Sludge Analysis

Physical and chemical analysis of sludge was conducted before the sludge was added to the leachate recirculation system. Moisture content (MC), total solids (TS), volatile solids (VS), pH, total carbon (TC), Total Kjeldahl nitrogen (TKN), and total phosphorus (TP) were measured according to "Standard Methods" (AWWA-APHA-WEF, 1998). Metal concentrations in the sludge were determined using the inductively coupled plasma spectroscopy (ICP) techniques.

CHAPTER 4

RESULTS AND DISCUSSION

In this chapter, the experimental results from the laboratory scale bioreactor models are reported and discussed. The colleted data are presented in the enclosed figures and tables. This section is divided into two parts that address the aerobic and anaerobic bioreactors performance.

4.1 Aerobic Bioreactors

4.1.1 Leachate Quality

4.1.1.1 pH

The pH variations of the leachate during the experimental period are presented in Figure 4.1. During the initial phase, from week 1 to 5, the pH of the leachate varied between 5.6 and 6.7 for all the bioreactors, with the exception of the AER2 and AER4 (the volumes of leachate recirculation were 855 mL/kg of MSW.d for both bioreactors, and sludge addition were 28.5 and 85.5 mL/kg of MSW.d) in which the pH was stabilized at approximately neutral (pH 7-7.5) after 3 weeks of operation. Whereas, it took 10 weeks for the leachate from bioreactors AER1 and AER3, which received 285 mL/kg of

MSW.d of leachate recirculation for both bioreactors and sludge addition of 28.5 and 85.5 mL/kg of MSW.d, to attain a pH of 7 to 7.5. The low pH observed in the early stages of MSW biodegradation process is related to the production and dissolution of CO_2 and organic acids. Further decomposition of these organic acids, production of NH_4^+ and release of CO_2, resulted in an increase of pH. Ideal pH for most microorganisms is in the neutral range of 6.5 to 7.5. Therefore, the pH of the recycled leachate was maintained consistently in the neutral range by adding a solution of sodium bicarbonate ($NaHCO_3$) 1.0 N as a buffer until the completion of the experiment (47 weeks).

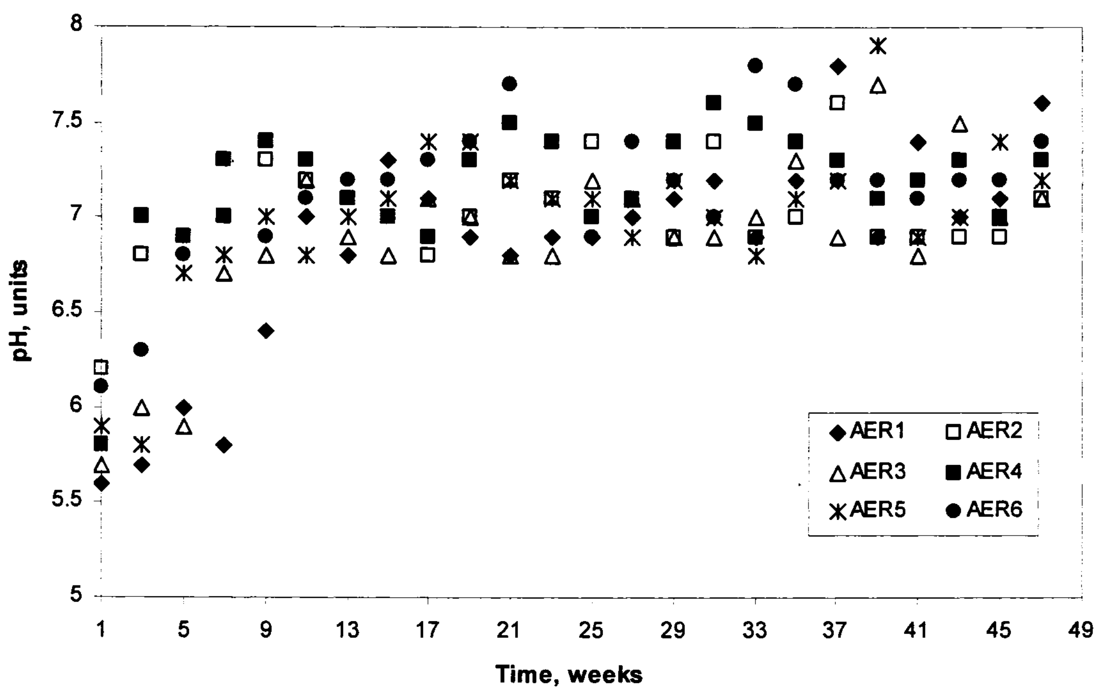

Figure 4.1 pH in leachate from aerobic bioreactors

4.1.1.2 COD

Commonly, the two parameters used to evaluate the effect of leachate recirculation and sludge addition on solid waste degradation are COD and BOD. Figure

4.2 depicts the concentration of COD in the leachate from the six bioreactors during the experimental study. All profiles had a similar trend; first a rising stage, followed by a peak concentration, and finally a declining COD concentration. The stabilization of MSW was assumed to have been attained when the COD concentration in the leachate was measured in the range of 1000 mg/L.

Initially, the COD concentration in the leachate from AER4 and AER2, where the volumes of leachate recirculation were 855 mL/kg of MSW.d for both bioreactors, and sludge addition were 85 and 28.5 mL/kg of MSW.d, respectively, increased slowly over a period of 3 weeks and generally remained below 5,000 mg/L. After this stage, the COD concentration increased exponentially and reached the peak values of approximately 38,000 mg/L for AER4 and 40,000 mg/L for AER2, after 11 and 13 weeks of operation, respectively. This stage was followed by a decrease at an almost constant rate until the 27th week. The COD trends demonstrated that MSW stabilization was achieved within 27 weeks in AER4 with a COD concentration in the leachate of approximately 1,000 mg/L, while the COD concentration in leachate from AER2 reached approximately 1,000 mg/L, after 31 weeks. This would appear to indicate that the higher sludge addition contributed to a faster MSW biodegradation process.

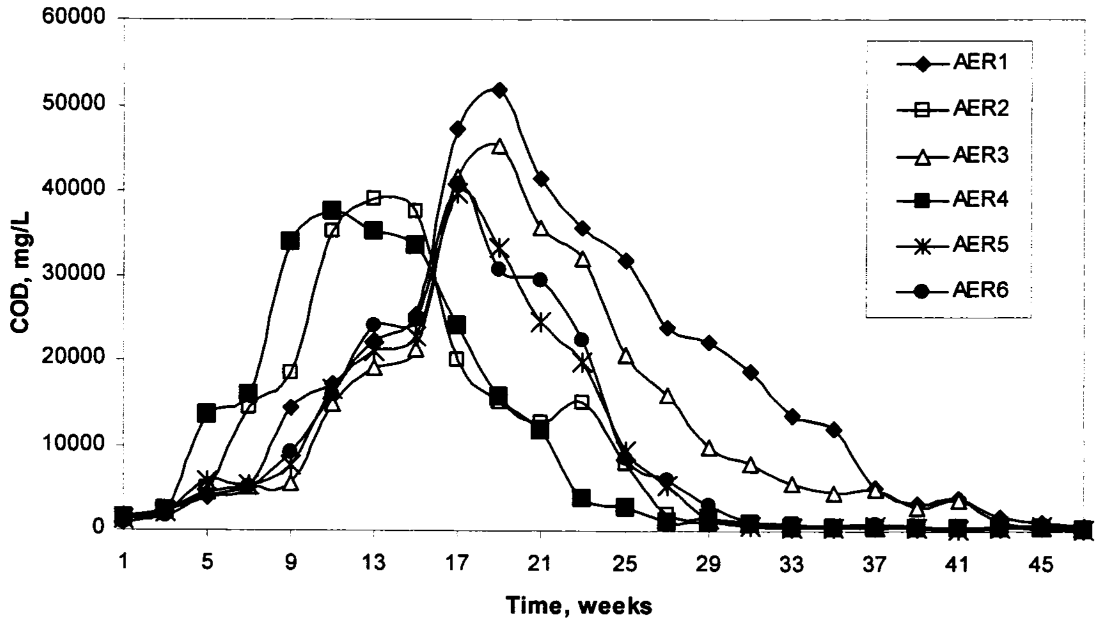

Figure 4.2 Concentrations of COD in leachate from aerobic bioreactors

The COD concentration in the leachate from AER5 and AER6 increased slowly up to the 7th week of operation, after which time it increased exponentially and reached a peak value of approximately 40,000 mg/L for both bioreactors after 17 weeks. Afterwards, the COD concentration decreased to approximately 5,500 mg/L after 27 weeks of operation. It should be noted that, for the same operating period, leachate from AER4 was approximately 1,000 mg/L. Therefore, the higher leachate recirculation and sludge addition the shorter time to reach maximum COD concentration and MSW stabilization in terms of lowering COD concentration in the effluent leachate to below 1000 mg/L.

As stated earlier, the two center point replicates, AER5 and AER6, were used as a control for the bioreactors operation. Profiles of the COD concentration in the leachate of these two replicates displayed similar trends and had very close behavior during the entire period of operation. These results strongly support the reproducibility of the MSW biodegradation process.

In bioreactors AER1 and AER3, where the volume of leachate recirculation was the lowest, 285 mL/kg of MSW.d. the COD concentration in the leachate increased slowly until the 7th week of operation; then it increased exponentially and reached peak values of approximately 51,000 and 45,000 mg/L, respectively, after 19 weeks of operation. Subsequently, the COD concentration in the leachate decreased rapidly to approximately 24,000 and 16,000 mg/L in the AER1 and AER3 bioreactors, respectively, after 27 weeks of operation. These results indicate that the MSW biodegradation in bioreactor AER3 was faster than in bioreactor AER1 mainly due to the higher rate of sludge addition. It is worthy to note that the COD concentration in the leachate from bioreactor AER3 reached approximately 1,000 mg/L after 43 weeks of operation, whereas it took only 27 weeks for bioreactor AER4 to attain the same results. For this reason, the rate of MSW biodegradation for AER4 was almost 2-fold faster than AER3, even though both bioreactors received the same volume of sludge addition. However, the volume of leachate recirculation in AER4 was 3-times higher than in AER3. Thus, it is a clear indication that the higher rate of leachate recirculation contributed significantly to the faster rate of MSW biodegradation.

Overall, biodegradation rates of MSW, measured in terms of COD released in the leachate, could be divided into two processes. The first was the ascending rates where COD was released from the solid phase into the liquid phase and decomposed biologically to a simple substrate, such as organic acids and the second was the descending rates where COD being released from the waste was stabilized and converted to mainly H_2O and CO_2 as a result of aerobic biodegradation processes. The COD concentration in the leachate was also affected by simulated rainfall. The effect of washing by simulated rainfall will increase COD at the beginning and decrease after reach peak concentration. The ascending and descending rates of COD concentration are shown in Table 4.1. The ascending rates (positive slopes) were calculated from the initial stage of the process to the maximum values of COD concentration as shown in Figure 4.2, and the descending rates (negative slopes) were calculated from the maximum point of COD concentration to the point just before the COD concentration tapered gradually. Again, for the biodegradation rates of AER5 and AER6, both ascending and descending rates were almost identical, which indicated that the physical and biochemical processes occurring in these bioreactors had a very high degree of reproducibility.

The highest ascending rate (567 mg/L.d) occurred in bioreactor AER4 which received the highest (855 mL/kg MSW.d) leachate recirculation and sludge (85.5 mL/kg of MSW.d) addition. The highest descending rate of 425 mg/L.d could be found in bioreactor AER5 and AER6 which was faster compared to the bioreactor AER4. This is probably because the maximum COD concentration (approximately 40,000 mg/L) in the leachate from bioreactor AER5 and AER6 were greater than in AER4 (approximately 38,000 mg/L). In addition, the time from the peak COD concentration to the point just before the rate tapered gradually for the leachate from bioreactor AER5 was also shorter (14 weeks) compared to AER4 (16 weeks).

Table 4.1 Ascending and descending rates of COD concentrations in the leachate from aerobic bioreactors

Bioreactors	Ascending rates (mg/L.day)	Descending rates (mg/L.day)
AER1	402	366
AER2	491	335
AER3	348	264
AER4	567	375
AER5	307	425
AER6	332	425

From the above results (Table 4.1), it is clear that the fastest MSW biodegradation occurred in bioreactor AER4. Conversely, the slowest MSW biodegradation was detected in AER1 which received the lowest (285 mL/kg of MSW.d) volume of leachate recirculation and sludge (28.5 mL/kg of MSW.d) addition. As such, it would appear that the higher rates of leachate recirculation and sludge addition contributed to the faster MSW biodegradation. This is due to the increased volume of leachate recirculation, which facilitated the exchange of substrate, nutrients, buffer, and dilution of inhibitors, as well as the spreading of microorganisms within the waste matrix. Whereas, the sludge served as a seed of microorganisms, source of nitrogen, phosporus, and other trace nutrients required for microbial metabolism. However, it should be noted that the volume of leachate recirculation was 10-fold higher than sludge addition and the leachate also contained microorganisms and nutrients. As a result, the addition of sludge likely had a lesser affect on MSW biodegradation as compared to the volume of leachate recirculation.

4.1.1.3 BOD

To study in greater detail the fraction of biodegradable organic carbon in the leachate, the concentrations of BOD were measured and the results are presented in Figure 4.3. Similar trends to the COD were observed for the BOD concentrations in the leachate from all bioreactors. Generally, the BOD increased slowly over a period of 9 weeks and remained below 10,000 mg/L. Around the same time (11 weeks), bioreactor AER4 attained the peak value of 24,500 mg BOD/L, while AER2 attained the peak value of 22,000 mg BOD/L after 15 weeks. For the remaining bioreactors, BOD peaks ranged from 30,000 to 35,000 mg/L and were attained after the 17^{th} to 19^{th} week periods. Peak values were followed by a sharp decrease, with a BOD of approximately 4,000 mg/L for AER2, 500 mg/L for AER4, and 7,000 mg/L for AER5 and AER6 which were attained after 27 weeks from the starting date of the experiment. At the same time, the BOD concentrations of the bioreactors AER1 and AER3 were approximately 18,000 and 12,000 mg/L, respectively.

Similar to that observed for COD concentration, the BOD concentration profiles from the two center point replicates, AER5 and AER6, coincided throughout the entire period of operation. These profiles also show that bioreactors AER5 and AER6 had very good reproducibility.

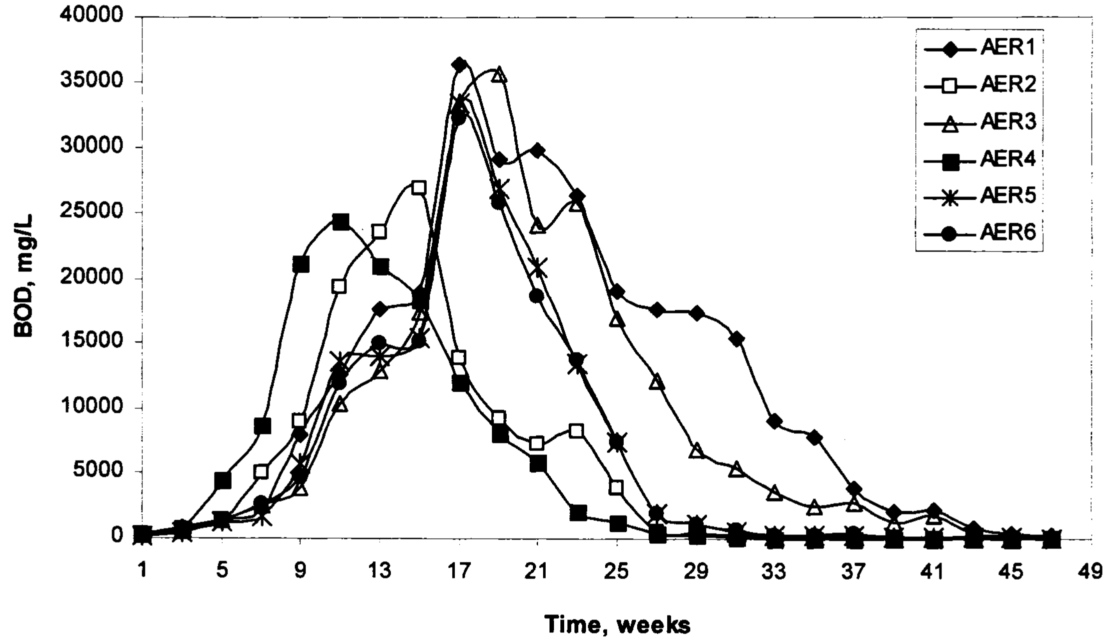

Figure 4.3 Concentrations of BOD in leachate from aerobic bioreactors

As previously indicated, the rates of MSW biodegradation in terms of BOD concentration were divided into two parts, ascending and descending rates. The ascending rate of AER4 was 344 mg/L.d and the descending rate was 237 mg/L.d. The ascending rate of AER1 was 210 mg/L.d and the descending rate was 189 mg/L.d. The ascending rate of AER4 was 1.6-fold greater than in AER1, and the descending rate was not significantly different. Even though descending rates were not significantly different, BOD concentration of approximately 1,000 mg/L was attained within 25 weeks for AER4, and 47 weeks for AER1. The peak values of BOD concentration of AER4 and AER1 were approximately 24,500 and 36,500 mg/L, respectively. The time to reach these values, however, was 11 and 17 weeks for AER4 and AER1, respectively. Bioreactor AER4 received 855 mL/kg of MSW.d of leachate recirculation and 85.5 mL/kg of MSW.d of sludge. In contrast bioreactor AER1 received 285 mL/kg of MSW.d of recycled leachate and sludge addition was 28.5 mL/kg of MSW.d. These results indicate that the volume of leachate and sludge added to the MSW bioreactors had a significant effect on the rate of waste biodegradation. Increasing the rate of leachate recirculation enhances contact between microorganisms and the biodegradable waste. Addition of sludge also increases the number of microorganisms and other nutrients. As a result, microbial activities increase inside the bioreactors and hence the rate of MSW biodegradation is enhanced.

Figure 4.4 shows the change in BOD/COD ratio of leachate collected from all the bioreactor models over the 47 weeks of operating time. During the first 9 weeks, the BOD/COD ratio increased from approximately 0.2 to 0.75 for all bioreactors. Then, the BOD/COD ratio varied between 0.6 to 0.85 for AER1, AER3, AER5 and AER6 until week 31st, which implies that most of effluent leachate still had highly biodegradable matter. After 13 weeks of operation, the BOD/COD ratio in AER4 decreased gradually from 0.65 to around 0.1 in 39 weeks. While in AER2, the BOD/COD ratio started to decrease from 0.7 after 15 weeks to 0.1 in 43 weeks. In the rest of the bioreactors, AER1, AER3, AER5 and AER6, the BOD/COD ratio decreased after 35 weeks of operation and ranged from 0.5 to 0.1. At the end of operation (47 weeks), the BOD/COD ratio ranged between 0.15 to 0.05 with the exception of AER1 and AER3.

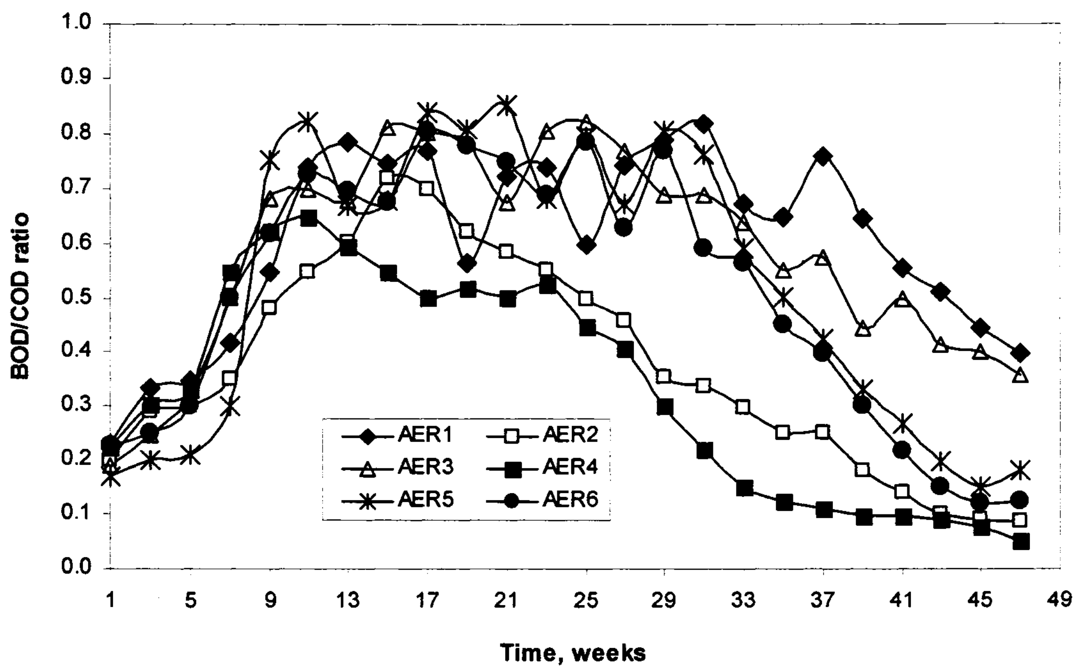

Figure 4.4 BOD/COD ratio in leachate from aerobic bioreactors

At the end of the experimental period, the lowest BOD/COD ratio of 0.05 was found in AER4. This is a clear indication that most of the biodegradable fraction of organic matter decreased significantly in AER4. Also, these results revealed that the MSW in AER4 was almost stabilized and would not require further oxygen demand. In contrast, the BOD/COD ratio remained above 0.35 for AER1 and AER3, indicating that the biodegradable fraction of organic matter in the leachate was still high compared to the non-biodegradable fraction and it would require some more time prior to complete stabilization. It is documented by Warith *et al* (1999), Townsend *et al.* (1996), Kjeldsen and Christensend (1989) that the BOD/COD ratio in the leachate lies from 0.02 to 0.87 for all stages of biodegradation, which compare favorably with the results obtained from the present experimental study.

4.1.1.4 Nitrogen

The Total Kjeldahl Nitrogen (TKN) and NH_3-N concentrations in the leachate were also measured to determine nutrient content released in the leachate from the MSW. Figure 4.5 illustrates the TKN results for leachate collected from all bioreactors over the experimental period. The highest nitrogen concentration was found in AER1, which also received the lowest leachate recirculation and sludge addition. At the beginning of the process the TKN concentration in AER1 was approximately 330 mg/L, increasing to a peak value of approximately 400 mg/L after 15 weeks, and then decreasing to a level of 260 mg/L by the end of the experimental period.

The TKN concentration profile in the bioreactors had similar trends: a rising curve, followed by a peak in concentration, and concluding with a declining curve. The sole exception being AER2, which only had a declining curve ranging from 230 to 90 mg/L over a period 47 weeks. With the exception of AER1, the TKN concentrations in the leachate from AER3, AER4, AER5 and AER6 were between 60 to 140 mg/L at the end of the experimental work. These results show that TKN concentration decreased over time primarily due to aerobic oxidation process. This is consistent with the work done by Leikam *et al.* (1999), who reported that nitrogen content can be reduced significantly in the simulated aerobic landfill reactor with TKN concentration from about 400 mg/L at the beginning to 70 mg/L after 200 days of operation.

Increasing TKN concentration in the leachate was primarily due to the breakdown of organic nitrogen in the bioreactors from solid phase into liquid phase. In addition, the breakdown of organic nitrogen at lower pH (≈ 7) in the leachate increases the NH_4^+-N concentration. This pH (≈ 7) is favorable for retaining ionized ammonium nitrogen in the aqueous form.

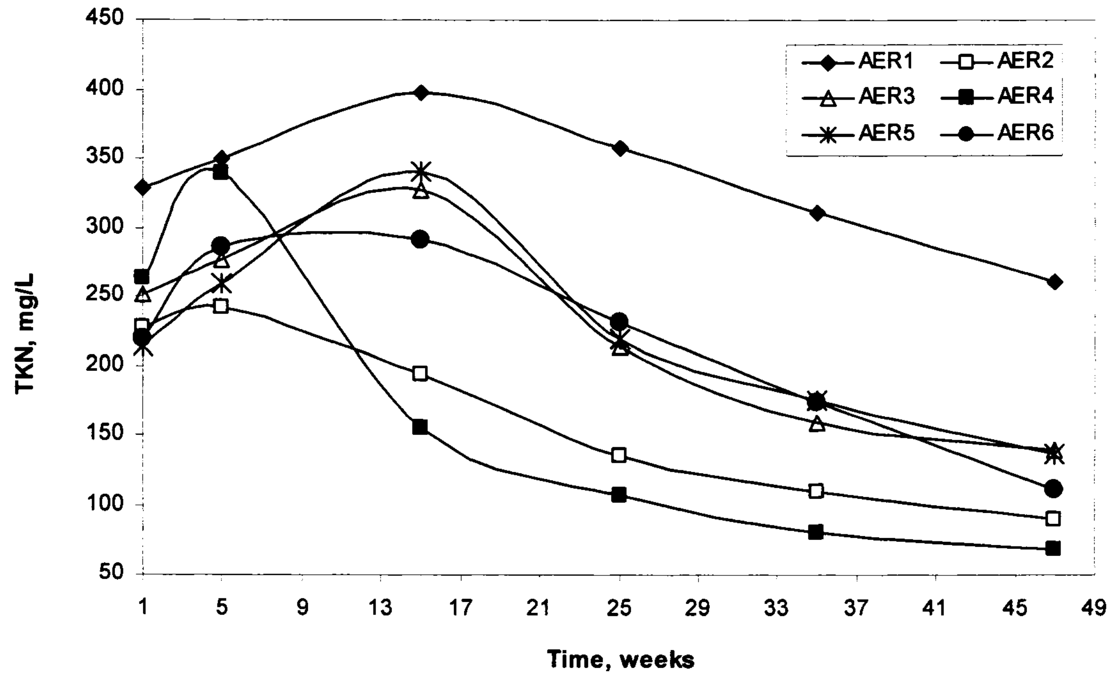

Figure 4.5 TKN Concentrations in leachate from aerobic bioreactors

The NH_3-N concentration in the leachate during this experimental work is presented in Figure 4.6. At the beginning of operation, the NH_3-N concentrations in the leachate for all bioreactors were between 130 to 230 mg/L, and then increased by about 180 to 280 mg/L until 15 to 17 weeks. Subsequently, the NH_3-N concentrations started to decline until the end of operation with concentrations between 60 to 130 mg/L for all bioreactors except AER1 with concentration of about 210 mg/L. This could be due to the amount of leachate recirculation and sludge addition in AER1, which was the lowest, thereby causing slower biochemical breakdown of organic nitrogen inside bioreactor

AER1 compared to other bioreactors. Therefore, the NH_3-N concentrations were attributed directly to the leachate recirculation management strategy within the bioreactors. The study reported by Sponza and Agdag (2004) and San and Onay (2001) showed that leachate recirculation increased opportunity for accumulation and/or removal through biological nitrogen assimilation. As a result, the decomposition rate of nitrogen compounds in AER1 was the slowest.

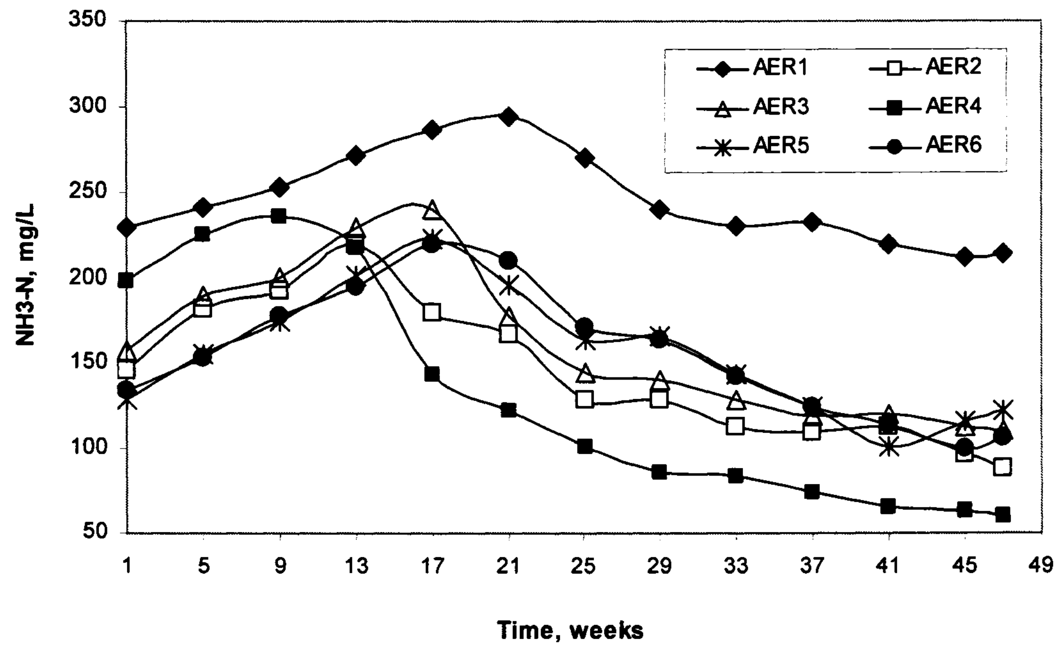

Figure 4.6 NH_3-N Concentrations in leachate from aerobic bioreactors

Measurement of NO_3^- was also conducted to investigate whether nitrification was taking place. Table 4.2 shows NO_3^- concentrations in the leachate from all bioreactors during the course of experiments. The NO_3^- in the leachate increased over the operating period for all bioreactors with the highest concentration (4.3 mg/L) in AER4, and AER1 for the lowest (2.1 mg/L) at the end of the experiment. This revealed that nitrification occurred to some extent in all bioreactors.

Table 4.2 Nitrate concentrations in the leachate from aerobic bioreactors

Week	NO_3^-, mg/L					
	AER1	AER2	AER3	AER4	AER5	AER6
5	0.9	2.1	1.1	2.3	1.9	1.8
15	1.2	2.5	1.4	2.5	2.3	2.2
25	1.4	3.1	1.5	3.2	2.6	2.5
35	1.7	3.9	1.8	4.1	2.7	2.8
47	2.1	4.1	2.5	4.3	3.2	3.3

4.1.1.5 Phosphorus

Concentrations of total phosphorus (TP) in the leachate from all bioreactors during the experiment are shown in Table 4.3. It can be seen that all TP concentrations in the leachate from all bioreactors had similar concentrations and were almost constant during the degradation process. These results indicate that there was no degradation process affecting the total phosporus concentration in the waste matrix. Similar studies were reported by Jackcobsen (1992) and Zhan (1993) who found that the total amount of phosphorus did not change during the composting process.

Table 4.3 Total Phosporus Concentrations in leachate from aerobic bioreactors

Week	Total phosphorus, mg/L					
	AER1	AER2	AER3	AER4	AER5	AER6
5	0.66	0.64	0.59	0.52	0.60	0.61
15	0.61	0.60	0.54	0.49	0.57	0.59
25	0.58	0.54	0.55	0.47	0.59	0.62
35	0.57	0.57	0.49	0.50	0.48	0.58
47	0.51	0.49	0.42	0.45	0.50	0.54

4.1.1.6 Metals

The concentration of heavy metals in the leachate is presented in Table 4.4. It can be seen that the concentration of heavy metals varied within a narrow range during the experimental study. Generally, these concentrations are low compared to the permissible maximum level (ML) defined by the Ontario Drinking-Water Quality Standards (ODWQS), Safe Drinking Water Act, 2002 and U.S. Drinking Water Standards (see Table 4.5) with the exception of Cd, Fe and Pb. There were no trends of an increase or a decrease of heavy metals concentrations in this experiment. Studies done by researchers on full-scale landfill, pilot scale, and laboratory test have shown that average metals concentrations in leachate are not a major concern (Kjeldsen and Christophersen, 2001; Revans *et al.*, 1999)

4.1.2 Solid Waste

The final product of MSW had physical characteristics of mature compost, i.e., earthy smell and dark brown colour. It had high total nitrogen ranging from 1.1 to 1.6 on dry weight basis, and the total phosphorus varied from 0.5 to 0.8 on dry weight basis. The MC of the stabilized MSW was almost the same at about 70%. This MC increased from the initial value which was only 20.2%. Simulated rainfall and leachate recirculation played an important role in increasing the final MC.

4.1.2.1 Characteristic of MSW

Physical and chemical properties of the final product of MSW from the aerobic simulated landfill bioreactors are presented in Table 4.6. These parameters are moisture content, volatile solids (VS), pH, total carbon, total nitrogen, and total phosporus. As previously discussed in section 3.2., the composition of MSW mostly consisted of biodegradable material.

Table 4.4 Concentrations of metals in leachate from aerobic bioreactors

Time (weeks)	B	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Zn
	Limit of Detection								
	0.0069	0.0114	0.0066	0.0048	0.0168	0.0015	0.0177	0.0723	0.0093
					AER1				
5	0.1613	BD	0.0061	0.0308	1.677	0.0534	0.0250	0.0240	0.0654
15	0.0887	0.0233	BD	0.0362	2.004	0.0163	0.0175	0.0286	0.0388
25	0.2912	0.0148	0.0083	0.0204	2.132	0.0338	0.0504	0.0275	0.1986
35	0.1929	0.0182	0.0092	0.0455	3.486	0.0278	0.0505	0.0161	0.4955
45	0.2128	0.0287	0.0136	0.0887	2.956	0.0261	0.0469	0.1553	0.2298
					AER2				
5	0.1828	BD	0.0087	0.0218	1.661	0.0261	0.0328	BD	0.1184
15	0.1854	0.0225	0.0074	0.0206	1.489	0.0267	0.0425	BD	0.1219
25	0.1847	0.0957	BD	0.0415	2.595	0.0258	0.0420	BD	0.1207
35	0.1848	0.0324	0.0069	0.0201	1.446	0.0253	0.0388	BD	0.1225
45	0.1825	0.0435	BD	0.0195	2.439	0.0254	0.0627	BD	0.1111
					AER3				
5	0.1636	0.0203	BD	0.0270	1.824	BD	BD	0.0846	0.0589
15	0.1050	0.0203	BD	0.0416	3.368	0.0408	0.0252	BD	0.0442
25	0.2867	0.0894	0.0079	0.0320	1.848	0.0265	0.0516	BD	0.0367
35	0.2959	0.0982	0.0125	0.0215	2.751	0.0434	0.0549	0.0824	0.2733
45	0.3132	0.1288	0.0428	0.0908	4.696	0.0539	0.0299	0.1321	0.4496
					AER4				
5	0.1839	BD	BD	0.0185	2.441	0.0019	0.0203	BD	0.0811
15	0.1822	0.0184	BD	0.0203	3.603	0.0127	0.0321	0.0854	0.2261
25	0.1832	0.0592	0.0412	0.0287	2.476	0.0283	0.0303	0.1652	0.123
35	0.1847	0.0825	0.0385	0.0391	3.548	0.0419	0.0997	0.1828	0.3475
45	0.1818	0.1263	0.0426	0.0489	4.484	0.0536	0.1568	0.1865	0.4356
					AER5				
5	0.0904	0.0203	BD	0.0367	2.376	0.0505	BD	BD	0.0435
15	0.1639	0.0363	0.0124	0.0297	1.469	0.0476	BD	BD	0.0542
25	0.2895	0.0329	0.0118	0.0248	2.942	0.0462	0.0385	BD	0.6331
35	0.2309	0.0224	0.0235	0.0398	2.556	0.0424	0.0566	0.1415	0.1571
45	0.1605	0.0284	0.0324	0.0448	2.066	0.0532	0.0363	0.1153	0.1024
					AER6				
5	0.1263	0.0265	0.0074	0.0519	2.822	0.0692	BD	0.0899	0.2051
15	0.1749	0.0335	0.0092	0.0212	2.165	0.0339	0.0865	BD	0.1607
25	0.2798	0.0288	0.0141	0.0232	2.039	0.0449	0.1032	BD	0.1584
35	0.2518	0.0295	0.0215	0.0426	2.044	0.0468	0.0564	BD	0.1514
45	0.1781	0.0275	0.0289	0.0481	2.006	0.0591	0.0372	0.1284	0.116

Note : 1. All concentrations are in ppm

2. BD = concentrations below the detection limit

Table 4.5 Ontario and US Drinking Water Quality Standards for metals

Metals	Maximum Level (ppm)		This study	
	Ontario	United States	Average (ppm)	Maximum (ppm)
B	5.0	-	0.1964	0.3132
Cd	0.005	0.005	0.0488	0.1288
Cr	0.05	0.1	0.0185	0.0428
Cu	1.3	1.3	0.0359	0.0908
Fe	-	0.3	2.5136	4.696
Mn	-	0.05	0.0374	0.0692
Pb	0.01	0.015	0.1029	0.1865
Zn	-	5.0	0.1827	0.6331

Source: Safe Drinking Water Act (2002), Ontario.
U.S. Drinking Water Standards (2001).

Table 4.6 Physical and chemical properties of MSW after end of operation from aerobic bioreactors

Component	Percent by weight					
	AER1	AER2	AER3	AER4	AER5	AER6
MC	69.9	70.5	70.1	70.2	69.8	69.9
Volatile Solids	91.4	89.1	90.1	87.8	89.9	89.5
pH[a]	7.4	7.1	7.2	7.1	7.3	7.2
Total carbon	21.4	18.9	21.7	18.4	20.3	20.7
Total nitrogen	1.6	1.2	1.4	1.1	1.5	1.4
Total phosporus	0.7	0.5	0.8	0.6	0.7	0.5

a: in 0.01 M $CaCl_2$

4.1.2.2 Temperature

Temperature was measured to gain a better understanding of the MSW degradation process in the bioreactor landfills. Figure 4.7 shows the average temperatures inside the bioreactors based on readings from the thermocouples inserted in each bioreactor. Since the temperature in the middle of bioreactors was between the temperature at the top and the bottom, the comparison of two samples was conducted to infer whether differences exist between the top and the bottom. In this case, a two-tailed variance ratio test (F) was used for the null hypotheses H_0: $\sigma_1^2 = \sigma_2^2$, where σ_1^2 and σ_2^2 are the variance of the top and the bottom temperatures, respectively. If F calculated is greater than F critical, the null hypothesis is rejected, and hence there is a significant difference between the two samples. However, the results of the paired F-test (α=0.05) in Table 4.7 show that there was no significant difference between the temperatures at the two sampling points of the individual bioreactors.

Table 4.7 Results of paired F-test (α=0.05) for temperatures from aerobic bioreactors

Parameter	Position	AER1	AER2	AER3	AER4	AER5	AER6
Mean	Top	30.06	35.07	36.94	31.47	32.34	32.72
	Bottom	37.41	34.53	36.39	31.01	31.92	32.3
Variance, σ^2	Top	200.37	172.63	188.19	178.71	121.39	129.46
	Bottom	197.07	172.47	188.63	177.88	121.98	129.66
Number of Sample, n	Top	47	47	47	47	47	47
	Bottom	47	47	47	47	47	47
Degree of Freedom, DF	Top	46	46	46	46	46	46
	Bottom	46	46	46	46	46	46
F calculation		1.017	1.001	0.998	1.005	0.995	0.998
F critical		1.69	1.69	1.69	1.69	1.69	1.69

From Figure 4.7 it can be observed that all bioreactors reach the thermopilic range, from 45^0 to 55^0 C, within 7 to 9 weeks of operation. Over the period of the first 7 weeks, temperatures rose rapidly to the thermophilic level (> 40 ^{0}C). This rapid increase in temperature was mainly due to an exothermic biochemical reaction in which microbial activities such as metabolism, respiration, and synthesis prevailed within the waste matrix. After 10 weeks, the peak values of 59, 56, 54 and 55 ^{0}C were recorded in AER2, AER4, AER5 and AER6, while at the same time temperatures in AER1 and AER3 continued to rise, reaching the peak values of approximately 59.5 and 59 ^{0}C after week 17 and 15, respectively.

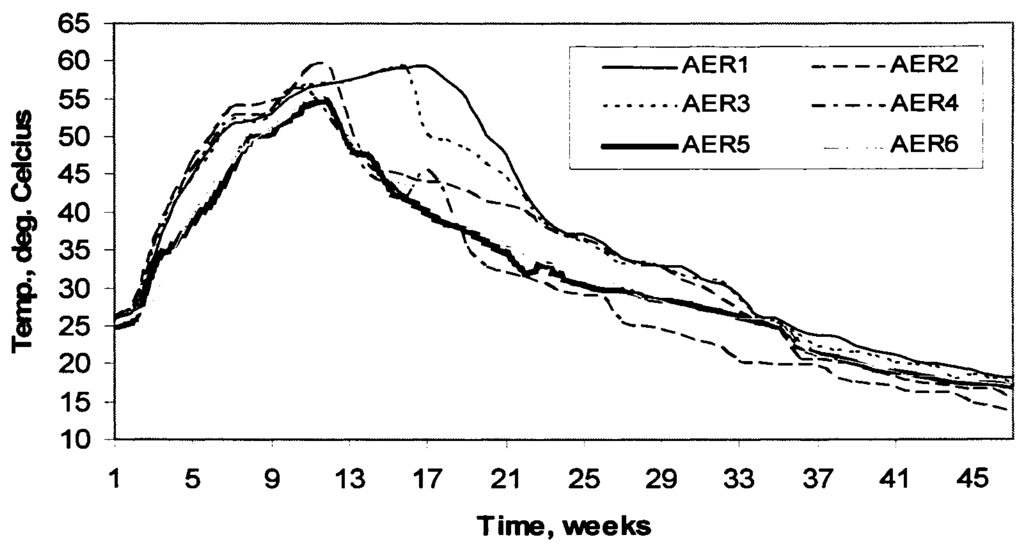

Figure 4.7 Average temperatures from aerobic bioreactors

The temperature profiles have a common crest type configuration, i.e., a rapid rise, a short peak duration for 1 week with the value around 55 to 59 ^{0}C, which is the temperature limit for sanitation purposes, followed by a sharp decrease to ambient temperature. It should be noted that the peak temperature in AER4 was not the highest recorded even though the rates of MSW stabilization in AER4 was highest in terms of COD and BOD concentration. Most likely, the higher volume of leachate recirculation reduced the temperature during the biodegradation process. The effect of air on the temperature could not be detected since the volume of air supplied in all bioreactors was the same.

In general, thermopilic temperatures (40 to 59 ^{0}C) were observed between weeks 7 to 23 of the experiment, followed by a temperature decrease to the mesophilic range (25 to 40 ^{0}C), finally ending below 15 ^{0}C for both AER4 and AER2, and between 17 to 19 ^{0}C for the rest of bioreactors. When temperature is in the thermophilic range, most pathogens are destroyed or killed. The US.EPA (1994) standard requires that wastes be sanitized at 55^{0}C for at least 3 days. For this reason, the aerobic degradation in landfills has an advantage over an anaerobic system, as the latter usually occurs in the mesophilic range and hence does not guarantee the destruction of pathogens.

4.1.2.3 Settlement

Volume reduction of MSW was investigated in each of the six bioreactors. Settlement was used to measure the volume reduction of the compacted MSW in the bioreactors. The height of the compacted MSW in bioreactors was measured to determine the settlement and was started from the beginning (day 0), after 9, 15, 25, 35 weeks of operation of bioreactors and at the end of the experiment (47 weeks). Results of these measurements, along with the calculated waste percentage settlement are presented in Table 4.8. The settlement of MSW ranged from 17 to 22 %. The highest degree of settlement was achieved in bioreactor AER4, which received the highest rate of leachate recirculation and sludge addition which contributed to an accelerated conversion of organic matter to CO_2 and H_2O. It is also possible that compaction was greater under the burden at a larger mass of water. In contrast, bioreactor AER1, which received the lowest rate of leachate recirculation and sludge addition, also had the lowest degree of settlement.

Table 4.8 Results of settlement measurements from aerobic bioreactors

Bioreactors	Original waste height (mm)	Height after weeks of operation (mm)					Final settlement (%)
		9 wk	15 wk	25 wk	35 wk	47 wk	
AER1	400	389	377	360	348	330	17.5
AER2	400	382	369	354	336	318	20.5
AER3	400	384	371	358	346	328	18.0
AER4	400	375	362	340	318	311	22.3
AER5	400	379	366	351	337	320	20.0
AER6	400	377	363	349	340	321	19.8

4.2 Anaeorbic Bioreactors

4.2.1 Leachate Quality

4.2.1.1 pH

Figure 4.8 shows the pH variations in the leachate during the experimental period of 65 weeks. At the beginning of operation, from week 1 to 9, the pH in the leachate varied from 5.6 to 6.5 for all bioreactors, with the exception of the ANA4 for which the pH was stabilized at about neutral (pH 7 to 7.2) after 7 weeks of operation. The pH in the leachate from the rest of bioreactors reached from 7 to 7.2 after 19 weeks. As in the aerobic bioreactors, the pH of the recycled leachate was maintained consistently in the neutral range by adding a solution of $NaHCO_3$ (1N) until the end of the experiment.

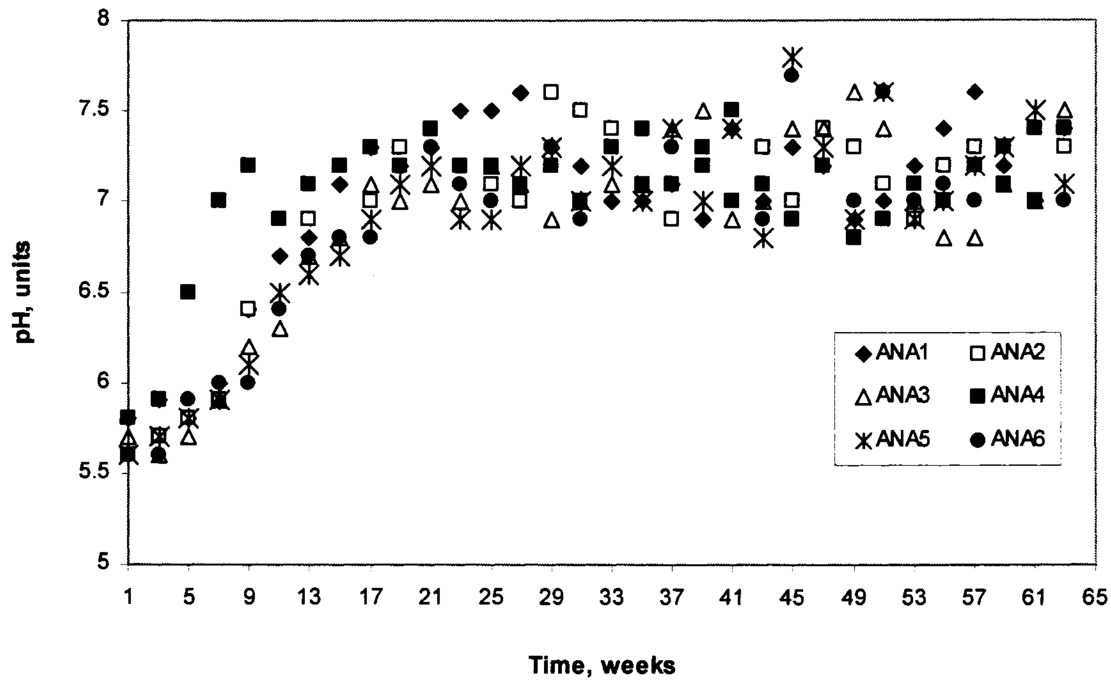

Figure 4.8 pH in leachate collected from anaerobic bioreactors

4.2.1.2 COD

Figure 4.9 depicts the concentration of COD in the leachate from the six bioreactors. At the initial stage, the COD concentration in the leachate from all bioreactors increased slowly over a period of 9 weeks and generally remained below 5,000 mg/L. After this stage, between the weeks of 33 to 39, the COD concentration increased exponentially and reached the peak value of about 54,000 mg/L for ANA5 and ANA6, 63,000 mg/L for ANA3, and 65,000 mg/L for ANA1. After 9 weeks of operation, the COD concentration increased exponentially and reached a peak value of about 45,000 mg/L for ANA4 (week 21st), and 52,000 mg/L for ANA2 (week 25th). Subsequently, this was followed by a decrease in the COD concentration at an almost constant rate until 35 weeks at which point no significant changes were observed in the COD concentration in the leachate. It can bee seen that all curves had a similar trend first a rising curve, followed by peak concentration, and finally a declining curve. The COD concentration in the leachate from ANA4 was about 1000 mg/L after 41 weeks of the operation. While during the same period, the COD concentrations in the leachate from the rest of the bioreactors were still above 7,000 mg/L. The trends of COD concentration showed that the MSW stabilization was achieved within 43 weeks in ANA4 and ANA2, where the volumes of leachate recirculation were 855 mL/kg MSW.d and the volumes of sludge addition were 1.285 mL/kg MSW.d and 57.0 mL/kg MSW.d, respectively.

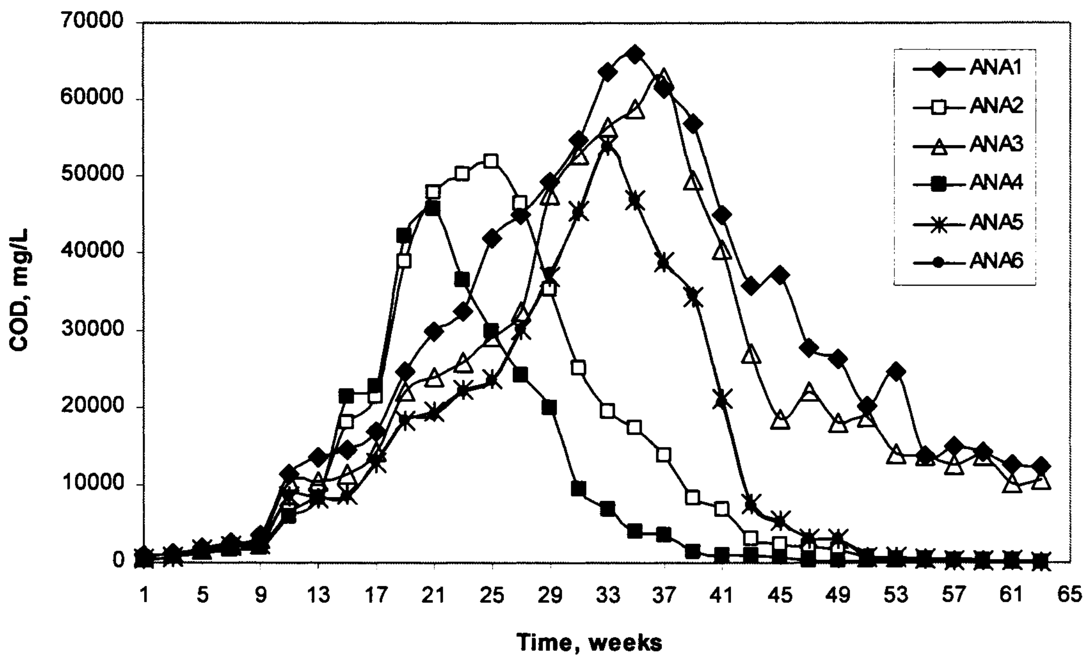

Figure 4.9 Concentrations of COD in leachate from anaerobic bioreactors

As in the aerobic bioreactors, the two stages of the overall biodegradation were ascending and descending rates. The ascending and descending rates of COD concentration are presented in Table 4.9. The highest ascending rate occurred for ANA2 (362 mg/L.d) which received the leachate recirculation and sludge addition of 855 and 28.5 mL/kg of MSW.d. While, the highest descending rate of 462 mg/L.d was in the ANA5.

Table 4.9 Ascending and descending rates of COD concentrations in the leachate from anaerobic bioreactors

Bioreactors	Ascending rates (mg /L.day)	Descending rates (mg/L.day)
ANA1	293	365
ANA2	362	354
ANA3	265	319
ANA4	311	322
ANA5	219	462
ANA6	219	460

The overall time to reach the peak COD concentration and then from the peak to the tapered line was used as a basis to calculate the biodegradation rates. According to Figure 4.9, it can be seen that, for ANA4, the time to attain the maximum COD concentration was 21 weeks, and within 41 weeks the MSW was considered stabilized since the COD concentration in the leachate was relatively constant (1000 mg/L). In comparison, ANA1 attained maximum COD concentration in 37 weeks and COD concentration was still above 12,000 mg/L after 63 weeks. The results obtained indicate that the highest rate of MSW biodegradation occurred in ANA4 and the slowest rate was in ANA1. The rate of MSW biodegradation for ANA4 was about 2-fold faster than the one in ANA1. In this case, the higher the rates of leachate recirculation, the faster waste biodegradation can be attained.

4.2.1.3 BOD

Results of measurement of BOD concentration are presented in Figure 4.10. Similar trends were observed for the BOD concentrations in the leachate from all bioreactors. Also, it was noted that the BOD data had a similar to the COD concentration trends, i.e., a rising, followed by peak, and a declining curve.

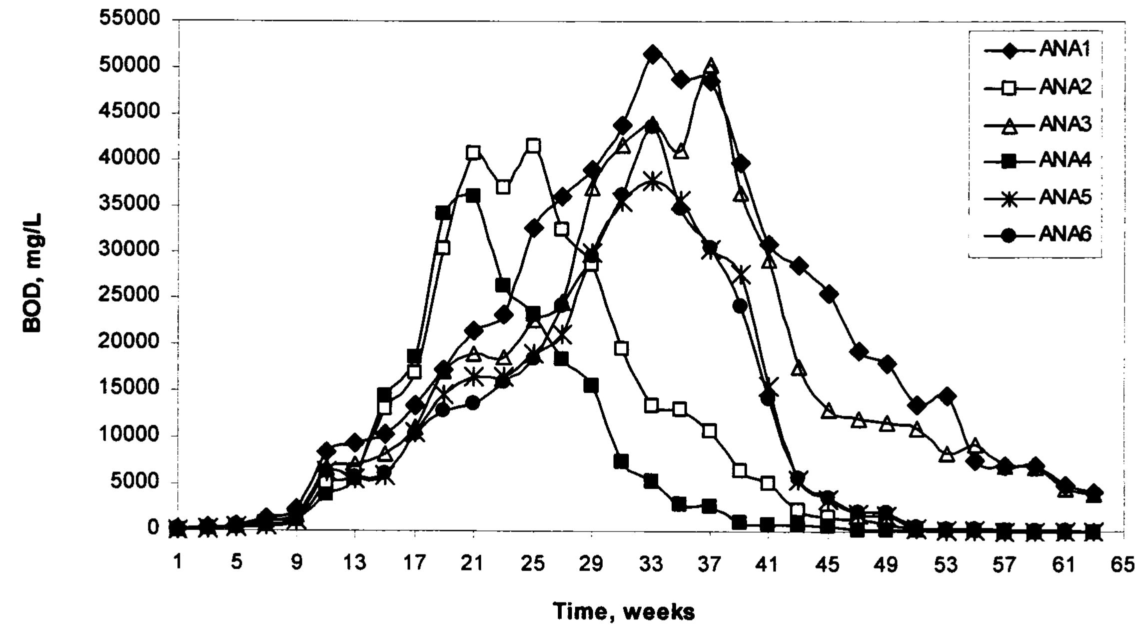

Figure 4.10 Concentrations of BOD in leachate from anaerobic bioreactors

At the beginning, BOD concentration in the leachate from all bioreactors increased slowly for a period of 9 weeks and generally remained below 2,000 mg/L with the exception of bioreactor ANA1 which reached about 2,500 mg/L.

The BOD concentration in ANA4 reached its peak value of about 36,000 mg/L after 21 weeks, and 41,000 mg/L for ANA2 after 25 weeks. The rest of the bioreactors had a peak value between 43,000 and 52,000 mg/L within a period of 35 to 39 weeks. These peak values were followed by a sharp decrease in BOD concentration until approximately 41 weeks. However, ANA1 and ANA3 required 57 weeks to reach the same low level of BOD concentration as in the other bioreactors. After 39 weeks of operation, BOD concentration decreased to approximately 1000 mg/L for ANA4, 6000 mg/L for ANA2, and was above 24,000 mg/L for the rest of bioreactors. Moreover, in ANA1 and ANA3, the BOD concentration was still above 36,000 mg BOD/L after a period of 39 weeks. As in the case of COD concentrations, the results indicate that the MSW stabilization was first achieved in ANA4, where the volume of leachate recirculation was 855 mL/kg MSW.d and the volume of sludge addition was 85.5 mL/kg

MSW.d. It can be concluded that a higher volume of leachate recirculation and sludge addition contributed to a higher rate of waste biodegradation in the simulated bioreactor landfill, which had a significant effect on the rate of waste biodegradation.

A BOD/COD ratio is used to study the fraction of the biodegradable organic matter in the leachate. Figures 4.11 show the change in BOD/COD ratio in leachate collected from all bioreactors over time. The BOD/COD ratio ranged from 0.05 to 0.85, where the lowest and the highest values were found in ANA4 and ANA5, respectively. This is consistent with work done by some researchers (Kjeldsen and Christopherson, 2001, Townsend *et al.*, 1996; Pohland, 1980) that the BOD/COD ratio in the leachate varied from 0.02 to 0.87 for all stages of biodegradation in anaerobic landfills. A ratio of 0.4-0.80 implies the presence of highly biodegradable components in the leachate. At the initial stage, the BOD/COD ratio ranged from about 0.15 to 0.6 for all bioreactors until 9 weeks, then the BOD/COD ratio was scattered within the range of about 0.65 to 0.8 until 45 weeks with the exception of ANA4 in which the BOD/COD ratio was below 0.65 after 37 weeks. The BOD/COD ratio decreased after 45 weeks of operation, indicating that the biodegradable fraction was decreasing compared to the non-biodegradable fraction.

At the end of operation (63 weeks), the BOD/COD ratio ranged from 0.15 to 0.20 with the exception of ANA1 and ANA3 where it was about 0.35 to 0.4 which indicated that the biodegradable fraction of organic matter in the leachate was still high compared to the non-biodegradable fraction. The lowest BOD/COD ratio was found in ANA4 at the end of operation. Clearly, these results indicate that most of the biodegradable organic matter was utilized in ANA4, which received the highest leachate recirculation and sludge addition.

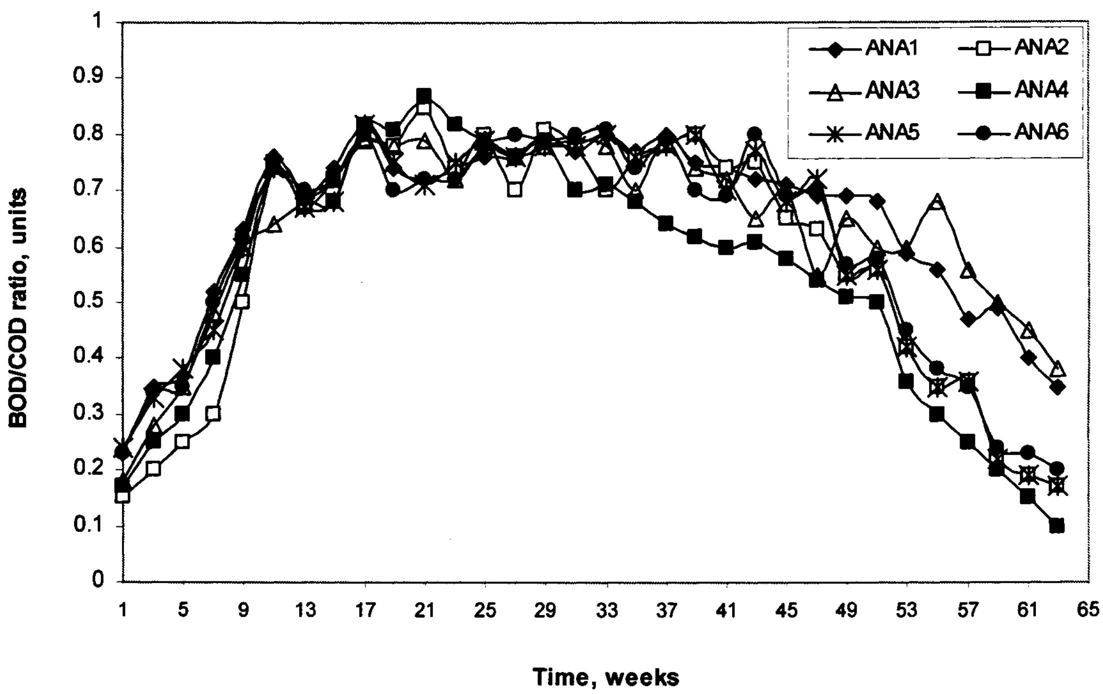

Figure 4.11 BOD/COD ratio in leachate from anaerobic bioreactors

4.2.1.4 Nitrogen

The nitrogen concentration, TKN and NH_3-N, in the leachate was measured to know the nutrient content of the MSW under anaerobic biological decomposition. Figure 4.11 shows the TKN concentration in the leachate collected from all bioreactors over time. The highest nitrogen concentration was found in ANA1 which received the lowest leachate recirculation and sludge addition. At the beginning of the experiment, the TKN concentration in ANA1 was about 330 mg/L, then increased to a peak value of about 400 mg/L after 15 weeks, and was 260 mg/L at the end of the operation. The profile of TKN concentration in all bioreactors had a similar trend: first a rising curve, followed by peak concentration, and finally a declining curve, with the exception of ANA2 which only displayed a declining curve. At the end of the experiment, the TKN concentrations in the leachate from the bioreactors, ANA1 and ANA3, were about 250 and 180 mg/L, respectively. While, for the rest of bioreactors these were in the range of 80 to 130 mg TKN/L.

In general, the trend shows that TKN concentration decreased over time for all 6 bioreactors. It can be concluded, therefore, that the nitrogen concentration can be reduced by anaerobic decomposition. This is supported by the work of Leikam *et al.* (1999) which reported that nitrogen content can be reduced significantly (about 70%) in the anaerobic landfill simulation reactor after 450 days of the experimental work.

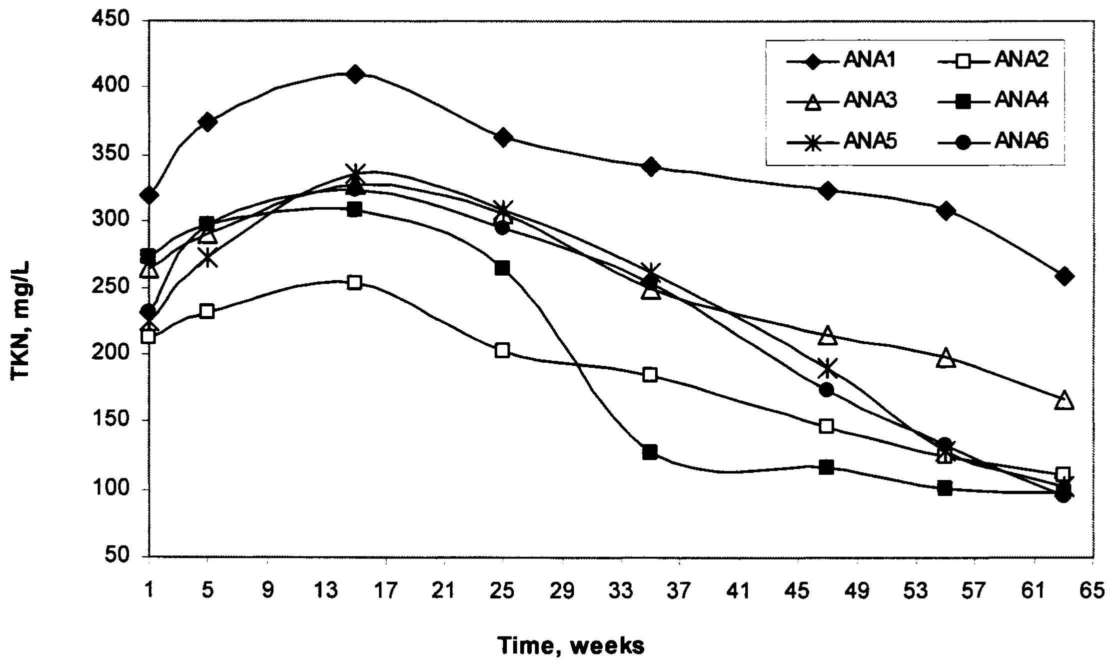

Figure 4.11 TKN Concentrations in Leachate from Anaerobic Bioreactors

Figure 4.12 shows the NH_3-N concentration in the leachate collected from all bioreactors over time. Similar to the TKN concentration, the highest NH_3-N concentrations was found in the ANA1, which received lowest leachate recirculation and sludge addition.

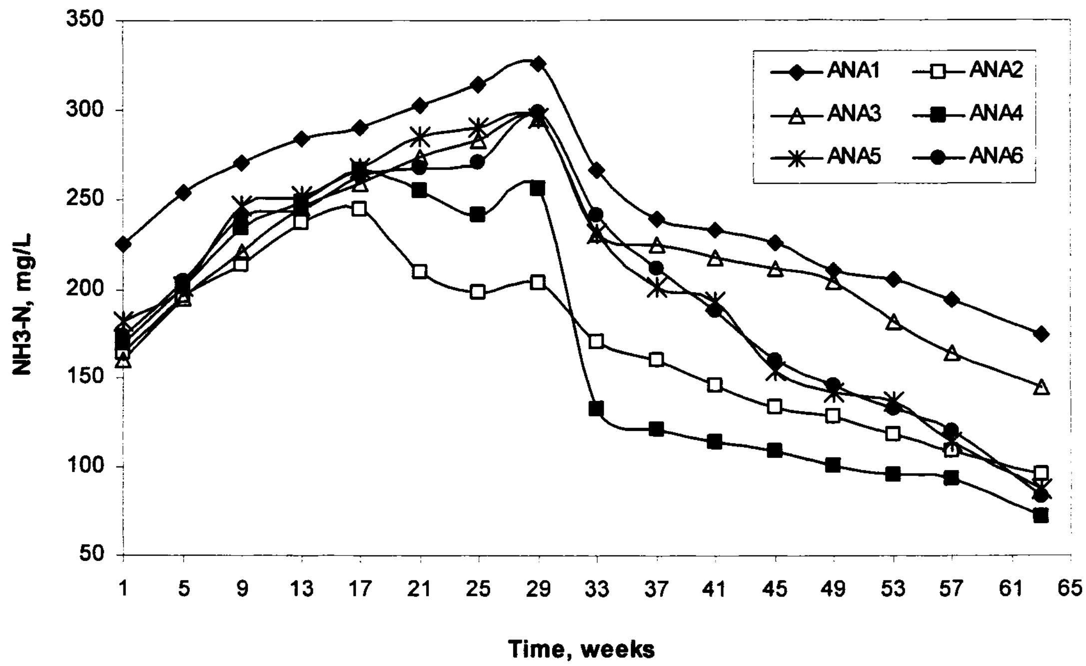

Figure 4.12 NH_3-N Concentrations in leachate from anaerobic bioreactors

As in aerobic bioreactors, measurement of NO_3^- was also conducted to examine whether nitrification was taking place. The NO_3^- concentrations in the leachate from all bioreactors during the course of experiment is presented in Table 4.10. It can be seen that in all bioreactors, concentration of NO_3^- in the leachate can only observed in the first 5 weeks after the experiment with the highest NO_3^- concentration in ANA4 (1.6 mg/L) and ANA1 (0.8 mg/L) for the lowest. This revealed that nitrification did not occur or it was below the detection limit from week 15 until the end of operation which is indication that the anaerobic conditions were achieved in all 6 bioreactors.

Table 4.10 **Nitrate concentrations in leachate from anaerobic bioreactors**

Week	NO_3^-, mg/L					
	ANA1	ANA2	ANA3	ANA4	ANA5	ANA6
5	0.8	1.2	0.9	1.6	1.0	0.9
15	ND	ND	ND	ND	ND	ND
25	ND	ND	ND	ND	ND	ND
35	ND	ND	ND	ND	ND	ND
47	ND	ND	ND	ND	ND	ND

Note:
ND: Non detectable

4.2.1.5 Phosphorus

Phosphorus is an essential nutrient that is used by microorganisms in the degradation process. Total phosphorus (TP) concentrations in the leachate from all bioreactors during the experiment are shown in Table 4.11. The TP concentrations were almost constant during the biodegradation process within the range of about 0.4 to 0.6 mg/L. This result indicated that there was no substantial removal of the total phosphorus present in the waste matrix.

Table 4.11 Total phosphorus concentrations in leachate from anaerobic bioreactors

Week	Total phosphorus, mg/L					
	ANA1	ANA2	ANA3	ANA4	ANA5	ANA6
5	0.67	0.69	0.61	0.55	0.63	0.59
15	0.62	0.65	0.59	0.48	0.59	0.61
25	0.63	0.60	0.58	0.52	0.57	0.63
35	0.59	0.63	0.55	0.54	0.55	0.57
47	0.61	0.67	0.60	0.46	0.48	0.51

4.2.1.6 Metals

The concentration of heavy metals in the leachate is presented in Table 4.12. As in aerobic bioreactors, the heavy metals in anaerobic bioreactors varied within a narrow range during the course of the experiment. These concentrations are low compared to Ontario Drinking-Water Quality Standards (ODWQS), Safe Drinking Water Act, 2002 and U.S. Drinking Water Standards. There were no trends indicating an increase or decrease of heavy metal concentrations. Several studies based on full-scale landfill, test-cell, and laboratory test have shown that average metal concentrations are fairly low compared to the U.S. Drinking Water Standards (Kjeldsen and Christophersen, 2001; Revans *et al.*, 1999).

4.2.2 Solid Waste

4.2.2.1 Characteristic of MSW

At the end of experiment, the final product of MSW from the simulated anaerobic bioreactor landfills was analyzed and presented in Table 4.13. As previously showed in section 3.2 (see Table 3.4), compositions of MSW mostly contained readily biodegradable material. Physical and chemical properties of MSW at the end of experiment including MC, VS, pH, total carbon, nitrogen, phosporus, and potassium are presented in Table 4.14.

The final product of MSW had physical characteristics of mature compost, i.e., earthy smell and dark brown colour. It had high total nitrogen ranging from 1.3 to 1.7% on a dry weight basis, and the total phosphorus varied from 0.5 to 1.1% on a dry weight basis. The final MC of the anaerobically composted MSW was almost the same at about 70% for all bioreactors. This MC increased from the initial value which was only 20.2%. Simulated rainfall and leachate recirculation played an important role in increasing the final MC of the waste product

Table 4.12 Concentrations of heavy metals in leachate from anaerobic bioreactors

Time (weeks)	B	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Zn
	Limit of Detection								
	0.0069	0.0114	0.0066	0.0048	0.0168	0.0015	0.0177	0.0723	0.0093
	ANA1								
5	0.1723	BD	0.0087	0.0298	1.784	0.0495	0.0240	0.0812	0.0596
15	0.0294	0.0203	0.0142	0.0424	2.146	0.0264	0.0210	0.0944	0.0485
25	0.2387	BD	0.0091	0.0358	2.781	0.0348	0.0489	0.1297	0.2024
35	0.1882	0.0199	0.0175	0.0405	2.993	0.0302	0.0524	0.0983	0.3958
45	0.1241	0.0394	0.0128	0.0746	2.940	0.0273	0.0276	0.1496	0.2473
55	0.1547	0.0325	0.0135	0.0762	2.729	0.0361	0.0423	0.1358	0.2812
	ANA2								
5	0.1496	0.0195	0.0149	0.0198	2.250	BD	0.0232	0.0796	0.1184
15	0.1453	BD	0.0093	0.0272	1.563	0.0084	0.0385	BD	0.1219
25	0.2345	0.0156	BD	0.0165	1.684	0.0152	0.0267	0.0859	0.1207
35	0.1375	0.0245	0.0174	0.0154	1.965	0.0228	0.0283	0.0934	0.1225
45	0.1512	0.0172	0.0186	0.0248	2.820	0.0423	0.0421	BD	0.1111
55	0.1534	0.0182	0.0135	0.0231	2.754	0.0412	0.0326	0.0824	0.1546
	ANA3								
5	0.2556	0.0224	BD	0.0245	2.450	0.0018	BD	0.0864	0.0762
15	0.2105	0.0812	0.0072	0.0368	2.563	0.0426	0.0495	0.1480	0.2058
25	0.3235	0.0745	0.0388	0.0356	3.628	BD	0.0546	0.0954	0.3572
35	0.3145	0.0976	0.0429	0.0421	3.456	0.0585	0.0617	BD	0.4467
45	0.3211	0.1053	0.0462	0.0484	4.568	0.0669	0.1723	0.1420	0.4364
55	0.3026	0.1627	0.0398	0.0423	3.856	0.0557	0.1346	0.1385	0.4235
	ANA4								
5	0.2839	0.0153	BD	BD	2.826	0.0020	0.0188	0.0845	0.0834
15	0.1822	0.0149	0.0243	0.0322	2.261	0.0542	0.0357	BD	0.3653
25	0.2832	0.0663	0.0346	0.0352	3.524	0.0459	0.0525	0.1276	0.3340
35	0.2847	0.0985	0.0475	0.0496	3.348	0.0527	0.1897	BD	0.4245
45	0.3818	0.1161	0.0341	0.0445	4.265	0.0616	0.1943	0.1763	0.5481
55	0.2684	0.1247	0.0383	0.0385	3.561	0.0537	0.1384	0.1687	0.5163
	ANA5								
5	0.0915	0.0203	0.0121	BD	1.172	BD	0.0194	BD	0.0035
15	0.1878	0.0384	0.0098	0.0174	2.883	0.0264	BD	BD	0.0673
25	0.2452	0.0387	0.0091	0.0198	1.254	0.0218	0.1248	BD	0.1284
35	0.2078	0.0297	0.0399	0.0233	2.597	0.0347	0.1382	BD	0.2821
45	0.2385	0.0294	0.0257	0.0264	2.556	0.0325	0.1616	0.1169	0.2686
55	0.2164	0.0195	0.0271	0.0311	2.658	0.0324	0.1734	0.1216	0.2487
	ANA6								
5	0.1228	0.0240	0.0082	0.0056	1.427	0.0021	0.0225	BD	0.1524
15	0.1543	0.0139	0.0096	0.0152	2.198	BD	0.0362	0.0876	0.1020
25	0.2358	0.0391	0.0134	0.0202	1.261	0.0295	0.1623	BD	0.2346
35	0.2027	0.0257	0.0285	0.0239	2.654	0.0342	0.1426	BD	0.2058
45	0.2364	0.0286	0.0212	0.0281	2.562	0.0364	0.1785	BD	0.3286
55	0.2345	0.0204	0.0512	0.0352	3.124	0.3517	0.1324	0.1324	0.3194

Note : 1. All concentrations are in ppm
2. BD = concentrations below the detection limit

Table 4.13 Physical and chemical properties of MSW at the end of operation from anaerobic bioreactors

Parameter	Percent by weight (dry basis)					
	AER1	AER2	AER3	AER4	AER5	AER6
MC[a]	70.1	70.5	70.1	71.4	70.3	70.1
Volatile Solids	91.6	88.8	90.1	89.5	90.2	90.3
pH[b]	7.1	7.2	7.2	7.4	7.5	7.3
Total carbon	23.9	20.0	21.7	19.5	21.5	21.9
Total nitrogen	1.7	1.4	1.4	1.3	1.5	1.4
Total phosporus	1.1	0.7	0.8	0.6	0.8	0.9

a: percent by wet basis
b: in 0.01 M $CaCl_2$

4.2.2.2 Temperature

As in the aerobic bioreactors, the *F*-test was used to determine if there were significant differences between the temperatures at the top and the bottom of the bioreactors. Results of the paired F-test (α=0.05) show that there was no significant difference between the temperatures at those two sampling points of the individual bioreactors.

Temperatures inside anaerobic bioreactors are presented in Figure 4.13. It can be seen that all bioreactors reached the mesophilic range from 30^0 to 35^0 C, within 15 to 29 weeks of operation. From the beginning until the 25^{th} week of operation, the temperatures rose linearly to the mesophilic level. Microbial activities such as metabolism, respiration, and synthesis prevailed within the waste matrix and increased the temperature. Between 25 and 27 weeks of operation, the peak values of 39 and 38 ^{0}C were recorded for ANA2 and ANA4, while during the same time temperatures in ANA1, ANA3, ANA5 and ANA6 continued to rise gradually reaching the peak values of approximately 38 ^{0}C in about 40 to 45 weeks. At the end of operation, temperatures varied between 20 and 32 ^{0}C. Unlike in aerobic bioreactors, thermopilic temperatures (40 to 59 ^{0}C) were never observed in anaerobic bioreactors. As a result, the pathogens kill would not have occurred under these anaerobic conditions.

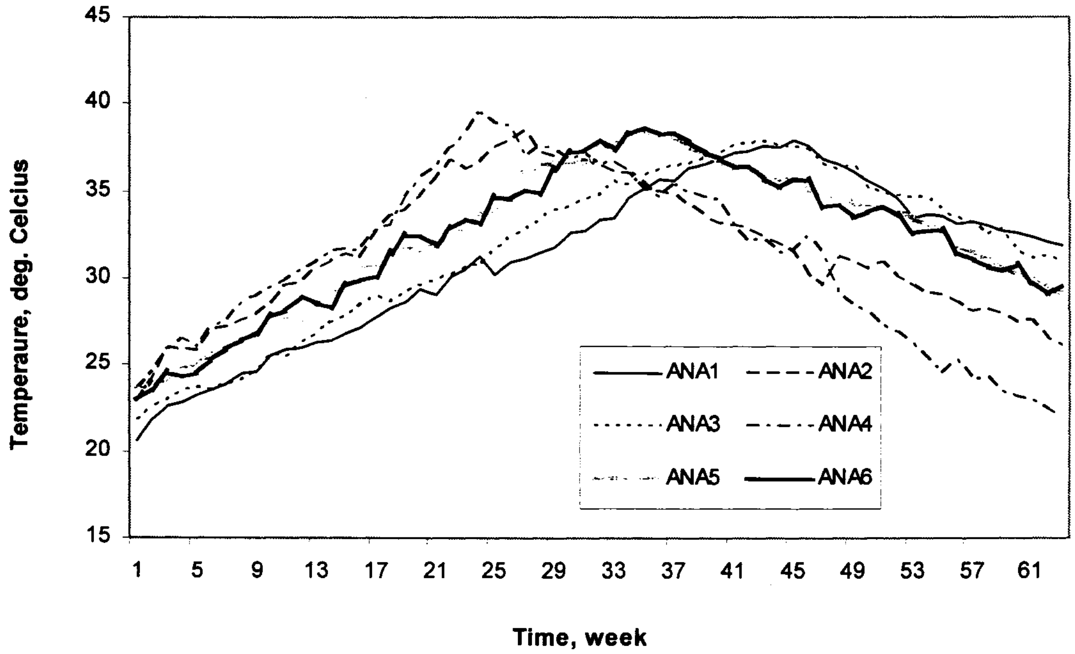

Figure 4.13 Average temperatures from anaerobic bioreactors

4.2.2.3 Gas Production

Figure 4.14 shows the trends of weekly biogas production from the simulated anaerobic bioreactors. It can be seen that biogas production started after 10 and 12 weeks of the experiment for ANA2, ANA4, ANA5 and ANA6. While for ANA1 and ANA3, the biogas production started after 18 weeks of the experiment. This indicates that the development of the anaerobic microorganisms may not occur before the 10 weeks of the experiment. During this period, the pH in the leachate was less than 6.5 which inhibited the development of the anaerobic microorganisms. These results agree with those obtained from the study carried out by Al-Yousfi (1992), and Christensen and Kjeldsen (1989). Their study concluded that high acidic leachate with pH less than 6.5 may contain a high concentration of fatty acids that does not support the development of anaerobic microorganisms.

The weekly gas production trends showed that the biogas production rate was higher for ANA4, where the volume of leachate recirculation was 855 mL/kg MSW.d and the volume of sludge addition was 85.5 mL/kg MSW.d. In ANA4, the maximum biogas production rate of about 3.2 L/wk was reached after 31 weeks of the experiment. While for the rest of anaerobic bioreactors, the maximum biogas productions varied from 1.8 to 2.9 L/wk. This reveals that higher rate recirculation and sludge addition will cause higher development of methanogenic activity. As a result, the biogas production rate will also increase. This work consistent with the work of several researchers that found MSW biodegradation can be achieved by increasing leachate recirculation content to the waste (Warith *et al.*, 1999, Bae *et al.*, 1998, Chugh *et al.* 1998, and Barlaz *et al.*, 1992). Similarly, Reinhart and Al-Yousfi (1996) reported that reactors with leachate recirculation showed a 12-fold increase in biogas production compared to reactors without leachate recirculation.

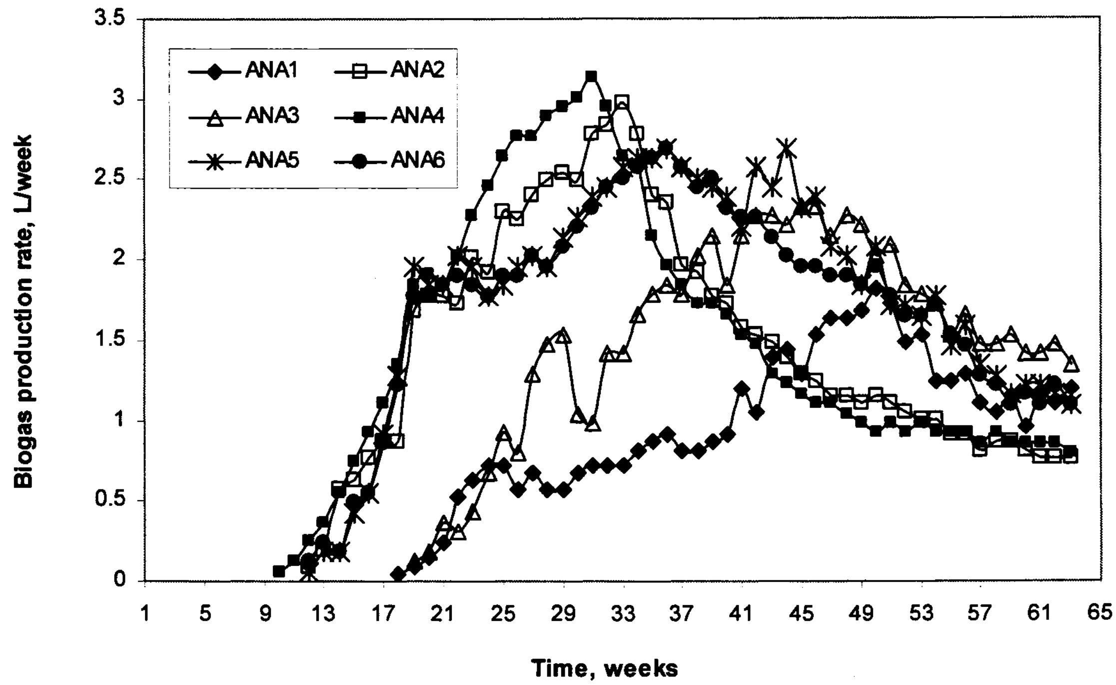

Figure 4.14 Biogas productions from anaerobic bioreactors

The composition of biogas generated from anaerobic bioreactors was measured on a biweekly basis. The results obtained from six bioreactors indicated that the biogas was composed of 54 to 60% methane, 40 to 50% carbon dioxide and about 1% nitrogen on volume basis (Appendix G). These compositions are comparable to typical value for biogas generated from landfills (Reinhart and Townsend, 1998, U.S. EPA, 1994, Tchobanoglous *et al.*, 1993).

4.2.2.4 Settlement

During the course of the experiment, measurements were taken to calculate the volume reduction of the MSW in all bioreactors. The settlement was determined based on the height of the MSW in the bioreactors. These measurements were taken from the beginning until the end of operation (see Table 4.14). Figure of settlement are presented in Appendix D. The height reduction due to settlement of MSW ranged from 16 to 20.0 %. In bioreactor ANA4, which received the highest rate of leachate recirculation (855 mL/kg of MSW.d) and sludge addition (85.5 mL/kg of MSW.d), the highest degree of settlement (20%) was achieved. In contrast, bioreactor ANA1 which received the lowest rate leachate recirculation (285 mL/kg of MSW.d) and sludge addition (28.5 mL/kg of MSW.d) had the lowest degree of settlement (16%). In previous laboratory and bench-scale studies, MSW settlement by aerobic biodegradation has been observed to be 30% and greater (Stessel and Murhy,1992) and lead to extensive landfill life extension with minimal LFG management.

Table 4.14 Results of settlement measurements in the anaerobic bioreactors

Bioreactor	Original waste height (mm)	Height after weeks of operation (mm)							Final settlement (%)
		9 wk	15 wk	25 wk	35 wk	45 wk	55 wk	63 wk	
ANA1	400	398	381	372	361	356	345	336	16.0
ANA2	400	385	372	359	348	337	332	325	18.8
ANA3	400	390	376	367	354	346	340	332	17.0
ANA4	400	380	369	355	340	334	329	320	20.0
ANA5	400	383	370	355	342	334	331	323	19.3
ANA6	400	381	368	560	343	336	334	324	19.0

4.3 Statistical Empirical Model

A statistical empirical model (Equation 3.1) was used to describe the effects of operating variables on the COD concentration in the leachate. In this model the data were fitted to the first order linear model using a factorial design. The model parameters were estimated using data collected according to a 2^2 factorial experimental design including two center point replicates from which reproducibility and lack of fit were evaluated.

4.3.1 Aerobic Bioreactors

The empirical model for COD concentration as a function of the two coded variables (leachate recirculation and sludge addition) is defined by parameter estimates. Due to the nature of the process, i.e. unsteady state conditions, the empirical model was fitted to a particular time of the bioreactor operation using all data collected from the experimental work. Based on the COD concentration profiles (Figure 4.2), three particular times were chosen to fit the model, i.e. an initial stage (5 weeks) consisting of lag phase followed by release on production of COD, the second stage (19 weeks) describes the declining concentration of COD found in the leachate and finally the third stage (27 weeks) where leachate is stabilized and minimum change in COD concentration. It should be noted that the COD concentration of bioreactor AER1 and AER3 reached the peak value in 19 weeks. The COD concentration of bioreactors AER2, AER4, AER5 and AER6 were at a final tapered stage where minimal changes occur after 27 weeks.

A pure error of variance for the different responses was examined using the results from the two runs of the center point replicates. This pure error variance was used to calculate the true value of each parameter estimate $\hat{\beta}_i$ in equation 3.1 that has a 95% confidence level. Table 4.15 shows the parameter estimates for the response of COD concentration in the leachate based on a 95% confidence level. These parameters, then, were used to build the empirical model based on equation 3.1 as follows:

Table 4.15 Parameter estimates from the 2^2 factorial design for the COD concentration responses from aerobic bioreactors

Parameter	95% confidence level		
	5 weeks	19 weeks	27 weeks
β_0	6108.17 ± 1606.98	31995.98 ± 1347.06	9328.25 ± 3073.34
β_1	2514.09 ± 1968.14	-16641.48 ± 1649.81	-9164.92 ± 3764.06
β_2	2275.79 ± 1968.14	-1424.19 ± 1649.81	-2175.66 ± 3764.06
β_{12}	1999.16 ± 1968.14	1817.63 ± 1649.81	1794.91 ± 3764.06

5 weeks: $\hat{Y} = 6108.17 + 2514.09\,x_1 + 2275.79\,x_2 + 1999.16\,x_1x_2$ **(4.1)**

19 weeks: $\hat{Y} = 319995.98 - 16641.48\,x_1 + 1817.63\,x_1x_2$ **(4.2)**

27 weeks: $\hat{Y} = 9328.25 - 9164.92\,x_1$ **(4.3)**

Applying a 95% confidence level to each of the parameter estimates in equation 4.1, reveals that zero is not plausible for all parameters. This indicates that leachate recirculation (x_1), sludge addition (x_2) and combination between leachate recirculation and sludge addition (x_1x_2) affected positively the MSW biodegradation process. A ratio of parameter estimate of leachate recirculation and sludge addition (β_1/β_2 ratio) was 1.11, which indicates that the effect of leachate recirculation was 1.11 times higher than sludge addition.

Since a 95% confidence level reveals that zero is a plausible value for the parameter estimate β_2, the term of x_2 (sludge addition) was deleted from the fitted model equation 4.2. It is seen that the COD concentration in the leachate was affected by leachate recirculation (x_1), and the combination of leachate recirculation and sludge addition (x_1x_2). For this operating stage (19 weeks), increasing the rate of leachate recirculation will decrease the COD concentration which is indicated by a negative sign of the parameter estimate β_1. In contrast, decreasing the rate of leachate recirculation will increase the COD concentration in the effluent leachate.

Applying a 95% confidence level to each of the parameter estimates in Table 4.15 for the data of week 27, it can be seen that that zero is plausible for the parameters β_2 and β_{12}. However, zero is not plausible for the parameter estimate β_1, the leachate recirculation. This indicates that the COD concentration in the leachate generated during the declining phase of the bioreactor operation, i.e. after 27 weeks of the experiment period was affected by leachate recirculation (x_1), but was not affected by the sludge addition (x_2) as well as the combination of leachate recirculation and sludge addition (x_1x_2). In this stage (27 weeks), increasing the rate of leachate recirculation will decrease the COD concentration which is indicated by a negative sign of parameter the estimate β_1.

Based on equation 4.1, 4.2 and 4.3, it can be statistically confirmed that leachate recirculation, sludge addition and combination of leachate recirculation and sludge addition plays a significant role in the MSW biodegradation during the ascending stage of the bioreactor landfill system. On the other hand, only leachate recirculation has a positive effect during the descending stage. This indicates that leachate recirculation and sludge addition support microbial activity during the biodegradation process. However, at the final stage (after 27 weeks), sludge addition had no effect on the biodegradation process.

4.3.2 Anaerobic Bioreactors

As in the case of the aerobic bioreactors, the empirical model for COD concentration as a function of the two variables, i.e. leachate recirculation and sludge addition, is defined by parameter estimates. The COD concentration profiles (Figure 4.9) was used to fit the model. Three particular times were selected to fit the model: first at 15 weeks for an increasing stage, followed by a declining stage (39 weeks), and finally at final stage 49 weeks. During the final stage (49 weeks), the COD concentration in the leachate from bioreactors ANA2, ANA4, ANA5 and ANA6 changed at a minimum level, while the COD concentration in the leachate from bioreactors ANA1 and ANA3 still decreased during this operation stage.

A 95% confidence level of the parameter estimates for the response of COD concentration in the leachate from anaerobic bioreactors is presented in Table 4.16. Based on equation 3.1 and the parameters in this table, the empirical models can be fitted as follows:

Table 4.16 Parameter estimates from the 2^2 factorial design for the COD concentration responses from anaerobic bioreactors

Parameter	95% confidence level		
	15 weeks	39 weeks	49 weeks
β_0	13804.17 ± 838.16	30869.37 ± 3675.42	8733.92 ± 5634.42
β_1	3352.43 ± 865.72	-24206.92 ± 4501.45	-10586.16 ± 6900.73
β_2	1398.85 ± 865.72	-3602.20 ± 4501.45	-2372.02 ± 6900.73
β_{12}	1224.56 ± 865.72	147.8 ± 4501.45	1775.45 ± 6900.73

15 weeks: $$\hat{Y} = 13804.17 + 3352.43\, x_1 + 1398.85\, x_2 + 1224.56\, x_1 x_2 \qquad \textbf{(4.4)}$$

39 weeks: $$\hat{Y} = 30869.37 - 24206.92\, x_1 \qquad \textbf{(4.5)}$$

49 weeks: $$\hat{Y} = 8733.92 - 1775.45\, x_1 \qquad \textbf{(4.6)}$$

From parameters calculated in Table 4.16, it can be seen that zero is not plausible for all parameters β_1, β_2, and β_{12} after 15 weeks of experimental work. Therefore, at this time, the leachate recirculation (x_1), sludge addition (x_2) and combination between leachate recirculation and sludge addition (x_1x_2) are fitted in equation 4.4, and it these shows that this three factors affected positively the MSW biodegradation process. The β_1/β_2 ratio was 2.39, which indicates that the effect of leachate recirculation was 2.39 times-fold higher than sludge addition. During this operation stage (11 weeks), increasing the rate of leachate recirculation and sludge addition will increase the COD concentration in the leachate which is indicated by the positive signs of parameter estimate β_1 and β_2. On the other hand, decreasing the rate of leachate recirculation and sludge addition will decrease the COD concentrations.

Applying a 95% confidence level to each of the parameter estimates in Table 4.14 for the operations time of 39 weeks, it is seen that zero is plausible for parameters β_2, and β_{12}. Therefore, only the x_1 term are fitted in equation 4.5. This reveals that the COD concentration in the leachate was affected by leachate recirculation. The ratio of parameter estimate of leachate recirculation and sludge addition (β_1/β_2 ratio) was 2.42, which indicates that the effect of leachate recirculation was 2.42 times higher than sludge addition. Negative signs of parameter β_1 and β_2 show that an increase in the rate of leachate recirculation and sludge addition will decrease the COD concentration in the leachate.

It can be seen in Table 4.16 that applying a 95% confidence level reveals that zero is plausible for the parameter estimate β_2 and β_{12} at 49 weeks of bioreactor operation. Consequently, the term of x_2 (sludge addition) and x_1x_2 (leachate recirculation and sludge addition) were deleted from the fitted model in equation 4.6. It is seen that the COD concentrations in the leachate was affected by leachate recirculation (x_1) only. Since the parameter estimate β_1 has a negative sign, it indicates that an increase in the rate of leachate recirculation will decrease the COD concentrations in the leachate. Conversely, decreasing the rate of leachate recirculation will increase the COD concentration in the leachate.

From the above results, it is clear that leachate recirculation, sludge addition and the combination of leachate recirculation and sludge addition plays a significant role in the anaerobic MSW biodegradation during the ascending and descending rates at a 95% confidence level. This indicates that leachate recirculation and sludge addition support microbial activity during the biodegradation process. However, at the final stage (after 49 weeks), sludge addition had no effect on the biodegradation process.

4.4 Aerobic vs. Anaerobic Biodegradation

The COD concentrations in the leachate from simulated aerobic and anaerobic bioreactors were used to compare the rates of the MSW biodegradation. Two operating conditions in the bioreactors which received the lowest and highest leachate recirculation and sludge addition, i.e. AER1, AER4, ANA1 and ANA4, were picked up for the comparison. Figure 4.15 depicts the concentration of COD concentration in the leachate from four bioreactors during the experimental study. A major factor in comparing the performance of the two processes is the time to reach stabilization.

It can be seen that the time needed to reach the stabilization stage (approximately 1,000 mg/L) was 27 weeks in AER4. On the other hand, the COD concentration in the leachate from ANA4 reached approximately 1,000 mg/L after 39 weeks. Both bioreactors AER4 and ANA4 received leachate recirculation and sludge addition of 855 and 85.5 mL/ kg of MSW.d. In bioreactor AER1, the COD concentration in the leachate reached approximately 1,000 mg/L after 45 weeks. On the other hand, the COD concentration in the leachate from ANA1 was approximately 12,500 mg/L at the end of operation. Therefore, it can be confirmed that the aerobic biodegradation is about 1.4-fold faster compared to the biodegradation under anaerobic operation.

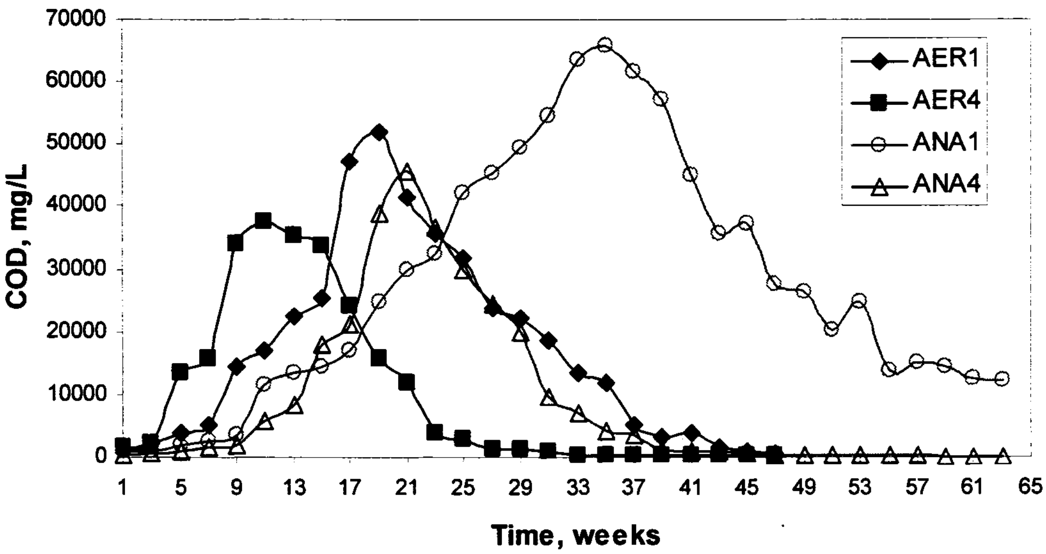

Figure 4.15 Concentrations of COD in leachate from aerobic and anaerobic bioreactors

A statistic procedure, F-test, was used to determine whether or not there is a significant difference between the aerobic and anaerobic biodegradation. The COD concentrations in the leachate from simulated aerobic and anaerobic bioreactors AER2 and AER4, and ANA2 and ANA4 at 27 weeks were used to calculate the F-test. This stage was a final stage (27 weeks) in the aerobic bioreactors in which the COD concentrations in the leachate change at minimum level. On the other hand, the COD concentrations in the leachate from anaerobic bioreactor were still in the declining stage at 27 weeks. The results of the paired F-test (F calculation>F critical) at a 95%

confidence level (α=0.05) in Table 4.15 show that there was a significant difference between the aerobic and anaerobic bioreactors. These results indicate that the addition of air affected positively the biological activities and consequently enhanced the MSW biodegradation process.

Table 4.17 Results of paired on F-test (α=0.05) for the COD concentration in the leachate from aerobic and anaerobic bioreactors in 27 weeks

Parameter	Bioreactors	
	AER2 and AER4	ANA2 and ANA4
Mean, μ	1523.004	35372.989
Variance, $\sigma 2$	902270.704	244384446.303
Number of sample, n	2	2
Degree of freedom, DF	1	1
F calculation	270.85	
F critical	160	

4.5 Analysis of Variance

Analysis of variance (ANOVA) was used to examine the effect leachate recirculation as well as sludge and air addition for the biodegradation of MSW in this experimental study. In this analysis, the statistical software "R 2.1.1" (2005, The R. Foundation for Statistic Computing) was used to calculate the ANOVA table based on the data obtained from the experimental work. These data include the maximum COD concentration from Figure 4.2 and 4.9 (aerobic and anaerobic bioreactors), ascending and descending rates from aerobic bioreactors (Table 4.1) as well as ascending and descending rates from anaerobic bioreactors (Table 4.9).

In the ANOVA test, it was assumed that the three groups of data (ascending and descending rates and maximum COD concentrations), was normally distributed and followed a linear model. The normal distribution is important because it is used to examine the validity of the statistical test. This means that many kinds of the statistical tests can be derived for normal distributions. In this research, a linear model from model was tested using residuals and normal Q-Q plots.

4.5.1 Analysis for Ascending Rates

The results of the ANOVA tables from the ascending rates and the treatments used are shown in Table 4.18 and Table 4.19.

Table 4.18 ANOVA for the ascending rates from the 12 simulated bioreactors

Factors	DF	SSE	MSE	F value	Pr(>F)
Sludge addition	2	33037	16518	9.4703	0.0113923*
Leachate recirculation	1	22428	22428	12.8584	0.011561*
Air addition	1	50371	50371	28.8791	0.001705**
Leachate recirculation + sludge addition	1	1456	1456	0.8350	0.396064.
Residuals	6	10465	1744		

Notes:

DF: degree of freedom
SSE: sum square error
MSE: mean square error
Significant codes of confidence level: 0'***', 0.001'**', 0.01'*', 0.05'.', 0.1' '.

As shown in Table 4.18, it can be seen that leachate and sludge addition had a significant effect (99.0% confidence level) on the biodegradation of MSW during the entire time of ascending stage based on the significance codes. In addition, combination of leachate recirculation and sludge addition also had a significant effect on the MSW biodegradation (90% confidence level). Also, as previously described in empirical models (equation 4.1 to 4.6), these same two factors and combination of the two had a significant effect on the biodegradation of MSW at a specific time during the ascending stage. The two results, the ANOVA and the empirical models, prove that leachate recirculation and sludge addition play an important role on the MSW biodegradation process during the ascending stage.

The effect of air addition on the MSW biodegradation was very significant at a 99.90% confidence level during the ascending stage (see Table 4.18) based on the significance code. This indicates that air addition increases microbial activities and enhances the biodegradation of MSW process.

The difference effect of treatment on the MSW biodegradation rate was examined using ANOVA. In this case, treatment means the factorial design which represents the rate of leachate recirculation and sludge addition based on the 2^2 factorial designs that was used in this experimental work. Therefore, a total of 12 treatments from aerobic and anaerobic bioreactors, i.e. AER1 to AER6 and ANA1 to ANA6, were analyzed using the ANOVA. The results of treatment (Table 4.19) show that there was a strong effect of treatment factors (99.0% confidence level) on the biodegradation of MSW rates. This indicates that the rate of leachate recirculation and sludge addition play a significant role in the MSW biodegradation process.

Table 4.19 ANOVA for the treatment analysis from the 12 simulated bioreactors

Factors	DF	SSE	MSE	F value	Pr(>F)
Treatment	4	56921	14230	8.1585	0.013238*
Residuals	6	10465	1744		

Notes:
DF: degree of freedom
SSE: sum square error
MSE: mean square error
Significant codes of confidence level: 0'***', 0.001'**', 0.01'*', 0.05'.', 0.1' '.

To verify the hypothesis for normal distribution and linear model, residuals and normal Q-Q plots from the ANOVA were used to confirm model fit and to diagnose the cause of lack of fit. A plot of the residual $(y-\bar{y})$ against the fitted model can also be used to identify outlier on the data observed. These residuals plots should not show any particular trends for the data point presented. The relatively random scattering of data points for the residuals and root of standardized versus fitted response shown in Figure 4.16 (a and b) suggest that there are no outlier present. Therefore, the data observed in this experiment have a normal distribution. This indicates that the ANOVA test has no lack of fit and the data follow a linear model.

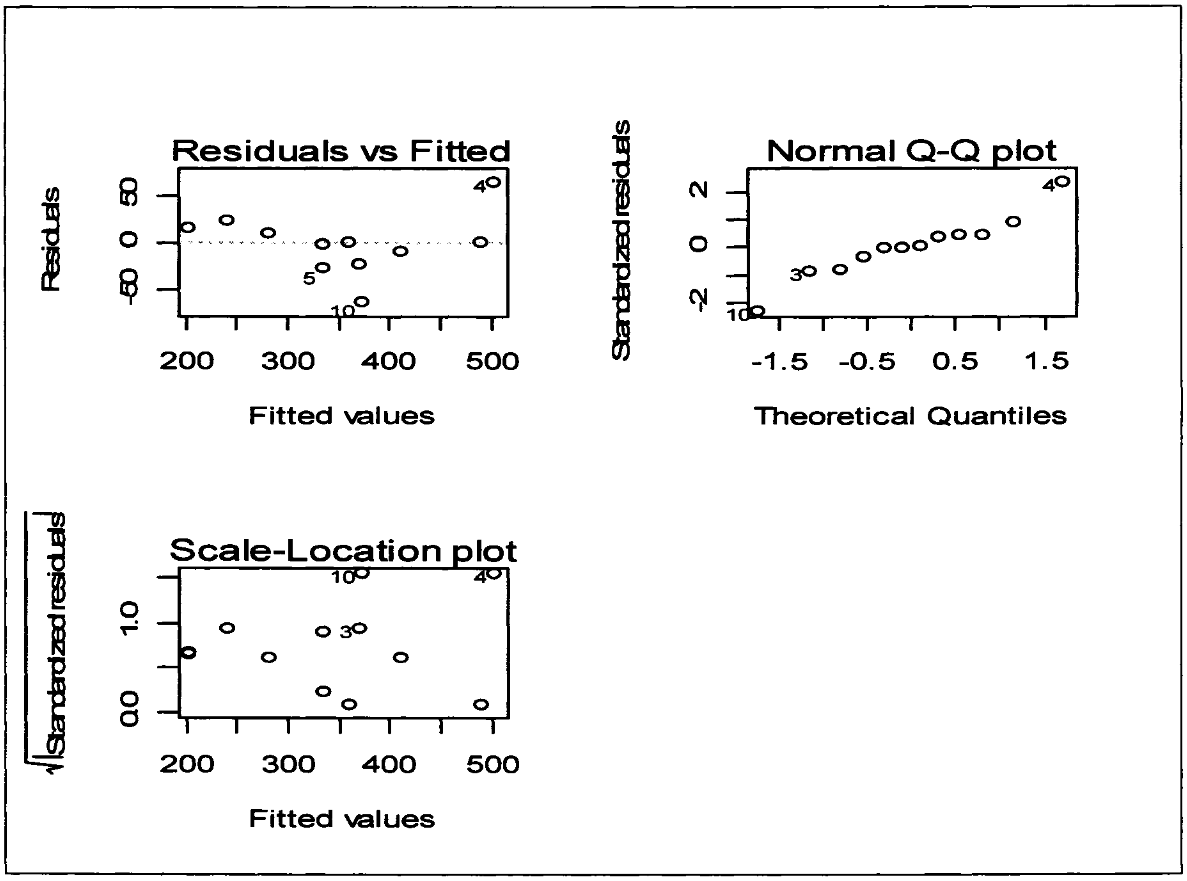

Figure 4.16 Residuals and Q-Q plots from the ascending rates

Another useful tool for assessing normality is a quantile quantile or Q-Q plot. The normal Q-Q plot graphically compares the distribution of a given variable to the normal distribution (represented by a straight line). From a normal Q-Q plot (Figure 4.16c), it can be seen that a scatter plot with the quantile on the horizontal axis and the expected normal on the vertical axis (standardized residuals) show a straight diagonal line. It can also be seen from Figure 4.16c that no outliers appear in the pattern. This plot indicates that the experimental obtained follow a normal distribution.

4.5.2 Analysis for the Maximum COD Concentration

Similar to previous case of ascending rate, the effect of leachate recirculation and sludge addition on the MSW biodegradation process was analyzed using data from the COD concentrations at the maximum level, i.e. a peak value of the COD concentration profiles (Figure 4.2 and 4.9). The corresponding process response (leachate, sludge, and air) is presented in the ANOVA table (Table 4.20).

Table 4.20 ANOVA for the maximum COD concentration from the 12 simulated bioreactors

Factors	DF	SSE	MSE	F value	Pr(>F)
Leachate recirculation	2	356657478	178328739	44.7086	0.0002486***
Sludge addition	1	37107618	37107618	9.3032	0.0225085*
Air addition	1	531331135	531331135	133.2095	2.544e-05***
Leachate recirculation + sludge addition	1	281894	281894	0.0707	0.7992576
Residuals	6	23932127	3988688		

Notes:

DF: degree of freedom
SSE: sum square error
MSE: mean square error
Significant codes of confidence level: 0'***', 0.001'**', 0.01'*', 0.05'.', 0.1' '.

The effect of leachate recirculation and air addition was very strong (100% confidence level) on the biodegradation of MSW as indicated in Table 4.18. Sludge addition also had a significant effect (99% confidence level) on the MSW biodegradation. The results of the ANOVA test indicate that leachate recirculation, sludge and air addition have a positive effect on the biodegradation of MSW. These three factors increase microbial activities in the bioreactors. Therefore, the rate of MSW biodegradation can be enhanced by manipulating the rate of leachate recirculation and sludge addition as well as air addition.

To evaluate and diagnose the normality of data and linearity of the model, residuals and normal Q-Q plots were used in the analysis. If there are any particular patterns and/or trends for the data point presented, the residuals plots indicate a lack of fit of the model. It is expected that the residual plot will not show any particular pattern and/or trend. It can be seen in Figure 4.17a & 4.17b that a plot of the residuals $(y - \bar{y})$ against the fitted values and root of standardized versus fitted values show a relatively random scattering of data points. It is also seen that there was no particular trend or pattern in these residual plots which indicate the normality of data observed in the experiment and follow the linear model.

A normal quantile quantile or Q-Q plot is a very useful visual tool for assessing whether the distribution for sample data follows a normal distribution. The normal Q-Q plot graphically compares the distribution of a given variable to the normal distribution (represented by a straight line). Figure 4.17c shows that a scatter plot with the quantile on the horizontal axis and standardized residuals on the vertical axis formed a straight line. This plot indicates that data observed in the experiment follows a the normal distribution.

4.5.3 Analysis for Descending Rates

Descending rates data of the COD concentrations in the leachate from Table 4.1 and 4.9 were used to calculate the ANOVA test. The corresponding process response (leachate recirculation, sludge and air addition) is presented in the ANOVA table (Table 4.19).

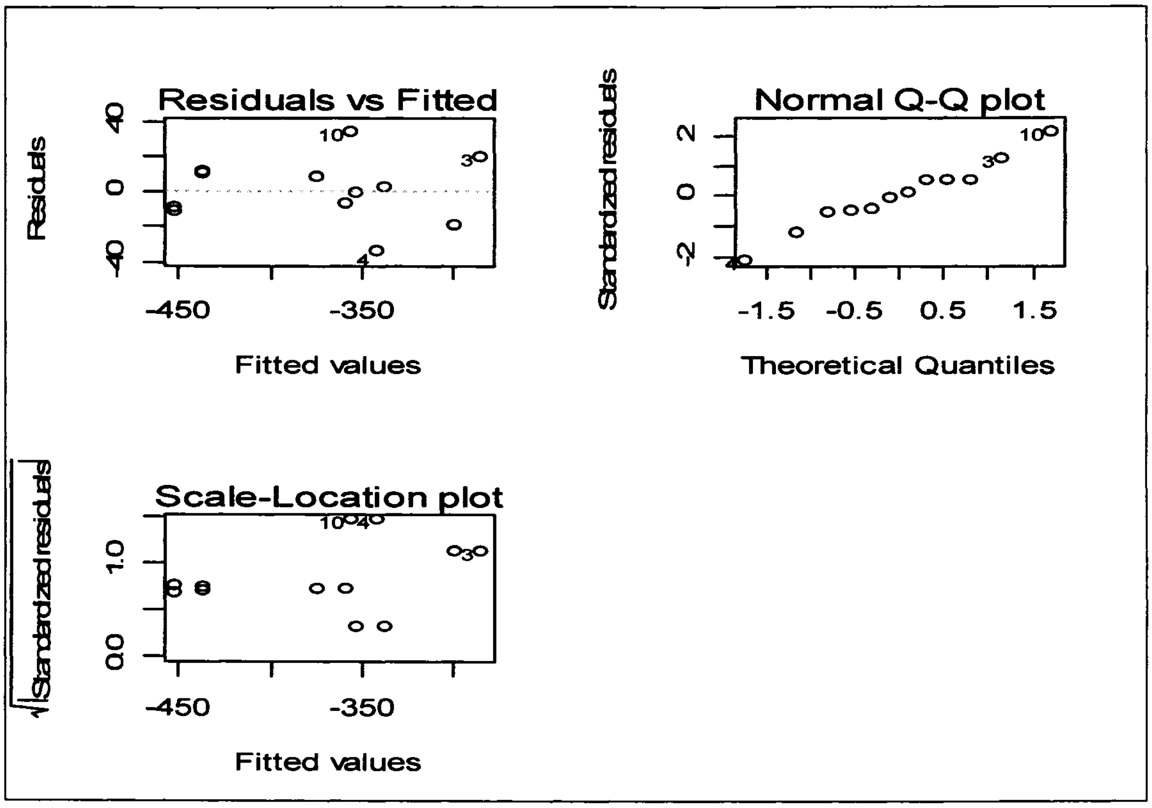

Figure 4.17 Residuals and Q-Q plots from the maximum COD concentration

From Table 4.21, it can be seen that there was a significant effect of leachate recirculation, sludge addition and their combination (99.9%, 95%, and 95% confidence level) on the biodegradation of MSW during the descending stage. As mentioned in empirical model for aerobic bioreactor (equation 4.2), leachate recirculation and its combination with sludge addition had a significant effect on the biodegradation of MSW at a particular time during the descending stage. On the other hand, empirical model for anaerobic bioreactors (equation 4.5), only leachate recirculation had an effect to MSW biodegradation at a particular time during the descending stage. These two results, the ANOVA and empirical models, prove that leachate recirculation play an important role on the MSW biodegradation process. This factors increase microbial activities and enhance the MSW degradation process.

Table 4.21 ANOVA for the descending rates from the 12 simulated bioreactors

Factors	DF	SSE	MSE	F value	Pr(>F)
Leachate recirculation	2	30284.1	15142.1	25.1579	0.001209**
Sludge addition	1	2496.9	2496.9	4.1485	0.087835
Air addition	1	701.8	701.8	1.1659	0.321719
Leachate recirculation + sludge addition	1	3031.5	3031.5	5.0367	0.065960.
Residuals	6	3611.3	301.9		

Notes:

DF: degree of freedom
SSE: sum square error
MSE: mean square error
Significant codes of confidence level: 0'***', 0.001'**', 0.01'*', 0.05'.', 0.1' '.

The effect of air addition on the MSW biodegradation was not significant during the descending rate (see Table 4.21) based on the significance code. This may indicate that air addition did not significantly increase microbial activities because most of the organic matter was consumed during the ascending rates.

Residuals plots presented in Figure 4.18 show that the relatively random scattering of data points for the residuals and root of standardized residuals versus the fitted value suggests that there were no trends or patterns for the data point. In addition, the normal Q-Q plot also shows that a scatter plots with the quantile on the horizontal axis and standardized residuals on the vertical axis formed a straight line. These results reveal that the data used in the ANOVA test had a normal distribution.

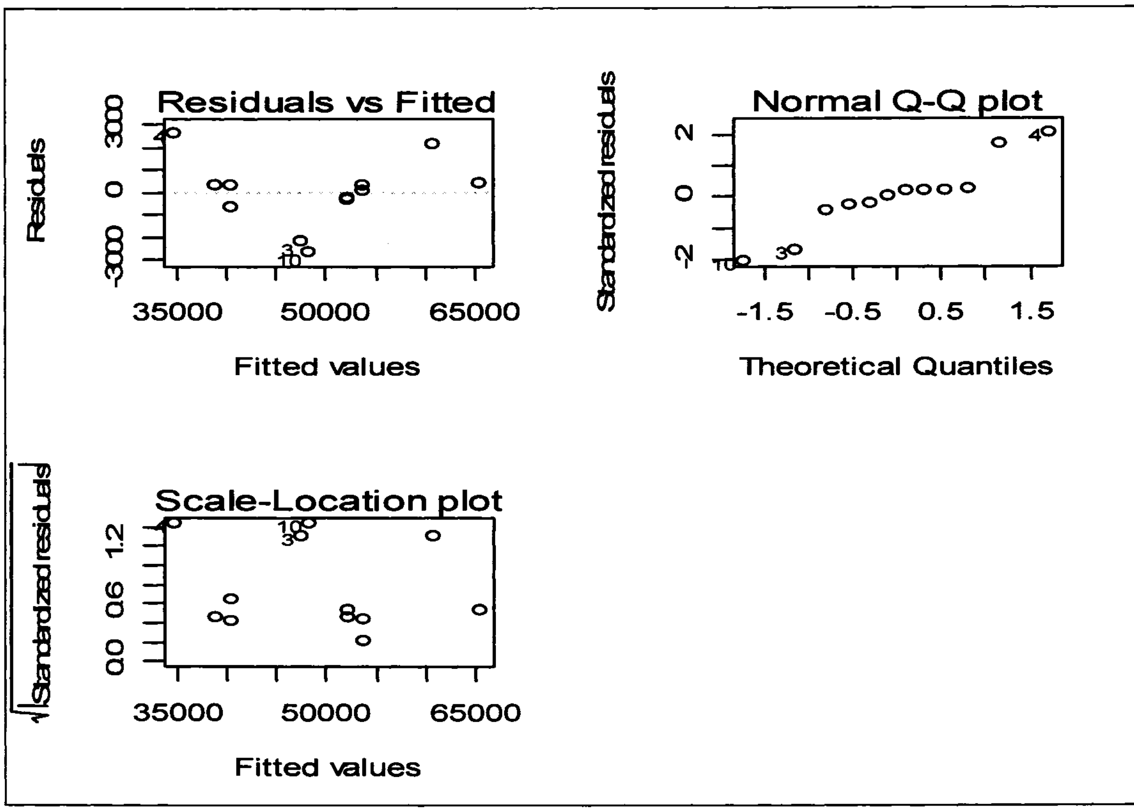

Figure 4.18 **Residuals and Q-Q plots from the descending rates**

4.5 Summary of Statistical Analysis

As discussed from sub chapter 4.3 to 4.5, statistical methods, i.e. statistical empirical model, F test for two sample hypotheses (aerobic and anaerobic), and ANOVA, were used to analyze the effect of leachate recirculation, sludge and air addition on the MSW biodegradation in simulated bioreactor landfills. It was confirmed that all these three factors affect the rate of MSW biodegradation during the ascending stage. In addition, the ANOVA test also show that these three factors also affect the rate of MSW biodegradation during the descending stage. In contrast, in the statistical empirical model, sludge addition did not show any effect on the MSW biodegradation during the descending stage. The differences of the results between the empirical model and ANOVA may be caused by the difference time that was selected for modeling of COD concentration in the leachate during the descending stage. In empirical model, only one particular time was selected to calculate model parameters, but in ANOVA the entire time was used to calculate the ANOVA table. Therefore, the ANOVA results were relatively more accurate compare to empirical model for analyzing the rate of MSW biodegradation in simulated bioreactor landfills.

CHAPTER 5

MODELLING OF MSW BIODEGRADATION

One of the important elements of design, operation and maintenance at landfill is predicting leachate and gas quantity and quality. A number of mathematical models have been used to simulate the biochemical processes, the quantity and quality of leachate as well as the generation of LFG in landfills (White *et al.*, 2004, Pareek *et al.*, 1999, Lay, J-J. *et al.*, 1998, El-Fadel *et al.*, 1997). Other models such as statistical modeling have also been used to simulate landfill biodegradation (Zacharof and Butler*et al.*, 2004). Both mathematical and statistical models were developed based on approaches such as linear control, non-linear control, discrete control, adaptive control, self-tuning system, and optimal control.

Kinetic model was used by Straub and Lynch (1982) to describe biodegradation of solid waste. The first model of landfill reactor was simulated as a series of fully mixed reactors. The model applied 1st order kinetic for solubilization from Monod's kinetic. Good results were obtained when the system was depicted as 4 fully mixed reactors in series. Then the second model of landfill reactor was simulated as unsaturated flow in porous media. Using previously kinetic found in the first model, there was no significant difference between the two models. Korfiatis and Dematropolous (1984) used unsaturated flow model for predicting leachate quality. These authors employed a different numerical scheme to solve the flow equation and performed a sensitivity analysis. However, their results were not verified with an independent set of experimental data from other researcher. Katsiri et al. (1999) assumed that landfill operates as a single reactor is a single fully mixed compartment with the volume equal to field capacity. They assumed 1^{st} order kinetic to run the model. The results obtained from the model were in a good agreement with the experimental results. However, there was no verification of the model.

According to above analysis it appears that due to very slow rates of solid waste biodegradation in relation with flow rate, the use of sophisticated flow equation to depict the movement of contaminant through lysimeter or landfill is not justified. In addition, the assumption that lysimeter is a fully mixed reactor like in wastewater treatment is not true. Moreover, for a model to have practical application, it would be better if it was simple and easy to apply in the real landfill. The models require extensive data input and have limited practical simulation because of complex formulations. The fuzzy logic approach can offer advantages in attempting to deal with these types of systems (Ibrahim, 2004).

A number of fuzzy logic applications of current interest include automobile transmission control, camcoder autofocusing, automatic train operations, water quality control, and structural design (Moller and Beer, 2004, Yan *et al.*, 1994). Several advantages of fuzzy logic are: not numerical, variables are used, making it similar to the way humans think; simplicity, allows the solution of previously unsolved problems because they do away with complex analytical equations used to model traditional control systems; rapid prototyping is possible because a system designer doesn't have to know everything about the system before starting work; cheaper to make than conventional systems because they're easier to design, they have increased robustness (Ibrahim, 2004).

To date, there is no information regarding the use of fuzzy logic approach as a method to simulate the biodegradation process at a landfill. Therefore, in this study, the fuzzy logic approach was employed to develop a MSW biodegradation model. The relevant elements governing the fuzzy logic approach for the model simulation are presented and discussed in the following section.

5.1 Fuzzy Logic Controllers

A fuzzy logic controller (FLC) is built to accomplish the objectives as follow (refer to Figure 5.1):

- Describe the biodegradation processes of MSW based on COD measurement in the leachate and the temperature in the bioreactors, and
- Control the process output from exceeding minimum and maximum level of specified constraint.

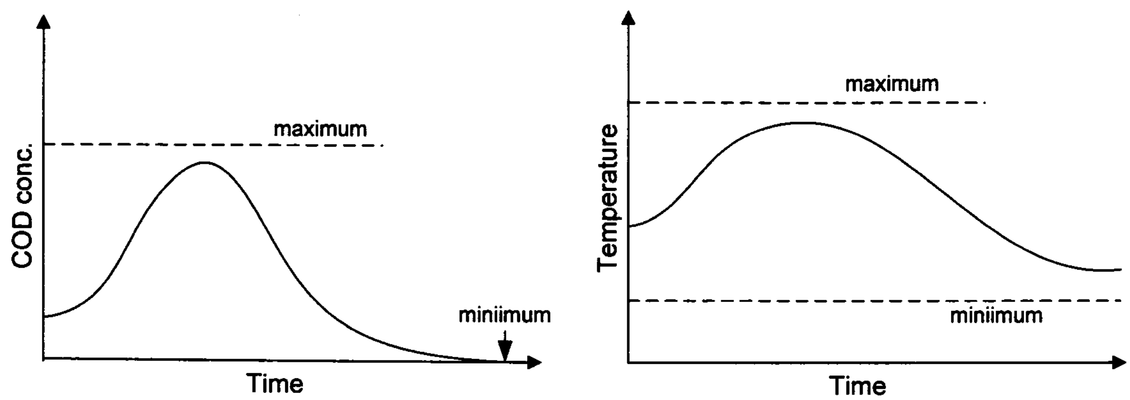

Figure 5.1 Basic control strategies

The control strategies are required to achieve the output (COD and temperature) in the range of the minimum and maximum levels. In this model, the strategies include input, fuzzyfier, fuzzy interference engine (FIE), defuzzyfier, and output.

A 4-input-3-output fuzzy logic controller structure is proposed for the model of MSW biodegradation and presented in Figure 5.2.

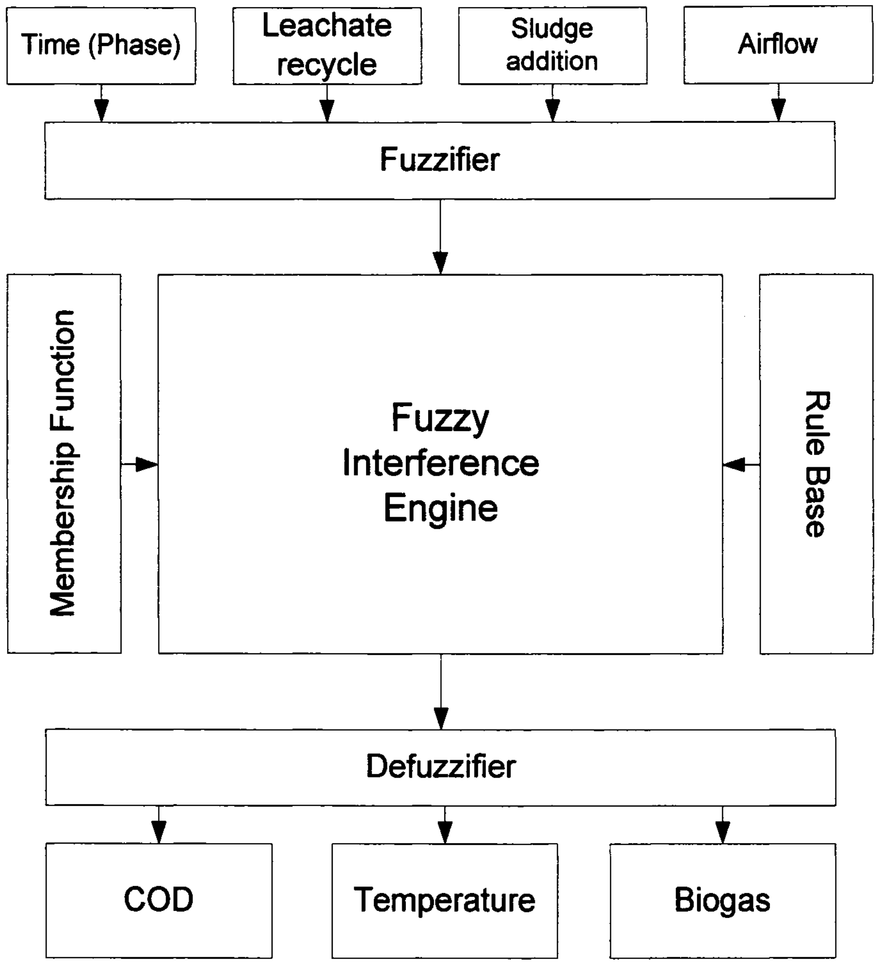

Figure 5.2 FLC structure for the MSW biodegradation model

The FLC structure as illustrated in Figure 5.2 consists of the following components:

1. Input: time (phase), recycle rate, sludge addition and airflow,
2. Input Fuzzyfication Module (Fuzzyfier),
3. Rule Base,
4. Membership Function,
5. Fuzzy Inference Engine
6. Output Defuzzification Module (Defuzzyfier)
7. Output: COD, and temperature and biogas production rate.

The following sections explain these components.

5.1.1 Input Module

The first step of fuzzy control strategies is to map the observed inputs to fuzzy sets in the universe of discourse (fuzzification). This process includes determination of the level of inputs to which they belong to each of the appropriate fuzzy sets. The observed data inputs are usually crisp and therefore fuzzification is required to map these crisp inputs into corresponding fuzzy values. Furthermore, the mapped data is transformed into linguistic variables to make it compatible fuzzy sets representation. This fuzzification can be expressed as (Yan *et al.*, 1994):

$$x = \text{fuzzyfier}\ (x_i) \tag{5.1}$$

Where x_i is a vector of crisp value of one input variable from the process, and x is a vector of fuzzy sets defined for the variable.

5.1.2 Membership Function

Membership functions must be defined for each input variable in order to convert crisp inputs into fuzzy inputs. Once membership functions are defined, a real data input, e.g. leachate recycle, is used to produce fuzzy inputs through membership function. The membership function is represented by a real number ranging between 0 and 1 (see Figure 5.3). This figure illustrate that the fuzzy sets (e.g. Early Degradation, ED , and Full Stabilization, FS) are functions that map a value, i.e. 1.2 to a number between zero and one (e.g. 0.8 of ED and 0.3 of FS), which indicates that the actual degree of membership, i.e. 1.2 = 80% ED and 30% FS (Cox, 1999). The degree of one means 100% the value belong to the fuzzy sets, whereas degree of zero does not belong to the fuzzy sets.

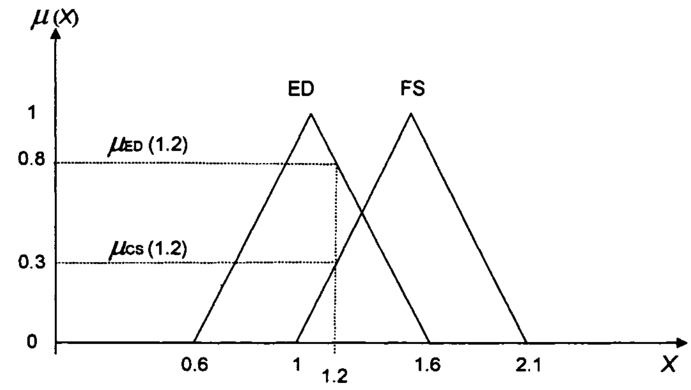

Figure 5.3 Graphical illustrations of two triangular fuzzy set operations

Gaussian membership functions are used to model of MSW biodegradation in order to increase accuracy of calculation results. One advantage using this membership function is that the result of the graphic simulation is smoother when compare to other such as triangular membership functions.

Ten fuzzy sets are used for the first inputs (i.e. phase) of the anaerobic model (refer to Figure 5.4). Other fuzzy set inputs, i.e., leachate recirculation, sludge addition and airflow are presented in Appendix E.

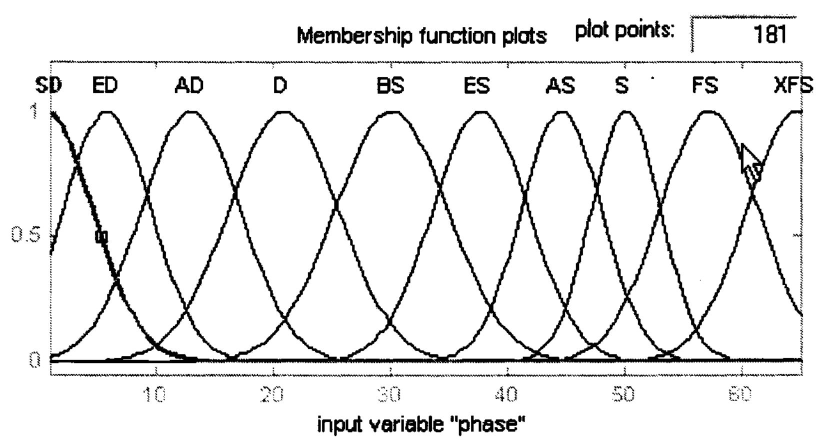

Figure 5.4 Gaussian membership function inputs of phase

The X-axis is phase (time) and Y-axis is membership value, and these inputs are consists of the following components:

1. SD: Start Degradation,
2. ED: Early Degradation,
3. AD: Almost Degradation,
4. D: Degradation,
5. BS: Begin Stabilization,
6. ES: Early Stabilization,
7. AS: Almost Stabilization,
8. S: Stabilization,
9. FS: Full Stabilization,
10. XFS: Extra Fully Stabilization.

5.1.3 Rule Base

The rule base module is very sensitive during the process of simulation in the fuzzy inference engine. Therefore, this module is one of the most crucial components of the FLC structure. The sets of fuzzy rules define the system behavior and replace the mathematical modeling of the system. These rules, then, are used in the fuzzy sets input systems to determine actions. Each of the rules can be denoted as an **IF** (process state) – **THEN** (control output) statement that describes the action to be processed in response of various fuzzy inputs. The following rule is an example of the fuzzy control rule base,

> **IF** phase is SD **AND** recycle is LR **AND** sludge is SL **THEN** COD is XXL **AND** TEMPERATURE is MW **AND** BIOGAS is NG

where SD, LR, SL represent system input variables and COD and TEMPERATURE denotes system output variables with the degree of XXL and MW; while AND is the fuzzy operator. The **IF** statement is called an antecedent and **THEN** is called consequent.

5.1.4 Fuzzy Inference Engine

A fuzzy inference engine is required to determine the fuzzy output and to compute the rules along with the membership function of the fuzzy input. A MAX-MIN fuzzy inference method is commonly used to compute a numerical value representing the aggregate effect of all that was triggered by an input value. Fuzzyfication in the fuzzy inference engine calculate the observed data inputs as fuzzy singletons (Yan *et al.*, 1994). This can be demonstrated as follows.

Assume there is a fuzzy control rule base with two rules:

Rule 1	IF x is A_1 and y is B_1 THEN z is C_1
Rule 2	IF x is A_2 and y is B_2 THEN z is C_2

If the COD concentration in the leachate of the i^{th} rule be denoted by η_i, the rule base of η_1 and η_2, with crisp inputs x_0 and y_0, can be expressed as:

$$\eta_1 = \mu_{A1}(x_0) \vee \mu_{B1}(y_0) \tag{5.2}$$

$$\eta_2 = \mu_{A2}(x_0) \wedge \mu_{B2}(y_0) \tag{5.3}$$

The membership function of the inferred consequences C of the i^{th} rule is given by:

$$\mu_C(w) = (\eta_1 \wedge \mu_{C1}(w)) \vee (\eta_2 \wedge \mu_{C2}(w)) \tag{5.4}$$

According to Cox (1999) intersection and union of fuzzy sets of A and B in the membership function can be written as follows:.

Intersection	$A \cap B$	Min $[\mu_A(x), \mu_B(x)]$	(5.5)
Union	$A \cup B$	Max $[\mu_A(x), \mu_B(x)]$	(5.6)

One of MAX-MIN methods that is commonly used to compute the inputs in the fuzzy inference engine is Mamdani's technique. This technique has the following advantages (Gulley and Jang*et al.*, 1995, and Driankov *et al.* 1996):

- Reduce computational time.
- The continuity of outputs could be guaranteed.
- Works well with linearier, optimization and adaptive techniques.
- Disambiguous and plausible.

The MAX-MIN inference process for the crisp input values x_0 and y_0 are illustrated in Figure 5.5 in this research.

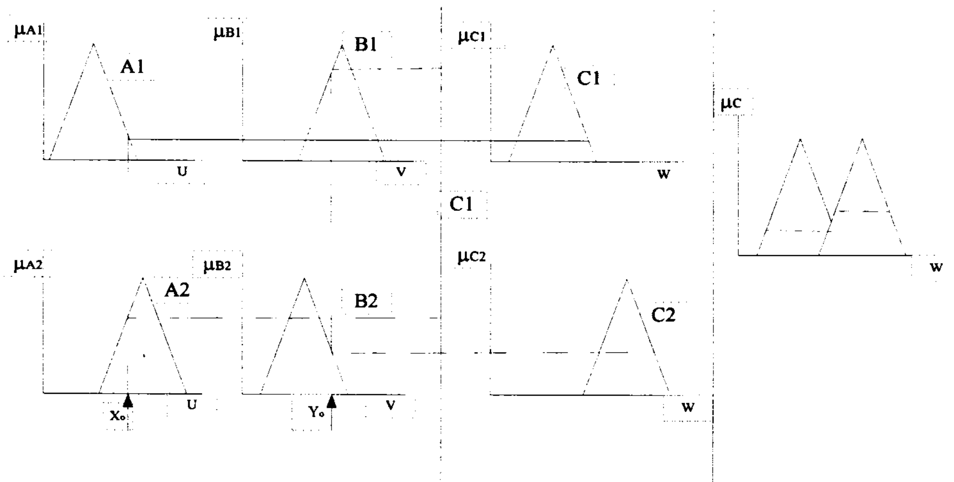

Figure 5.5 MAX-MIN fuzzy inference work under two crisp inputs.

5.1.5 Defuzzification Module

The defuzzification module is the last step of the design process for the fuzzy logic control systems. The purpose of the defuzzification is to produce a non-fuzzy (crisp) control action that best represents the possibility of distribution of the inferred fuzzy control action (Moller and Beer, 2004). The resulting non-fuzzy, which defines the action taken by fuzzy controller, is not arbitrary. This can be written as:

$$y_0 = \text{deffuzifier}\ (y) \tag{5.7}$$

where y is the fuzzy control action; y_0 is the crisp control action, and deffuzifier is the defuzzification operator.

A number of deffuzification methods leading to distinct results were proposed in the literature. One of the methods is called the *centre of gravity method* or *centroid method* (Klir and Yuan*et al.*, 1995). In this method, the defuzzified value, ***u***, can be calculated as follows.

$$\boldsymbol{u} = \frac{\sum_{i=1}^{r} \mu_{Ci}.C_i}{\sum_{i=1}^{r} \mu_{Ci}} \tag{5.8}$$

where r is the total number of rules, C_i is the value associated with the peak of output fuzzy set i, i.e., the suggested output value associated with the peak of fuzzy set i, and μ_{Ci} is the height illustrated in Figure 5.5 which is given by

$$\mu_{Ci} = \text{Min}\ [\mu_{Ai}\ (\text{x}),\ \mu_{Bi}\ (\text{x})] \tag{5.9}$$

In general, for a problem with r rules, and k inputs

$$\mu_{Ci} = \text{Min}\ [\mu_{1i}\ ,\ \mu_{2i}\ ,\ \ldots \mu_{ki}] \tag{5.10}$$

5.2 Model Development

As mentioned in the previous chapter, the goal of this study is to develop the MSW biodegradation model capable of describing leachate quality and temperature. To achieve this task, the MATLAB 5.3 program was used to build the system with the fuzzy logic toolbox. The following discussion is the steps how to build the system.

5.2.1 The Fuzzy Inference System (FIS) and Membership Function Editor

General information about a fuzzy inference system is displayed in the fuzzy inference system (FIS) Editor. The system consists of individual input and output variables and inference method. This editor associated with the Membership Function Editor which is used to edit all of the membership functions associated with all of the input and output variables for the entire fuzzy inference system. The FIS Editor for the MSW biodegradation modeling is presented in Figure 5.6.

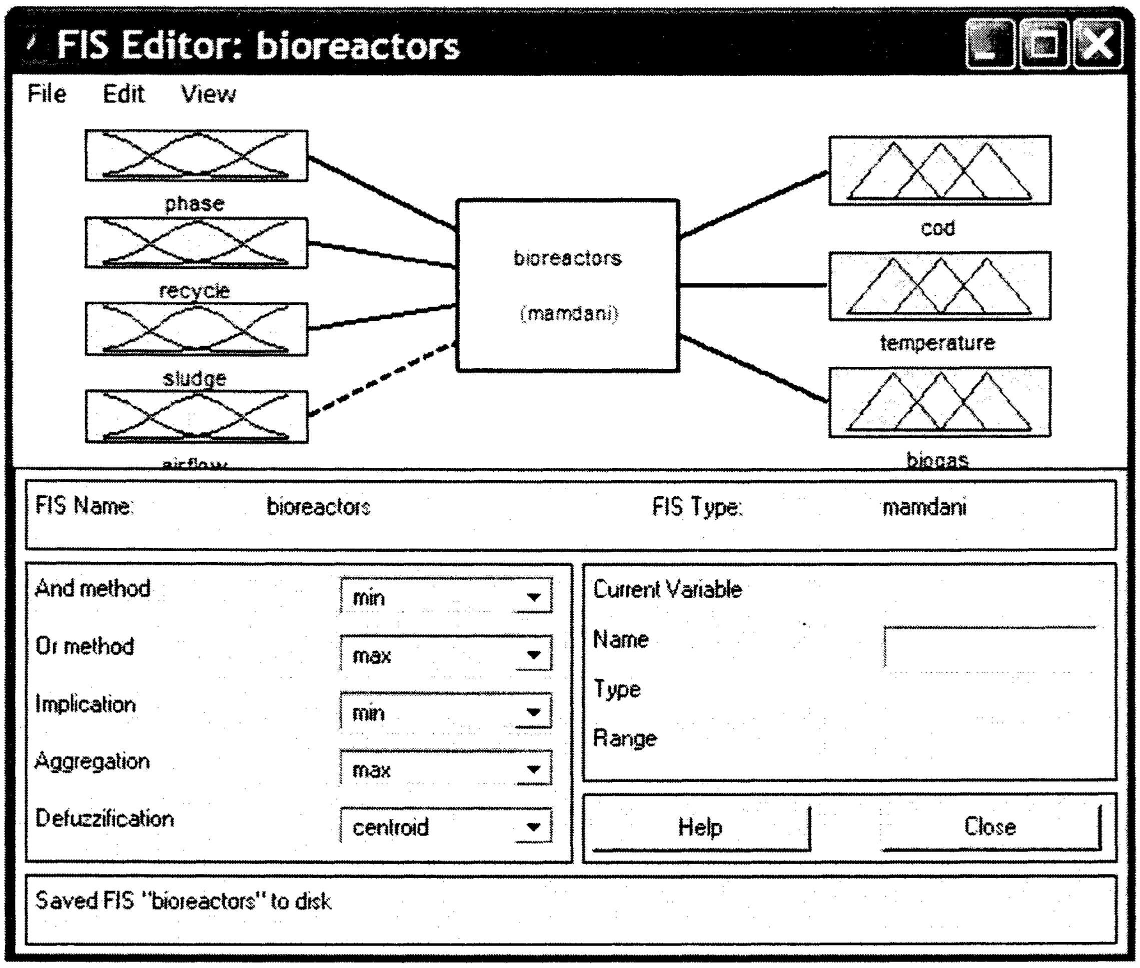

Figure 5.6 The FIS editor for the simulation

From Figure 5.6 it can be seen that the input variables of the model were phase (time), leachate recycle, sludge addition, and airflow. The COD concentrations and temperatures were the output of the model. Based on the individual input and output, a membership function was defined defined to formulate the rule base. One membership function is shown in Figure 5.4. In this study, the Mamdani technique in the FIS Editor was used as inference fuzzy engine to process the inputs.

5.2.2 Rule Editor

When the FIS and Membership Function Editor have been established, the Rule Editor must be used to construct the rule statements. The number of rule statement depends on each input variable and the number of system input variables in the membership function. In the anaerobic bioreactor, the input was 4 variables and the systems input were 10, 3, 3 and 0 for phase, leachate recycle, sludge addition, and airflow respectively. Therefore, the total number of rule base was 60 statements. While, in the aerobic bioreactors, the total number of rule base was 72 statements. The rule base of the model is presented in Appendix F.

5.2.3 Model Simulation

Any successful attempt to predict leachate quality and temperature must be constructed in a sufficient model simulation. In MATLAB 5.3, the Fuzzy Logic Toolbox is designed to work flawlessly with Simulink, the software which is used to simulate a model. A flowchart of the model simulation must be designed properly to achieve the result. Once the flowchart and fuzzy set system are available, the Simulink can be run to simulate the model. In this research, the flowchart for the simulation is presented in Figure 5.7.

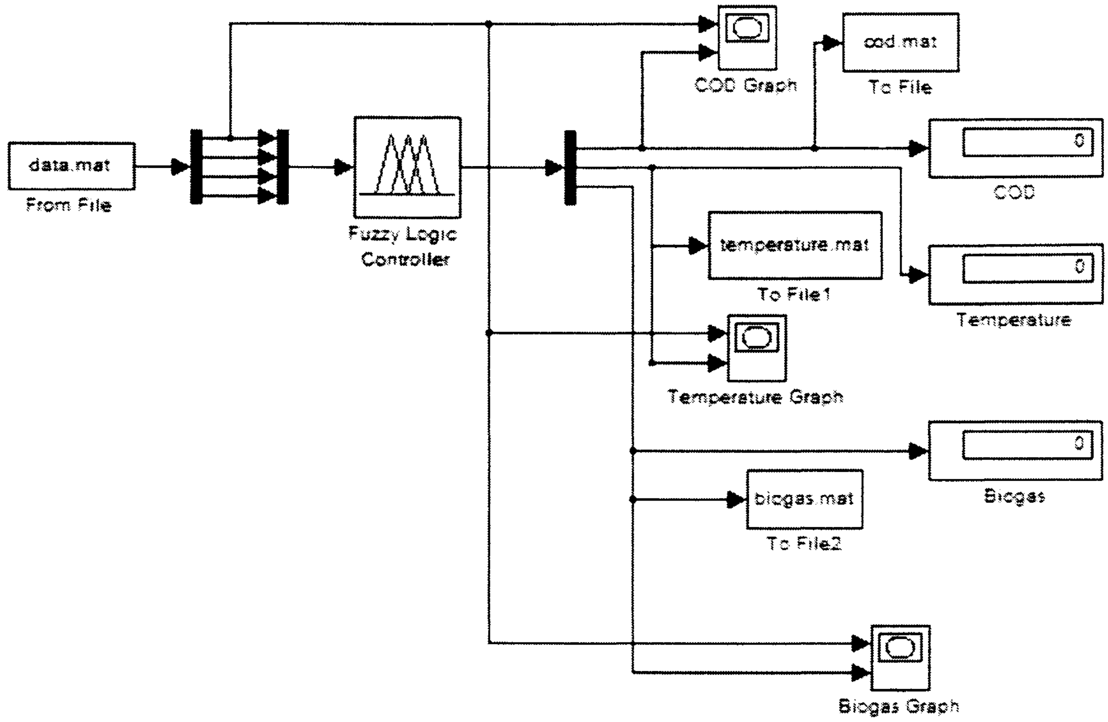

Figure 5.7 Flowchart of the model simulation

5.3 Results and Discussion

The applicability of the model was accomplished by comparing the model predictions with experimentally obtained data. This section examines the modeling capability of the fuzzy logic approach in term of fitting adequacy to COD concentration in the leachate and temperature in the bioreactors. The comparison between the model simulations and the experimental data was performed to examine the predictive capability of the developed model. A statistic procedure, F-test, was used to determine whether or not there is a significant difference between the experimental results and simulations.

5.3.1 Aerobic Bioreactors

5.3.1.1 COD

As presented in Chapter 4, all profiles of COD concentration in the leachate had a similar trend; first a rising stage, followed by a peak concentration, and finally a declining stage and constant stage afterward. Based on data obtained in the experiment, the prediction of COD concentrations was conducted using the fuzzy logic control. Figures 5.8 and 5.9 compare the COD concentrations in AER2 and AER 4, which received high rate of leachate recirculation, between observation and the prediction.

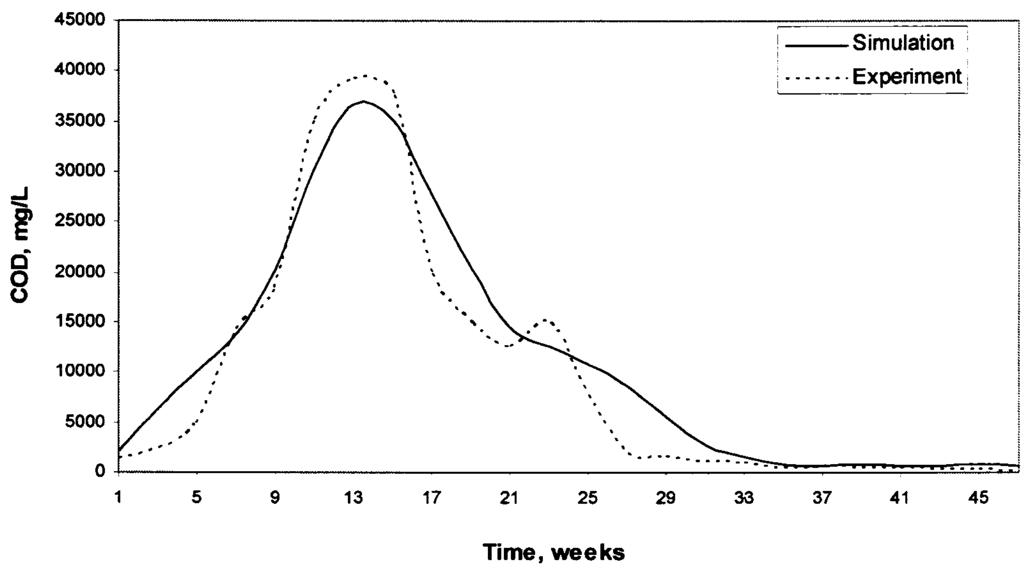

Figure 5.8 Measurements and simulations of COD in leachate from AER2

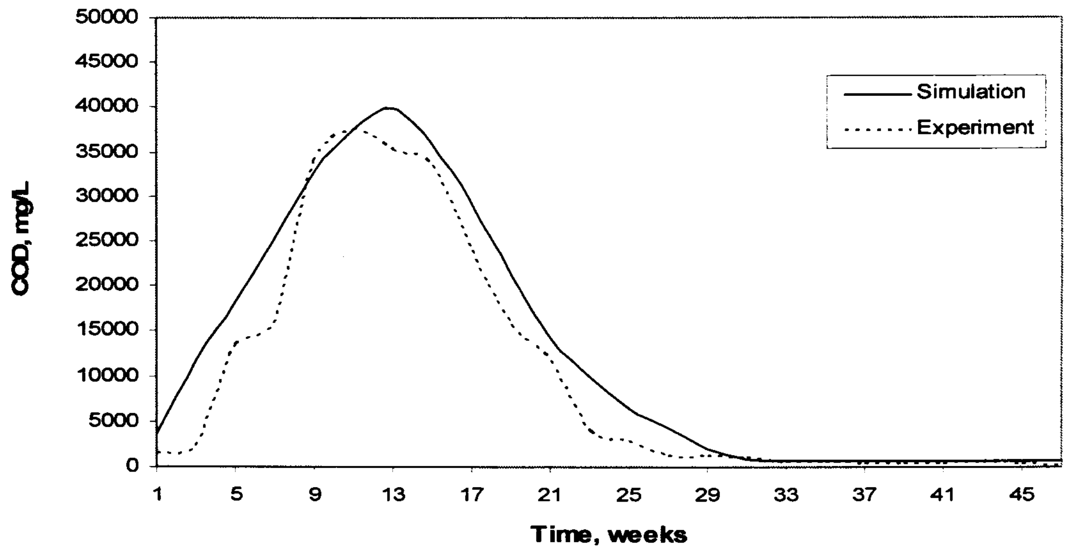

Figure 5.9 Measurement and simulations of COD in leachate from AER4

It can be seen that the simulation results for the COD concentrations were relatively attached to accurately superimposed on the experimental data, as shown in Figures 5.8 and 5.9. The results of the paired F-test (F calculation<F critical) at a 95% confidence level (α=0.05) show that there was no significant difference between the experimental results and simulations. The The salient trend of the experimental data was closely reproduced by the simulation for both the bioreactors AER2 and AER4. Slight deviations between the predicted and measured concentrations of COD occurred, however, they are within an acceptable range of confidence. The simulation exhibited a rapid increase of the COD concentration for bioreactors AER2 and AER4, which were slightly higher in magnitude than measured during the course of the experiment. The differences of the COD concentrations between observed data and the simulation can be estimated for about 3,000 to 5,000 mg/L before stabilization. In the AER2, the maximum COD concentration from the simulation was about 2,500 mg/L lower compared to the experimental data. In contrast, the maximum COD concentration from the simulation was about 2,500 mg/L higher compared to the experimental data obtained from AER4. Such a deviation was on the order of 2,000 to 5,000 mg/L, perhaps this was due to overestimating the input of the fuzzy sets during this period.

The COD concentrations from AER1 and AER3, which received low rate of leachate recirculation, are shown in Figures 5.10 and 5.11, respectively. A comparison between the experimental and simulation data indicated that the COD concentration in the leachate was reasonably reproduced by the model in both cases. The results of the statistical F-test (F calculation<F critical) at a 95% confidence level (α=0.05) also show that there was no significant difference between the experimental results and simulations. The simulation results for the COD concentrations were super imposedsuper impose on the measurement results (see Figures 5.10 and 5.11). However, in general, the simulation curves were lower than the experimental one. In addition, the maximum COD concentrations of the computer simulations were also lower than the measurement. Discrepancy between the observed data and simulation of the COD concentrations were in the range from 500 to 3,000 mg/L with the exception of about 5,000 mg/L for the maximum concentration in AER3.

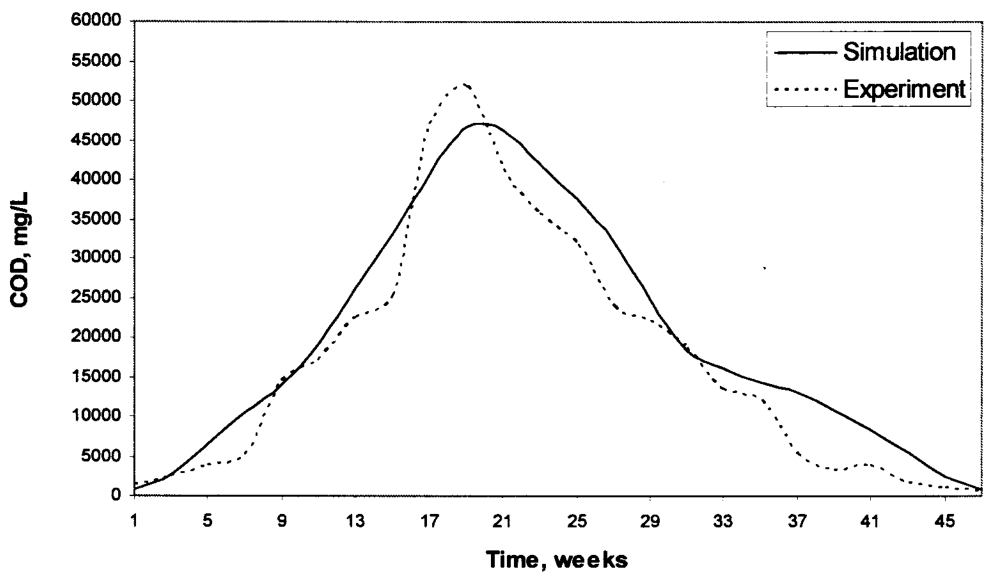

Figure 5.10 Measurement and simulations of COD in leachate from AER1

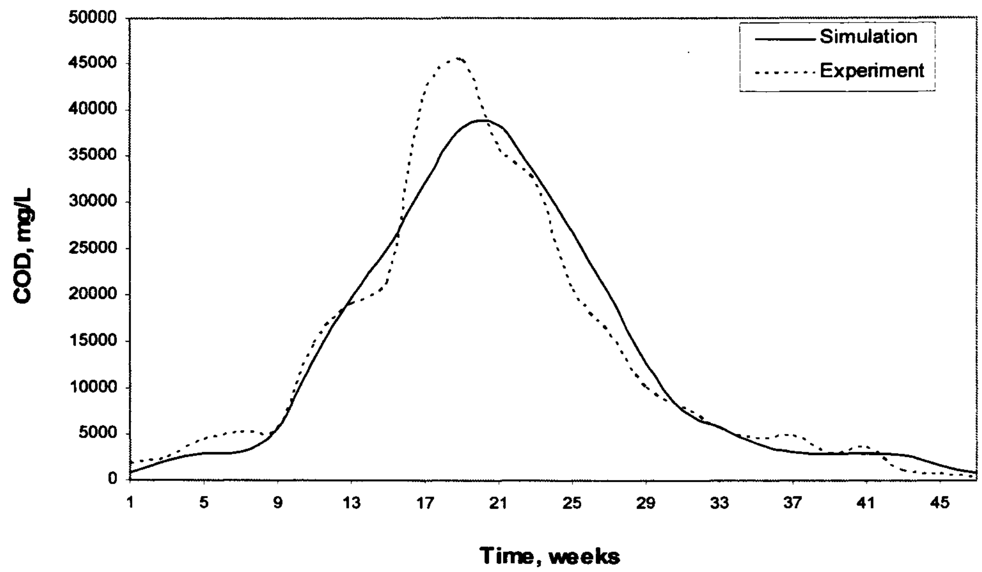

Figure 5.11 Measurements and simulations of COD in leachate from AER3

Figure 5.12 represents the measured and the simulated COD concentration data in AER6. Since the bioreactors AER6 and AER5 were replicates, the computer simulation was conducted once with the same input parameters. In general, the trend of COD concentration measurement was reasonably mirrored by the model simulation and there was no significant difference between experimental results and simulation based on statistical F-test (F critical<F calculation) at 95% confidence level. . However, slight disagreements were also indicated, especially by underestimating COD peaks and time of occurrence. At the beginning (from week 1 to 9), the simulation exhibited a sharp increase to a peak of 35,000 mg/L and then decline back to about 1,000 mg/L COD in week 31. On the other hand, a gradual increase in the COD concentration was depicted initially from week 1 to 9 of the observed data. The deviation between experimental data and the computer simulations was in the range from 500 to 5,000 mg/L.

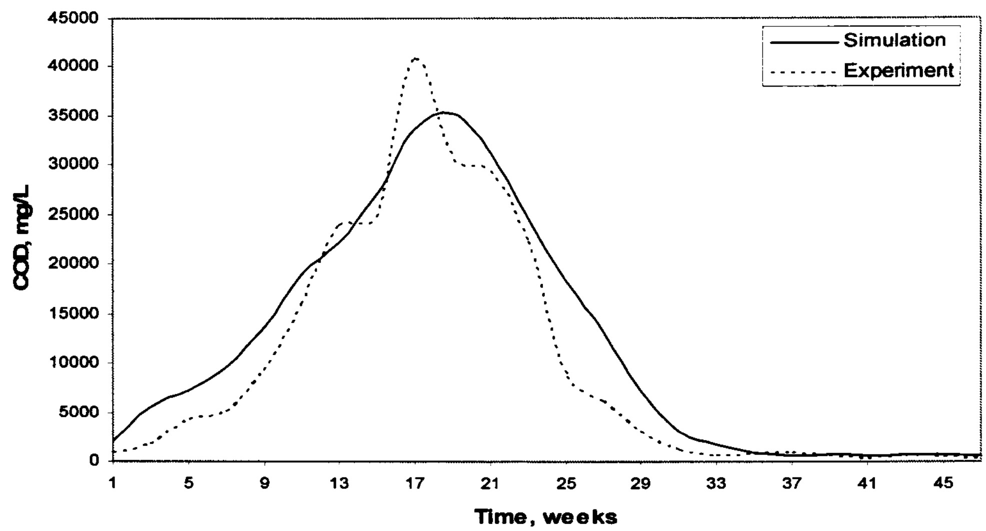

Figure 5.12 Measurements and simulations of COD in leachate from AER6

According to the simulation results, the overall model demonstrated reasonable agreement with the experimental observations, indicating the model has ability to capture the pertinent features of the MSW biodegradation process. Indeed, time to reach stabilization obtained from the model closely coincided with the experimental data for all bioreactors.

5.3.1.2 Temperature

The second parameter to be discussed is the model simulation of temperature in the simulated bioreactor landfills. The temperatures observed during experiment and estimations by model simulation are presented in Figures 5.13 and 5.14 for the bioreactor AER2 and AER4. Figure 5.13 shows that behavior and shape of both experimental and simulated curves are very similar. The results of the paired F-test (F calculation<F critical) at a 90% confidence level (α=0.10) show that there was no significant difference between the experimental results and simulations. However, the model simulation predicted higher temperatures than had been observed by measurement of about 1 to 5 ^{0}C with the exception of weeks 7 to 13 of the experimental study. At the end of operation, the deviation between the model simulation and experimental data was about 8 ^{0}C. A similar graph was observed in the bioreactor AER4 (refer to Figure 5.14). The

temperatures observed and prediction was increased exponentially from the 1^{st} week until a peak at the 11^{th} week of the experimental study. After reaching peak temperature, both measured and simulated temperatures curves exhibited an identical steep decline to the end of the experimental study.

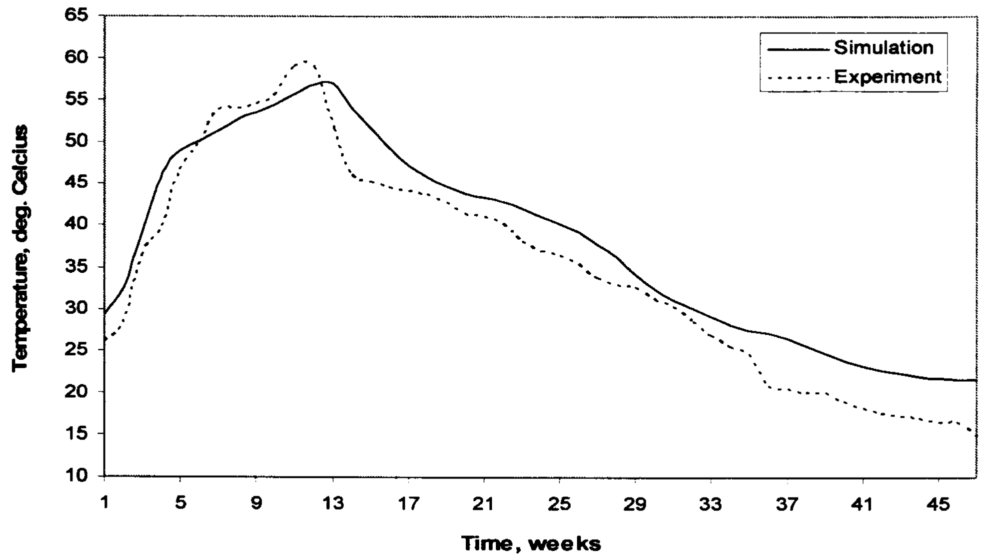

Figure 5.13 Measurements and simulations of temperature in AER2

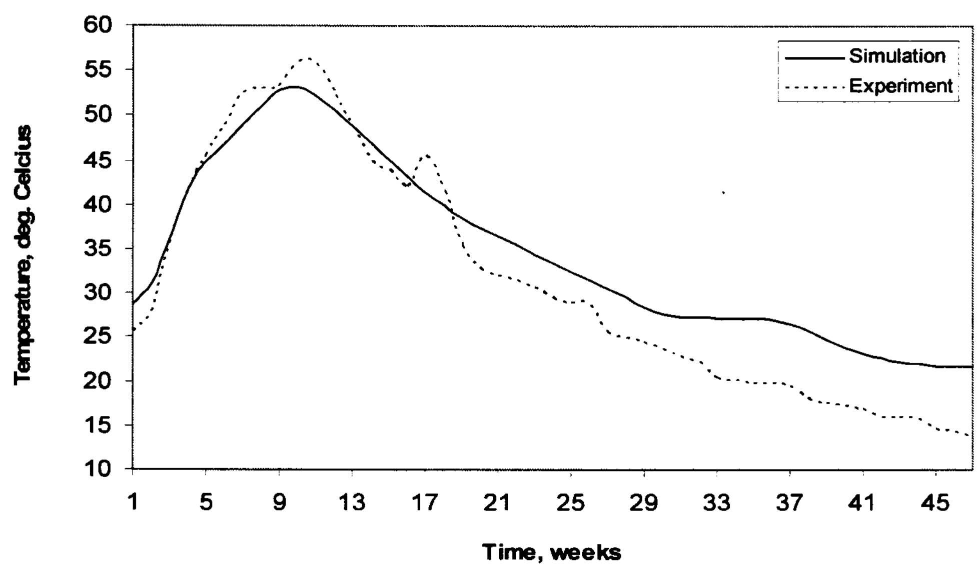

Figure 5.14 Measurements and simulations of temperature in AER4

The comparison between the temperatures experimental data and model simulation data for AER1 and AER3 are shown in Figures 5.15 and 5.16, respectively. Comparing the modeling results with the experimental measurements indicated that the predicted simulation was credible in capturing the behavior of the obtained data. A statistical procedure F-test (95% confidence level) also shows that there was no significant difference between experimental results and simulations. As can be seen in Figure 5.15, in the AER1, the model simulation had best fit the experimental data compared to the model simulation in other bioreactors. There was minimal difference at a peak temperature between the simulation and the experimental data. In addition, the graphs were practically overlaid from the beginning until week 18 of the experimental study. Minimal difference was noted between weeks 20 to the end of operation (Figure 5.15).

As in the AER1, the predicted simulation of temperature in the bioreactor AER3 (refer to Figure 5.16) was super imposed on the experimental data, but slight variation were also noted. These deviations may be due to inaccurate fuzzy set input of the membership function in the model for the bioreactor AER3. Furthermore, the temperature may be quickly released to the atmosphere during the biodegradation process.

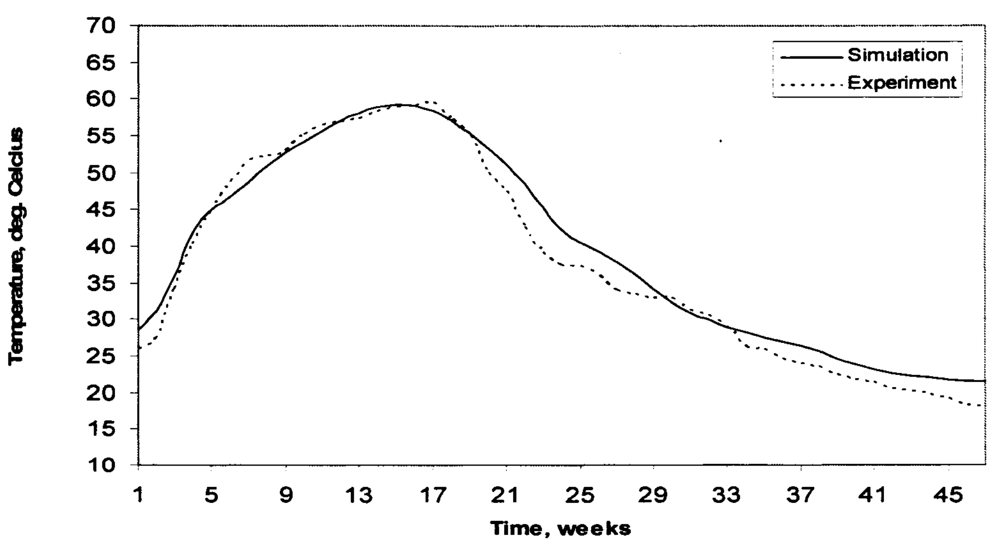

Figure 5.15 Measurements and simulations of temperature in AER1

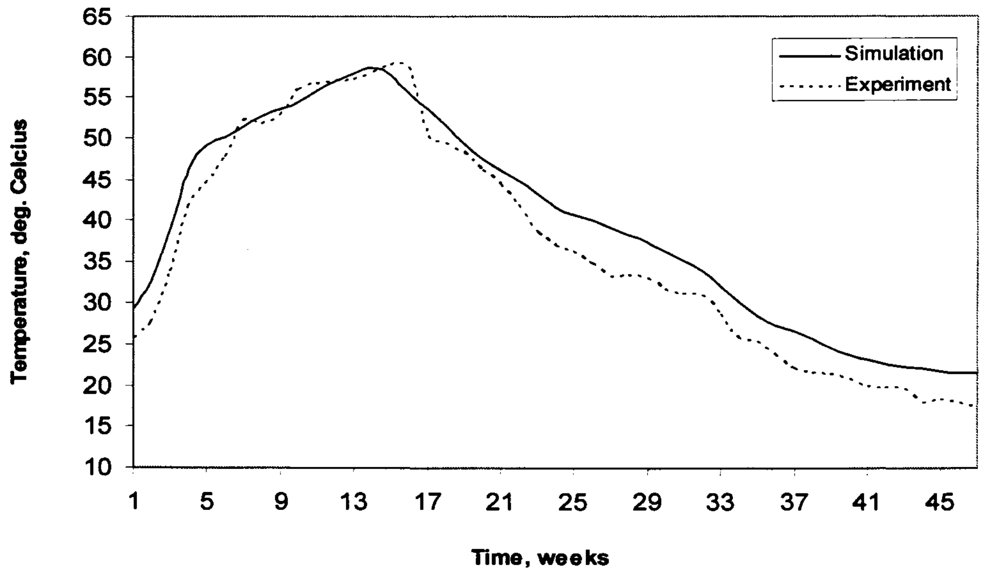

Figure 5.16 Measurements and simulations of temperature in AER

Figure 5.17 representsrepresent the measured and the simulated temperature data in AER6. As mentioned earlier, the bioreactor AER6 and AER5 were replicates; therefore, the simulation was run once for both bioreactors. The curve of the experimental data temperature was closely duplicated by the model simulation. At the beginning and until reached peak at the 13^{th} week of operation, both the simulation and measured data increased sharply to a peak of 50 and 55 ^{0}C. Furthermore, the temperature gradually declined until the end of the operation.

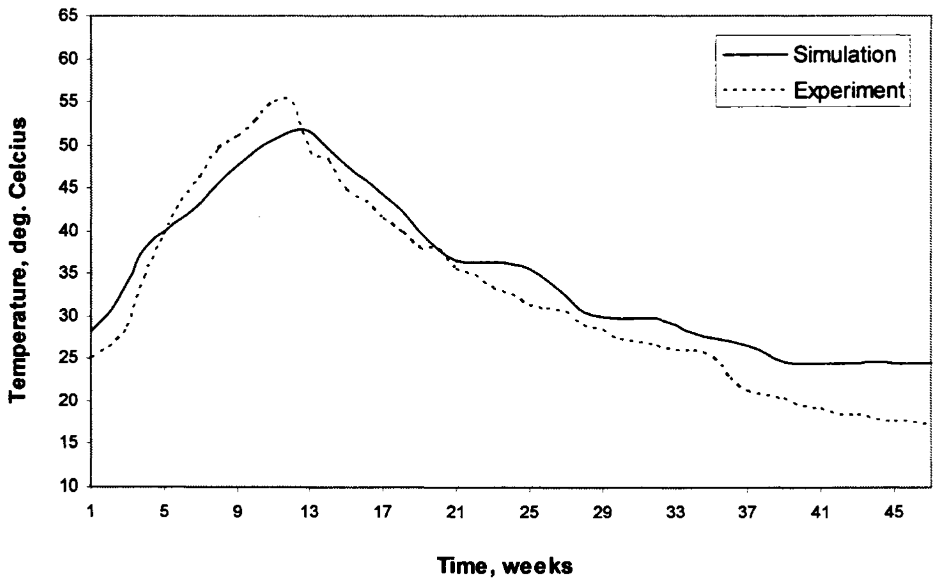

Figure 5.17 Measurements and simulations of temperature in AER6

5.3.2 Anaerobic Bioreactors

5.3.2.1 COD

Similar to the aerobic bioreactors, all profiles of COD concentrations in the leachate started with an increasing concentration until a peak value, followed by declining phase and constant afterward. A comparison between the predicted and experimental data of the COD concentrations in ANA2 and ANA4 are presented in Figure 5.18 and 5.19, respectively.

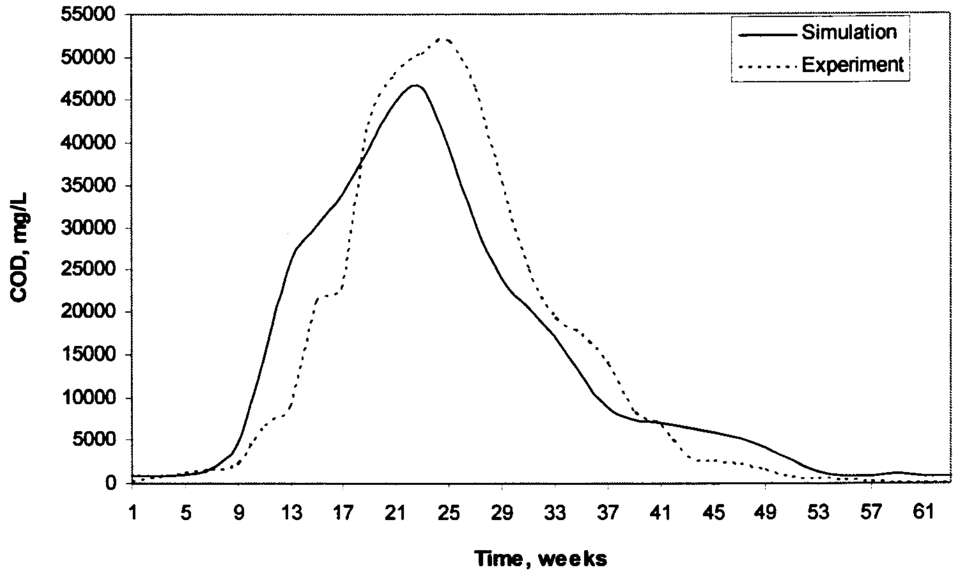

Figure 5.18 Measurements and simulations COD in leachate from ANA2

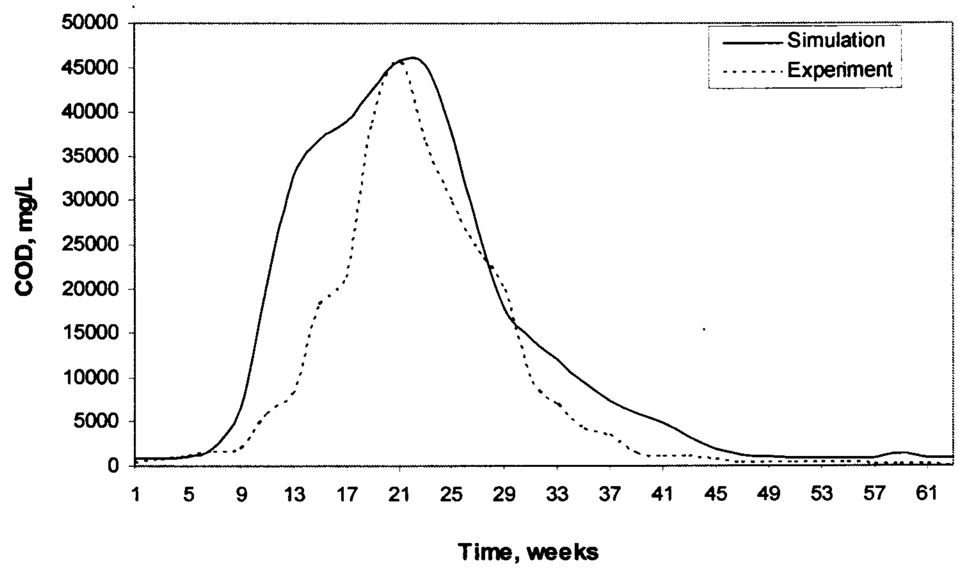

Figure 5.19 Measurements and simulations COD in leachate from ANA4.

The predicted simulation trends of the COD concentrations were similar to the trend of the experimental data (see Figures 5.18 and 5.19). In both bioreactors, the

ANA2 and ANA4, the major trend of the experimental data was closely reproduced by the model simulation. In the ANA2, the model simulation was over estimate the exhibited higher COD concentrations than the measured datain (weeks 1 to 19). in the range from 1,000 to 10,000 mg/L. After this period, the simulation data was lower compared to the experimental data (weeks 21 to 41) in the range from 1,000 to 5,000 mg/L. As in the ANA2, similar COD concentration curves were also indicated in the ANA4, but the model simulation showed higher COD concentrations along the period of the experiment. Discrepancies between the simulation and measured data were about 1,000 to 5,000 mg/L with the exception of weeks 13 to 17 which had a deviation of about 15,000 mg/L. In general, however, the model simulation in the ANA2 and ANA4 were in a good general agreement with the experimental data. This is also supported by statistical F-test that there was no significant difference between experimental results and simulations (F calculation < F critical) at 90% confidence level.

Figures 5.20 and 5.21 show the simulated and measured data of COD concentrations from ANA1 and ANA3, respectively. In the ANA1, the model simulation resulted in a slightly higher COD concentrations but lower at the peak than the datameasured data, followed by a close agreement between the simulated and experimental curves started from week 41 until the end of the experiment. The differences in the COD concentrations between the model simulation and the data were in the range from 1,000 toand 5,000 mg/L. As in the ANA1, similar profile between the predicted model and experimental data was observed in the bioreactor ANA3. However, the predicted simulation overestimated the COD concentrations from weeks 9 to 31 by an order about 10,000 mg/L. After the maximum COD concentration was obtained, the model closely coincided with the experimental data.

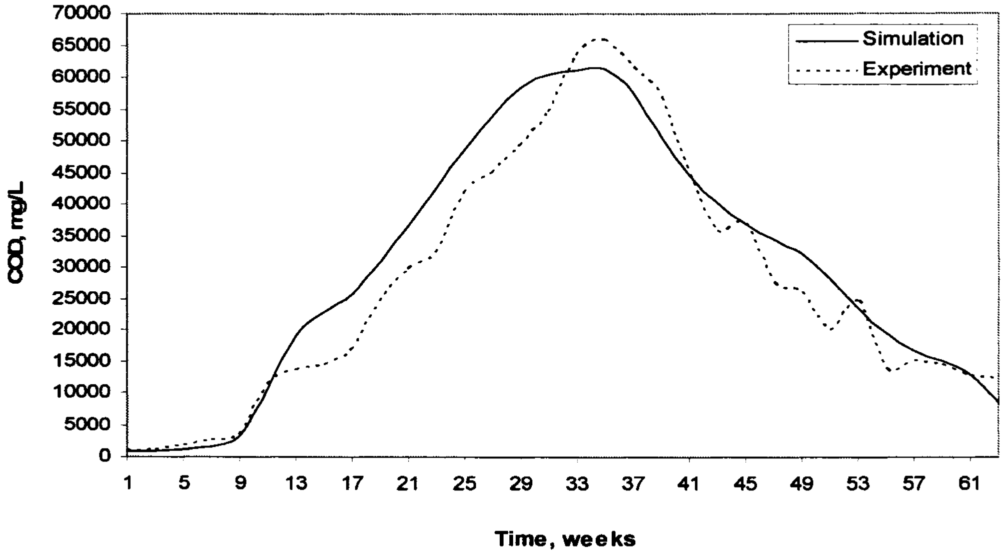

Figure 5.20 Measurements and simulations COD in leachate from ANA1

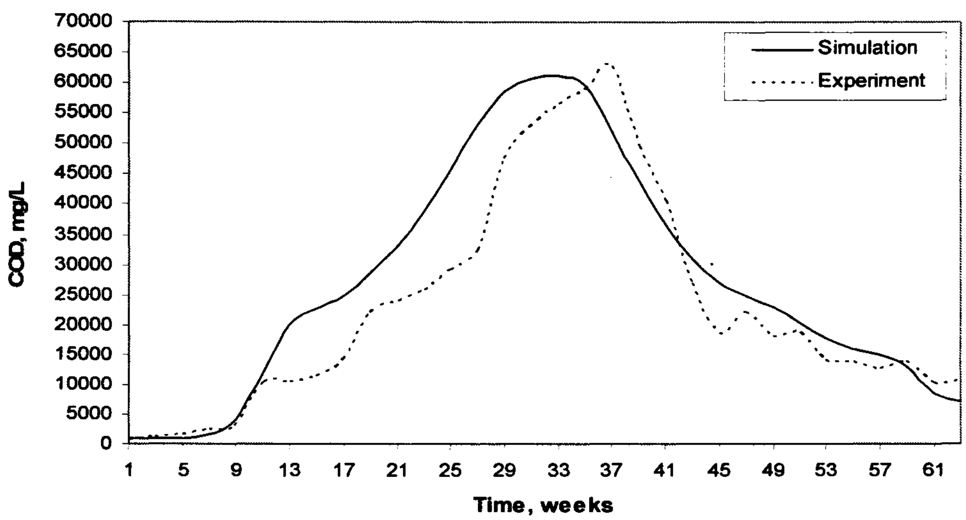

Figure 5.21 Measurements and simulations COD in leachate from ANA3

The COD concentrations in the leachate from the bioreactor ANA6 versus time obtained from the model simulation and experimental data are plotted in Figure 5.22. The computer simulations were conducted at once for the bioreactor ANA6 and ANA5 due to the fact that these bioreactors were replicates. It can be seen that the predicted simulation closely imitated the magnitude and the shape of the measurement data. Although there were discrepancies, in general, the trend of COD concentrations measurement was reasonably duplicated by the model simulation. At the beginning (from week 1 to 9), both the model simulations and the experimental data exhibited a gradual increase, followed by sharp increase to a peak of about 52,000 mg/L, and then dramatically decreased thereafter to about 5,000 mg/L in week 49, with a continuing trend until the end period of the experiment. The difference between experimental measurements and the model simulations was in the range from 1, 000 to 5,000 mg/L and occurred before and after the peak was reached.

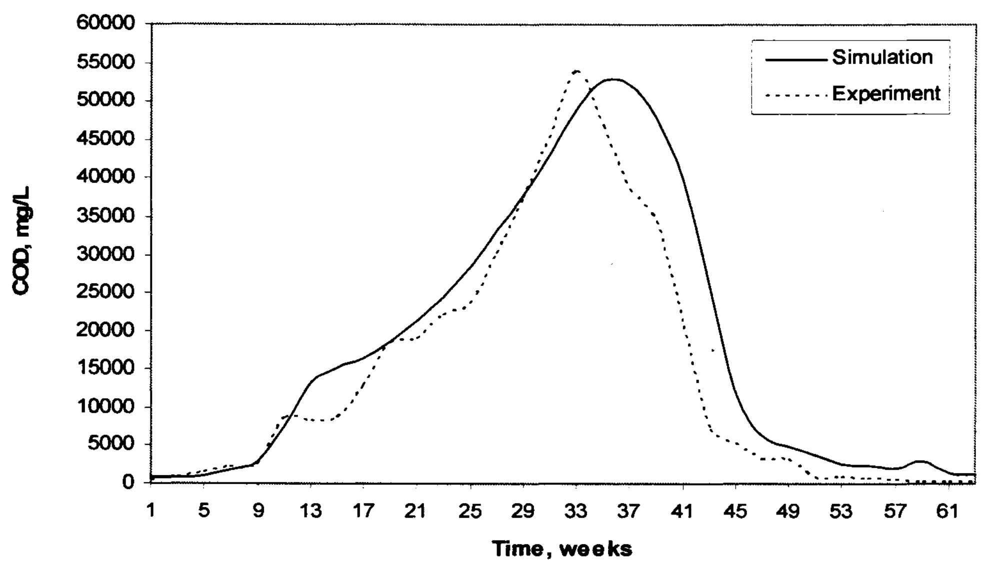

Figure 5.22 Measurements and simulations COD in leachate from ANA6

5.3.2.2 Temperature

Unlike in the aerobic bioreactors, the temperatures obtained in the experiment and estimated by model simulation in the anaerobic bioreactors increased and decreased gradually during the course of the operation. Figures 5.23 and 5.24 presented the model simulations and the experimental data for the bioreactor ANA2 and ANA4, respectively. In theFor bioreactor ANA2, the temperature simulation results for the temperature were closely superimposed followed on the experimental data (refer to Figure 5.23). Slight deviations, however, can be seen after week 29th to the end of experiment. In this phase, the deviation between the simulation and experimental data was about 2 ^{0}C. As in the ANA2, the temperature curves between predicted simulation and observed data in the bioreactor ANA4 were very close to each other (see Figure 5.24). This indicated that the trend of the temperature measurements was plausibly duplicated by the model simulations. Slight discrepancies between the predicted and measured temperature occurred after week 27th; however, discrepancies of about 2 ^{0}C were within an acceptable range of confidence (F calculation <F critical).

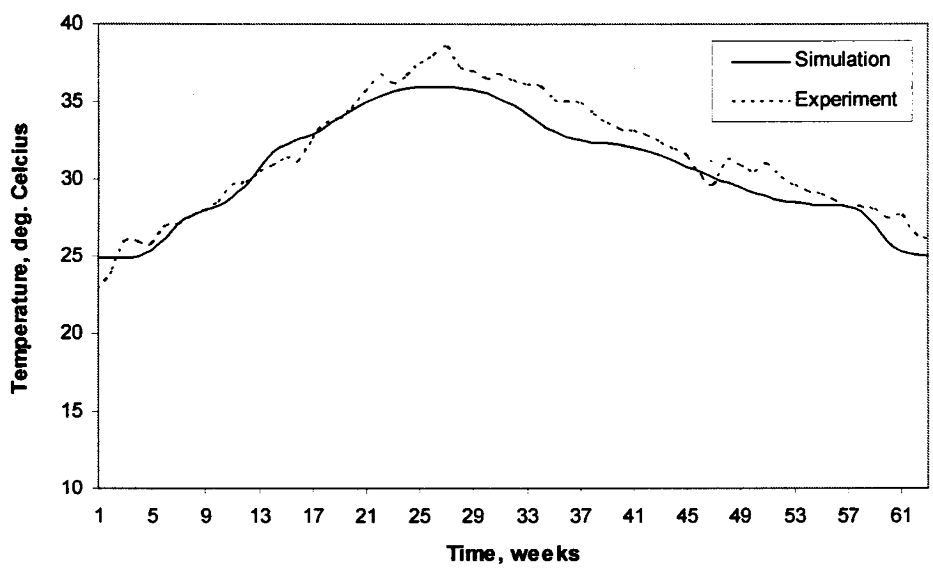

Figure 5.23 Measurements and simulations temperature in ANA2

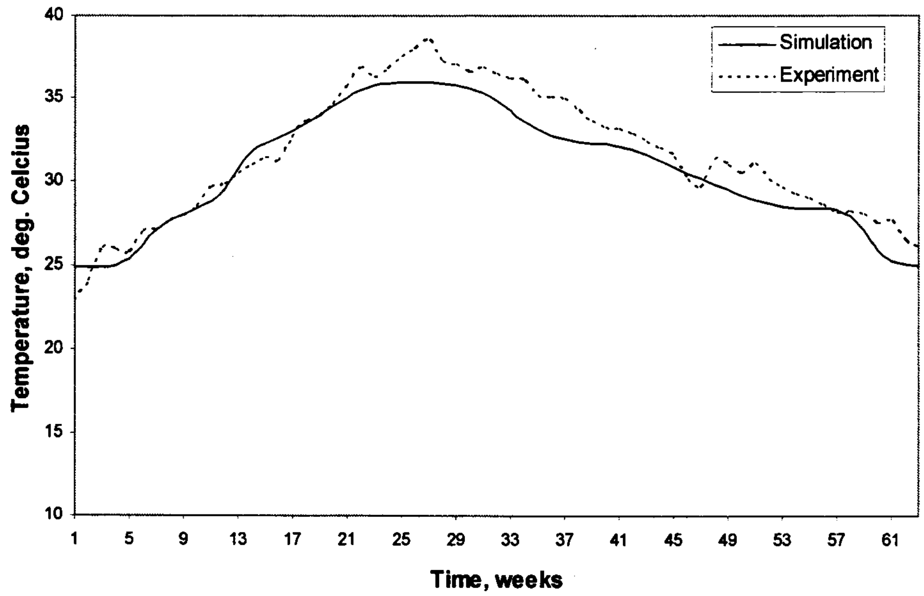

Figure 5.24 Measurements and simulations of temperature in ANA4

The comparison between the temperatures experimental data and model simulations for ANA1 and ANA3 are shown in Figures 5.25 and 5.26, respectively. It both cases, in the ANA1 and ANA3, the predicted results of temperatures indicated that the model developed was reliable in capturing the temperature performance of the reactors. obtained data during the experiment. The differences of temperature between the simulation and the experimental data was about 2 ^{0}C. As in ANA1, Tthe predicted simulations of temperature in the bioreactor ANA3 (see Figure 5.25) were also quite close the experimental data, but slight variations were also indicatedobserved. During the course of the experiment, the temperatures in the predicted simulations were about 3 ^{0}C higher of about 3 ^{0}C compared to the experimental data with the exception of week 17th to 37th. In this period, the simulation results fluctuated under and over the experimental data with the difference of about 1 ^{0}C. The results of the statistical F-test (F calculation<F critical) at a 95% confidence level (α=0.05) also show that there was no significant difference between the experimental results and simulations. In general, the simulation results for ANA1 and ANA3 were in a good agreement with the experimental data and can be used for predicting the temperature in biodegradation process.

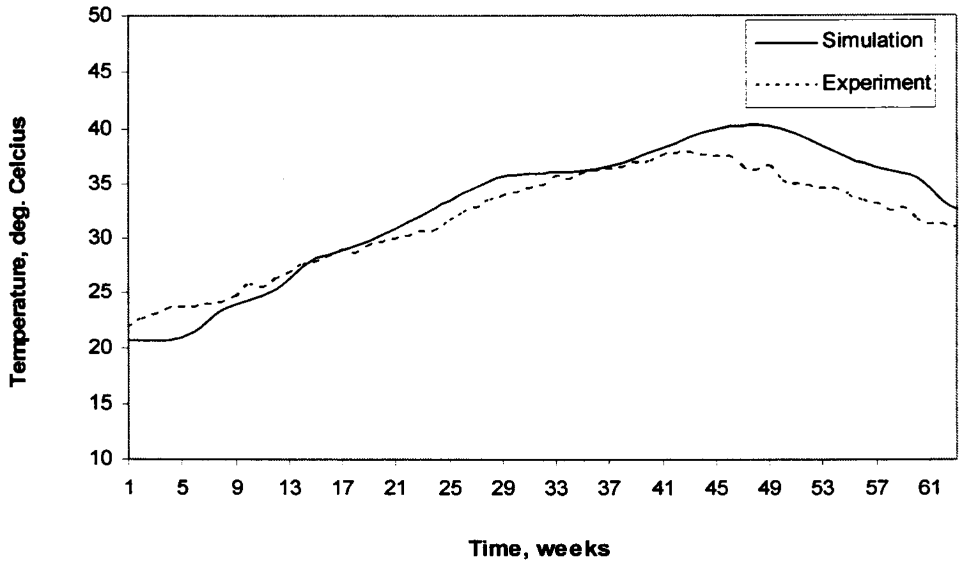

Figure 5.25 Measurements and simulations of temperature in ANA1

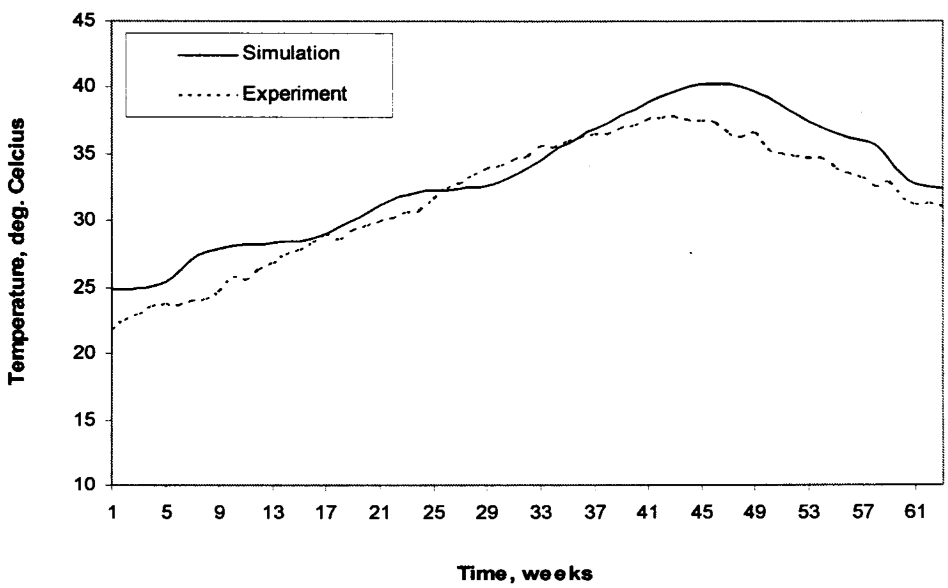

Figure 5.26 Measurements and simulations of temperature in ANA3

Figure 5.27 represents the measured and the simulated temperature data in ANA6. The model developed for ANA5 and ANA6 was the same due to the fact that these bioreactors were replicates. The temperature profile of the model simulation was over lay to the experimental data which indicated that the model developed was able to predict the actual data. In the model simulations and experimental measurements, temperatures increased gradually to their peak, and then decreased thereafter until the end of the operation. From 1st week to 33rd, temperature in the simulation results were about 1 to 3 ^{0}C higher compared than to the experimental data. Furthermore, from week 35 to 63, the predicted temperature was about 1 to 2 ^{0}C lower compared to the measured data.

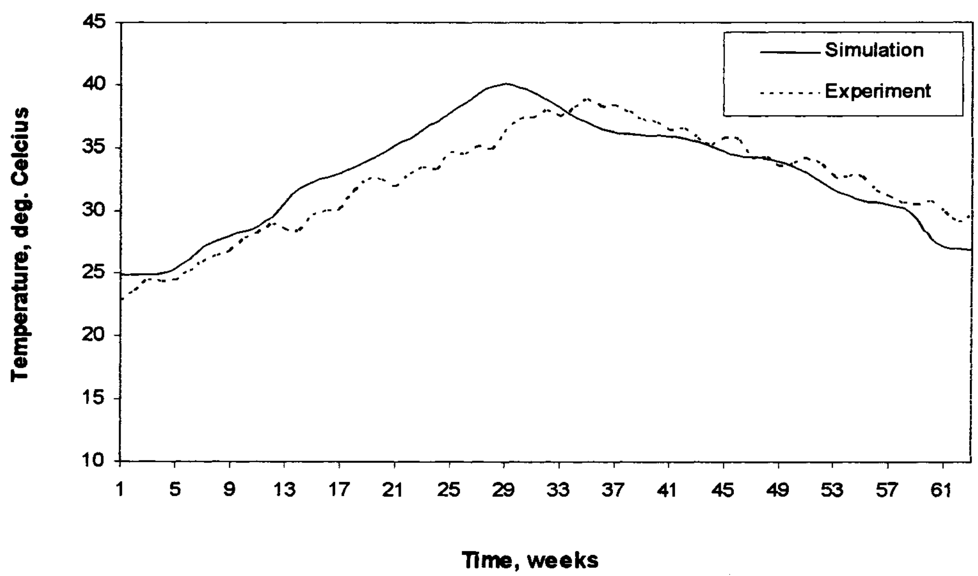

Figure 5.27 Measurements and simulations of temperature in ANA6

<u>5.3.2.3 Biogas</u>

The biogas production rates from the anaerobic bioreactors were previously discussed in Chapter 4. The biogas production rates obtained from the experiment and estimated by model simulation in the anaerobic bioreactors increased and decreased gradually during the course of the operation. Figures 5.28 and 5.29 present the model simulations and the experimental data for the bioreactor ANA2 and ANA4, respectively.

In the bioreactor ANA2, the simulation results for the biogas production rate were similar on the experimental data with the exception at the peak (refer to Figure 5.28). Deviations, however, can be seen after week 29th to 37th of the experiment. In this phase, the deviation between the simulation and experimental data was about 0.5 to 1.5 L/week. As in the ANA2, the biogas production rate curves between simulation and observed data in the bioreactor ANA4 were very close to each other (see Figure 5.29). This indicated that the trend of the biogas production rate measurements was plausibly duplicated by the model simulations. Slight discrepancies between the predicted and measured biogas production rate occurred after week 29th; however, this discrepancies of about 0.5 L/week were within an acceptable range of confidence based on paired F-test (F calculation<F critical) at a 95% confidence level (α=0.05).

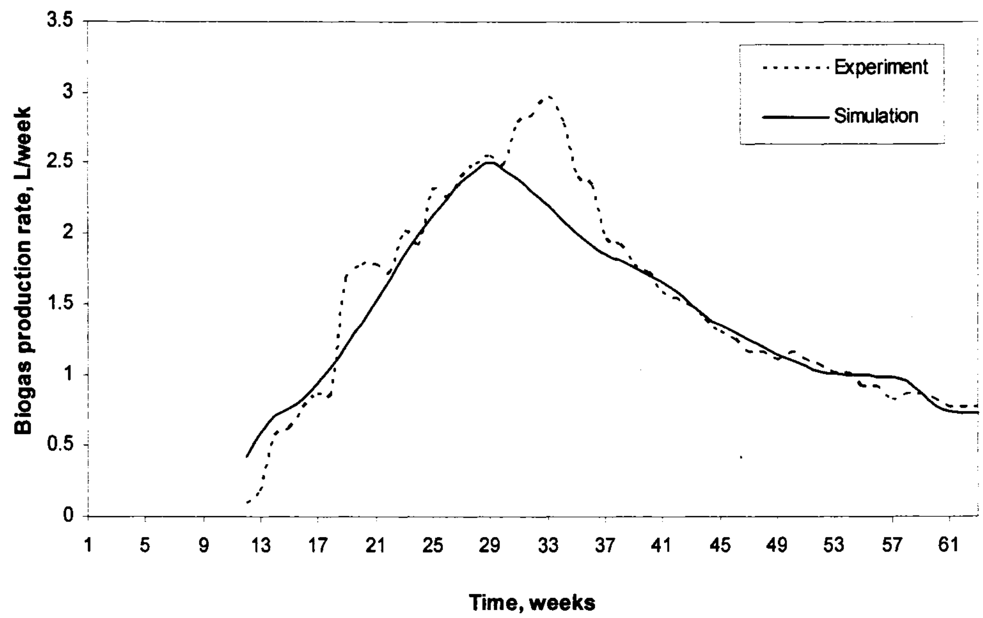

Figure 5.28 Measurements and simulations of biogas from ANA2

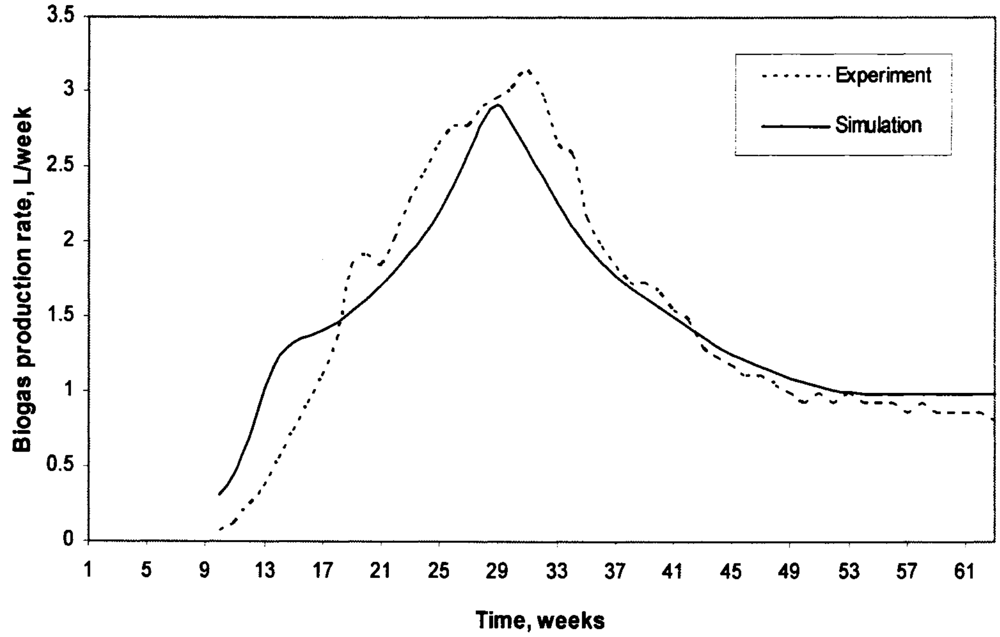

Figure 5.29 Measurements and simulations of biogas from ANA4

The comparison between the biogas production rates from experimental data and model simulation for ANA1 and ANA3 are shown in Figures 5.30 and 5.31, respectively. It both cases, in the ANA1 and ANA3, the predicted results of biogas production rate indicated that the model developed was reliable in capturing the performance of the obtained data during the experiment. The differences of biogas production rate between the simulation and the experimental data was from 0.25 to 5 L/week. The predicted simulations of biogas production rate in the bioreactor ANA3 (see Figure 5.31) were quite close the experimental data, but variations were also indicated. During the course of the experiment, the biogas production rate in the simulations were about 0.5 mL/week lower before week 41st and 0.25 L/week higher after week 43rd to 55th compared to the experimental data. In general, the simulation results for ANA1 and ANA3 were in a good agreement with the experimental data based on paired F-test (F calculation<F critical) at a 95% confidence level (α=0.05). Therefore the model can be used for predicting the biogas production rate in biodegradation process.

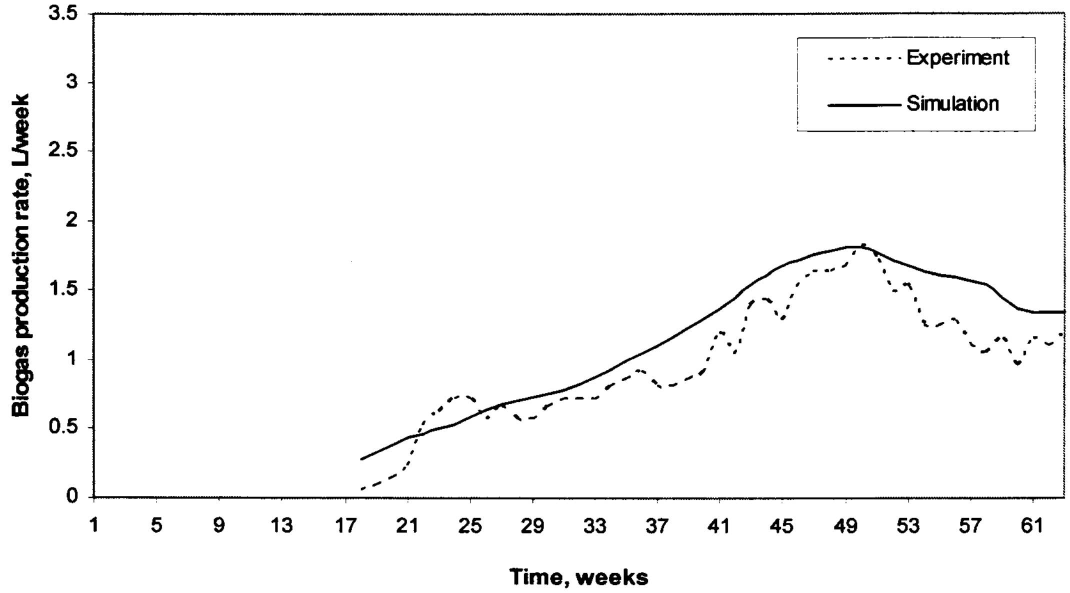

Figure 5.30 Measurements and simulations of biogas from ANA1

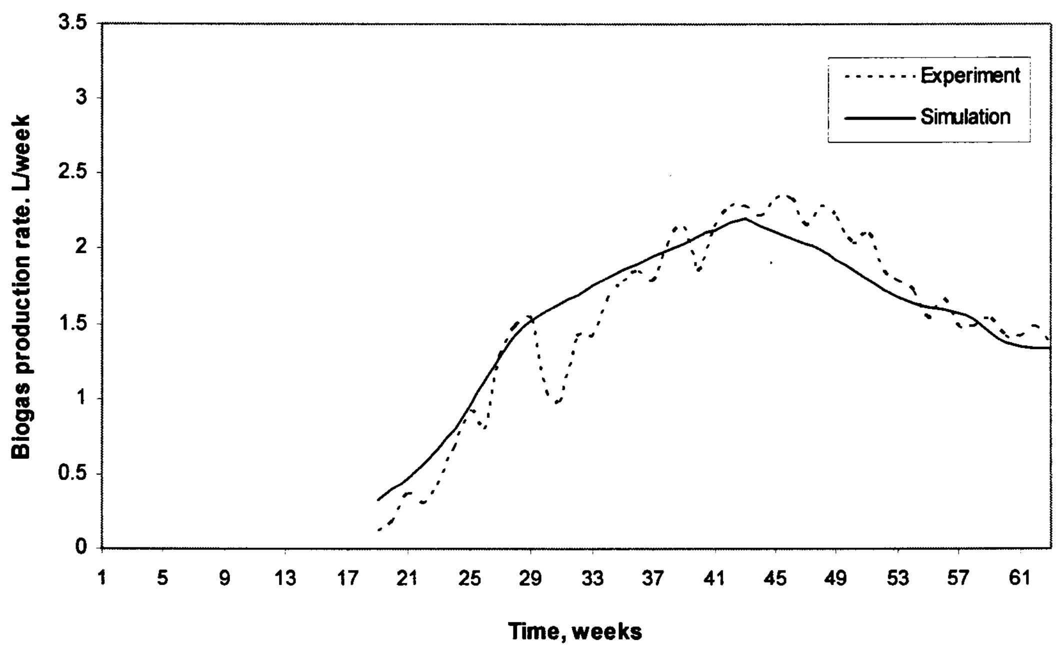

Figure 5.31 Measurements and simulations of biogas from ANA3.

Figure 5.32 represents the measured and the simulated biogas production rate in ANA6. The model developed for ANA5 and ANA6 was the same due to the fact that these bioreactors were replicates. The biogas production rate profile of the model simulation was similar and attached to the experimental data which indicated that the model developed was able to predict the actual data. In the model simulations and experimental measurements, biogas production rate increased exponentially to their peak, and then decreased gradually thereafter until the end of the operation. From 10th week to 35th, biogas production rate in the simulation results were about 0.5 L/week higher compared to the experimental data with the exception of week 19th to 23rd. Furthermore, from week 37 to 63, the predicted biogas production rate was about 0.25 L/week higher compared to the measured data.

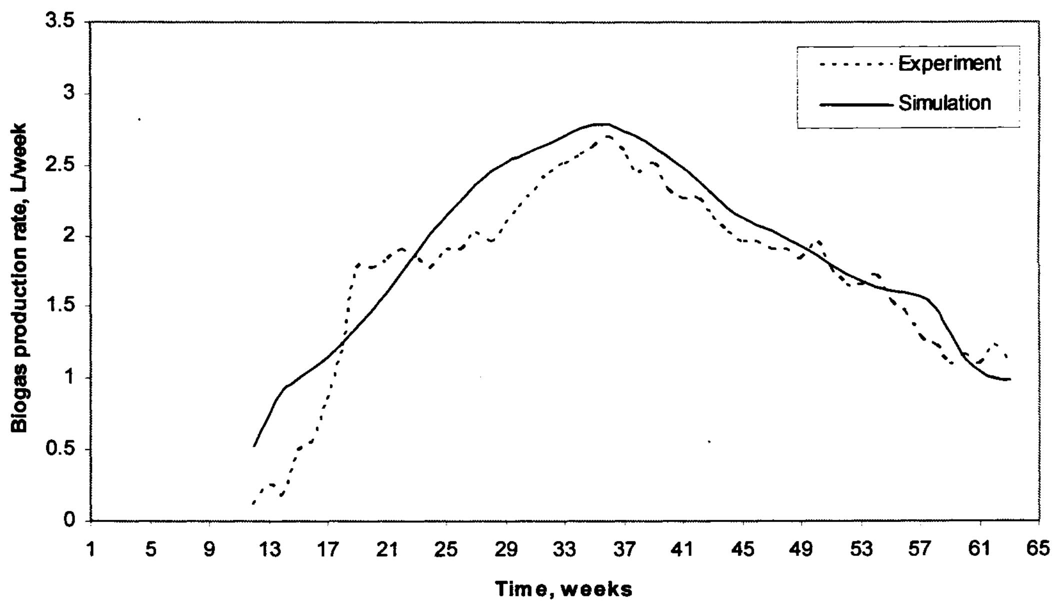

Figure 5.32 Measurements and simulations of biogas from ANA6.

5.3.3 Model Verification

Model verification was used to examine the applicability of the model simulation under a range of conditions. In selecting and developing a fuzzy model, one must consider the database availability to support and test the model. Most common measurements assembled from landfill leachate investigations include some basic information on leachate organic strength, such as COD and BOD, with impediment considerations to landfill biodegradation process.

Data used for model verification was COD concentration results from laboratory-scale simulated landfill bioreactors published by Sponza and Agdag (2004). Three anaerobic simulated landfill bioreactors consisted of cylinders of 30-cm in diameter and 100-cm in height (for a total volume of 71 L) were operated for the experiment. All bioreactors received a domestic waste which consisted approximately 75 to 95% of organic waste, 2 to 6 % of paper, and 1 to 2% of plastic, glasses, and textiles. The first bioreactor was filled up with a 26.5 kg of waste and was operated without leachate recirculation (single pass bioreactor). The second bioreactor was filled up with a 29 kg of waste and was operated with leachate recirculation at the rate of 9 L/day (310 mL/kg of waste.d). The third bioreactor was filled up with a 24 kg of waste and was operated with leachate recirculation at the rate of 21 L/day (875 mL/kg of waste.d). The particular characteristic of these experiments are presented in Table 5.1.

Table 5.1 Characteristic of simulated landfill bioreactors

Parameter	Simulated bioreactors		
	Single pass	Reactor9	Reactor21
Quantity of waste (kg)	26.5	29	24
Recirculation rate (L.d)	Without	9	21
Water content (%)	75	77	86
Organic matter (%)	76	85	96
Carbon (%)	42	47	53

Of all the leachate and gas data reported in this experiment, the COD concentration was the parameter utilized herein to examine the predictive capabilities of the fuzzy model. Leachate COD concentrations from measured data experiment and simulated model for the single pass and recirculation bioreactors are shown in Figure 5.33. In the verification model, it should be noted that the simulation result of COD concentration in the leachate was multiply by factor 1.46. This is due to the fact that the COD concentration in the model was developed based on experimental data from the anaerobic bioreactors ANA1 to ANA6. While the COD concentration in the leachate from simulated landfill bioreactors published by Sponza and Agdag (2004) was about 1.46 times higher compared to the bioreactors ANA1 to ANA6.

It can be seen from Figure 5.33 that the trend of the experimental data was closely reproduced by the model simulations for both single pass and recirculation bioreactors. For all bioreactors, both the model simulations and the experimental data exhibited a sharp increase, followed by a gradual increase a peak, and then decrease until the end period of the experiment. In the single pass bioreactor, the difference between experimental measurements and the model simulations was in the range from 500 to 10,000 mg/L. In the recirculation bioreactor, reactor 9, the model simulation exhibited lower COD concentrations than the measured data (weeks 1 to 6) in the range from 1,000 to 10,000 mg/L. After this period, the COD concentrations of the model simulation were higher compared to the experimental data in the range from 1,000 to 5,000 mg/L.

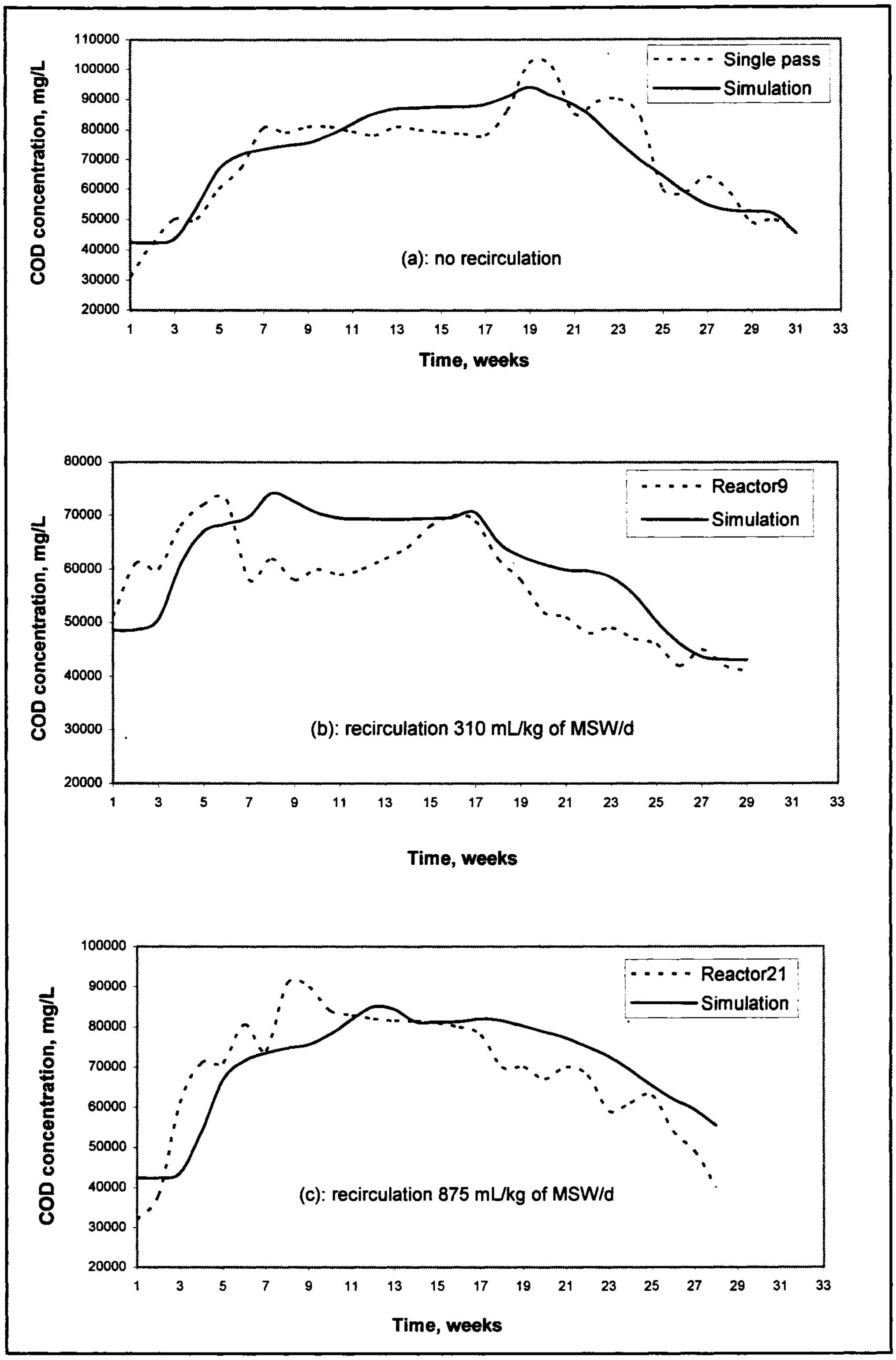

Figure 5.33 Model verification using experimental data from Sponza and Agdag.

The time delay to reach peak COD concentration can be estimated by 2 weeks period. As in the reactor 9, similar COD concentration curves were also indicated in the reactor 21, but the model simulation showed higher COD concentrations after week 13^{th} of the experiment. Discrepancies between the simulation and measured data were about 1,000 to 15,000 mg/L at various time intervals (Figure 5.33).

In the case of the single pass reactor and reactor 21, the prediction of maximum COD concentrations fell short in estimating the actual values of COD concentrations of about 10,000 mg/L. The reason of high COD concentrations at the single pass and reactor 21 may be due to accumulation of VFA that increase the COD concentrations. Although there was deviations between the experimental and predicted COD concentrations in the leachate, the overall model simulation demonstrated reasonable agreements with the experimental results, indicating the model ability to capture the pertinent features of MSW biodegradation process.

In the case of the single pass reactor and reactor 21, the prediction of maximum COD concentrations fell short in estimating the actual values of COD concentrations of about 10,000 mg/L. The reason of high COD concentrations at the single pass and reactor 21 may be due to accumulation of VFA concentrations that increase the COD concentrations.

Although there was deviations between the experimental and predicted COD concentrations in the leachate, the overall model simulation demonstrated reasonable agreements with the experimental experiment, indicating the model ability to capture the pertinent features of MSW biodegradation process.

5.3.4 Sensitivity Analysis

To assess the relative influence of reasonable value of model parameter from leachate recirculation and sludge addition, sensitivity analysis was performed in this experiment. In this case, model parameter was λ and γ for leachate recirculation and sludge addition, respectively. The following formulas were used to predict the value of the parameter,

$$LR_p = \lambda\ LR_{max} \tag{5.11}$$

$$SL_p = \gamma\ SL_{max} \tag{5.12}$$

where, LR_p = predicted leachate recirculation rate (mL/kg of MSW.day)

LR_{max} = maximum leachate recirculation rate from the experiment (855 mL/kg of MSW.day)

SL_p = predicted sludge addition rate (mL/kg of MSW.day)

SL_{max} = maximum sludge addition rate from the experiment (85.5 mL/kg of MSW.day)

Figure 5.34 and 5.35 illustrate the sensitivity of the model of varying leachate recirculation and sludge addition. The sensitivity of the model was evaluated using the COD concentrations in the leachate and temperature from aerobic bioreactors as measures of the system's response.

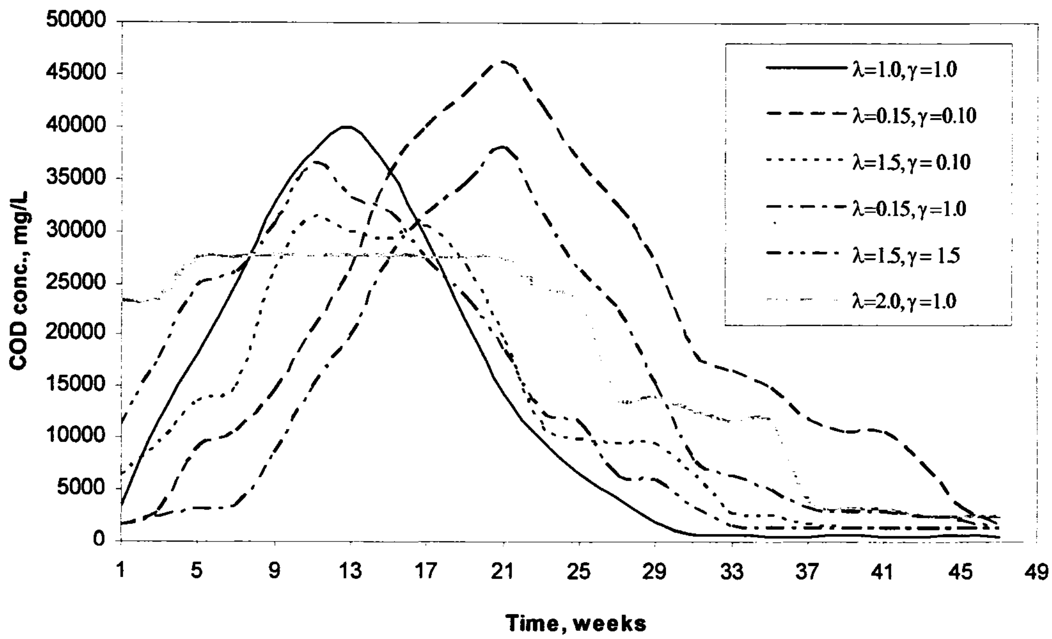

Figure 5.34 Sensitivity analysis for the COD concentration of model simulation

The maximum COD concentration in the leachate is reached about 45,000 mg/L with λ = 0.15 and γ = 0.10 after 21 weeks. Increasing λ to 1.5 and maintaining γ of 0.10 decreased the COD maximum concentration to about 31,000 mg/L after 11 weeks. The value of λ was found to influence significantly the maximum COD concentration in the leachate and time to reach MSW stabilization. Increasing parameter γ from 0.1 to 1.5 and maintaining λ of 1.5 was also found to have effect on COD concentrations. The overall influence of these parameters, λ and γ, were found to influence the maximum of COD concentration. By varying λ of 2.0 and keeping λ of 1.0 resulted in changes of the curve's shape. It can be seen that the sensitivity model predicted was found to be disagreement with the shape of other curves.

Figure 5.35 depict the sensitivity analysis for the temperature model simulation. Decreasing λ to 0.15 and γ to 0.10 increased the temperature to maximum of about 58°C after 17 weeks. Similar trends of temperature model simulation were found for all value of parameter λ and γ with the exception of λ of 2.0 and γ of 1.0. These results show that temperature model was sensitive to the rate of leachate recirculation and sludge addition.

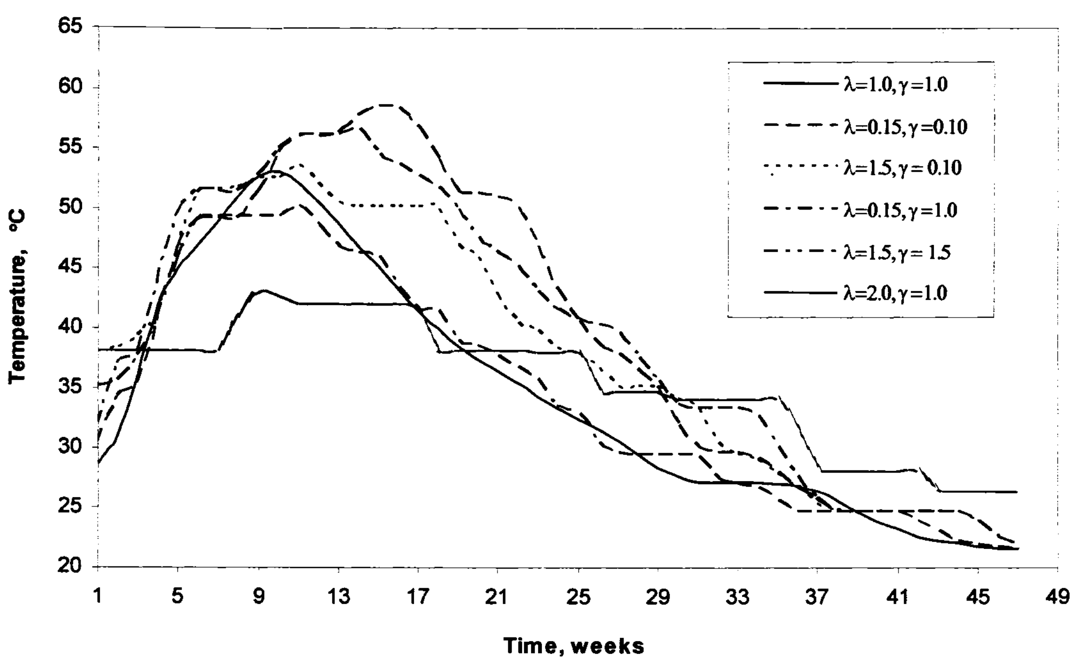

Figure 5.35 Sensitivity analysis for the temperature of model simulation

5.3.5 Summary of COD Concentration, Temperature, and Biogas Models

As discussed previously, the model simulation for the COD concentrations in the leachate, temperature, and biogas production rates in the bioreactors demonstrated reasonable agreement with the experimental observations. Although there were deviations between the model simulation and measured data, the simulation results can closely predict the trend of the process. The overall evaluations of the simulation also confirmed the efficiency and reliability of the model to reproduce pertinent characteristics and important features of MSW biodegradation under leachate recirculation and sludge addition with or without air addition. In addition, the model has the ability to capture the pertinent features of the MSW biodegradation process. Indeed, time to reach stabilization obtained from the model closely coincided with the experimental data for all bioreactors. Accordingly the MSW biodegradation process can be accurately simulated using the fuzzy logic control approach.

CHAPTER 6

CONCLUSIONS, RECOMMENDATIONS, AND CONTRIBUTIONS

6.1 Conclusions

The biodegradation of municipal solid waste (MSW) in simulated bioreactor landfills was investigated over a period of operated until 47 and 63 weeks under for aerobic and anaerobic conditions, respectively. The experimental results have shown that the aerobic bioreactors were stabilized within 27 to 45 weeks and it took 39 to 63 weeks for aerobic andand the anaerobic bioreactors. The results of the experimentthis research study could be concluded as follows:

1. Leachate recirculation and sludge addition at the rate of 855 and 85.5 mL/kg of MSW.d were the optimum rate for operating aerobic and anaerobic bioreactors.

2. Leachate recirculation and sludge addition enhanced conditions such that there was a rapid increase which was followed by a rapid decrease of COD and BOD concentrations in the leachate. The combination of leachate recirculation and sludge addition at the rate of 855 and 85.5 mL/kg of MSW/d for AER4 and ANA4, which was the highest rate, exhibited the fastest biodegradation of MSW, as compared to the medium rate of 570 and 57 mL/kg of waste/d for AER5 and AER6, and ANA5 and ANA6. The slowest biodegradation was from AER1 and ANA1, which was given the

lowest rates of leachate recirculation and sludge addition (285 and 28.5 mL/kg of MSW/d). Therefore, the optimal conditions for operating bioreactor landfills can be achieved by combination of the highest rate of leachate recirculation and sludge addition.

3. In the aerobic bioreactors, the fastest time to reach the maximum concentration of COD in the leachate (38,000 mg/L) was was 11 weeks for the bioreactor AER4 which received the highest rate of leachate recirculation and sludge addition with the longest being 19 weeks for both AER1 (52,000 mg/L) and AER3 (42,000 mg/L). Both bioreactors AER1 and AER3 received the lowest rate of leachate recirculation and sludge. In addition, it took between 27 and 47 weeks to reach the stabilization of MSW, with COD concentration in the leachate being approximately 1,000 mg/L. While in the anaerobic bioreactors, the fastest time to reach the maximum concentration of COD in the leachate (45,000 mg/L) was 21 weeks after operation for the ANA4, and it was even longer the latest was (359 and 37 weeks) for both ANA1 and ANA3. These bioreactors had a maximum COD concentration in the leachate of about 66,000 and 63,000 mg/L, respectively.. In additionMoreover, the fastest time to reach the MSW stabilization, with the COD concentration of about 1000 mg/L, was between 39 weeks and the latest was 63 weeks. It could be concluded that the time to reach attain stabilization for under anaerobic is 1.5-fold slower compared to aerobic bioreactors.

4. Addition of air flow at the rate of 84 L/kg of MSW.d /d to the simulated aerobic bioreactors landfill was found to have a positive effect on the rate of biodegradation of the MSWwaste. It was proven that the performance of aerobic and anaerobic bioreactors are statistically different. The aerobic conditions increased the bioreactors temperatures of the bioreactors and reachedin the mesophilic/ thermophilic ranges (40 to 55 ^{0}C) within 7 weeks of operation. While in the anaerobic bioreactors, only the bioreactor ANA4 temperatures reached mesophilic thermophilic temperature (40 ^{0}C), and that was range after 23 weeks of operation. and only in ANA4. In the bioreactors AER2 and AER4, and ANA2 and ANA4 which received the highest rate of leachate

recirculation (855 mL/kg of MSW.d), temperatures were below 15 ^{0}C at the end of the operation. This indicated that the higher rate of leachate recirculation can resulted in faster waste stabilization and finally cooling the bioreactor treatment system..

5. The highest total Kjeldahl nitrogen (TKN) concentration of approximately 400 mg/L was found in AER1 and 300 mg/L in ANA1 after 15 weeks of operation. On the contrary, theThe lowest TKN concentration of 90 mg/L was detected in AER2 and ANA2 at the end of experimental work.operation. In all aerobic bioreactors, the TKN concentrations decreased over the time and reached between 90 to 130 260 for aerobic bioreactors and 60 80 to 140 250 mg/L for aerobic and anaerobic bioreactors at the end of the operation. In both aerobic and anaerobic bioreactors, the total phosphorus (TP) concentrations were almost constant during the biodegradation process within the range of about 0.4 to 0.6 mg/L. These results indicate that there was no substantial removal of the total phosphorus present in the waste matrix. These results indicated that nutrients nitrogen and phosphorus were not a limiting factor for this experiment.

6. The Sstatistical empirical models were used to describe the effects of leachate recirculation and sludge addition. The values of estimate β_1 and β_2 in the models indicated that the leachate recirculation and sludge addition had a significant effect on the leachate COD concentration which was used as an indicator of MSW biodegradation. TSince the statistical empirical model with 95% confidence level were presented in equation as follows:

   Aerobic:

   5 weeks: $\hat{Y} = 6108.17 + 2514.09\,x_1 + 2275.79\,x_2 + 1999.16\,x_1x_2$

   19 weeks: $\hat{Y} = 319995.98 - 16641.48\,x_1 + 1817.63\,x_1x_2$

   27 weeks: $\hat{Y} = 9328.25 - 9164.92\,x_1$

   Anaerobic:

   15 weeks: $\hat{Y} = 13804.17 + 3352.43\,x_1 + 1398.85\,x_2 + 1224.56\,x_1x_2$

   39 weeks: $\hat{Y} = 30869.37 - 24206.92\,x_1$

49 weeks: $\hat{Y} = 8733.92 - 1775.45\, x_1$

For aerobic bioreactors, estimate parameters of β_1 were 2514.09, 16641.48 and 9164.92, was larger than β_2 were 2275.79, 0, and 0 and β_{12} were 199.16, 1817.63, and 0. While in anaerobic bioreactors, estimate parameters of in equation 5 β_1 were 3352.43, 24206.92, and 1775.45, β_2 were 1398.85, 0, and 0, and β_{12} were 1224.56, 0 and 0. These values showthis means that leachate recirculation had a more significant effect as compared to sludge addition as well as combination of leachate recirculation and sludge addition.

7. The ANOVA test indicated that leachate recirculation, sludge addition and their combination had a significant effect on biodegradation of the waste at the confidence level of 99.0%, 99.0%, and 90%, respectively during ascending stage. During the descending stage, the confidence levels were 99.9%, 95%, and 95%, respectively. The effect of air flow addition on the MSW biodegradation was very significant at a 99.90% and 90% confidence level during the ascending and descending stages.

8. A fuzzy logic model was developed to simulate the effect of leachate recirculation and sludge addition on MSW biodegradation in landfills. The COD concentration in the leachate, temperature inside the bioreactors, and biogas production rates were used to build the model.

9. A fuzzy logic model was developed to demonstrate the effect of leachate recirculation and sludge addition on MSW biodegradation in landfills. The COD concentration in the leachate and temperature inside the bioreactors is used to build the model. The model show that the effect of leachate recirculation at the rate of 855 mL/kg of waste/d and sludge addition at the rate of 85.5 mL/kg of waste/d resulting in the fastest biodegradation of MSW.

10. A sensitivity analysis show that model parameter $\lambda = 1.5$ and $\gamma = 1.5$ were the maximum rate of leachate recirculation and sludge addition, respectively that fuzzy logic model capable simulating relatively well the COD in the leachate, temperature, and biogas production.

6.2 Recommendations for further research

Further studies in the following directions may be carried out to improve the system.

1. *Further examination of the proposed system.* The proposed system was tested using 7-L bioreactors and only handled 2.5 kg of MSW. To further examine the performance of the system, additional test should be conducted using in situ landfill site.

2. *Investigation using different air flow rates.* Air The air flowrate is used at constant rates in this study. The effect of air on the MSW biodegradation under aerobic conditions can be conducted using different scenario of air flowrate.

3. *No buffer addition.* Buffer addition was applied to maintained control the pH in the range of 6.5 pH in all bioreactors. Further research should be conducted to examine the effect of buffer addition.

4. *Higher sludge addition rate.* As mentioned earlierThe , the effect effect of sludge addition was found to be not less significant compared to leachate recirculation. Further research could be conducted using higher sludge additiondifferent ratios of sludge addition to leachate recirculation. rate compared to the system proposed.

6.3 Contribution to Knowledge

Several researchers used kinetic model to simulate biodegradation rate of landfill under anaerobic conditions. There is limited information in the literature regarding the use of aerobic landfill as a method of solid waste disposal. As a result, there is no kinetic model developed to simulate biodegradation rate of aerobic landfill. In addition, comparative study of biodegradation rate of municipal solid waste under aerobic and anaerobic conditions that based on statistical experimental design was also limited. Therefore, the use of aerobic and anaerobic conditions for the rapid stabilization of MSW in simulated bioreactor landfills was analyzed in this research.

Presently, there is no fuzzy logic model used to simulate the biodegradation of municipal solid waste under aerobic and anaerobic conditions. In this research, therefore, fuzzy logic model was developed and used to simulate of COD concentrations in the leachate, temperature, and biogas production that represent biodegradation process of municipal solid waste.

The experimental results show that there is significant different between aerobic and anaerobic bioreators in term of biodegradation rate. In addition, fuzzy logic model can simulate the biodegradation rate of municipal solid waste under aerobic and anaerobic conditions. Therefore, there are scientific contributions based on this research.

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APPENDIX A

SAMPLE CALCULATION OF PARAMETER ESTIMATE OF STATISTICAL EMPIRICAL MODEL

The empirical models for COD concentration as a function of the two coded variables (leachate recirculation and sludge addition) were carried out. Matrix calculation of the aerobic bioreactor at 5 weeks (model 4.1) was performed using Microsoft Excel as follows:

The model is expressed as follows:

$$\hat{Y} = \beta_0 + \beta_1 x_1 + \beta_2 x_2 + \beta_{12} x_1 x_2$$

The least square estimates of the parameters are given by

$$\beta = (X^T X)^{-1} X^T Y$$

where:

$$X = \begin{bmatrix} 1 & -1 & -1 & 1 \\ 1 & 1 & -1 & -1 \\ 1 & -1 & 1 & -1 \\ 1 & 1 & 1 & 1 \\ 1 & 0 & 0 & 0 \\ 1 & 0 & 0 & 0 \end{bmatrix} \qquad Y = \begin{bmatrix} 3872 \\ 4902 \\ 4426 \\ 13452 \\ 5809 \\ 4185 \end{bmatrix}$$

$$X^T = \begin{bmatrix} 6 & 0 & 0 & 0 \\ 0 & 4 & 0 & 0 \\ 0 & 0 & 4 & 0 \\ 0 & 0 & 0 & 4 \end{bmatrix} \qquad X^TX = \begin{bmatrix} 6 & 0 & 0 & 0 \\ 0 & 4 & 0 & 0 \\ 0 & 0 & 4 & 0 \\ 0 & 0 & 0 & 4 \end{bmatrix}$$

$$(X^TX)^{-1} = \begin{bmatrix} 0.16667 & 0 & 0 & 0 \\ 0 & 0.25 & 0 & 0 \\ 0 & 0 & 0.25 & 0 \\ 0 & 0 & 0 & 0.25 \end{bmatrix} \qquad X^TY = \begin{bmatrix} 36649.04 \\ 10056.34 \\ 9103.17 \\ 799.63 \end{bmatrix}$$

$$\beta = (X^TX)^{-1}\ X^TY$$

$$\begin{bmatrix} \beta_0 \\ \beta_1 \\ \beta_2 \\ \beta_{12} \end{bmatrix} = \begin{bmatrix} 0.16667 & 0 & 0 & 0 \\ 0 & 0.25 & 0 & 0 \\ 0 & 0 & 0.25 & 0 \\ 0 & 0 & 0 & 0.25 \end{bmatrix} \begin{bmatrix} 36649.04 \\ 10056.34 \\ 9103.17 \\ 799.63 \end{bmatrix}$$

$$\begin{bmatrix} \beta_0 \\ \beta_1 \\ \beta_2 \\ \beta_{12} \end{bmatrix} = \begin{bmatrix} 6108.17 \\ 2514.09 \\ 2275.79 \\ 1999.16 \end{bmatrix}$$

Thus the fitted model is:

$$\hat{Y} = 6108.17 + 2514.09\, x_1 + 2275.79\, x_2 + 1999.16\, x_1 x_2$$

Quantitative Lack of Fit Test:

The quantitative lack of fit test compares an estimate of lack of fit plus pure error variance to an estimate of pure error variance from replicated runs. If lack of fit test is insignificant then the expected value of the ratio of these two variances is one. The significant test of the ratio under null hypothesis is $R = \sigma_{LF}^2 / \sigma_{PE}^2 = 1$.

Estimate σ_{PE}^2 from the center point replicates at $x_1 = x_2 = 0$

AER5: Y = 5809.33

AER6: Y = 4185.27

Variance = σ_{PE}^2 =1318794.16

Standard deviation = σ = 1148.39

Mean = μ = 4997.30

Degree of freedom = DF = 1

Calculate the lack of fit ratio, R, from six runs (AER1 to AER6)

Sum Square Error = SSE = 5020906.55

Number of run, n = 6

Number of parameter = 4

σ_{LF}^2 = [SSE – DF. σ_{PE}^2]/[n-p-DF]

= [5020966.55 – 1 (1318794.16)]/[6-4-1]

= 3702172.39

Therefore:

$R = \sigma_{LF}^2 / \sigma_{PE}^2$

= 3702172.39 / 1318794.16

= 2.81

Compare this with the F critical value. From the F table (Zar, 1999).

$F(1,1,0.05) = 161$

Since R < 161, **lack of fit is not significant.**

Determine 95% confidence level for the true values of the parameters.

Confidence levels of 95% for the parameters are bounded by:

$$\beta \pm t_{2,0.025}\sqrt{V(\beta)}$$

$$\text{where } V(\beta) = \begin{bmatrix} \sigma^2/6 & 0 & 0 & 0 \\ 0 & \sigma^2/4 & 0 & 0 \\ 0 & 0 & \sigma^2/4 & 0 \\ 0 & 0 & 0 & \sigma^2/4 \end{bmatrix}$$

Since the model displays no lack of fit, the pure error variance will be estimate using:

$\sigma^2 = 836817.76$

Consequently,

$$V(\beta) = \begin{bmatrix} 139469.63 & 0 & 0 & 0 \\ 0 & 209204.44 & 0 & 0 \\ 0 & 0 & 209204.44 & 0 \\ 0 & 0 & 0 & 209204.44 \end{bmatrix}$$

95% confidence levels, for which $t_{2,0.025} = 4.303$, are bounded by:

$$\beta_0 \pm (4.303)\sqrt{139469.63} = 6108.17 \pm 1606.98$$

$$\beta_1 \pm (4.303)\sqrt{209204.44} = 2514.09 \pm 1968.14$$

$$\beta_2 \pm (4.303)\sqrt{209204.44} = 2275.79 \pm 1968.14$$

$$\beta_{12} \pm (4.303)\sqrt{209204.44} = 19996.16 \pm 1968.14$$

APPENDIX B

TWO-TAILED PAIRED SAMPLE F-TEST

Two tailed paired-sample F-test was carried out to determine whether or not there is a significant difference between the aerobic and anaerobic biodegradation. The COD concentrations in the leachate from simulated aerobic and anaerobic bioreactors AER2 and AER4, and ANA2 and ANA4 at 27 weeks were used to calculate the F-test. For this test, the two-tailed variance ratio for the null hypothesis H_0: $\sigma_1^2 = \sigma_2^2$ was tested against the alternate hypothesis H_A: $\sigma_1^2 \neq \sigma_2^2$, where σ_1^2 and σ_2^2 are the variance population of aerobic and anaerobic bioreactors.

Two tailed paired-sample F-test for the COD concentration data presented in Table 4.15 is as follows:

H_0: $\sigma_1^2 = \sigma_2^2$

H_A: $\sigma_1^2 \neq \sigma_2^2$

$\alpha = 0.05$

	Aerobic		Anaerobic
AER2:	1903.75	ANA2:	46427.05
AER4:	1142.25	ANA4:	24318.93

Number of sample,	n = 2	n = 2
Variance, σ^2	$\sigma_1{}^2 = 902270.704$	$\sigma_1{}^2 = 244384446.303$
Degree of freedom	DF = 1	DF = 1

$F = \sigma_2{}^2 / \sigma_1{}^2 =$ 270.85 → F calculation

$F_{0.05\,(2),\,1,\,2,\,2} = 160$ → F critical was obtained from F-table in Zar, 1999)

F calculation > F critical

Therefore, reject the null hypothesis H_0.

APPENDIX C

SAMPLE CALCULATION OF ASCENDING AND DESCENDING RATES OF COD CONCENTRATIONS

The ascending rates (positive slopes) were calculated from the initial stage of the process to the maximum values of COD concentration and the descending rates (negative slopes) were calculated from the maximum point of COD concentration to the point just before the COD concentration tapered gradually. The COD data from the aerobic bioreactor AER1 was used as an example for this calculation.

Time (weeks)	COD concentration (mg/L)	Time (weeks)	COD concentration (mg/L)
1	1348.60	19	51869.05
3	2397.50	21	41411.58
5	3872.89	23	35764.54
7	5256.06	25	31764.56
9	14490.40	27	23823.42
11	17125.02	29	22058.72
13	22394.26	31	18750
15	25286.16	33	13500
17	47237.88	Slope (mg/L.wk)	-2565.18
19	51869.05		
Slope (mg/L.wk)	2817.22		

Ascending Rate = 2817.22 mg/L.wk = 402.46 mg/L.d

Descending Rate = 2565.18 mg/L.wk = 366.45 mg/L.d

APPENDIX D

1. CALCULATION OF WASTE SETTLEMENT

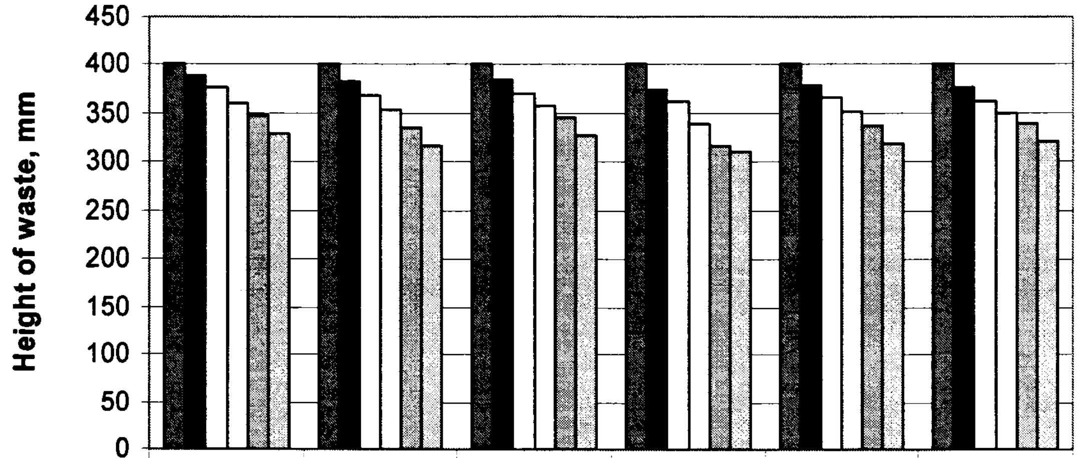
The waste settlement was used to measure the volume reduction of the compacted MSW in the bioreactors. For sample calculation, the initial and final height of the compacted MSW in the aerobic bioreactor AER1 was used to determine the settlement.

$$\text{Settlement} = \frac{\text{(Initial - Final) height}}{\text{Initial height}} \times 100\ \%$$

$$= \frac{400\ \text{mm} - 330\ \text{mm}}{400\ \text{mm}} \times 100\%$$

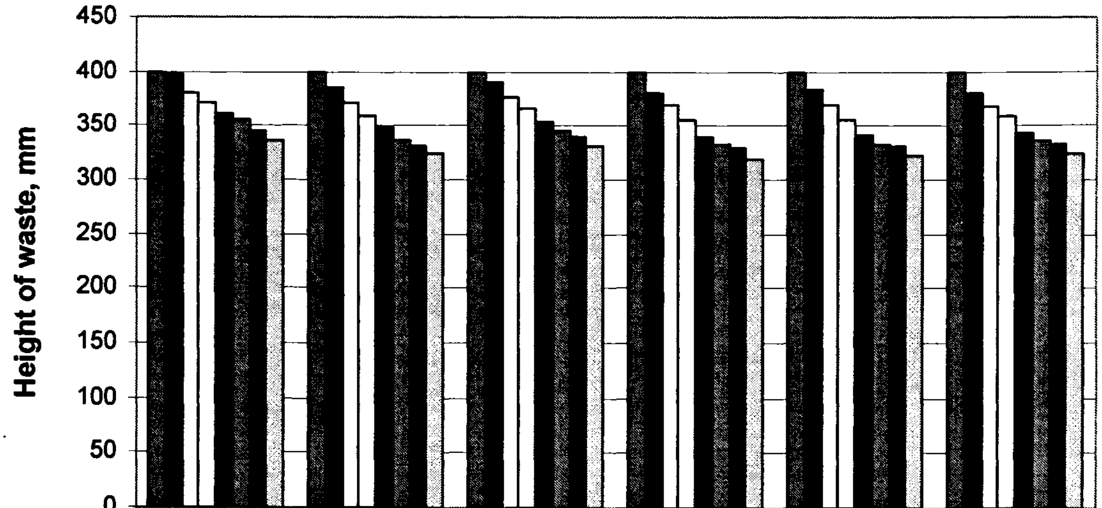
$$= 17.5\ \%$$

2. GRAPHICS OF SETTLEMENT MEASUREMENT



	AER1	AER2	AER3	AER4	AER5	AER6
Week1	400	400	400	400	400	400
Week9	389	382	384	375	379	377
Week15	377	369	371	362	366	363
Week25	360	354	358	340	351	349
Week35	348	336	346	318	337	340
Week47	330	318	328	311	320	321

SETTLEMENT FROM AEROBIC BIOREACTORS



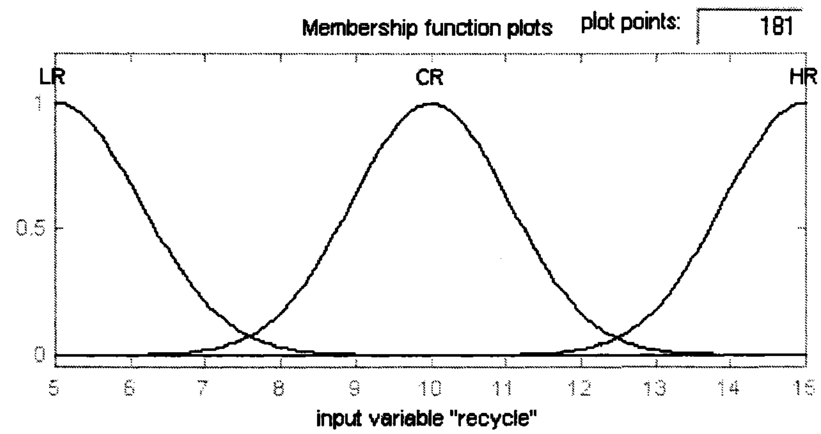
	ANA1	ANA2	ANA3	ANA4	ANA5	ANA6
Week1	400	400	400	400	400	400
Week9	398	385	390	380	383	381
Week15	381	372	376	369	370	368
Week25	372	359	367	355	355	360
Week35	361	348	354	340	342	343
Week45	356	337	346	334	334	336
Week55	345	332	340	329	331	334
Week63	336	325	332	320	323	324

SETTLEMENT FROM AEROBIC BIOREACTORS ANAEROBIC BIOREACTORS

APPENDIX E

DETERMINATION OF GAUSSIAN MEMBERSHIP FUNCTION OF INPUTS

Gaussian membership functions are used to model of MSW biodegradation in order to increase accuracy of calculation results. Fuzzy set inputs, i.e., leachate recirculation, sludge addition and airflow are presented below.



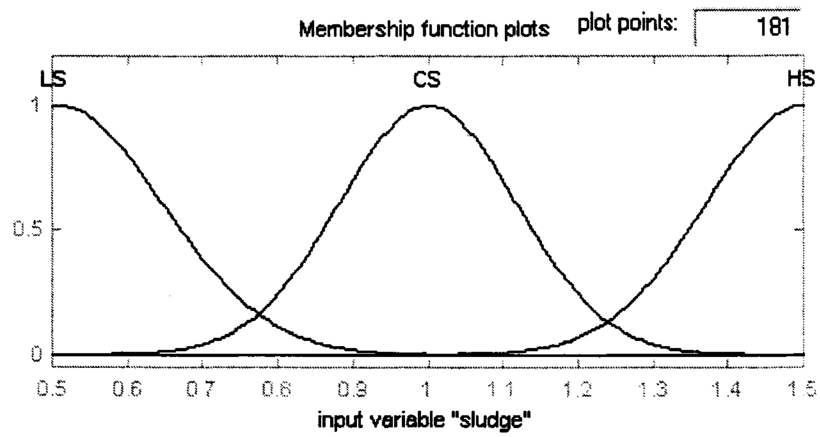
Gaussian Membership Function: Inputs of Leachate Recirculation

Where:

LR = Low rate of leachate recirculation (5 L/wk)

CR = Medium of leachate recirculation (10 L/wk)

HR = High of leachate recirculation (15 L/wk)



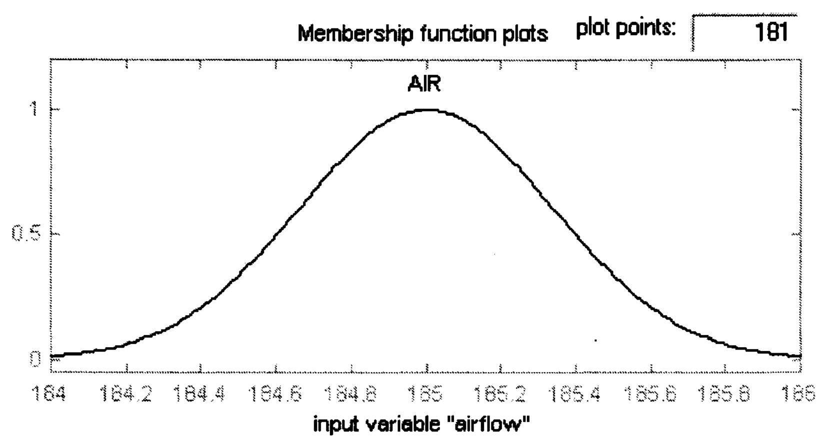
Gaussian Membership Function: Inputs of Sludge Addition

Where:

LS = Low rate of sludge addition (0.5 L/wk)

CS = Medium rate of sludge addition (1.0 L/wk)

HS = High rate of sludge addition (1.5 L/wk)



Gaussian Membership Function: Inputs of Airflow

Where:

AIR = Air flow at constant rate of 185 L air/day

APPENDIX F

RULE EDITOR FOR THE MODEL DEVELOPMENT

For the model development, the rule editors were used to construct the rule statements. Each input variable and the number of system input variables in the membership function were set-up to make statements. The rule statements were created using computer program MATLAB 5.3 and are presented as follow.

1. If (phase is SD) and (recycle is LR) and (sludge is LS) then (cod is XL)(temperature is C)(biogas is NG) (1)
2. If (phase is ED) and (recycle is LR) and (sludge is LS) then (cod is XL)(temperature is MW)(biogas is NG) (1)
3. If (phase is AD) and (recycle is LR) and (sludge is LS) then (cod is ML)(temperature is W)(biogas is GXL) (1)
4. If (phase is D) and (recycle is LR) and (sludge is LS) then (cod is H)(temperature is MHt)(biogas is GML) (1)
5. If (phase is BS) and (recycle is LR) and (sludge is LS) then (cod is XH)(temperature is Ht)(biogas is GL) (1)
6. If (phase is ES) and (recycle is LR) and (sludge is LS) then (cod is XXH)(temperature is Ht)(biogas is GMH) (1)
7. If (phase is AS) and (recycle is LR) and (sludge is LS) then (cod is MH)(temperature is VHt)(biogas is GVH) (1)
8. If (phase is S) and (recycle is LR) and (sludge is LS) then (cod is M)(temperature is VHt)(biogas is GVVH) (1)
9. If (phase is FS) and (recycle is LR) and (sludge is LS) then (cod is L)(temperature is Ht)(biogas is GVH) (1)
10. If (phase is XFS) and (recycle is LR) and (sludge is LS) then (cod is VL)(temperature is MHt)(biogas is GH) (1)
11. If (phase is SD) and (recycle is HR) and (sludge is LS) then (cod is XL)(temperature is MW)(biogas is NG) (1)
12. If (phase is ED) and (recycle is HR) and (sludge is LS) then (cod is XL)(temperature is W)(biogas is GXL) (1)
13. If (phase is AD) and (recycle is HR) and (sludge is LS) then (cod is M)(temperature is MHt)(biogas is GL) (1)
14. If (phase is D) and (recycle is HR) and (sludge is LS) then (cod is VH)(temperature is Ht)(biogas is GVVH) (1)
15. If (phase is BS) and (recycle is HR) and (sludge is LS) then (cod is ML)(temperature is Ht)(biogas is GXH) (1)
16. If (phase is ES) and (recycle is HR) and (sludge is LS) then (cod is VL)(temperature is MHt)(biogas is GVVH) (1)
17. If (phase is AS) and (recycle is HR) and (sludge is LS) then (cod is VL)(temperature is MHt)(biogas is GVH) (1)
18. If (phase is S) and (recycle is HR) and (sludge is LS) then (cod is XL)(temperature is W)(biogas is GMH) (1)
19. If (phase is FS) and (recycle is HR) and (sludge is LS) then (cod is XL)(temperature is W)(biogas is GMH) (1)
20. If (phase is XFS) and (recycle is HR) and (sludge is LS) then (cod is XL)(temperature is MW)(biogas is GL) (1)
21. If (phase is SD) and (recycle is LR) and (sludge is HS) then (cod is XL)(temperature is MW)(biogas is NG) (1)
22. If (phase is ED) and (recycle is LR) and (sludge is HS) then (cod is XL)(temperature is W)(biogas is NG) (1)
23. If (phase is AD) and (recycle is LR) and (sludge is HS) then (cod is ML)(temperature is W)(biogas is GXL) (1)
24. If (phase is D) and (recycle is LR) and (sludge is HS) then (cod is MH)(temperature is MHt)(biogas is GML) (1)
25. If (phase is BS) and (recycle is LR) and (sludge is HS) then (cod is XH)(temperature is MHt)(biogas is GVH) (1)

26. If (phase is ES) and (recycle is LR) and (sludge is HS) then (cod is XXH)(temperature is Ht)(biogas is GVVH) (1)
27. If (phase is AS) and (recycle is LR) and (sludge is HS) then (cod is M)(temperature is VHt)(biogas is GVXH) (1)
28. If (phase is S) and (recycle is LR) and (sludge is HS) then (cod is ML)(temperature is VHt)(biogas is GVVH) (1)
29. If (phase is FS) and (recycle is LR) and (sludge is HS) then (cod is L)(temperature is Ht)(biogas is GVH) (1)
30. If (phase is XFS) and (recycle is LR) and (sludge is HS) then (cod is VL)(temperature is MHt)(biogas is GH) (1)
31. If (phase is SD) and (recycle is HR) and (sludge is HS) then (cod is XL)(temperature is MW)(biogas is NG) (1)
32. If (phase is ED) and (recycle is HR) and (sludge is HS) then (cod is XL)(temperature is W)(biogas is GXL) (1)
33. If (phase is AD) and (recycle is HR) and (sludge is HS) then (cod is H)(temperature is Ht)(biogas is GH) (1)
34. If (phase is D) and (recycle is HR) and (sludge is HS) then (cod is VH)(temperature is VHt)(biogas is GVVH) (1)
35. If (phase is BS) and (recycle is HR) and (sludge is HS) then (cod is L)(temperature is Ht)(biogas is GEEH) (1)
36. If (phase is ES) and (recycle is HR) and (sludge is HS) then (cod is VL)(temperature is Ht)(biogas is GVVH) (1)
37. If (phase is AS) and (recycle is HR) and (sludge is HS) then (cod is XL)(temperature is MHt)(biogas is GH) (1)
38. If (phase is S) and (recycle is HR) and (sludge is HS) then (cod is XL)(temperature is MHt)(biogas is GMH) (1)
39. If (phase is FS) and (recycle is HR) and (sludge is HS) then (cod is XL)(temperature is W)(biogas is GMH) (1)
40. If (phase is XFS) and (recycle is HR) and (sludge is HS) then (cod is XL)(temperature is MW)(biogas is GMH) (1)
41. If (phase is SD) and (recycle is CR) and (sludge is CS) then (cod is XL)(temperature is MW)(biogas is NG) (1)
42. If (phase is ED) and (recycle is CR) and (sludge is CS) then (cod is XL)(temperature is W)(biogas is GXL) (1)
43. If (phase is AD) and (recycle is CR) and (sludge is CS) then (cod is L)(temperature is MHt)(biogas is GMH) (1)
44. If (phase is D) and (recycle is CR) and (sludge is CS) then (cod is ML)(temperature is Ht)(biogas is GVVH) (1)
45. If (phase is BS) and (recycle is CR) and (sludge is CS) then (cod is MH)(temperature is VHt)(biogas is GXH) (1)
46. If (phase is ES) and (recycle is CR) and (sludge is CS) then (cod is XH)(temperature is Ht)(biogas is GEH) (1)
47. If (phase is AS) and (recycle is CR) and (sludge is CS) then (cod is VL)(temperature is Ht)(biogas is GVXH) (1)
48. If (phase is S) and (recycle is CR) and (sludge is CS) then (cod is XL)(temperature is Ht)(biogas is GVVH) (1)
49. If (phase is FS) and (recycle is CR) and (sludge is CS) then (cod is XL)(temperature is W)(biogas is GVH) (1)
50. If (phase is XFS) and (recycle is CR) and (sludge is CS) then (cod is XL)(temperature is MW)(biogas is GMH) (1)
51. If (phase is SD) and (recycle is CR) and (sludge is CS) then (cod is XL)(temperature is MW)(biogas is NG) (1)
52. If (phase is ED) and (recycle is CR) and (sludge is CS) then (cod is XL)(temperature is W)(biogas is GXL) (1)
53. If (phase is AD) and (recycle is CR) and (sludge is CS) then (cod is L)(temperature is MHt)(biogas is GMH) (1)
54. If (phase is D) and (recycle is CR) and (sludge is CS) then (cod is ML)(temperature is Ht)(biogas is GVVH) (1)
55. If (phase is BS) and (recycle is CR) and (sludge is CS) then (cod is MH)(temperature is VHt)(biogas is GXH) (1)
56. If (phase is ES) and (recycle is CR) and (sludge is CS) then (cod is VH)(temperature is Ht)(biogas is GEH) (1)
57. If (phase is AS) and (recycle is CR) and (sludge is CS) then (cod is VL)(temperature is Ht)(biogas is GVXH) (1)
58. If (phase is S) and (recycle is CR) and (sludge is CS) then (cod is XL)(temperature is MHt)(biogas is GVVH) (1)
59. If (phase is FS) and (recycle is CR) and (sludge is CS) then (cod is XL)(temperature is MHt)(biogas is GVH) (1)
60. If (phase is XFS) and (recycle is CR) and (sludge is CS) then (cod is XL)(temperature is W)(biogas is GMH) (1)

Where:

The X-axis is phase (time) and consists of the following components:

1. SD: Start Degradation,
2. ED: Early Degradation,
3. AD: Almost Degradation,
4. D: Degradation,
5. BS: Begin Stabilization,
6. ES: Early Stabilization,
7. AS: Almost Stabilization,

8. S: Stabilization,
9. FS: Full Stabilization,
10. XFS: Extra Fully Stabilization.

Y-axis is COD concentration, temperature and biogas production.These outputs are consists of the following components.

For COD concentration:

1. XL: Extra Low
2. VL: Very Low
3. L: Low
4. ML: Medium Low
5. M: Medium
6. MH: Medium High
7. H: High
8. VH: Very High
9. XH: Extra High
10. XXH: Extra Extra High

For Temperature:

1. VC: Very Cold
2. C: Cold
3. MW: Medium Warm
4. W: Warm
5. MHt: Medium Hot
6. Ht: Hot
7. VHt: Very Hot
8. XHt: Extra Hot

For biogas production Rate:

1. NG: No Gas
2. GXL: Gas Extra Low
3. GML: Gas Medium Low
4. GL: Gas Low
5. GMH: Gas Medium High
6. GH: Gas High
7. GVH: Gas Very High
8. GVVH: Gas Very Very High
9. GVXH: Gas Very Extra High
10. GXH: Gas Extra High
11. GEH: Gas Extremely High
12. GEEH: Gas Extremely Extremely High

APPENDIX G

MEASUREMENT OF BIOGAS COMPOSITIONS FROM ANAEROBIC BIOREACTORS

Table G-1 Composition of biogas generated from anaerobic bioreactors

	ANA1			ANA2			ANA3		
Week	CH_4 (%)	CO_2 (%)	N_2 (%)	CH_4 (%)	CO_2 (%)	N_2 (%)	CH_4 (%)	CO_2 (%)	N_2 (%)
13	N/G	N/G	N/G	N/A	N/A	N/A	N/G	N/G	N/G
15	N/G	N/G	N/G	N/A	N/A	N/A	N/G	N/G	N/G
17	N/G	N/G	N/G	56.18	42.8	1.02	N/G	N/G	N/G
19	55.74	43.24	1.02	55.45	43.59	0.96	55.68	43.17	1.15
21	56.72	42.23	1.05	55.65	43.31	1.04	56.45	42.52	1.03
23	57.28	41.73	0.99	57.24	41.64	1.12	57.25	41.79	0.96
25	58.65	40.23	1.12	59.64	39.22	1.14	58.67	40.32	1.01
27	59.61	39.34	1.05	58.66	40.32	1.02	58.14	40.83	1.03
29	57.86	41.19	0.95	60.21	38.61	1.18	59.46	39.49	1.05
31	54.56	44.42	1.02	N/A	N/A	N/A	57.65	41.21	1.14
33	57.42	41.59	0.99	N/A	N/A	N/A	56.24	42.74	1.02
35	55.31	43.67	1.02	N/A	N/A	N/A	55.37	43.51	1.12
37	55.64	43.24	1.12	N/A	N/A	N/A	57.45	41.6	0.95
39	56.46	42.49	1.05	N/A	N/A	N/A	56.24	42.64	1.12
41	57.18	41.67	1.15	58.95	40.03	1.02	55.37	43.58	1.05
43	56.28	42.77	0.95	55.24	43.61	1.15	55.63	43.29	1.08
45	58.37	40.57	1.06	56.99	41.83	1.18	54.96	43.96	1.08
47	56.19	42.73	1.08	58.65	40.3	1.05	55.82	43.06	1.12
49	54.67	44.28	1.05	55.68	43.17	1.15	55.67	43.29	1.04
51	56.85	42.07	1.08	56.75	42.19	1.06	54.81	44.11	1.08
53	55.68	43.2	1.12	57.24	41.8	0.96	56.97	41.9	1.13
55	54.26	44.8	0.94	59.84	39.14	1.02	57.53	41.33	1.14
57	59.65	39.2	1.15	58.74	40.24	1.02	59.65	39.27	1.08
59	55.86	43.08	1.06	56.75	42.21	1.04	58.61	40.31	1.08
61	N/A	N/A	N/A	59.65	39.23	1.12	N/A	N/A	N/A
63	N/A	N/A	N/A	54.65	44.32	1.03	N/A	N/A	N/A

Note:
All compositions are percent by volume
N/G: no biogas generated
N/A: not applicable due to the machine was broken.

Table G-1 Composition of biogas generated from anaerobic bioreactors (cont.)

	ANA4			ANA5			ANA6		
Week	CH_4 (%)	CO_2 (%)	N_2 (%)	CH_4 (%)	CO_2 (%)	N_2 (%)	CH_4 (%)	CO_2 (%)	N_2 (%)
13	N/A	N/A	N/A	55.86	43.06	1.08	N/A	N/A	N/A
15	N/A	N/A	N/A	55.61	43.34	1.05	N/A	N/A	N/A
17	56.75	42.23	1.02	57.83	41.05	1.12	55.82	43.13	1.05
19	58.67	40.26	1.07	58.64	40.23	1.13	57.83	41.03	1.14
21	60.02	38.95	1.03	56.67	42.29	1.04	56.64	42.3	1.06
23	57.45	41.49	1.06	57.45	41.43	1.12	57.95	40.93	1.12
25	59.94	38.98	1.08	59.34	39.62	1.04	58.42	40.55	1.03
27	56.71	42.25	1.04	56.47	42.57	0.96	59.34	39.6	1.06
29	57.32	41.56	1.12	60.23	38.74	1.03	60.34	38.71	0.95
31	N/A	N/A	N/A	58.62	40.31	1.07	N/A	N/A	N/A
33	N/A	N/A	N/A	57.25	41.63	1.12	N/A	N/A	N/A
35	N/A	N/A	N/A	56.46	42.49	1.05	N/A	N/A	N/A
37	N/A	N/A	N/A	55.84	43.01	1.15	N/A	N/A	N/A
39	N/A	N/A	N/A	56.27	42.65	1.08	N/A	N/A	N/A
41	57.75	41.2	1.05	57.36	41.5	1.14	56.13	42.82	1.05
43	58.67	40.18	1.15	56.43	42.52	1.05	57.46	41.52	1.02
45	55.95	43.1	0.95	55.28	43.76	0.96	57.25	41.77	0.98
47	57.24	41.74	1.02	55.25	43.73	1.02	56.38	42.6	1.02
49	57.99	40.88	1.13	57.65	41.23	1.12	56.67	42.19	1.14
51	56.48	42.4	1.12	60.28	38.6	1.12	58.62	40.3	1.08
53	58.95	40.04	1.01	56.99	41.93	1.08	56.24	42.73	1.03
55	60.65	38.31	1.04	55.96	43.01	1.03	54.75	44.11	1.14
57	56.47	42.45	1.08	54.68	44.18	1.14	56.67	42.21	1.12
59	57.23	41.61	1.16	57.26	41.71	1.03	57.81	41.11	1.08
61	55.67	43.31	1.02	N/A	N/A	N/A	56.53	42.42	1.05
63	58.64	40.24	1.12	N/A	N/A	N/A	55.68	43.26	1.06

Note:

All compositions are percent by volume

N/G: no biogas generated

N/A: not applicable due to the machine was broken.