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EFFECT OF AERATION ON FRESH AND AGED MUNICIPAL SOLID WASTE

by

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EFFECT OF AERATION ON FRESH AND AGED MUNICIPAL SOLID WASTE

Graham Jon Takata

Master of Environmental Applied Science and Management

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ABSTRACT

Under the anaerobic conditions of conventional sanitary landfill, entombed municipal solid waste (MSW) is slow to stabilize necessitating long-term monitoring and pollution control. Although anaerobic conditions can provide revenue through energy generation, aerobic stabilization may offer several advantages including reduced fugitive greenhouse gas emissions, accelerated landfill stabilization, and increased landfill airspace recovery.

Air injection was applied to bench-scale bioreactor landfills in order to determine the potential for active aeration to accelerate municipal solid waste stabilization and settlement in both new and pre-existing landfills. Fresh and aged wastes were used to represent newly constructed and existing landfill matrices over 130 days.

In the fresh MSW bioreactors, aeration reduced the time to stabilization of leachate pH by 44%, TSS by 25%, TDS by 54%, BOD_5 by 38% and COD by 59%. Ammonia concentrations stabilized after 129 days of aeration, but remained problematic in the anaerobic bioreactors at the study conclusion. Final leachate concentrations were consistently lower in the aerobic bioreactors than in their anaerobic counterparts. Physical settlement also improved, resulting in a 21.5% recovery of landfill airspace in the aerobic fresh waste bioreactors. Aeration had a similar but reduced influence in the aged waste bioreactors since they were near stabilization at the study inception.

The results of this study indicate that aeration significantly accelerates stabilization of MSW with greatest influence on fresh waste with a high biodegradable organic fraction.

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1. INTRODUCTION

1.1 BACKGROUND

1.1.1 Generation and Disposal of Municipal Solid Waste

Increasingly stringent regulations and design requirements have raised the costs of landfill siting and facility operation. Additionally, increased public opposition to landfill siting has compounded the issue by frequently impeding the construction of new waste disposal facilities (Stessel and Murphy 1992). In 1988, 7924 landfills were accepting waste in the United States. By 1999, only 2216 were in operation, requiring that new facilities be larger than previous landfills to accompany the same volume of waste generation (USEPA 1999). Over the same period, the total production of Municipal Solid Waste (MSW) in North America has increased annually. In 1997, 217,000,000 tons of MSW was produced in the U.S. alone, 8,000,000 tons more than the previous year (USEPA 1998). In 1994, Canadians disposed of 715 kg of solid waste per capita, totaling approximately 22,000,000 tons nationwide (CCME 2002). As existing sites reach their final capacity, and plans for new sites are obstructed by NIMBY mind-sets, disposal of MSW will become increasingly problematic.

Moreover, sanitary landfills have operational, liability, and environmental costs. Landfilled municipal solid wastes are slow to stabilize, generate leachate, and emit large volumes of greenhouse gases. A landfills potential for environmental impact endures far beyond the landfill's usable lifetime, and can be a source of pollution for several decades or even a century after closure. Heyer and Stegmann (1997) estimate that landfill leachate can take up to 300 years to reach harmless levels based on field-scale studies. As a result, long term monitoring and environmental control systems are required to prevent groundwater contamination and mitigate gas emissions. Furthermore, the financial burden of post-closure maintenance and pollution control must be carried while the landfill no longer generates revenue from waste disposal (Sullivan and Stege 2000).

1.1.2 Landfill Leachate

Despite landfill cover, final capping, and other techniques implemented in conventional "dry tomb" landfills to dissuade the infiltration of rainwater, water from rainfall eventually permeates through the landfill cover and percolates through the waste media. The result is a sometimes highly contaminated liquid containing organic material and various dissolved constituents called leachate. The chemical composition of leachate is a function of the contents of the landfill, time, site hydrology and the rate of infiltration. As such, concentrations of chemical constituents frequently vary in orders of magnitude such that typical concentrations cannot be reported with any assurance (Barlaz and Ham 1993). Commonly, a high BOD_5, COD, suspended solids content and ammonia concentration are characteristic of a landfill leachate, with dissolved metals varying with the pH (Barlaz and Ham 1993). Conditions in a conventional landfill are anaerobic, with oxygen only available to the top 1m. Under these conditions, acetogenic bacteria and fermentive organisms produce alcohols, carboxylic and acetic acids. Accumulation of these byproducts in the leachate results in an acidic leachate pH. Acidification of the leachate in this manner also helps to dissolve metals, which may require specific treatment to remove.

1.1.3 Landfill Gas

Gases generated by the anaerobic degradation of the biodegradable organic fraction (BDOF) of municipal solid waste (MSW) by acetogenic, fermentive, and methanogenic bacteria are also problematic. Landfill gas (LFG) is predominantly composed of carbon dioxide and methane, with low concentrations of VOCs and sulphides. Typically, LFG can contain as much as 40% to 60% methane by volume making it the largest single source of anthropogenic methane in Canada (OCETA 1998), where between 1.0 and 1.2 megatonnes of methane are produced annually from nearly 10,000 active or closed landfills (Neitzert et al. 1999). As such, landfill methane accounts for approximately 23% of the total methane emissions in Canada, of which only approximately one quarter is captured by LFG collection systems (Natural Resources Canada 2000).

Methane gas has a 100 year global-warming potential 21 times greater than carbon dioxide (WMO/UNEP 1996). The annual methane emissions from landfills in Canada account for 21 megatonnes CO_2 eq with respect to global warming protocols (WMO/UNEP 1996). Comparatively, it would take over 6 million cars, or 40% of all of the passenger vehicles in the country, to match the global warming potential of landfill emissions (Environment Canada 1999).

In addition, the annual emissions of methane per ton of deposited MSW are only a fraction of its total methane potential. Overall, each metric ton of MSW has the potential to release 50-110m^3 of carbon dioxide and 90-140 m^3 of methane over the duration of its entombment (Vieitez and Ghosh 1999), such that emissions continue for decades after burial. Consequently, landfill methane emissions are expected to increase annually due to the cumulative methane-producing potential of old and newly emplaced MSW. Natural Resources Canada (2000) projects that by 2020, landfill methane emissions will have increased by 15%. For dry entombment, post-closure gas collection could require 45 to 60 years (Sullivan 2000).

1.2 PROBLEM STATEMENT

Two major problems with our current waste disposal practices have become evident that place us on the verge of a waste management crisis. First, our generation of municipal solid waste is increasing annually, while our present landfill capacity and ability to dispose of it diminishes. Secondly, the pollution potential of existing landfills can easily extend decades after closure, requiring pollution control systems, monitoring, and maintenance long after they have exhausted their ability for waste acceptance.

As a result, there is considerable interest in landfill management techniques that can prolong the usable lifetime of existing landfills and reduce the duration of post-closure monitoring.

One design concept that addresses both is the bioreactor landfill.

1.3 LANDFILL BIOREACTORS

1.3.1 Definition of a Bioreactor

A bioreactor is an enclosed system that supports organisms used for their capacity to convert substrates through biological processes. In the pharmaceutical and biotechnological industries, they are for the production of drugs, antibodies, and vaccines. In waste management, the bioreactor concept can be applied to landfills in order to enhance the bioconversion of organic waste. Bioreactor landfills reduce the strength of landfill leachate (Townsend et al. 1996), increase the capacity of a landfill (Wall et al. 1995; Warith et al. 2001), decrease the time to landfill stabilization (Pacey et al. 1999), and allow for more rapid reclamation of the site (Pacey 2001).

The most prominent and common bioreactor enhancement is the controlled addition of moisture to the system. Though there are no specific elements of design that are integral to the definition of a bioreactor, bioreactors are typically maintained with high moisture content and include leachate recirculation systems to ensure an even distribution of moisture within the landfill. The bioreactor concept also incorporates other research developments that are not commonly reflected in current sanitary landfill practices.

Studies have shown that the addition and circulation of nutrients, buffer solutions, and sludges can enhance MSW biodegradation (Warith et al. 1999). Other practices include implementing waste shredding, reduced compaction densities, modified lift designs, and temperature controls. Typically, bioreactor designs will incorporate a number of these enhancements in addition to leachate recirculation.

Presently, most bioreactors operate under anaerobic conditions. However, aerobic and semi-aerobic bioreactors have recently drawn attention for their ability to even further accelerate the biodegradation of MSW. For the purposes of this paper, a bioreactor incorporates leachate recirculation with an elevated moisture content, and aerobic and anaerobic bioreactors will be discussed.

1.3.2 Comparison of Conventional Landfills and Bioreactors

Decomposition in conventional sanitary landfills occurs very slowly, due to a design allowing for minimal infiltration of oxygen and moisture by encapsulating and storing of waste in cells or lifts. While encapsulation of MSW has managed to minimize leachate production, such "dry tomb" practices are synonymous with long post-closure monitoring and potential for environmental impact. In essence, while these sites attempt to reduce the risk of groundwater contamination by restricting leachate production, they instead prolong the risk of contamination. Recently, growing concerns with our present landfill practices have resulted in a fundamental shift in landfill management philosophy, from the more passive leachate minimization associated with conventional landfill operation towards approaches that actively promote in-ground microbial activity.

Biodegradation rates in landfill bioreactors are aggressive in comparison to conventional dry tomb landfills. Where a conventional landfill will typically only attain partial stabilization of its organic constituents in 15 to 80 years, an anaerobic bioreactor can achieve nearly complete stabilization of the BDOF in only 5 to 10 years, and aerobic bioreactors may only require a few years to stabilize (Pacey 2001).

Stabilization is achieved when the MSW has undergone sufficient biodegradation, such that the remaining BDOF does not give rise to increased leachate constituent concentrations or gas production levels over time. Instead, the quality of the leachate and gas evolution remains constant.

Differences in gas production trends are also significant. The potential for landfill gas production is never eliminated in dry entombment landfills, which generate methane slowly long after closure. In contrast, an anaerobic bioreactor limits methane generation to an anticipated 10 to 15 years post-closure, with a greater peak production and recovery potential within this period. This allows for greater and more efficient waste-to-energy conversion and a reduced duration and expense of landfill gas control. In aerobic bioreactors, methane production and the associated odour and hazards are minimal. While methane collection from both conventional landfills and anaerobic bioreactors has

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often been thought of as a lucrative method to offset maintenance costs; a reduced duration for monitoring, liability, and pollution risk has generated renewed interest in aerobic bioreactor technology.

Rapid biodegradation and stabilization in bioreactors also results in rapid settlement and recovery of airspace. This in turn allows for a greater waste acceptance capacity, especially in aerobic bioreactors (Sullivan 2000).

1.3.3 Benefits of Bioreactors

In comparison to conventional landfills, the benefits of implementing anaerobic or aerobic bioreactor technologies are numerous. Bioreactors:

1. Increase the final acceptance capacity of a landfill and prolonging its operational lifetime.
2. Reduce the long-term financial risk associated with long-term liability.
3. Reduce the long-term requirement for maintenance, operational, and gas-collection responsibilities.
4. Reduce the risk of future containment breaches.
5. Reduce requirement for leachate treatment during operation.
6. Enhance the production and capture of LFG in anaerobic bioreactors.
7. Reduce the long-term emission potential of a landfill by transferring it to a state of low biological activity in an accelerated timeframe.
8. Provide a potential for future landfill mining of recyclables and humic materials for use as landfill cover from aerobic bioreactors.
9. Allow for earlier site reclamation.

1.4 OBJECTIVE OF THIS STUDY

The objective of this study is twofold:
1. To examine the potential of air injection to further accelerate MSW stabilization in newly constructed bioreactor landfills, compared to anaerobic bioreactor designs.
2. To determine if implementation of aerobic bioreactor technology into existing closed landfill sites provides comparable benefits.

These objectives will be achieved by examining fresh and aged municipal solid waste used to represent new and retrofitted bioreactor landfills, and a comparison of the extent and time to stabilization of MSW will be made based on leachate quality for the tests bioreactors under both aerobic and anaerobic conditions. The leachate pH, TSS, TDS, Ammonia, BOD_5, COD and BCR will be determined to be stabilized when the respective concentrations for each analytical parameter remain near constant over time. The degree of MSW settlement will also be examined.

It is hoped that this study will demonstrate the potential for aerobic bioreactor technology in solid waste management, and aid in the decision-making for the viability of its implementation into new or retrofitted landfills.

1.5 HYPOTHESIS

Active aeration will improve the extent of MSW biodegradation and reduce the time to stabilization in both fresh and aged MSW, as determined by the leachate pH, TSS, TDS, Ammonia concentration, BOD_5, COD and BOD/COD Ratio. The influence of aeration will be more pronounced in the fresh MSW, which has a greater biodegradable organic fraction.

2. BACKGROUND AND LITERATURE REVIEW

The following background and literature review will discuss the stages of biodegradation in both aerobic and anaerobic landfill lifetimes and present the state of knowledge with regards to leachate recirculation and air injection.

2.1 PHASES IN LANDFILL BIODEGRADATION

Solid waste degradation in sanitary landfills proceeds sequentially through a number of phases, denoted by the microbiological activity and the byproduct concentrations within the landfill media. The number of these phases, as well as their inclusion criteria, differs from investigator to investigator based on the particular research interest (Warith and Sharma 1998; Barlaz and Ham 1993; Barlaz et al. 1989; Pohland and Harper 1986; Pohland 1980). A four-phase model of MSW refuse decomposition in an anaerobic landfill are described and characterized here and illustrated in Figure 1. The phases are:

1. Aerobic Phase
2. Anaerobic Acid Phase
3. Accelerated Methane Production Phase
4. Decelerated Methane Production Phase

The initial aerobic phase in conventional landfills and anaerobic bioreactors is a short-lived stage where oxygen provided during the emplacement of MSW is consumed. This phase, which lasts in the order of days, is restricted to the topmost lift of a landfill. Sugars provide the main source of carbon during aerobic decomposition, and carbon dioxide is generated. Aerobic biodegradation progresses at a rate greater than anaerobic decomposition, and is capable of breaking down more difficult organics. When methane production is desired, which requires anaerobic conditions, this phase is not promoted.

Leachate from this stage will have a high organic content as a result of primary compaction of the waste and resultant liquid production. However, leachate collected

from the bottom of the landfill during this phase will typically reflect conditions of the MSW beneath it, having passed through solid waste undergoing anaerobic biodegradation.

The aerobic conditions rapidly give way to a state of oxygen depletion. In the anaerobic acid phase, facultative anaerobes convert insoluble biological polymers such as cellulose, hemicellulose, proteins and fats into soluble sugars, amino acids, long-chain fatty acids and glycerol (Burton and Watson-Craik 1998). These byproducts are subsequently hydrolysed and fermented into fatty acids, carboxylic acids, alcohols, hydrogen and carbon dioxide resulting in a decrease in the leachate pH (Warith and Sharma 1998). In addition, it is likely that a wide variety of complex intermediates and by-products are also produced, which may be converted into acetic acid by acetogenic bacteria. Fermentative and acetogenic processes generate high levels of carbon dioxide, which slowly gives way to methane production. This transition may take months to years without buffering, moisture attenuation or leachate recycling, due to inhibitory pH levels. Large amounts of metals may dissolve into the leachate, and ammonia levels may increase due to hydrolysis of proteinaceous matter (Doedens and Cord-Landwehr 1989).

In the accelerated methane production phase, increases in the population of methanogenic bacteria bring on a rapid increase in methane production. Concentrations of 50-70% are typical, with the remainder being mostly carbon dioxide. The concentration of carboxylic acids present in the landfill leachate decreases from increased consumption, with a subsequent rise in pH observed. Little hydrolysis of the MSW solids is observed, as the methanogenic bacteria utilize dissolved organics in the leachate.
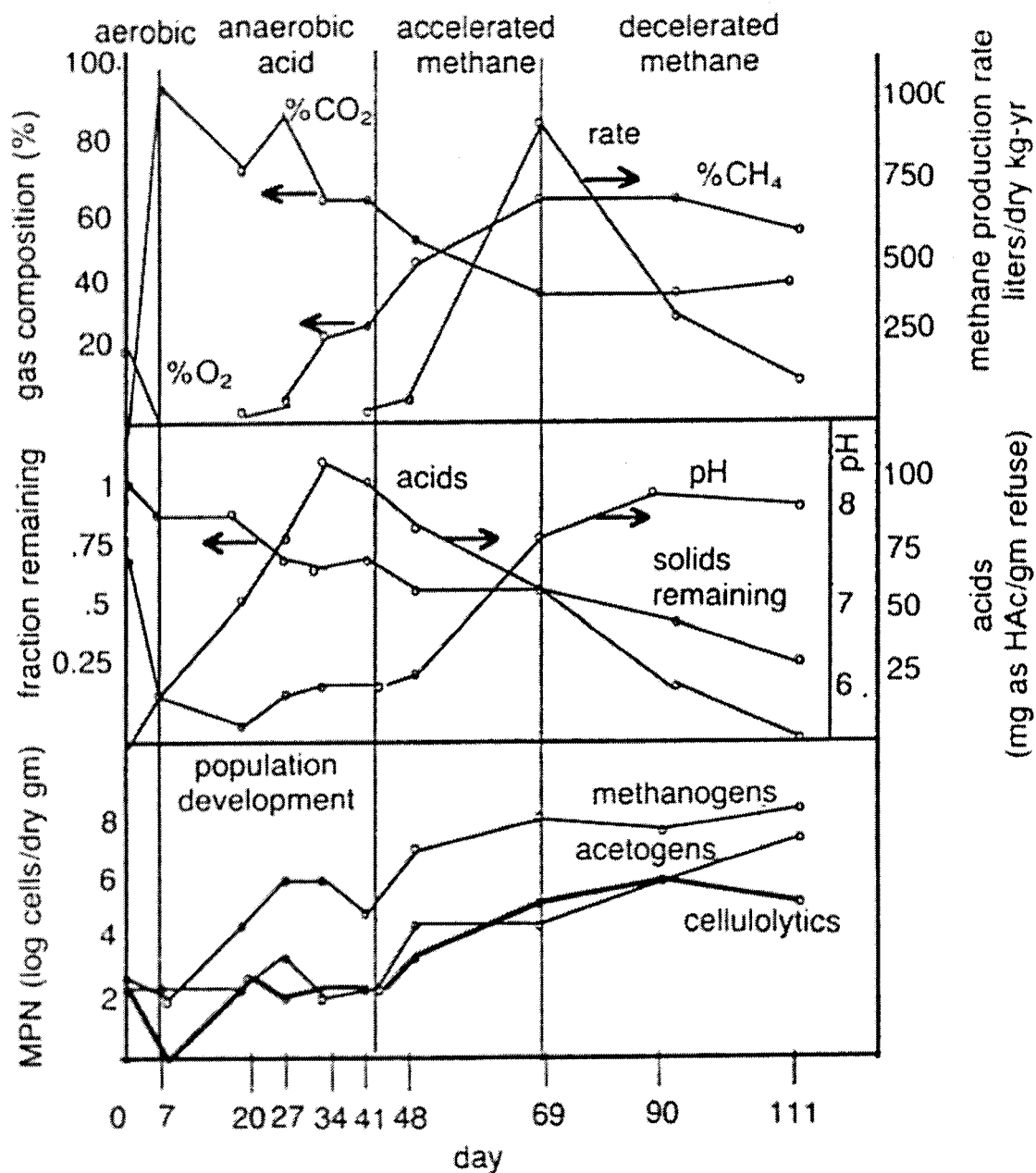
Figure 1. Observed Trends in Anaerobic Biodegradation in a Leachate Recycle Bioreactor: Four Phases of Refuse Decomposition. (Barlaz et al. 1989, with permission).

The leachate pH in the decelerated methane production phase continues to increase, due to continued utilization of carboxylic acids by methanogenic bacteria. The concentration of carboxylic acids in the leachate become depleted, typically to concentrations below 100mg/L. Depletion of the accumulated organic acids results in a decrease in the rate of methane production, which becomes limited by the rate of solids hydrolysis. Because the readily degradable portion of the solid substrate has already been significantly hydrolyzed, the remaining solid waste is composed of less degradable compounds such as humic materials, including lignin.

This phase, which may last for decades, is symptomatic of ongoing pollution control requirements and low methane production at rates. Waste-to-energy conversion is uneconomical, yet methane capture and flaring are still required for emissions control.

2.1.1 Stages of Degradation in Anaerobic Bioreactors

In an anaerobic bioreactor, the landfill would also undergo the above stages, with a few notable differences. Considerable reduction in the duration of each degradation phase should be evident, with the onset of methanogenesis occurring earlier than in conventional landfills. The leachate pH would also acidify and neutralize more rapidly. Methane production would also occur at a more rapid rate, achieving a higher peak production within the methanogenic period. This would result in a more rapid utilization of the BDOF by the microbial community, such that methane production in the final deceleration phase would be lower and the pollution potential of the site would be reduced in the post-closure period. This is discussed in greater detail in Subsection 2.3.1.

2.1.2 Stages of Degradation in Aerobic Bioreactors

In an aerobic bioreactor, an even greater acceleration of the MSW stabilization is anticipated. Though there has not been an aerobic bioreactor in use long enough to detail the complete lifetime of an aerobic landfill, preliminary results are promising. In an aerobic bioreactor, the four stages of refuse decomposition would be replaced by an ongoing aerobic biodegradation phase. Methane production is suppressed, which eliminates methane capture for energetic purposes. The benefits of methane suppression are further discussed in Subsection 2.4.2, *Infrastructural Benefits of Air Injection*. In two

aerobic bioreactors described by Read et al. (2001), air injection was shown to reduce the pollution potential of landfill gas significantly. Methane production was reduced by 50 to 99% with an inversely proportional rise in carbon dioxide production. The strength of landfill gas odour was also noticeably reduced.

The leachate BOD_5 and dissolved volatile organic compound concentrations were also improved, and the overall volume of leachate requiring treatment was also lessened (Read et al. 2001). Furthermore, acidification of the leachate would not be expected, since aerobic biodegradation does not produce excessive amounts of alcohols and acids associated with fermentation processes. Ammonia concentrations are also reduced (Leikam et al. 1999).

2.2 COMPOSITIONAL VARIATIONS IN MSW OVER TIME

The composition of MSW has varied significantly over time, such that MSW disposed today is compositionally different from MSW as it was discarded in previous decades. This is attributed to increased waste sorting and recycling initiatives, which have diverted significant amounts of discarded plastics, metals, and glass. As a result, landfills are more biodegradable than they were 30 years ago (OCETA 1998). Comparing the fractions of MSW discarded in the United States from 1980 to 1997, the glass fraction has been reduced from 10.0% to 5.5% and the total metals have dropped from 10.2% to 7.7% by mass. Though diversion of plastics has increased since 1980, the total mass of plastics discarded increased to a greater extent such that the plastics fraction of MSW has risen from 4.5% in 1980 to 9.9% in 1997. This is attributed to increased usage of plastics in the United States (USEPA 1998).

Once entombed, the percent composition of the MSW shifts towards relatively inert items such as metals, glass, and plastics with the ensuing biodegradation of the biodegradable organic fraction, namely food waste, yard waste, and paper refuse. A significantly aged MSW sample may contain far less organic material, with only the recalcitrant organics

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remaining. However, this depends not only on the length of entombment, but the degree to which biodegradation was promoted or discouraged.

As a result, we must acknowledge that implementing the bioreactor concept to present landfills and construction of new landfills will involve considerably different MSW both chemically and compositionally. Additionally, results from studies on fresh MSW are limited in their applicability towards aged MSW in pre-existing landfills.

2.3 LEACHATE RECIRCULATION

2.3.1 Benefits of Leachate Recirculation

Conventional sanitary landfills based on the "dry tomb" concept are engineered to minimize the infiltration of water into the landfill by providing daily cover and encapsulating waste into cells. This practice of strict moisture control is motivated by the principle that by restricting the infiltration of water, the production of leachate is also reduced; subsequently reducing leachate treatment costs. Unfortunately, low moisture levels within the landfill result in low biological activity, leading to slow biodegradation and a prolonged pollution potential.

Contrary to conventional designs, incorporation of a leachate recirculation system encourages the permeation and accumulation of moisture within a landfill. In doing so, several advantages over conventional designs become prevalent:

2.3.1.1 Distribution and Dilution effects

Leachate recirculation is not simply the re-distribution of moisture, but the accompanying inoculate, organic matter, and nutrients as well. With leachate recirculation, subsurface zones that are nutrient deficient, lack the appropriate microbiological community, or simply lack adequate moisture become more susceptible to biodegradation (Warith and Sharma 1998). Furthermore, regions with high concentrations of inhibitory compounds or amassed toxic byproducts become diluted, such as in studies by Stegmann and

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Splendin (1989) where recirculation reduced the leachate pH and improved microbial activity. Spatial variations in landfill temperature are also reduced, as the leachate recirculation moderates the internal temperature to a more uniform state through heat transfer. Undesirably high operating temperatures from accelerated microbial activity are prevented in the bioreactor. In summer months, evapotranspiration can result in a significant reduction in the recirculation volume (Robinson and Barr 1999).

2.3.1.2 Benefits of Moisture Augmentation

Typically, leachate recycle systems operate with elevated moisture contents in the bioreactor. Leachate recycle systems obtain additional moisture by encouraging infiltration of precipitation in field-scale studies, and from the controlled addition of water in bench-scale models. In practice, precipitation and controlled water additions provide additional moisture. In aerobic bioreactors, maintaining a high moisture content is necessary. Studies with composting systems have shown that aerobic biodegradation can produce very high temperatures, well into the thermophilic range ($40 - 65°C$), which can ultimately inhibit the breakdown of MSW by microorganisms (Sesay et al. 1998). In one landfill study where aeration was combined with an insufficient recirculation volume, a fire occurred (Mertz and Stone 1970).

Implementation of leachate recycling with elevated moisture contents has also demonstrated increased MSW biodegradation and accelerated reduction of the organic load in several studies (Warith et al. 2001; Chugh et al. 1998; Townsend et al. 1996). Leachate stabilization is also accelerated, which decreases the duration of leachate treatment and post-closure monitoring (Townsend et al. 1996). However, ammonification can be accelerated under leachate recirculation regimes, and can result in toxic ammonia levels and inhibit degradation processes (Burton and Watson-Craik 1998). Indeed, the principal route for the removal of nitrogen is by ammonification and solubilization in the leachate, which then requires external treatment. With no apparent anaerobic oxidative pathway to nitrogen gas, Burton and Watson-Craik (1998) suggest collecting and aerating leachate in lagoons prior to recirculation. Methanogenesis is also

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promoted under moist conditions. (See Subsection 2.3.2, *Infrastructural Benefits of Leachate Recirculation*).

2.3.2 Infrastructural Benefits of Leachate Recirculation

2.3.2.1 Accelerated subsidence

Accelerated biodegradation of the organic fraction of MSW improves the subsidence of municipal solid wastes, allowing valuable airspace to be recovered and prolonging the landfill's usable lifetime. Though Wall and Zeiss (1995) observed no significant difference in MSW settlement of sanitary landfills and anaerobic bioreactors in the initial year of implementation, extrapolation of their results indicates that differences would become significant in the long term. Warith et al. (2001) better demonstrated this; where a recovery of 30% of the landfill airspace over 8 years resulted due to leachate recycle. Accelerated biodegradation also allows for a more uniform and predictable landfill settlement. Due to prior biodegradation and settlement, less cracking in the final cover is expected after final closure (Chugh et al. 1998).

2.3.2.2 Enhancement of Methanogenesis

In comparison to moisture-starved landfills, increasing the moisture content reduces the lag time to the onset of methane production in anaerobic bioreactors. In a study by Chugh et al. (1998), recirculation of 30% w/w achieved a peak methane production of approximately $1.5m^3$ of methane per day from a 200L bioreactor in only 21 days. For the test bioreactor with only 2% recycle, 52 days were required. Similarly, in a study by Barlaz et al. (1989), increasing the moisture content from 45.2% to 60% resulted in methane production after 6 months, as opposed to 2 years. An increase in the rate of methane production was also observed. Rees (1980) experienced an exponential increase in methane production by raising the moisture content from 25% to 60%.

A more rapid onset of methanogenesis and increased methane production results in a more rapid consumption of the MSW biodegradable organic fraction. After a significant portion of the biodegradable materials has been consumed, the rate of methane

production in a bioreactor landfill declines below rates typical of conventional landfills (Barlaz and Ham 1993). Sullivan and Stege (2000) predict that significant LFG production in a bioreactor landfill will be limited to 15 to 20 years after site closure. This is also beneficial since post-closure methane production is commonly insufficient for profitable gas collection, and contributes methane to the atmosphere (Vieitez and Ghosh 1998). Incorporating infrastructural costs, a cost-benefit analysis by Clarke (2000) suggested anaerobic bioreactor landfills are more profitable than dry-tomb landfills in Australia. At the time of this research, a comparable economic analysis for aerobic bioreactors has not been published.

2.3.3 Design Variables for Leachate Recirculation

Leachate-recycling scenarios have been performed on waste samples in test bioreactors (Chugh et al. 1998; Warith et al. 1999), lysimeters (Stessel et al. 1992; Robinson et al. 1982), and field studies (Warith et al. 2001; Townsend et al. 1996). A description of design optimizations and relevant findings follows.

Three different operational parameters are considered in the design of leachate recirculation systems: moisture content, leachate recirculation volume, and recirculation regime.

On large scale sites where minimal landfill cover is used, moisture addition usually comes from rainfall, such as in studies by Townsend et al. (1996) and Deusto et al. (1998). Similarly, studies with lysimeters have also received moisture through precipitation. In more controlled studies, moisture contents were maintained and leachate recirculation volumes were standardized. Though there is no clear optimal moisture content for biodegradation, moisture contents set at field capacity are common. Field capacity is defined as the maximum amount of moisture a soil can hold with negligible losses after 24 hours of free drainage. When applied to MSW, field capacity can range from 30% to 80%, but are usually around 60% w/w. A moisture content of 20% is considered low (Barlaz and Ham 1993). Increasing the moisture content beyond field capacity did not significantly improve or impede the leachate quality (Buivid et al. 1981).

In past studies, the volume of leachate in relation to the amount of MSW varies considerably. For field scale tests, the leachate recirculation volume can vary seasonally, unless a leachate collection pond is used. Chugh et al. (1998) recirculated leachate recirculation volumes of 2%, 10%, and 30% of the MSW volume weekly. While recirculating 2% showed gradual improvement in the leachate pH, COD, and methane yield, recirculation of 10% did so much more rapidly. Methane production was even greater for the 30% recirculation; however, there was no significant difference in the rate of pH neutralization. Larger recirculation volumes also enhance the flushing of inhibitory compounds, either originally present or generated through microbial activity, such as volatile organic acids. Chugh et al. (1998) suggest that both low volume recirculation and high volume recirculation have applications in bioreactor technology, as low rates allow the adhesion and growth of beneficial bacteria, while higher rates wash away inhibitory compounds and allow for greater dissemination of inoculate, buffers, nutrients, and organic matter.

2.3.4 Other Results

It should be noted that there is some disagreement among researchers as to the benefits which leachate recirculation can be attributed to exclusively. While the physical benefits of leachate recirculation such as heat regulation, moisture and nutrient distribution are largely unchallenged, the impact on leachate quality and methane production have occasionally yielded contradictory results. While studies by Leckie et al. (1979) and Vieitez and Ghosh (1999) indicate a positive effect on methane production due to leachate recycling, Barlaz et al. (1989) and Leuschner (1989) did not experience increased methane production from leachate recirculation. It has been suggested that the pH was too low in these cases. Additionally, the length of the acetogenic phase preceding methanogenesis has presented opposing results (Komilis et al. 1999). Robinson and Barr (1999) consider leachate recirculation to be only a partial solution on its own, since ammonia, chloride, and metals require other means of treatment.

Never the less, it is widely accepted that the combination of leachate recirculation with other technologies has a positive influence on biodegradation of municipal solid wastes (Doedens and Cord-Landwehr 1989). In addition to moisture augmentation, the addition of nutrients, buffer solutions, and sludge to the recirculation process has demonstrated positive results (Warith et al. 1999). Experiments have also used other additives. Powdered activated carbon was added to leachate in order to reduce fluctuations in landfill COD concentrations, and polyethylenimine was added to increase biomass activity by Pouliot et al. (2000) with a small influence on the rate of COD utilization.

2.4 AIR INJECTION

2.4.1 Benefits of Air Injection

While anaerobic digesters have been well researched, active aeration for the treatment of landfilled municipal solid wastes has been performed in only a few studies. Instead, studies involving aerobic bioreactors are typically associated with composting or aerated static piles and not landfill attenuation (Murphy et al. 1995). However, application of this technology to landfill bioreactors provides benefits including the rate and extent of biodegradation, on a site management, site capacity and lifetime. As a result, aerobic in-ground digesters are an emerging technology. Air is most commonly delivered to the MSW through positive pressure injection, where vertical sparges force air down into the landfill strata. Achieving an adequate airflow rate using this approach in a full-scale landfill should not present technical problems, as air requirements are low (Leikam et al. 1999).

Other techniques of aeration have also been studied, such as suck and blow (Sesay et al. 1998), the Fukuoka method (Read et al. 2001a,b), and leachate aeration (Stessel and Murphy 1992). The "suck and blow" method alternates aeration between positive and negative air pressures. The benefit of this practice, as shown by Sesay et al. (1998) is a more even temperature distribution throughout the composting media as compared with aerobic static piles, reducing the risk of localized inhibitory temperatures. In the Fukuoka method, air is carried into the landfill through the headspace of the leachate

recirculation system. The leachate recirculation system, which is open to air, draws air into and out of the landfill based on the temperature differences in the landfill, creating a 'chimney' effect (Read et al. 2001a,b). Using leachate aeration in place of air sparging or other more aggressive aeration technologies was shown to inadequately aerate the bioreactor system resulting in anaerobic conditions (Stessel and Murphy 1992).

2.4.1.1 Rate and Extent of MSW Biodegradation

Aerobic biodegradation occurs at a much faster rate and to a greater extent than anaerobic conditions in a given time period (Murphy et al. 1995). This is because aerobic respiration is more efficient at generating energy for use by microorganisms than anaerobic respiration, resulting in greater biomass production. Yet in typical landfills, the aerobic zone of a landfill is usually confined to the top 1m of the landfill's surface (Christensen et al. 1989). Murphy et al. (1995) demonstrated that under aerobic conditions, greater biomass production and greater cellulolytic activity resulted in an increase in the degree, speed, and completeness of MSW stabilization. This accelerated biological activity requires installation of a leachate recycle system with sufficient moisture levels. Sesay et al. (1998) noted a drop in moisture content from 60% to 21% in only 13 days of operation in an aerobic bioreactor with insufficient leachate recycle, causing temperatures to exceed 50°C inhibiting microbial activity within the first 24 hours.

According to Pichler and Kogel-Knabner (1999), carbohydrate losses of 71 to 88% were observed under aerobic conditions, representing the majority of the organic losses. Protein and lipid losses were lower due to resynthesis while lignins remained relatively undegraded. After aerobic biodegradation, organic matter losses totaled 35% to 75% under various aeration schemes. The relative amounts of the chemical constituents in the landfilled MSW were primarily proteins, humic materials, and lignin indicating stabilization (Pichler and Kogel-Knabner 1999).

2.4.1.2 Enhanced Leachate Degradation

The more rapid and complete biodegradation of the landfill MSW in a bioreactor system is noted by a reduction in leachate strength. In a study by Leikam et al. (1999), BOD_5, COD, and ammonia concentrations were reduced considerably. Leikam et al. (1999) showed degradation of organic substances approximately five times higher than under anaerobic conditions. After 50 days of aeration, the leachate nitrogen concentration was also reduced to regulatory levels. Comparatively, 400 to 500 days longer was required in the anaerobic bioreactors to reach similar levels. The more toxic ammonia (NH_3) may have also been lost through volatilization in the form of ammonium (NH_4^+) dissolved in the condensate (Burton and Watson-Craik 1998). Ammonia generated continuously by microbial breakdown of nitrogen-containing and proteinaceous matter under anaerobic conditions typical of conventional landfills, where no mechanism is known for its removal in the absence of oxygen (Robinson 1995). When oxygen is available, ammonia oxidizers and nitrite oxidizers readily convert ammonia to nitrite and nitrite to nitrate respectively (Burton and Watson-Craik 1998). They are typically *Nitrosomomas* and *Nitrobacter sp.* (Sincero and Sincero 1996).

Because anaerobic landfill conditions frequently generate high concentrations of ammonia, they are prone to inhibition of biodegradation (Burton and Watson-Craik 1998). For anaerobic sanitary landfills, ammonia generation can be the most significant long-term pollution problem at landfill sites (Robinson 1995). Aerobic conditions can enhance ammonia removal, and as such, ammonia is being monitored closely in this study.

2.4.1.3 Leachate Neutralization and Reduced Metal Dissolution

Anaerobic biodegradation in landfills generates carboxylic and acetic acids, lowering the leachate pH. Acidification of the leachate can inhibit microbial activity and thus the rate of degradation (Wang and Banks 2000). Additionally, acidification of the leachate increases the dissolution and mobility of metals, increasing the pollution potential of the leachate and possibly increasing the requirements of leachate treatment.

Aerobic biodegradation of the organic fraction prevents the acidification of the landfill leachate. As a result, buffering of the leachate is not required and the potential for long-term heavy metal leaching is reduced (Murphy et al. 1995). Belevi and Baccini (1989) concluded that only a small fraction of heavy metals can be mobilized in 2000 years, assuming the leachate pH does not drop below pH 7. In a study by Leikam et al. (1999) leachate metal concentrations were extremely low. A combined effect of leachate recirculation, neutral pH due to MSW aeration, and the sorptive capacity of the MSW paper and fines fraction may have demobilized the metals by behaving as a metals "sink" (Flyhammar et al. 1998).

2.4.2 Infrastructural Benefits of Air Injection

From a managerial perspective, aerobic biodegradation is tremendously advantageous. The key advantages to aerobic biodegradation are discussed below.

2.4.2.1 Settlement and Final Capacity

Greater cellulolytic activity allows for greater biodegradation of cellulose-containing materials, such as paper and vegetative matter (Pichler and Kogel-Knabner 1999).

The accelerated and more complete biodegradation of the organic fraction causes significant volumetric reduction in landfill usage (Murphy et al. 1995) such that the lifetime of a landfill site may be extended (Stessel and Murphy 1992). The results of Stessel and Murphy (1992) indicated that only 20% of the volume of a landfill would remain after aerobic treatment, effectively increasing the final capacity of a single site by five times.

With accelerated leachate stabilization, the requirements and duration of post-closure monitoring are reduced (Leikam et al. 1999). In addition, the risk of cracks appearing in the landfill cap are also reduced; since more complete biodegradation structurally weakens the landfill matrix promoting settlement during the bioreactor landfill's operating lifetime (Stessel and Murphy 1992).

2.4.2.2 Inhibition of Methanogenesis

The predominant reason for a lack of aerobic bioreactor landfills presently in use is their inhibition of methanogenesis. Current sanitary landfills operating under anaerobic conditions collect methane generated by the anaerobic decomposition of the buried municipal solid waste. Anaerobic decomposition of MSW has the potential to release 50-110m^3 of carbon dioxide and 90-140m^3 of methane per metric ton of solid waste (Vieitez and Ghosh 1999). While a fraction of this methane can be used as a source of energy and revenue to offset landfill-monitoring costs, methane emissions from landfills contribute a significant volume of greenhouse gases to the atmosphere. Under these conditions, an estimated 100 to 300 years is required for fugitive methane emissions to drop to harmless levels (Leikam et al. 1999). According to Stessel et al. (1992), the benefits of landfill gas collection for energy purposes are likely to be small compared to the associated long-term site operational costs. Reported methane capture efficiencies vary considerably, with the most optimistic values reported by the USEPA (2002) as roughly 70% to 85% during peak production. However, this does not account for methane losses prior to the installation of the methane capture system and after methane production begins to decline. When methane production rates subside, the collection efficiencies drop considerably, such that the actual proportion of methane captured would be far less over the lifetime of a landfill. Landfills with methane capture systems are still large sources of atmospheric methane emissions. In addition, only approximately one quarter of methane produced by landfills in Canada is collected, and the remaining is released directly into the atmosphere (Natural Resources Canada 2000).

1997 figures indicate that of the 1.0 to 1.2 megatonnes of methane produced annually by about 10,000 active or closed landfills in Canada (Neitzert et al. 1999) only 250 kilotonnes is captured (OCETA 1998). Only 200 kilotonnes of the captured methane produces useable energy, and the remaining captured methane is flared off without electrical production to reduce the environmental impact (Natural Resources Canada 2000). This leaves an annual loss of 1.0 megatonne of methane, which is released into the atmosphere untreated accounting for approximately 23% of the total methane

emissions in Canada (Natural Resources Canada 2000) and representing a significant contribution to Canada's greenhouse gas emissions.

Methane gas has a 100-year global warming potential 21 times greater than carbon dioxide (WMO/UNEP 1996), such that the annual methane emissions from landfills in Canada account for 21 megatonnes CO_2 eq with respect to global warming protocols (WMO/UNEP 1996).

Other studies use different values for the impact of methane. Thermodynamically, methane has 30 times the heating potential of carbon dioxide (Vieitez and Ghosh 1999), and a global-warming potential 35 times greater than carbon dioxide by mass (Daskalopoulos et al. 1998) over a 20-year period.

Conversely, aerobic biodegradation does not produce methane and instead releases carbon as carbon dioxide, which is considerably less problematic. Carbon dioxide does not require gas collection (Murphy et al. 1995), has a far lesser impact on global warming than methane, and is removed from the atmosphere more readily via plant fixation. In addition, inhibition of methane and sulfide production greatly reduces landfill odour (Stegmann et al. 1989). The risk of explosion is also reduced for aerobic landfills.

Overall, the economic incentives of methane production derived from anaerobic landfilling are overstated. Only about one-fourth of the methane generated is captured (OCETA 1998) due to technical constraints and uncontrolled fugitive emissions. Old sites and sites with low methane production are also inadequate for thermal or energetic purposes, but still require methane capture and treatment by catalytic converters, combustion, or biofilters increasing the cost of landfill monitoring. Studies by Read et al. (2001a,b) suggest aerobic landfilling can present significant savings for many landfills due to reduced post-closure operating and maintenance costs.

Additionally, aerobic biodegradation of the organic fraction prevents the acidification of the landfill leachate that occurs under anaerobic conditions. As a result, expensive

buffering of the leachate is not required and the potential for long-term heavy metal leaching is reduced (Murphy et al. 1995). Residual humic materials can also be reused as soil cover and non-biodegradable materials can be mined (Stessel et al. 1992).

2.4.3 Design Variables

Various airflow rates have been studies in aerobic test systems. Stessel and Murphy (1992) used air flow rates ranging from $0.007m^3/s$ to $0.020m^3/s$ in an aerobic bioreactor study. At the low flow rate of $0.007m^3/s$, nitrogen levels increased, indicating greater activity by facultative bacteria and anaerobic landfill conditions. An air delivery rate of $0.0104m^3$ or greater was sufficient to provide the air requirements for 140kg of MSW, using leachate pH as an indicator of aerobic conditions. In a subsequent study with smaller test bioreactors, an airflow rate of 420L/min ($0.007m^3/s$) for 5.91kg of MSW was sufficient, but dropped during the experiment due to increased static pressure from airflow obstruction resulting from MSW settlement (Murphy et al. 1995).

Leikam et al. (1999) used much lower airflow levels, with various aeration intervals and frequencies ranging from four times a week to twice a month. The airflow levels coincided with the maximum respiration activity described by Leikam et al. (1999) as 25 to 32 mg O_2/g dry matter for a total air demand of $85 - 105$ m^3 of air per metric ton of dry municipal solid waste. These figures do not account for unutilized air, but strictly the requirements for microbial activity. To account for this, a greater actual air delivery rate should be maintained (Leikam et al. 1999).

3. EXPERIMENTAL APPROACH

3.1 OVERVIEW

This study was designed to determine the potential for biodegradative enhancement by forced air injection on newly constructed and pre-existing landfills. Fresh and aged municipal solid waste (MSW) samples were used to simulate newly constructed and retrofitted landfills matrices respectively. A series of bioreactors were constructed to mimic bioreactor landfills containing either fresh MSW or aged MSW and were subjected to either forced aeration or left in an anaerobic state.

Determination of the performance benefits of air injection are characterized by the physical analysis of the municipal solid waste at the beginning and end of the study, and through chemical analysis of the leachate during the study period. From this analysis, the degree of stabilization and biodegradation of the MSW was determined and the relative benefits of air injection for both new and retrofitted landfills was ascertained.

The apparatus used in this study consisted of eight test bioreactors, each containing approximately 50L of municipal solid waste subject to closed-loop leachate recycling systems. Of these eight bioreactors, four bioreactors received excess air delivered by an air compressor for thirty minutes daily, as discussed in Section 3.4. The remaining four bioreactors were not aerated, and remained under constant anaerobic conditions. Two bioreactors of each treatment contained sufficiently aged MSW, while the remaining two contain fresh MSW, such that each treatment had one replicate. Fresh MSW was collected from the curbside in Toronto, and the aged MSW was obtained from the Brock West Landfill, Toronto, Canada.

The eight test bioreactors were assigned the designations AF1, AF2, AO1, AO2, NF1, NF2, NO1, and NO2 based on their respective treatment, matrix, and replicate number of the test bioreactor. The treatment and matrix for these samples were as follows:

Table 1. Laboratory Bioreactor Description: Bioreactor Identification, Treatment Condition and Matrix.

Identification	Treatment Condition	Bioreactor Matrix
AF1	Aerated	Fresh MSW
AF2	Aerated	Fresh MSW
AO1	Aerated	Aged MSW
AO2	Aerated	Aged MSW
NF1	Anaerobic	Fresh MSW
NF2	Anaerobic	Fresh MSW
NO1	Anaerobic	Aged MSW
NO2	Anaerobic	Aged MSW

Due to the inherent heterogeneity of MSW and the limited sample size of the test bioreactors, physical preparation of the MSW involving sorting, shredding, and compaction of the MSW was required. Preparation in this manner provided better uniformity between bioreactor replicates while maintaining a percent composition representative of the original waste samples. Compaction densities reflected typical landfill conditions.

Each test bioreactor was kept at field capacity with respect to its moisture content throughout the study. Field capacity was defined as the maximum amount of water emplaced MSW can hold after 24 hours of free drainage with negligible losses. Field capacity was maintained through daily recirculation of leachate and provisions for any excess recirculated leachate to drain into a secondary collection stage were made. On average, field capacity was approximately 58% in the aged waste and 95% in the fresh waste.

Initial analyses performed on the MSW included density, percent composition, and percent moisture. These parameters were necessary in order to compare the physical attributes of the waste and to draw general conclusions about initial landfill conditions. The internal temperature of the bioreactors were monitored over time, and the final settlement of the waste was measured at the conclusion of the study.

Analysis of the MSW leachate included pH, TSS, TDS, COD, BOD_5, and ammonia. In order to provide adequate information about the MSW while minimizing leachate

dilution from leachate removal, sampling for chemical analysis was performed weekly for 19 weeks.

From the results of these parameters, overall determinations of the MSW leachate quality and occurrence of system stabilization were made. The occurrence of stabilization was identified as a state of steady pH and BCR, and uniform leachate TSS, TDS, Ammonia, BOD_5, and COD concentrations. The study concluded when the chemical analysis of the aerobic MSW leachate has indicated MSW stabilization.

3.2 DESIGN AND CONSTRUCTION OF THE APPARATUS

The apparatus consisted of eight test bioreactors. The four aerated bioreactors were designed in the manner presented in Figure 2. The anaerobic bioreactors were identical except they lacked the air sparging system. A photograph of the apparatus is presented in Appendix E. The apparatus is described in the following sections, broken down by its major components.

1. Bioreactor Design and Contents
2. Leachate Recirculation System
3. Air Injection and Exhaust Systems

3.2.1 Bioreactor Design and Contents

3.2.1.1 Bioreactor Dimensions

Eight bioreactors were constructed to model the design of bioreactor landfills, complete with leachate recirculation. Each bioreactor was cylindrical in shape and constructed from polyvinyl chloride having an average diameter of 0.41m, height of 0.56m, and a total capacity of 74L. A cylindrical shape was selected to allow for better settlement and to promote plug flow conditions for the leachate. A port was installed in the base of each bioreactor to prevent leachate pooling and complete saturation of the MSW. The mouth of the port was 2" in diameter with steel meshing to prevent clogging or MSW loss. A ¾"

diameter PVC pipe led from the port into a separate leachate collection basin. To allow for temperature readings, a second port was installed 25cm from the base, at the approximate center of the MSW sample. During standard operation, this port was sealed with an airtight cap to prevent air infiltration and leachate losses.

3.2.1.2 Structural Integrity

It was important for the apparatus to maintain structural integrity throughout the study; despite MSW settlement, increasing pressures delivered by air injection, and gradual chemical deterioration from acidic leachate. To ensure this, all joints and seams in the apparatus were sealed to make the system both air and water tight. Sealants were selected based on their water tightness, acid resistance, and binding strength. Carlon Standard Grade Solvent Cement (Carlon Canada Ltd., Canada) was used for ABS piping, including the air sparges and the MSW drainage port. Marine Goop (Eclectic Products, USA) was used to seal the vinyl tubing and the bioreactor seams, which were composed of PVC plastic. Tubing was also held in place with 3/8" tie clamps and pipe threads were wrapped in Teflon™ tape. In addition, the aerobic bioreactors required 1/8" rivets to be mounted along the lip of the bioreactor to maintain a secure seal during aeration. The rivets were spaced approximately 2" apart.
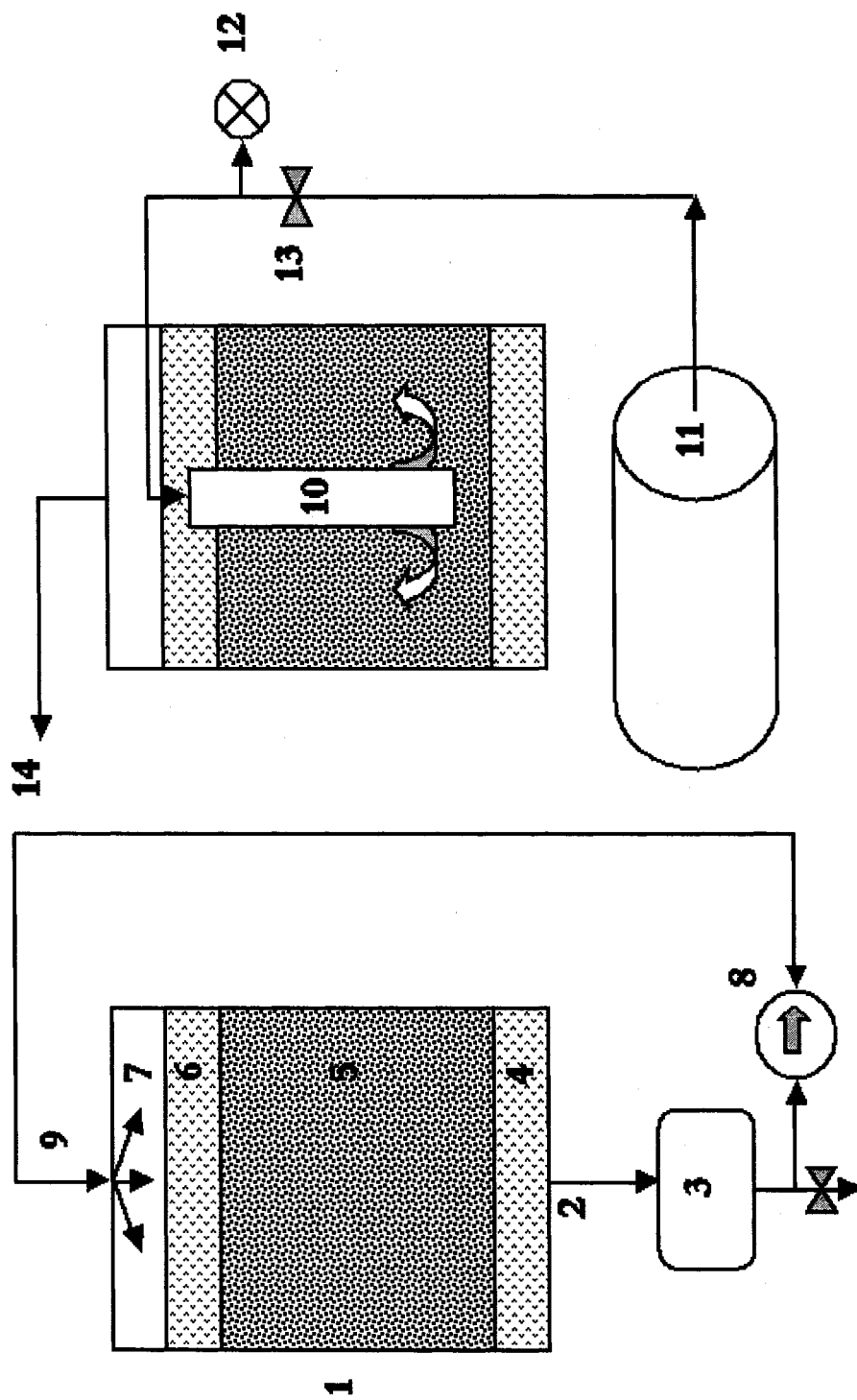
Figure 2. Schematic Illustrating the Experimental Leachate Recycle (1-9) and Aeration Systems (10-14): 1 Bioreactor; 2 Drainage Port; 3 Leachate Collection Reservoir; 4 Gravel Filter; 5 MSW Sample; 6 Gravel Filter; 7 Headspace; 8 Leachate Pump; 9 Leachate Distributor; 10 Air Sparge; 11 Air Compressor; 12 Pressure Gauge; 13 Airflow Control Valve; 14 Air Vent.

3.2.1.3 Contents of the Bioreactor

In order to prevent clogging and mobilization of the MSW, a mesh geotextile (Fiberglass Screen Model 4U1, Phifer Wire Products, Inc., USA) was placed directly on top of a 5cm thick layer of gravel lining the bottom of the test bioreactors. Prior to emplacement of the fresh and aged MSW into the bioreactors, sample preparation was required. This was due to the nature of the aged samples, which were irrevocably intermingled with landfill cover (topsoil). In order to achieve as much similarity as possible between the fresh and aged sample matrices, the fresh waste samples were incorporated with a proportional mass of soil based on percent composition.

Aged MSW samples were taken from Brock West Landfill and were highly intermingled with daily cover, such that pure MSW samples were unobtainable. The aged waste samples were expected to be approximately 10 years old. The actual age of the waste was determined to be in the order of 23 to 24 years old, as noted by newspaper articles imbedded within the sample (Appendix E). This was significantly older than anticipated, as the Brock West Landfill staff believed the location where the samples were taken from would yield less mature samples. Further research into the site history showed that some MSW had been translocated to the Brock West Landfill from the Brock North Landfill, which closed in 1978. Official documentation able to confirm the origins of the sampled aged MSW was not available, but it is near certain that the Brock North Landfill was the original source of the aged MSW. In consideration of this new information, it was decided to proceed using these aged samples since samples of this age are usually very difficult to obtain.

Fresh waste samples were collected in residential Toronto from the curbside and were representative of MSW as disposed, prior to shredding and compaction. Fresh MSW samples were then intermingled with soil to an approximate 38% composition by mass in order to liken them to the condition of the aged samples. Although the fresh MSW samples were initially pure samples, it was felt that having both fresh and aged MSW samples subject to the same preparatory conditions was more important than maintaining virgin material.

After the fresh MSW had been mixed with topsoil, such that the soil amounted to 38%
composition by mass, packing of the bioreactors began. Initial MSW compaction
densities before the addition of soil ranged from 321.6kg/m^3 to 358.0kg/m^3 for the fresh
waste samples. MSW was added to the bioreactors in 15cm thick strata and compacted.
Since the aged MSW had a greater initial density than the fresh MSW, equivalent masses
of fresh and aged MSW occupied slightly different volumes in the bioreactors even after
significant compaction. Sample masses were maintained as constant as possible.

On average, the fresh MSW bioreactors AF1, AF2, NF1 and NF2 were filled with 28.1kg
of MSW and soil to a depth of 0.40m, and compressed to a density of 531.1kg/m^3. This
thickness occupied 52.8L of space within the bioreactors. Likewise, the aged bioreactors
AO1, AO2, NO1 and NO2 contained an average 28.1kg of aged waste reduced to 47.5L
with an average density of 590.7kg/m^3. The aged waste bioreactors were filled to an
average depth of 0.36m, 0.04m less than the fresh waste bioreactors, as 6L of aged waste
had to be removed to maintain similar sample masses. The individual mass, compacted
thickness, volume, and density data are summarized in Table 2. Complete data for the
solids analysis can be found in Appendix C.

Table 2. Mass, MSW Depth, Volume, and Density Data For The Eight Bioreactors.

Bioreactor	Mass (kg)	MSW Depth (m)	Volume (L)	Density (kg/m^3)
AF1	28.8	0.40	52.8	545.3
AF2	26.6	0.40	52.8	503.7
NF1	28.8	0.40	52.8	545.3
NF2	28.0	0.40	52.8	530.2
Average	28.1	0.40	52.8	531.1
AO1	27.2	0.36	47.5	572.3
AO2	27.0	0.36	47.5	568.1
NO1	29.7	0.36	47.5	624.9
NO2	28.4	0.36	47.5	597.5
Average	28.1	0.36	47.5	590.7

Overall, the compaction densities in both the aged and fresh MSW bioreactors were consistent with real-life design, representing moderate-density landfill conditions (Sincero and Sincero 1996). The compacted MSW was capped with a final layer of washed gravel to a thickness of 5cm for the fresh waste bioreactors and 9cm in the aged, such that 6cm of headspace remained in both. Both the gravel layer and headspace were necessary to facilitate the recirculation of leachate and encourage its even distribution throughout the bioreactor.

3.2.2 Leachate Recirculation System

A series of self-timers (Intermatic Standard Timer, Intermatic Inc., USA) were used to automate the leachate recirculation system such that the contents (2L) of the Leachate Collection Reservoirs (LCRs) were recirculated once daily for 20 minutes. Recirculation of the leachate was performed at the same time each day. Since the internal temperature can fluctuate throughout the day due to the cooling influence of leachate recirculation, subsequent temperature measurements and leachate sampling also followed a schedule. The LCRs were constructed from 2.5gal chemical resistant carboys positioned below the test bioreactors in order to capture the leachate after it had percolated through the MSW. Vinyl tubing (1/2" OD) was affixed between the bottom of the LCRs and a peristaltic pump (Laguna Utility Pump 3 #PT-310, Brite-Lite Indoor Garden Centre, Toronto, ON). Each pump drew the leachate from the leachate collection reservoir located below each test bioreactor, and re-introduced it to the top of the test bioreactor through a network of vinyl tubing terminating at a fixture capable of distributing the leachate evenly above the

gravel cover. Percolation through the gravel served to promote an even distribution of moisture within the system.

A manometer fitted into the LCR was used to determine the volume of leachate contained in the LCR, and was graduated to a 2L capacity. A sampling port was also installed between the LCR and the pump.

Previous studies have utilized leachate recirculation systems where 2%, 10%, 15%, 30% and 35% of the sample volume were recirculated as leachate on a weekly basis (Stessel and Murphy 1992; Chugh et al. 1998; Warith et al. 1999). In this study, 2L of leachate was recirculated daily in each bioreactor, totaling of 14L of leachate was per week. This is equivalent to 26.4% of the sample volume (Stessel and Murphy 1992; Chugh et al. 1998; Warith et al. 1999).

3.2.3 Air Injection and Exhaust Systems

The four bioreactors receiving forced air (AF1, AF2, AO1, and AO2) had additional design requirements. Air was delivered to these bioreactors through vertical sparges installed within each bioreactor. Each sparge was made from a 1 ½" ABS pipe, with air vents radiating out horizontally into the MSW. Geotextile was fitted over the vents to prevent blockage and a hole was installed in the bottom of the sparge to allow for any accumulated leachate to drain out. Air was delivered to each of these sparges from a centralized distribution arm used to control the airflow to the bioreactors. The distribution arm consisted of a 1 ½" OD iron pipe with 5 fitments. One fitment received air from an air compressor, and delivered it evenly to the four test bioreactors by means of ¾"valves (B&K® Elk Grove Village, IL) and pressure gauges (0-100psi economy gauge, Winters Canada, Canada) located on the remaining four fitments. The valves were used to adjust the airflow to the individual bioreactors such that all four received the same influx of air. This compensation was necessary to ensure continuous and even air delivery to the bioreactors in lieu of MSW settling and heterogeneity. Each of the four aerated bioreactors received air at a rate of 0.63L/s for thirty minutes daily, at 6pm. Air

sparging was automated using self-timers (Intermatic Standard Plug-in Timer, Intermatic Inc., USA). A photograph of the distribution arm appears in Appendix E.

Gases generated by fermentative and methanogenic bacteria were allowed to escape from the non-aerated bioreactors via one-way ventilation tubes, located on the top of the individual test bioreactors. Bioreactors receiving air had a separate ventilation system from the anaerobic exhaust system such that no exhaust air could enter the anaerobic bioreactors through their ventilation tubes. Gases from both the anaerobic and aerobic bioreactors were expelled through ½" vinyl tubing terminating in a fume hood.

3.3 PREPARATION AND CALIBRATION OF THE BIOREACTOR

Preparation of the MSW and calibration of the bioreactor leachate recirculation and aeration units was required.

3.3.1 MSW Preparation

Space and design limitations restricted each bioreactor to a maximum MSW content of 53L for this experiment (Stegmann and Spendlin 1989; Stessel and Murphy 1992; Chugh et al. 1998; Warith et al. 1999). As a result, prior sorting and shredding of the MSW was required in order to achieve a more representative and uniform sample distribution among test bioreactors. The MSW samples were sorted and oversized inert items in the MSW were removed from the sample. Remaining large pieces of MSW were broken down to 7.5cm across or less in order prevent clogging and promote leachate circulation (Warith et al. 1999). MSW shredding has also been shown to enhance the rate of degradation (Komilis et al. 1999b).

The percent composition of the MSW by weight was determined using a simplified ASTM method D5231-92. In order to compare the matrices on a dry weight basis, ASTM method D4959-00 was employed to determine the MSW moisture content. This was necessary since water will be added to the test bioreactors in order to bring them to

field capacity by the leachate recirculation system (Chugh et al. 1998). These methods are discussed in Section 4.2, *Methods For Physical Analysis Of MSW.*

3.3.2 Standardization of the Leachate Recirculation Volume

All eight bioreactors were brought to field capacity with respect to moisture content. Water was added to the system and recirculated daily for the duration of the first week, by means of the leachate recirculation pumps. When the amount of water in the leachate collection reservoir (LCR) was approximately equal to the amount of water added the previous day, the test bioreactors were considered to be at field capacity. Water was then added to bring the total leachate volume in the leachate collection tank to 2L, which was maintained throughout the study duration by weekly water additions.

3.4 INITIAL BIOREACTOR CONFIGURATION

Both the initial leachate pumping rate and the rate of airflow to the aerated test bioreactors were measured in order to configure the experimental apparatus.

3.4.1 Leachate Pumping Rate and Volume

The leachate-pumping rate for all eight test bioreactors was standardized to 8L/h. This was determined by measuring the time required to displace 2L of water through the bioreactor models prior to packing the bioreactors with MSW. Reducing the channel diameter upstream of the pump and increasing the water column height to 1.60m made the flow rate and recirculation volume comparable to other leachate recirculation studies (Stessel and Murphy 1992; Chugh et al. 1998; Warith et al. 1999).

Field capacity was maintained in the bioreactors by daily recirculation of leachate, which was allowed to percolate through the MSW media. This was achieved by pumping the contents (2L) of the leachate collection reservoir located below the bioreactor into the top of the bioreactor chamber. As the leachate percolated through the MSW matrix, field capacity conditions were re-established within the bioreactor, and excess leachate was allowed to drain back into the LCR. This prevented saturation of the waste in the bottom

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of the bioreactor. A small amount of the recirculated leachate was utilized by microorganisms in the bioreactor, such that the LCR required the weekly addition of water to maintain a 2L recirculation volume.

3.4.2 Rate of Airflow

The rate of airflow delivered by the compressor (DeVILBISS® Tradesman Model DAC-7118, DeVILBISS, Canada) was determined with a Sho-Rate™ model #8-50-1 rotameter (Brooks Instrument Division, Emerson Process Hatfield, PA). The total rate of airflow to the distribution arm remained constant at 2.5L/s as provided by a compressor, which was reduced to 0.63L/s per aerated test bioreactor. Air was delivered for thirty minutes per day, such that the total volume of air passing through each individual bioreactor each day was 1134L, enough to displace the volume of the bioreactors 15 times. Thirty minutes was chosen as the duration of aeration because it assures a complete exchange of the air within the cell. Constant aeration was not required since the airspace within each bioreactor was greater than the amount that could be consumed in 24 hours at peak microbial respiration.

A 30kg sample of fresh MSW with a moisture content of 50% and at its peak respiration activity would require a maximum flow rate of 0.0155L/s based on the 96 hour peak respiration activity of 80mg O_2/g dry matter for fresh MSW as determined by Leikam et al. (1999). Typically, the total air demand over 96 hours lower, at 25 to 32 mg O_2/g dry matter (Leikam et al. 1999). In either case, the 0.63L/s rate of delivery used in this study exceeds this value considerably. Leikam (1999) also states that the total air demand for complete biodegradation of municipal solid waste is 85 – 105 m^3 of air per metric ton of dry MSW. Using these figures as a guideline, a 30kg sample of MSW with 50% moisture content would require between 1.70m^3 – 2.10m^3 of air in one year to achieve complete biodegradation and ignoring blow-through losses. The amount required in this experiment should be significantly lower since the results of Leikam et al. (1999) were for pure MSW whereas this experiment includes 38% Soil Cover.

This rate of oxygen use for this 30kg sample of MSW would require only 4.65L – 5.75L of air per day including the soil cover as MSW. The volume of the headspace (7.9L) combined with the MSW pore space (10.6L, assuming 20% by volume) totals to 18.5L, indicating that ample air has been supplied and oxygen is not limited within the aerobic bioreactors. This does not include the volume of air in the air sparge or in the gravel layers. Finally, the air requirements for wastes containing higher moisture contents and aged wastes would be significantly lower. Since the compressor provides oxygen in excess to the rate of oxygen consumption and the airspace is greater than the daily air requirements the rate of biodegradation should not be affected by minor changes in the rate of air delivery, which are assumed to have little effect on the rate of degradation.

An increase in air delivery pressure throughout the study was expected over time due to MSW settling. The compressor used in this study is capable of delivering an output pressure of 689kPa and handled this increasing resistance easily, operating consistently throughout the experiment regardless of this pressure increase. The period of aeration is specific to this study.

3.5 STUDY DESIGN

3.5.1 Study Timeline

Addition of water to the bioreactor systems commenced on January 28, 2002, bringing the bioreactors to field capacity with respect to moisture content on February 3, 2002. Analytical data was collected over the following 19-week study period, ending on June 5, 2002. An automated system was used to time leachate recirculation for 20 minutes a day throughout the study. Each recirculation began at 4am and involved 2L of leachate, totaling 14L per week. On February 14, 2002 aeration of the aerobic bioreactors began. An air compressor provided air daily at 6:00pm, providing excess air for 30 minutes to the test bioreactors. A timeline of the events is represented in Table 3.

Table 3. Bioreactor Operational Timeline.

Date	01/28/02	01/29/02	02/03/02	02/14/02	06/05/02	06/06/02
Day Number	**1**	**2**	**7**	**18**	**129**	**130**
Physical MSW Analysis	Untreated MSW					Treated MSW
Leachate Recirculation						
Leachate Sampling						
Air Injection						

3.5.2 Schedule of Sampling Events

Analysis of the solid MSW was performed at the beginning and end of the study, and included percent composition, percent moisture, and settlement. The methods used and sample sizes are summarized in Table 4 below, and are further discussed in Section 4.2, *Methods For Physical Analysis Of MSW.*

Table 4. Sample Size and Method for Physical Analysis of MSW.

Parameter	Sample Size (kg)	Method	Reference
% Composition	480	ASTM D5231-92	(ASTM 2001)
% Moisture	0.400	ASTM D2216-98	(ASTM 2001)
Settlement	0	Direct Measurement	N/A

Chemical analysis of the leachate was performed weekly throughout the study. Leachate sampling required 250mL of leachate from the LCR per test bioreactor. The bioreactor temperature was also taken at this time. Chemical analysis of the leachate followed, and was performed in the order as they appear in Table 5.

Table 5. Order of Operations for Leachate Analysis Including Sample Volume and Method.

Parameter	Sample (mL)	Method	Reference
pH	0	Accumet BASIC method	(Fisher Scientific Ltd. Whitby, ON)
TSS	20	SM 2540D	(APHA 1989)
TDS	20	SM 2540C	(APHA 1989)
Ammonia	0.01 to 10	Orbeco-Hellige Indophenol Method	(Fisher Scientific Ltd. Whitby, ON)
BOD_5	0.05 to 50	ASTM 5210B	SM 5210B (APHA 1989)
COD	0.5 to 2.5	Accu-TEST™ method	(Bioscience Inc. USA.)
BOD_5/COD Ratio	0	ASTM 5210B / Accu-TEST™ method	N/A

3.5.3 Bioreactor System Maintenance

Water was added to the LCR on Mondays at 9am to bring the volume of leachate in the LCR back up to 2L after sampling, biological, and evaporative losses. Following this, the distribution of the airflow to the four aerated bioreactors was tested with the rotameter and adjusted as necessary (See Section 4.1). The proper operation of the leachate recirculation system was also confirmed and the BOD_5 analysis initiated the previous Wednesday was concluded.

3.5.4 Chemical Analysis Overview

Because the source and composition of MSW varies regionally, MSW used in similar studies have yielded large variations in analytical results often ranging in orders of magnitude. Consequently, predeterminations of the initial leachate quality based on previous studies are unreliable. Initial analysis of the leachate samples in this study therefore required multiple runs at different dilutions to ensure initial results were within analytical limits.

Compositional differences in MSW samples also imply that a final numeric endpoint for the chemical parameters cannot be made prior to experimentation. Instead, the experiment was concluded when no further changes in the pH, ammonia, TSS, TDS, COD, and BOD_5 are observed. At this point, the leachate as a whole was judged to be stabilized.

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4. METHODOLOGY

4.1 METHODS FOR SYSTEM MAINTENANCE AND OPERATIONS

4.1.2 Confirmation of Constant Airflow Rate

By adjusting the air pressure delivered to each of the four bioreactors by a series of valves located on the air distribution arm, the rate of airflow to each test bioreactor was kept constant at 0.63L/s. A rotameter (Sho-Rate™ model #8-50-1, Brooks Instrument Division, Emerson Process Hatfield, PA) was used to measure the airflow rate at the bioreactor exhaust ports, and adjustments were made to airflow rates using the valves until all four bioreactors had the same airflow rate. This process was repeated weekly to ensure an equal rate of flow in all four aerobic test bioreactors.

4.1.3 Confirmation of Constant Leachate Recycling Rate

Manometers were installed in the LCRs to measure the change in volume after the pumps recirculated the collected leachate. The rate of recycling was confirmed weekly by noting the time it took for each bioreactor to displace the 2L contents of the LCR, which was approximately 15 minutes. The duration of pumping was 30 minutes, such that the 2L contents of the LCRs were easily displaced. The residence time of this recirculated leachate in the MSW was long enough that percolation back into the LCR did not occur within the 30 minute timeframe.

4.1.4 Confirmation of Safe Internal Temperature

Mercury thermometers (Fisher Brand #14-986C, Fisher Scientific Ltd., Whitby, ON) were used to record the ambient air and bioreactor temperatures. The thermometers were inserted into the approximate center of the MSW to take temperature measurements via airtight ports installed in the side of the reactors.

Under moisture conditions lower than applied in this study, bioreactor temperature is a good indicator of the level of bioactivity within the bioreactor. The temperature may rise

significantly from aerobic conditions due to exothermic activity of the bacteria. However, the correlation between biological activity and temperature was diminished in this study due to maintenance of field capacity and the implementation of leachate recirculation. Instead, the internal bioreactor temperature was monitored to ensure that the MSW did not heat to a dangerous extent, such as in the case of Mertz and Stone (1970).

4.2 METHODS FOR THE PHYSICAL ANALYSIS OF MSW

Physical MSW characteristics measured in this study include percent composition, percent moisture, and settlement. These parameters help describe the nature of the waste samples and provide bulk information of its degree of stabilization as a whole.

4.2.1 Percent Composition by Mass

Percent composition is used to characterize the municipal solid waste, and can give an overall notion of it's suitability for biodegradation. The relative proportions are based on mass, and not volume. This is an important factor to recognize as some low-density objects can have a large influence in a bench-scale bioreactor environment. For example, plastic bags are low in mass but can restrict the flow of moisture in the waste bioreactor, or encapsulate material that would readily biodegrade in the presence of air and oxygen. Materials shredding should have minimized this effect. Percent composition was determined with ASTM method D5321-92 (ASTM 2002). Percent newsprint, corrugated cardboard, and fine paper were simplified into one combined category. The category identified as "other inorganic" includes the soil cover and "other organic" includes textiles and leather materials.

4.2.2 Percent Moisture and Field Capacity

ASTM Method 2216-98 (ASTM 2001) was used to determine the moisture contents. Two modifications were made to the ASTM procedure. According to ASTM method 2216-98 (ASTM 2001), sample heating should be reduced from 110°C suggested for rock materials to 60°C for samples containing high organic contents. Secondly, the sample

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size was increased to total approximately 400g in order to account for the inherent heterogeneity of municipal solid waste. This required that each sample be split into three components for analysis since the upper analytical limit of the balance was 200g. Quantifying the moisture content at the beginning and end of the study is necessary to provide a basis for comparisons between the samples. At the start of the study, the moisture content describes the initial nature of the sample and determines the mass as dry matter. Since the moisture content increases to field capacity by the end of the study, mass comparisons between the MSW as discarded and as saturated are inadequate. Instead, the MSW samples should only be compared when they have identical moisture contents. This can be either on a dry weight basis where the moisture content is negligible, or at field capacity.

4.2.3 Settlement of the MSW Over Time

Settlement in the bioreactors was determined by measuring the change in the distance between the top of the bioreactor to the surface of the emplaced MSW before and after the study. Several factors can influence the extent of MSW settlement in addition to biodegradation. Physical characteristics such as the MSW mass, volume, and density also influence settlement. Particle size and relative pore space are also important characteristics of the MSW to consider. Prior MSW sorting and shredding were assumed to achieve uniform pore space within the fresh and aged waste bioreactors. Settlement is a good bulk measurement and approximation of biodegradation in large-scale studies. It is important to note for its influence on final landfill capacities, since enhanced settlement allows for the acceptance of more materials prolonging the landfills lifetime.

4.3 METHODS FOR THE CHEMICAL ANALYSIS OF LEACHATE

In order to minimize the potential for a dilution effect that may arise as a result of reoccurring leachate sampling, analyses of the leachate were performed weekly, requiring 250mL per week (Leikam et al. 1999). The parameters and their results are presented below.

4.3.1 pH

The pH was monitored in this study to determine if pH levels occur that are inhibitory to biodegradation and to confirm that the aerobic bioreactors are sufficiently aerated. pH was measured using an Accumet Basic pH Meter (Fisher Scientific Ltd., Whitby, ON) following the method provided with its documentation. Leachate pH can decrease due to accumulation of carboxylic acids produced by fermentative processes under anaerobic conditions. If the pH becomes too low, biodegradation may be suppressed (Wang and Banks 2000). Aerobic biodegradation pathways do not tend to produce acids, and a more neutral pH is typically observed. The pH meter's electrode was cleaned after use and stored in the KCI solution. Fresh standards were prepared at the start and 2 months into the study to ensure the pH meter was accurately calibrated.

4.3.2 Suspended Solids (TSS and TDS)

Analysis of the Total Suspended Solids (TSS) and Total Dissolved Solids (TDS) give an impression of the nature and the amount of organic material present in the leachate. TSS and TDS are analyzed here as a general parameter to aid in the characterization of the leachate and to determine the time of leachate stabilization.

TSS was determined by quantification of the amount of solids removed by a 0.45μm Whatman 934-AH™ glass fiber filter (Fisher Scientific Ltd. Whitby, ON) from filtration of a fixed volume of leachate. The standard method 2540D (APHA 1989) was followed with one stipulation: In the event that a 20mL sample of leachate resulted in a TSS value of 0mg/L, the analysis was repeated with 50mL sample volumes. TSS samples were stored in desiccators and dried in a Fisher Scientific Isotemp Oven (Model 630G Fisher Scientific Ltd. Whitby, ON). An OHAUS® Precision Standard balance (GENEQ Inc. Montreal, PQ) was used to quantify the amount of suspended solids.

TDS was determined after the filtration of leachate samples through the 0.45μm Whatman 934-AH™ filter (Fisher Scientific Ltd. Whitby, ON) for TSS analysis. TDS was quantified by determining the amount of solids remaining in the filtrate after the TSS determination. The filtrate was dried using a Fisher Scientific Hotplate (Model 11-500-

7H Fisher Scientific Ltd. Whitby, ON) hotplate followed by a Fisher Scientific Isotemp Oven (Model 630G Fisher Scientific Ltd. Whitby, ON), as described in standard method 2540C (APHA 1989). An OHAUS® Precision Standard balance (GENEQ Inc. Montreal, PQ) was used to quantify the amount of dissolved solids remaining after the leachate was evaporated.

4.3.3 Ammonia Nitrogen

The Orbeco-Hellige Indophenol method used in this study for ammonia-nitrogen quantification was provided with the Orbeco-Hellige Aqua Analyzer 2 (Series 952, Fisher Scientific Ltd., Whitby, ON). The indophenol method is a colourimetric method similar to the phenate method SM 4500-NH3 D (APHA 1989) in which indophenol is produced from reaction of between ammonia, hypochlorite, and phenol. Aerobic conditions are required for ammonia removal, and as such, ammonia is being monitored closely in this study.

4.3.4 Biological Oxygen Demand (BOD$_5$)

BOD$_5$ measures the quantity of oxygen utilized for the biodegradation of oxidizable organic matter over a period of 5 days incubated at 20°C. The BOD$_5$ test is commonly used as a general indicator of the "biodegradability" of a test sample. Numerous serial dilutions were required to obtain the initial BOD$_5$, since like the COD; the initial BOD$_5$ can vary in orders of magnitude. The BOD test requires only a small amount of the sample diluted in a 300mL bottle when BOD expectations are high. For BOD$_5$ greater than 600 mg/L, less than 1 mL of leachate is required. Pacey et al. (1999) suggests that the BOD should be less than 100 mg/L when the leachate is stabilized.

ASTM method 5210B was used in this study and samples were incubated in a Fisher Scientific Isotemp Incubator (Model 637D, Fisher Scientific Ltd., Whitby, ON). Dissolved oxygen was determined with a YSI model 51B DO Meter (YSI Inc., USA). The leachate ranges in Appendix B were used to select the appropriate BOD$_5$ dilution factors. Due to the potential for a large variability in BOD$_5$, three test runs were performed per bioreactor. Dilution factors used for this analysis were bracketed above

and below the expected BOD_5 range. For example, if the BOD_5 was expected to be in the 300mg/L – 1050mg/L range, three test runs would consist of 1.0mL, 2.0mL, and 5.0mL of leachate diluted to 300mL with phosphate buffer solution. This would broaden the effective analytical range to 102mg/L – 2100mg/L.

4.3.5 Chemical Oxygen Demand (COD)

The COD of the MSW leachate was analyzed using the potassium dichromate ($K_2Cr_2O_7$) reflux kit provided by Bioscience Inc. for use with accu-TEST™ COD #174-318 and #174-326 vials (Fisher Scientific Ltd., Whitby, ON). The potassium dichromate reflux method is an oxidation-reduction reaction that estimates the portion of matter in the filtered sample that is susceptible to chemical oxidation. COD is also a good approximation of TOC for most waste streams that do not contain recalcitrant aromatic and highly stable components. Pacey et al. (1999) suggests that COD should be less than 1000 mg/L at stabilization.

A Unicam SP 1800 Ultraviolet Spectrophotometer (Pye Unicam Ltd. England) set to detect at 600nm was used for the majority of the study, but broke down between May 5, 2002 and May 22, 2002. During this time, samples were analyzed on the Aqua Analyzer 2 (Orbeco-Hellige series 952, Fisher Scientific Ltd. Whitby, ON). The Aqua Analyzer used a wavelength of 608nm and provided results as transmissivity, which were converted to absorbance and finally COD concentration.

4.3.6 BOD/COD Ratio (BCR)

Assuming that the COD is approximately equal to the TOC, comparison of the BOD_5 and COD of a filtered leachate sample gives an indication of how "biodegradable" the leachate is. As biodegradation proceeds, the BOD/COD ratio declines since easily biodegraded organics are oxidized and recalcitrant carbonaceous matter (predominantly humic and fulvic compounds) is retained. Ehrig et al. (1989) suggest that a BOD/COD ratio of 0.1 or below indicates stable leachate, while values over 0.4 are readily degradable. Initial BOD/COD ratios of 0.9 are not uncommon.

4.4 Stabilization of the Leachate

Due to the heterogeneity of municipal solid waste, stabilization endpoints are not discernable prior to analysis. Furthermore, changes in the MSW matrix cannot be adequately described by any one single parameter, and as such determining when stability is achieved requires monitoring of several chemical indicators. The analyses of BOD_5, COD, pH, TSS, TDS, ammonia-nitrogen and bioreactor temperature provide results which not only provide test-specific data, but when integrated describe the degree of stabilization for the matrix as a whole. For each analyses, stabilization is achieved when the respective concentrations for each analytical parameter remain near-constant over time. The MSW has undergone sufficient biodegradation such that the remaining BDOF does not give further rise to leachate constituent concentrations. It is implicitly understood to be a state of low toxicity and pollution potential, though this is not always the case. Only a few researchers (Ehrig 1989; Pacey et al. 1999) have published numerical limits to describe stabilization, and these values are typically specific to the conditions of the study or generalized in scope. As such, their application to other studies is limited, and such numerical endpoints should be taken strictly as guidelines.

4.5 Statistical Analysis

A replicate bioreactor for each treatment and matrix was constructed in order to improve the reliability of the results. Though the replicates were designed to be identical, this was not the case due to the natural heterogeneity of MSW. Unfortunately, this limits the application of statistical analysis. However, preparation of the samples by sorting, shredding, and weighing should have minimized differences between these replicates. In addition, physical and chemical analysis of each individual bioreactor involved repetitions. To account for MSW heterogeneity when analyzing the samples for moisture content, the sample size was increased and repeated three times. Biological Oxygen Demand was also repeated three times per replicate. Other analyses such as the colourimetric analysis of Ammonia and COD yield more accurate results, and were only

required once per leachate sample. All calculations and trends were plotted using plotted with Microsoft® Excel™ 2002.

5. RESULTS AND DISCUSSION

The following results and discussion are divided into two parts.

In the first chapter, the physical characteristics of the MSW are described including the initial nature of the MSW samples, the percent composition, and physical changes that occur over time including settlement. Changes in the physical characteristics of landfill waste can enhance settlement and increase the usable lifetime of the landfill in question.

This section is followed by a discussion of the chemical analysis of the leachate, broken down by test parameter. The analysis of the leachate gives the best information regarding the overall nature of the MSW both chemically and biologically, and its degree of stabilization.

5.1 RESULTS OF THE PHYSICAL ANALYSIS

In this section, three major topics are covered. First, the initial physical characteristics of the MSW are discussed, including the source characteristics and percent composition of the waste. The compositions of the MSW samples are compared with typical values for the year of emplacement. In the second section, the physical changes in the MSW from the start to conclusion of the study are discussed, including changes in mass, density, temperature, and compaction. Finally, moisture content and the rate of water utilization are discussed. Complete data for the physical parameters discussed here are presented in Appendix C.

5.1.1 Initial Characteristics of the MSW

5.1.1.1 Initial Waste Composition

Results from the analysis of waste composition are represented in Table 6. The data in Table 6 is discussed below where the percent composition of the aged MSW is compared

with typical 1980 values, the percent composition of the fresh MSW is compared with typical 1999 values, and the percent composition of fresh and aged MSW samples are compared. A description of the initial waste composition of the aged and fresh waste samples follows.

5.1.1.2 Percent Composition of Aged Waste

The percent composition of the aged waste was based on a 41.7kg sample as it was collected, and is presented in Table 6. The percent composition excluding the daily cover presented in Table 6 was determined from the waste component proportions in the remaining 61% after disregarding the "other inorganics" component.

The original aged waste was composed of primarily daily cover, newsprint, plastic bags, plastic containers, glass bottles, textiles, metals, wood and other such materials. Very little organic food waste was remaining. Remaining food matter was usually encapsulated in plastic bags or in sealed jars. The vast majority of the paper in the aged MSW sample was newsprint, with the remainder comprised of waxed paper cartons, paper packaging, and plasticized paper products such as labels. By volume, most of the plastics were plastic bags and showed very little change after 23 years of burial. Plastic bottles, including some bottles that originally contained household hazardous wastes such as motor oils and cleaning supplies were still in abundance and did not show any signs of breakdown. The plastics were mostly LDPE, PETE, and some HDPE objects. Organic matter remaining in the aged waste was predominantly vegetative husks and rinds. Other organic materials assumed to be food wastes were slimy in consistency and unidentifiable. Small shards of wood no longer than 7.5cm in length were kept in the MSW sample, while larger objects were rejected. The bulk of this category is due to textiles, as clothing was in abundance. The textiles included cotton, acrylic and polyester materials, which were still identifiable as sweaters, bathing suits, et cetera. Various objects in the waste samples were metallic in nature, with most falling into the ferrous metals category. Ferrous metals were comprised of crushed aerosol cans, tin cans, nails and machine parts. Aluminum cans were also in abundance. Heavy copper wiring with insulation was also present. A few broken glass bottles were present, as well as some

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small jars encapsulating organic residuals. This category was comprised mostly of the soil that made up the landfill's daily cover. It also included some inert objects like ceramics and less reactive materials like insulation and rubber, which accounted for less than 2%

5.1.1.3 Comparison of Aged Waste and MSW Typical of 1980

Unlike the comparison of the fresh and 1999 as-discarded MSW compositions in the following section, comparisons of the aged waste samples to 1980 as-discarded values are not as equitable. This is because the comparison is between MSW that has aged approximately 23 years and has undergone significant changes with respect to its initial composition. Thus, comparison of the aged MSW to typical values does not describe the difference between what the aged waste was like when it was discarded and typical values of that time, but rather how 23 years of entombment in a conventional landfill affects the waste compositionally. Since the true initial composition of the aged MSW as discarded is unknown, inferences drawn from this comparison are purely conjectural.

The aged waste had significantly higher proportions of plastics and "other inorganic" materials, excluding glass. Because these items are not biodegradable, while the actual amount remains the same their percent composition increases over time as a result of a decreasing organic fraction. The amount of glass in the aged waste was approximately one third of the typically disposed-of waste. This is likely because the glass was frequently in shards, with a significant amount incorporated into the "other inorganic" category with the soil cover. The percent yard waste was much lower in the aged MSW, and surprisingly the percent food waste composition was near as-discarded proportions. This was not expected, since the amount of food-waste in the aged MSW appeared low. The numerical data supporting this comparison are presented in Table 6.

5.1.1.4 Percent Composition of Fresh Waste

The data for the initial percent composition of the fresh waste was based on a 36.2kg sample prior to soil addition. The percent composition including the daily cover was calculated after a proportional mass of soil was added to the sample to make it

comparable to the aged waste with respect to "other inorganics" content. The initial percent composition of the fresh sample as discarded is similar to the results of 1999 values (USEPA 2001). The original fresh MSW and typical 1999 values are compared in Table 6. The percent composition of the fresh waste with increased daily cover is also presented. Soil was added to the fresh MSW to emulate the cover in the aged waste, resulting in an equal fraction of "other inorganics" to the aged waste composition.

The fresh waste had a much larger percent composition of food waste since it had not undergone any previous biodegradation as did the aged waste. The relative amounts of aluminum and plastic wastes were proportionately less, likely due to recycling programs and improved waste diversion compared with initiatives in place 23 years ago. However, the fresh waste still had a large amount of commonly recyclable material in it, such as aluminum cans, beverage bottles, and newsprint. The vast majority of the paper in the aged MSW sample was newsprint, with the remainder comprised of waxed paper cartons, paper packaging, and plasticized paper products such as labels. The paper fraction in the fresh samples was composed of nearly equal masses of newsprint and waxed paper products such as milk cartons. Fine paper and cardstock made up the remainder. Overall, less paper was present in the fresh waste than in the aged sample. The plastics in the fresh garbage were mainly LDPE plastic bags, PETE food containers, plastic bottles and packaging. Proportionally, the results were similar to the average waste composition reported for 1999 in the United States (USEPA 2001). The initial fresh waste had a much larger composition of food waste than the aged waste samples. The food waste accounted for 29.7% of the mass prior to soil addition. This fraction was approximately 30% putrescibles such as meat and dairy products, with vegetable matter accounting the remaining 70%. Yard waste accounted for only 1.5% of the original sample. When compared to typical waste compositions, the percent yard waste in the fresh sample was quite low. This was anticipated since the sample was gathered in January. Some heavy textiles were present, made from heavy cotton (denim) material. A wicker basket with negligible mass was also present. The sources of the metals in the fresh sample were aluminum foil, aluminum cans, and tin cans exclusively. No other metallic materials were present. Though this category had less metal than the aged waste, the amount of metals

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present was still quite high considering they could have been easily diverted by the consumer. Unbroken glass bottles accounted for the glass fraction of the fresh waste. More glass was present in the fresh waste than the aged waste. The "other inorganics" category originally included diapers and batteries, but were removed from the system prior to MSW shredding and bioreactor packing since they could not be evenly distributed between the four fresh MSW bioreactors.

5.1.1.5 Comparison of Fresh Waste and MSW Typical of 1999

The fresh MSW matched the typical 1999 MSW compositionally in percent paper, plastic, and "other organics". However, there were large differences in the percent yard waste, food waste, aluminum and wood contents. The percent yard waste in the sample was much lower, and virtually no wood products were present in the fresh sample. The percent food waste in the fresh MSW sample was much greater than the typical U.S. values presented in Table 6, and the percent aluminum was proportionally double.

5.1.1.6 Comparison of Fresh and Aged Waste with Equal Soil Content

Soil accounted for 38% of the overall composition by mass in both the aged and soil-amended fresh waste bioreactors, which have been included for this comparison. The composition of the fresh waste is as discarded with soil added to be proportionally similar to the soil cover in the aged waste, while the aged waste reflects over 20 years of biodegradation in a conventional landfill. Most non-degradable fractions such as total metals were similar in proportions, but the fresh waste had a much higher biodegradable fraction, including 13% more food waste. The aged waste had 4% more plastics.

Table 6. Comparison of Aged and Fresh MSW Samples by % Composition to Typical 1980 and 1999 Values. % Composition of the Aged and Fresh Waste Samples are Reported By Mass Including and Excluding Daily Cover.

Component	Aged MSW: % Composition Including Daily Cover	% Composition Excluding Daily Cover	Typical 1980 % Composition As Discarded[1]	Fresh MSW: % Composition Including Daily Cover	% Composition As Discarded	Typical 1999 % Composition As Discarded[2]
Paper	22.2	36.4	31.7	18.4	29.7	30.6
Plastic	12.6	20.7	5.0	8.5	13.8	13.8
Yard Waste	2.0	3.3	20.1	0.9	1.5	9.1
Food Waste	5.1	8.4	9.5	17.6	28.4	14.8
Wood	1.9	3.1	5.1	0.0	0.0	6.9
Other Organics Including Textiles	8.0	13.1	6.3	6.3	10.2	10.1
Ferrous Metals	5.1	8.4	8.9	4.1	6.6	5.3
Aluminum	2.2	3.6	1.5	2.2	3.6	1.7
Glass	1.9	3.1	10.5	3.9	6.2	5.8
Other Inorganics Including Soil	39.0	0.0	1.5	38	0.0	1.9
Total (%)	100.0	100.0	100.0	100.0	100.0	100.0

[1] 1980 values (USEPA 2001)
[2] 1999 values (USEPA 2001)

5.1.2 Physical Changes in MSW over Time

5.1.2.1 Percent Composition after Bioreactor Treatment

Comparison of the percent composition by mass after the 19-week study was not performed due to time constraints and the likelihood of misleading results from non-homogeneous water absorption across all components. Since individual fractions of the MSW absorbed different volumes of water before coming to field capacity, resulting measurements based on mass would likely be substantially skewed. The mass of components such as plastic bags would remain virtually unchanged with the addition of water, while items such as newspapers and textiles would increase in mass due to water adsorption, leading to an unwarranted proportional increase in the paper percent composition. However, a visual inspection was performed at the end of the study in which observations were compared to the percent composition of the MSW at the time of bioreactor preparation. The general observations are as follows.

5.1.2.2 Visible Changes in MSW Composition after Bioreactor Treatment

In the anaerobic fresh waste bioreactors, meat products were not found after the 19 week leachate recycling initiative, though they were likely still present in small amounts. Dairy products such as cheese and vegetative matter remained, though obvious physical and chemical degradation occurred. In the unaerated fresh waste bioreactors, some organic matter was converted to slimy residuals, and the odour was strong. The aerated fresh waste bioreactors had an earthy aroma, and some algae started to grow in AF1 and AF2, which suggests that the final leachate was low in toxicity.

Little change was noted in the aged waste, aside from the organic content. The organic constituents in both the aerated and anaerobic aged waste bioreactors were noticeably depleted, such that a negligible amount remained when the study concluded. The odour in the unaerated bioreactors was still detectible, while the aerated samples were odour-free.

5.1.2.3 Change in Mass, Compaction, and Density

After the MSW was packed into the bioreactors, the aged waste had an average initial density of 590.7kg/m^3. The fresh waste was brought to a similar initial density, averaging 531.1kg/m^3 after soil was added to make the fresh MSW comparable to the high soil cover content in the aged waste sample. The MSW samples were then rapidly brought to field capacity, with each bioreactor requiring a different amount of water to do so, which is discussed later in Subsection 5.1.3, *Moisture Content and Utilization.*

Based on the change in moisture content between the initial and final waste masses, calculations were performed to estimate the initial mass of the waste samples when field capacity was first achieved. Overall, the fresh MSW bioreactors contained MSW with an average density of 647.4kg/m^3 once the moisture content reached field capacity, while the aged samples had an average density of 672.2kg/m^3 under the same conditions.

Since the initial MSW masses, volumes, and densities within each bioreactor vary, changes in mass, compaction, and density are presented as percent change in the following discussion. Complete data sets appear in Appendix C.

The mass of MSW at the initiation and conclusion of the study were compared in attempt to quantify the amount of mass reduction that may have occurred through biodegradation in the bioreactors. The initial sample mass was the sum of its initial mass as-emplaced, plus the mass of water required to bring it up to field capacity, as discussed in Subsection 5.1.3. The final mass of MSW at field capacity was determined by weighing. The data presented in Table 7 represent the averaged values for each waste matrix and treatment, such that replicates have been averaged. For example, AF represents the average of the data from bioreactors AF1 and AF2.

Table 7. Loss of Mass (%) for Averaged MSW Replicates Saturated to Field Capacity.

Mass Reduction	AF	NF	AO	NO
Initial Mass[1] (kg)	32.48	35.90	31.23	32.68
Final Mass[1] (kg)	30.75	34.10	30.70	32.25
Change in Mass (kg)	-1.72	-1.80	-0.53	-0.42
Change in Mass (%)	-5.31	-5.01	-1.68	-1.30

[1] at field capacity

Mass losses (as %) in the aerated bioreactors were on average marginally greater than in the anaerobic bioreactors. An even greater difference occurred between the mass losses in fresh and aged MSW matrices. The fresh waste had a significantly higher biodegradable organic fraction, such that mass losses were greater in the fresh waste than in the aged waste.

Similarly, the change in volume over time was measured. The fresh waste underwent a greater reduction in volume than the aged wastes, and aeration also contributed to an increased volume reduction in Table 8. The final volume was calculated using the settlement data. Settlement is discussed below in under *Settlement of the MSW*.

Table 8. Change in Volume (%) for Averaged MSW Replicates Saturated to Field Capacity.

Change in Volume	AF	NF	AO	NO
Initial Volume (L)	52.81	52.81	47.53	47.53
Final Volume (L)	39.61	42.25	38.95	39.61
Change in Volume (L)	-13.20	-10.56	-8.58	-7.92
Change in Volume (%)	-25.00	-20.00	-18.06	-16.67

The fresh MSW bioreactors contained MSW with an average density of $647.4 kg/m^3$ once the moisture content reached field capacity, while the aged samples had an average density of $672.2 kg/m^3$ under the same conditions.

The change in density was quantified based on the initial and final MSW masses and volumes. As expected from the changes in mass and volume discussed above, increases in density were greater in fresh MSW samples, and under aerobic conditions. The changes in density are presented in Table 9.

Table 9. Increase in Density for Averaged MSW Replicates Saturated to Field Capacity.

Change in Density	AF	NF	AO	NO
Initial Density[1] (kg/m^3)	614.9	679.8	657.0	687.5
Final Density[1] (kg/m^3)	776.0	805.3	790.0	815.9
Change in Density (kg/m^3)	161.1	125.5	132.9	128.4
Change in Density (%)	26.2	18.5	20.2	18.7

[1] at field capacity

The increase in the density of the MSW is a result of settlement, and is further discussed in *Settlement of the MSW* below.

5.1.2.4 Changes in Internal Bioreactor Temperatures

The temperature in the bioreactors typically stayed within 1°C of the ambient laboratory temperature, but on occasion increased up to 3°C above the ambient laboratory conditions. The ambient lab temperature ranged from 21°C to 25°C throughout the study duration. Temperature data is presented in Appendix C.

In less-moist bioreactors, temperature is expected to increase significantly from accelerated microbial activity, before returning to more ambient temperatures when the rate of biodegradation slows down. This trend was not observed in this study due to temperature regulation achieved through maintenance of a high moisture content and daily leachate recirculation.

Because of this, the internal MSW temperatures do not reflect the rate of biological activity or stabilization, but demonstrate the potential for leachate recirculation to prevent temperature extremes. In a previous experiment in which the leachate recirculation system used a smaller exchange rate, temperatures rose uncontrollably leading to a fire (Mertz and Stone 1970).

5.1.2.5 Settlement of the MSW

Settlement in the bioreactors was determined by measuring the change in the distance between the top of the bioreactor to the surface of the emplaced MSW before and after the study. Several factors influence the extent of MSW settlement in addition to

biodegradation. Physical characteristics mentioned above such as the MSW mass, volume, and density also influences settlement. Particle size and relative pore space are also important characteristics of the MSW to consider.

In this study, MSW shredding and sorting were performed to ensure a more uniform particle size and percent composition. It is assumed that the variation in particle sizes and pore space within aged and fresh MSW bioreactors are negligible, although the aged and fresh MSW groupings may in fact differ from each other.

The degree of settlement in the test replicates was averaged, and the results are presented in Table 10.

Table 10. Average Settlement in Bioreactors.

MSW Settlement	AF	NF	AO	NO
Initial Headspace (cm)	6.0	6.0	10.0	10.0
Final Headspace (cm)	16.0	14.0	16.5	16.0
Change in Headspace (cm)	-10.0	-8.0	-6.5	-6.0
Change in MSW Depth (%)	-25.0	-20.0	-16.3	-15.0

Between fresh and aged samples, the fresh waste settled an average of 9.0cm, compared to 6.3cm in the aged bioreactors. This difference was expected, since the fresh wastes had a greater BDOF and was therefore more susceptible to mass reduction. The 2.7cm difference in settlement between the fresh and aged bioreactors is equivalent to a 6.9% height reduction and a volume of 3.56L. Aerated bioreactors also settled more than the anaerobic bioreactors, with an average 8.3cm drop in height for aerobic bioreactors compared to 7.0cm in the unaerated bioreactors. The 1.3cm difference in settlement between the aerated and anaerobic bioreactors is equivalent to a height reduction of 3.1% and a volume of 1.72L. Overall, aeration increased the degree of settlement in both fresh and aged MSW matrices.

The observations of settlement coincide with the results of volume and mass losses. Increases in density were due to a greater reduction in sample volume proportional to the sample mass. However, caution must be taken when applying individual parameters such

as settlement to overall MSW stabilization. It is not sufficient to estimate stabilization based on the settlement alone, since additional factors such as the pore volume and particle sizes in the waste matrix can impede settlement of highly degraded wastes. It is not sufficient to generalize settlement as an equitable measure of MSW mass reduction. Instead, the combination of bulk parameters such as mass reduction and landfill settlement can result in a valuable general description of the degree of landfill stabilization.

Reactors containing fresh MSW had a greater degree of settlement than the aged-waste bioreactors and also experienced a greater mass reduction. Similarly, the mass reduction and settlement in aerated bioreactors resulted in greater mass reductions and settlement than in the anaerobic reactors. This suggests that the fresh MSW underwent a greater degree of stabilization proportional to the aged waste, and aeration has positively affected the degree of physical stabilization. In concert with analysis of the bioreactor leachate, the extent of biodegradation and MSW stabilization can be well defined in both physical and chemical terms. The leachate analysis follows in Section 5.2.

5.1.3 Moisture Content and Utilization

5.1.3.1 Changes in Moisture Content

The moisture contents of the fresh and aged waste samples were calculated before and after the 19-week leachate recirculation and bioreactor aeration period. Even though larger-than-standard sample sizes were used for the determination of the MSW moisture content, these estimates still involve moderate uncertainty due to the heterogeneous nature of municipal solid waste.

The average initial moisture content in the aged bioreactors was 38.8%. The fresh MSW had a higher moisture content of 60.9% since it contained a more appreciable amount of organic matter, such as food wastes, which contain moisture.

The final moisture content of the municipal solid waste samples represents the moisture content at field capacity. The fresh waste had a moisture content of 95.1% at field capacity, 34.2% greater than its initial average moisture content of 60.9%. The average moisture content of the aged waste increased 19.2% from its initial moisture content of 38.8% to 58.0%. Fresh MSW required more water to bring it to field capacity.

The aged waste was not at field capacity in the Brock West Landfill. This is typical of conventional landfills, which are designed to dissuade moisture permeation by MSW encapsulation. As such, the landfill conditions were not optimal for biodegradation. A summary of the average changes in moisture content for the bioreactors appear in Table 11.

Table 11. Difference In Moisture Content Of Fresh And Aged MSW As Emplaced Compared To Field Capacity.

Moisture Content	AF	NF	AO	NO
Initial Moisture Content AE[1]	61.3	60.5	37.6	39.9
Final Moisture Content FC[2]	89.2	101.1	58.6	57.5
Increase in MC	27.9	40.6	21.0	17.6

[1] AE: As-Emplaced
[2] FC: Field Capacity

The cumulative amount of water added to the individual bioreactor systems is shown in Figure 3. It was determined that the system was at field capacity when the amount of water added each week was remained constant. Field capacity was achieved in the individual bioreactors at different times, from 4 days before the first sampling session to 9 days afterwards. In Figure 3, the time at which field capacity was reached appears as the beginning of a long linear trend with water consumption increasing at a constant rate over time.

Not all of the water added was absorbed or utilized by the MSW contents, such that the cumulative amount of water added at the point where field capacity was reached in Figure 3 is greater than the amount of water required to bring the moisture content of the encapsulated MSW to field capacity.

The volume of water required to bring each bioreactor to field capacity was determined as the total amount of water added to the system when field capacity was achieved less the amount stored in the bioreactor leachate collection system as well as any losses from sampling or leaks. The volumes of water taken up by the MSW after accounting for these factors are presented in Appendix C.

5.1.3.2 Rate of Water Utilization

After the bioreactors were brought to field capacity with respect to moisture content, weekly additions of water to the bioreactors were relatively constant, such that a linear relationship could be seen over time. The slope of this relationship gives us the average daily consumption of water by the bioreactors. Weekly sampling was already accounted for in Figure 4, which shows the rate of actual moisture utilization rather than the cumulative leachate usage including samples taken for analysis.

The average slopes for the replicates in Figure 4 are based on the data presented in Appendix C. Though the slopes appearing in Figure 4 are linear, the aerated bioreactors appear to suit an exponential curve. However, for the purpose of a general comparison, a linear rate of water consumption was used for comparison in the aerated bioreactors.
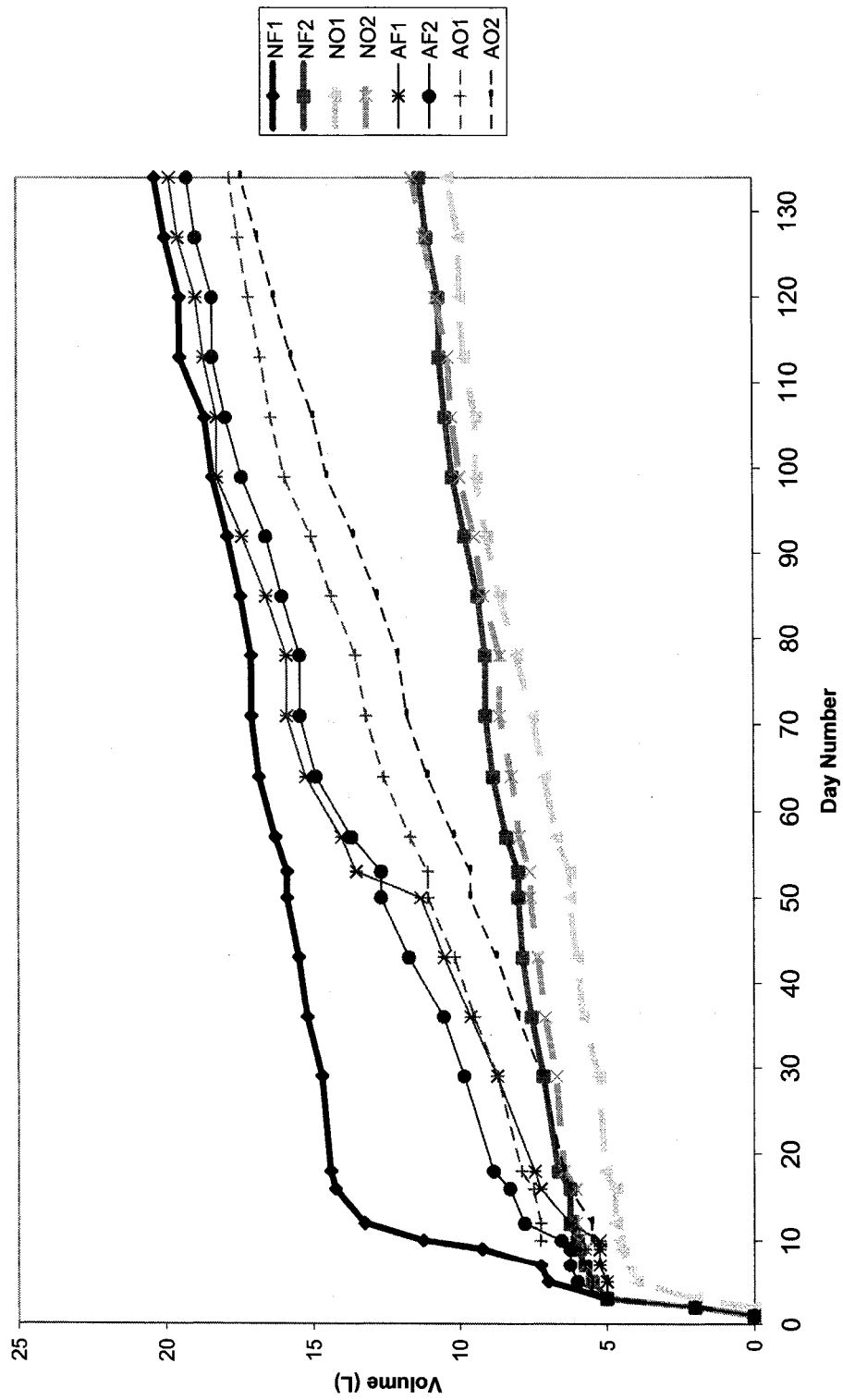
Cumulative Water Additions

Figure 3. Cumulative Water Additions to the Bioreactors Over Time.

Corrected Water Consumption Over Time
slopes = L/day

Legend:
- ◆ NF1 y = 0.0173x
- ■ NF2 y = 0.0076x
- ▲ NO1 y = 0.0144x
- ✕ NO2 y = 0.0087x
- ✱ AF1 y = 0.0800x
- ● AF2 y = 0.0617x
- + AO1 y = 0.0567x
- - A02 y = 0.0633x

X-axis: Day Number (0, 10, 20, 30, 40, 50, 60, 70, 80, 90, 100, 110, 120, 130)
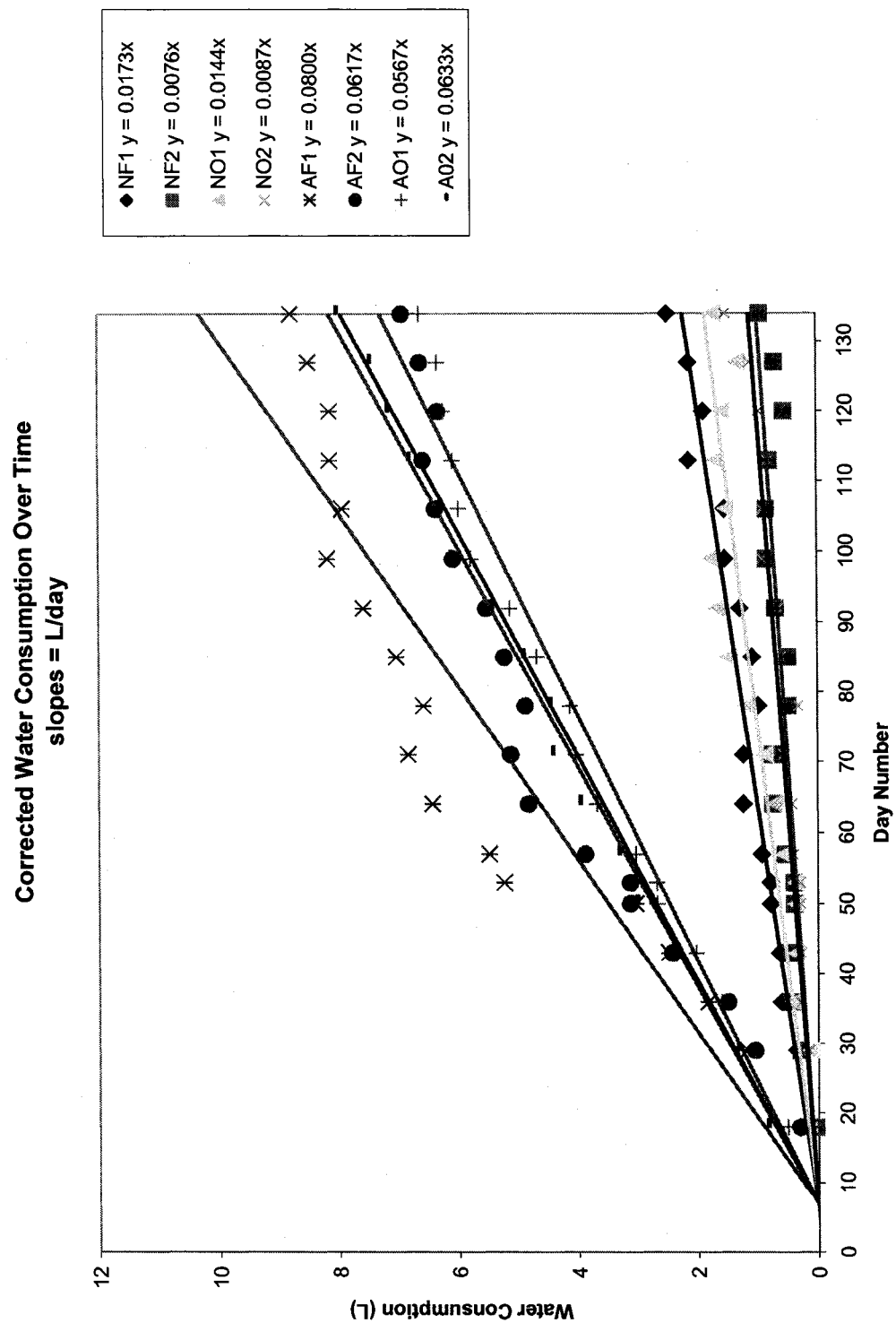Y-axis: Water Consumption (L) (0, 2, 4, 6, 8, 10, 12)

Figure 4. Rate of Water Consumption in the Aerated and Anaerobic Bioreactors.

5.1.3.3 Water Utilization and Biodegradation in Bioreactors

The fresh waste bioreactors used 1.16 times more water than the aged bioreactors, suggesting greater biological activity in the fresh waste bioreactors. Additionally, water consumption was considerably greater in the aerated bioreactors than in their anaerobic counterparts. However, due to the nature of the aeration system, a portion of the water used in the aerated bioreactors may have been lost due to enhanced evaporation through the ventilation system. If we assume that no evaporation occurred due to the compressed airflow, the aerated bioreactors utilized 5.45 times more water than the anaerobic bioreactors (Table 12). Comparatively, if we regard the water losses in the aged bioreactors to be due entirely to evaporation, the ratio of water consumption in the aerated bioreactors to the anaerobic reactors is reduced to 2.15 to 1. This assumes that no water usage in the aged bioreactors was for biological purposes. The actual microbial water consumption for the aerated bioreactors is likely to somewhere in between these two extremes. However, in either case it aeration utilized more water than the anaerobic bioreactors regardless of evaporative losses. This increased usage is assumed to be due to greater biological activity in the aerobic bioreactors.

Table 12. Relative Water Consumption within Treatments and MSW Matrices Assuming no Vapour Losses and Maximum Potential Losses.

	Fresh MSW (L/wk)	Aged MSW (L/wk)	Average Usage for Treatment (L/wk) No Vapour Losses	Average Usage for Treatment (L/wk) Maximum Losses
Aerated Bioreactors	0.496	0.420	0.458	0.181
Anaerobic Bioreactors	0.087	0.081	0.084	0.084
Average Usage for Matrix	0.292	0.250		
Ratio of Water Usage: Aerobic vs Anaerobic Treatment			5.45	2.15
Ratio of Water Usage: Fresh vs Aged Matrices			1.16	1.16

5.2 CHEMICAL ANALYSIS OF THE LEACHATE

The experimental results obtained from the weekly analysis of the bioreactors leachate are presented in this section. The first sample was taken on February 3, 2002, with

weekly samplings following on February 6, 2002 and ending on June 5, 2002. For the purposes of this thesis, the sample period is presented with "day numbers" instead of actual dates. The significant dates and their representative day number as they are described in this study were previously listed in Table 3.

A total of 1064 samples were taken over the 19-week duration of the study, excluding blanks and replicates. The analyses performed on the leachate included: pH, total suspended solids (TSS), total dissolved solids (TDS), ammonia-nitrogen, BOD_5, COD, and BOD_5/COD ratio. They were performed in the order presented in Table 5. As part of quality control, all analyses were performed in the same order and at the same time of day on the day of sampling. The leachate samples were kept in airtight containers and stored in the dark while not in use.

All analyses were initiated and completed on the day of sampling, with the exception of BOD_5. BOD_5 was initiated on the day of sampling, with results taken 5 days later.

5.2.1 Stabilization Endpoints

In the following results and analysis section, stabilization has largely been determined empirically based on the degree a parameter changes over time, the magnitude of fluctuations, and prior behavior in the analysis. As such, a leachate is considered stabilized when its respective concentration resists change over time, fluctuations are small, and notable biodegradation has previously occurred. When applicable, guidelines developed in other studies such as Ehrig (1989) and Pacey et al. (1999) have been applied. (See Section 4.4).

The results for the leachate analysis are presented and discussed in this section in the order they were analyzed.

5.2.2 pH

5.2.2.1 Initial pH Conditions

Several trends can be noted prior to aeration, which began on day 18 of the study. The initial pH values for the fresh test bioreactors were all lower than the pH of the aged test bioreactors. However, different trends were observed in the individual bioreactors in Figure 5.

The pH of AF2 and NF2 both decreased between day 7 and day 10 but increased on day 17. The pH of the leachate from AF1 rose throughout the same period, from 5.24 to 5.64, while the pH of NF1 dropped from 6.32 to 5.76.

Unlike the fresh samples, the aged samples NO1, NO2, AO1, and AO2 maintained a steady pH in the same period, showing little variation within each other. On day 7 the pH values for the aged samples varied from 6.38 to 6.67 and rose slightly to a range of 6.64 to 6.73 by day 17, just prior to aeration.

5.2.2.2 Effect of Aeration on pH in Aged MSW

A strong leachate neutralizing effect occurred in all aerobic bioreactors after aeration began on day 18. A comparison of AO1 and AO2 to NO1 and NO2 demonstrates this effect on the aged samples. Prior to aeration, the pH of the four test bioreactors were very similar, with pH values of 6.64, 6.64, 6.64, and 6.73 respectively. Within 3 weeks, the leachate pH in the aerated and non-aerated aged MSW bioreactors differentiated significantly. In this duration, the pH of AO1 and AO2 had both elevated above pH 7, where it remained after day 52 until the end of the study.
NO1 and NO2 experienced little change in the same duration, from pH 6.64 and 6.73 on day 17 to 6.68 and 6.55 respectively on day 38.

Little fluctuation was observed in the leachate pH for AO1 and AO2 in the timeframe of day 52 to day 129. In this period, pH values changed from 7.22 to 7.27 and from 7.28 to

66

7.38 respectively. Similarly, the leachate pH in the unaerated samples also showed little fluctuation in the same duration.

On day 52 the pH values for NO1 and NO2 were 6.80 and 6.68 respectively, and rose only slightly to 6.84 and 6.69 by the termination of the study. These values only show a small difference from the pH values of 6.38 and 6.50 for NO1 and NO2 from the initial measurements taken on day 7. From these data, stabilization of the leachate in AO1 and AO2 occurred on day 52, 5 weeks after aeration began. The resulting leachate pHs were neutral and above 7.0. These pH values were never obtained in the unaerated aged bioreactors, which remained below 7 with respect to leachate pH. However, the leachate pH in these bioreactors changed only slightly throughout the study, which indicates that the aged MSW in these bioreactors may have already been stable in an anaerobic environment.

5.2.2.3 Effect of Aeration on pH in Fresh MSW

A rapid neutralization of the leachate pH was observed in the aerated test bioreactors containing fresh MSW. By the third week of aeration, the leachate pH of bioreactors AF1 and AF2 rose from pH 5.64 and 6.18 to pH 6.95 and 6.97, and remained above pH 7 from day 52 to the end of the study period.

The pH of the leachate from unaerated bioreactors containing fresh MSW (NF1 and NF2) also progressed towards neutrality, at a less pronounced never less rapid rate reaching pH values of 6.55 and 6.72 for NF1 and NF2 respectively by day 45. Four weeks earlier, acidic pH values of 5.76 and 5.87 for NF1 and NF2 were measured on day 17. These values coincided with the pH of the likewise unaerated bioreactors NO1 and NO2. By day 80 NF1 and NF2 pH values became dissimilar to the non-aerated aged MSW bioreactors, with pH values just below and above pH 7. By the end of the study both NF1 and NF2 were pH > 7.

For this study, the pH was considered stabilized on the date after which a steady pH over time is observed. Using this estimation, stabilization of the aerated fresh-waste samples

occurred on day 52, when the leachate pH of both AF1 and AF2 rose and remained above pH 7 for the remaining 77 days. For the unaerated bioreactors NF1 and NF2, stabilization occurred on day 87 and 80 respectively. This does not indicate that the leachate pH did not change after this date, only that such fluctuations were negligible. Aerobic leachates stabilized above pH 7 while anaerobic bioreactors stabilized slightly below pH 7, but rose to above 7 as they neared the end of the study.

5.2.2.4 Overall Trends in pH and Leachate Stabilization

Regardless of aeration, the leachate pH in all bioreactors moved towards a more neutral pH condition, which was most likely encouraged by leachate recirculation. The rapid drop in leachate pH observed early in this study for the fresh MSW suggests that fresh MSW leachate has a large potential for acidification under conventional landfill conditions. The aged waste did not experience a notable drop in leachate pH, and was possibly stabilized prior to this study. The pH of aerated bioreactors stabilized at values above 7, while unaerated leachate pHs stabilized predominantly slightly below 7 in Figure 6.

Comparison of the time to stabilization for the aged waste samples is inconclusive. While the aerated bioreactors AO1 and AO2 stabilized on day 52 with pHs above 7, the unaerated aged bioreactors remained relatively unchanged and below pH 7. Stabilization for the aged waste may have already occurred before the inception of this study, and aeration only served to set a new stabilization pH.

In the fresh waste, the leachate pH was stabilized much more rapidly than in the unaerated bioreactors. For AF1 and AF2, stabilization occurred on day 52. Comparatively, it took 35 days longer on the unaerated bioreactors NF1 and NF2, which stabilized on day 87. Though conclusions from the aged-MSW data are inconclusive, aeration accelerated the neutralization of the fresh-MSW leachate.
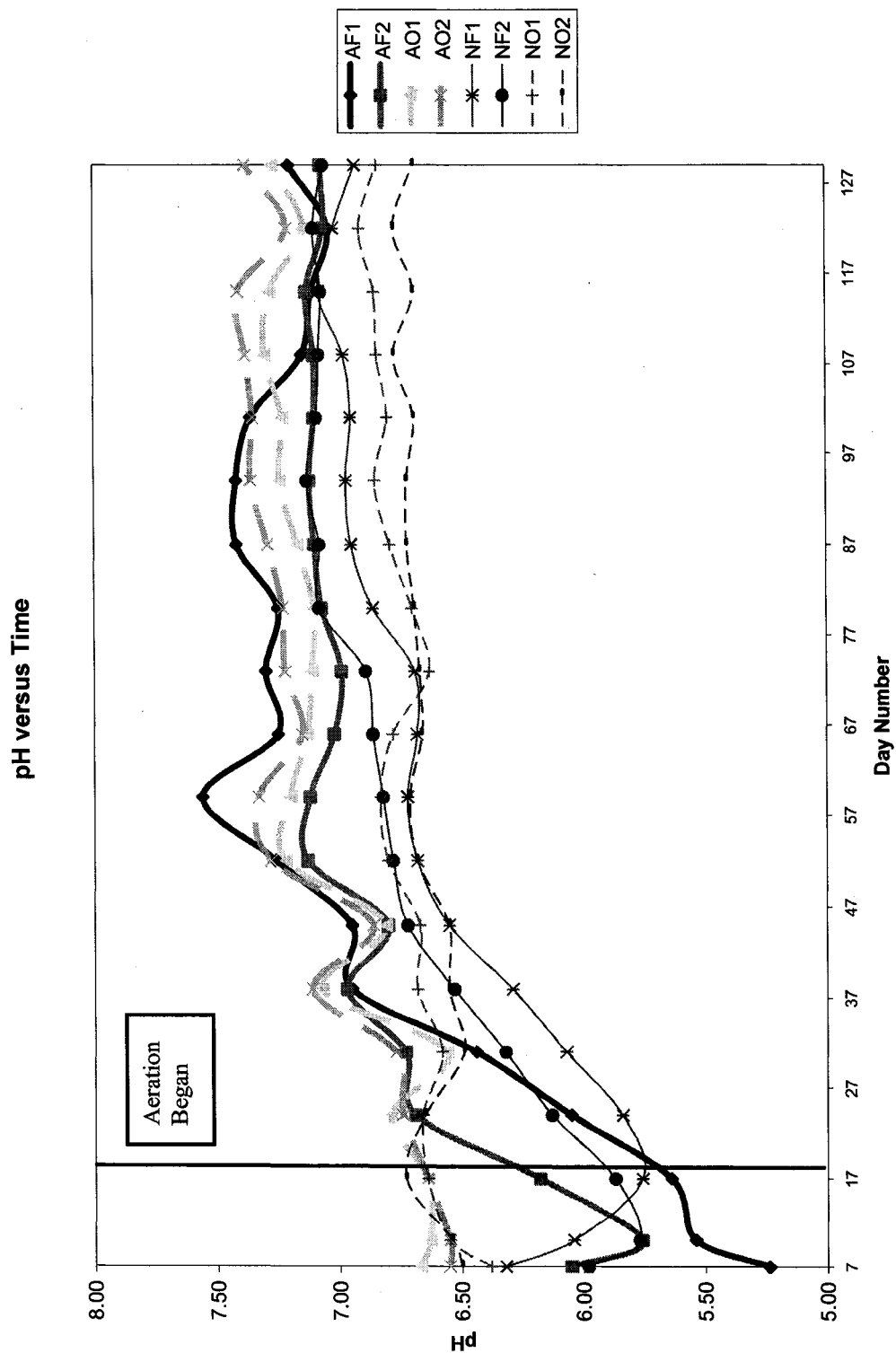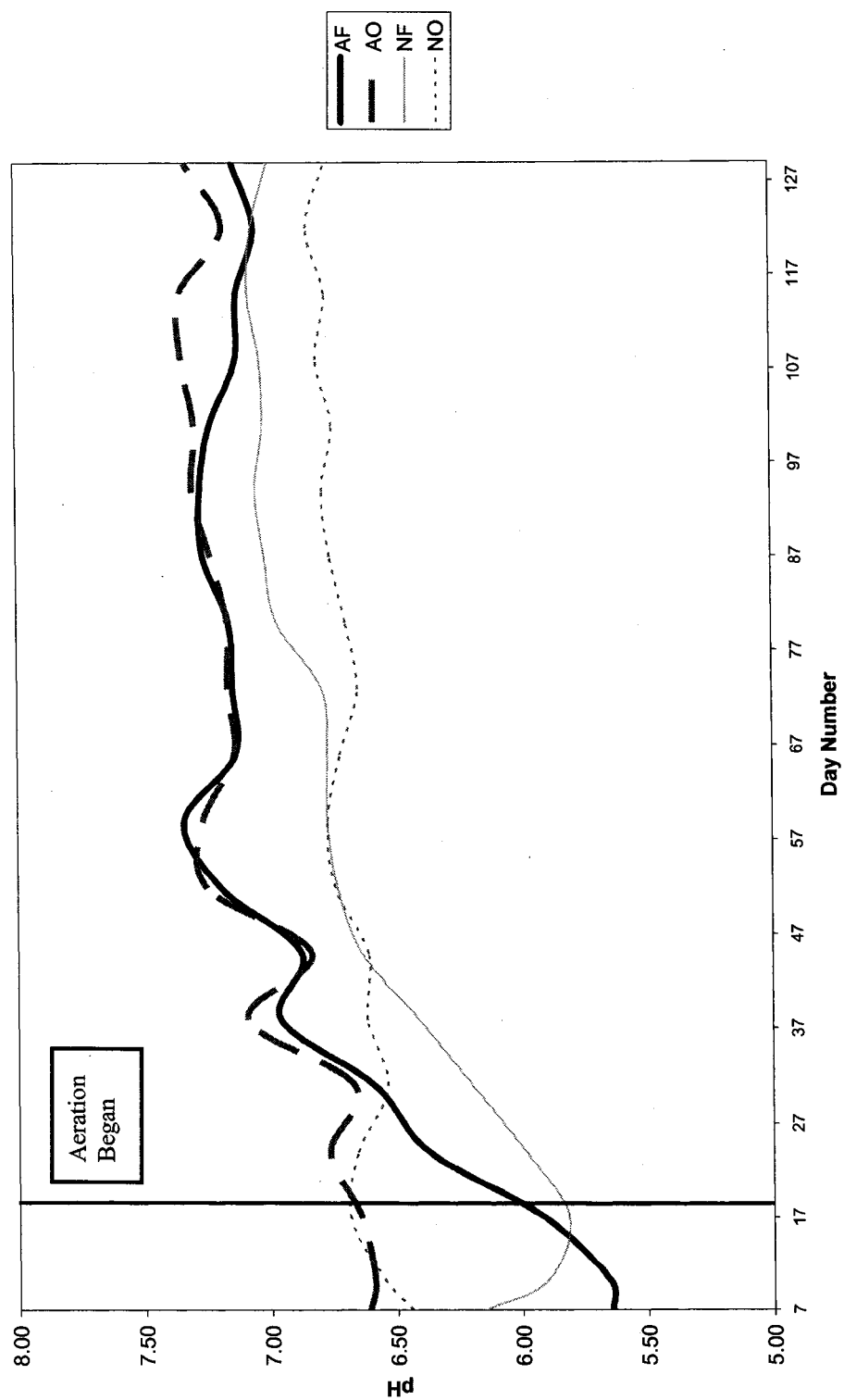
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Figure 5. Change in pH in the Individual Bioreactors.

Figure 6. Average Change in pH for Bioreactor Replicates.

5.2.3 Total Suspended Solids (TSS)

5.2.3.1 Initial Suspended Solids

Because the study began while the leachate was still relatively young, it had not yet reached an equilibrium TSS value with respect to the various bioreactors contents. As a result, TSS values in the initial period before aeration were highly erratic (Figure 7). The average TSS concentration of the fresh-MSW bioreactors AF1, AF2, NF1, and NF2 over the period spanning day 7 through day 17 inclusively yielded a higher TSS concentration than the average TSS of the aged-MSW bioreactors AO1, AO2, NO1, and NO2 prior to aeration.

In practice, erratic results are commonplace with leachate solids analysis, as the test itself typically yields widely fluctuating results. For this reason, it was imperative to have a large number of sampling events to ensure an adequate overall description of the leachate TSS concentrations was provided.

5.2.3.2 Effect of Aeration on Suspended Solids in Aged MSW

The suspended solids in AO1 and AO2 leachate remained somewhat erratic, with TSS concentrations increasing for the first week after aeration, with peak TSS concentrations of 152mg/L and 184mg/L reached on day 24 and day 31 respectively. This trend reversed rapidly and on day 24 and day 31 the TSS for AO1 and AO2 began a steep decline. On day 45, the TSS concentration of both AO1 and AO2 were not detectable using method 2540D (APHA 1989) on a balance accurate to ± 1mg. The TSS for these samples only deviated slightly through the remainder of the study.

The leachate from bioreactor NO1 performed similarly to bioreactor AO2, peaking on day 31 with a TSS of 160mg/L, and stabilizing on day 45. However, the TSS at stabilization was notably higher, averaging 35mg/L for the remainder of the study. NO2 declined steadily, achieving TSS concentrations comparable to NO1 on day 66. Similarly, NO2 stabilization TSS concentrations were higher than the aerated bioreactors AO1 and AO2, averaging 39mg/L over the remaining 63 days.

Both AO1 and AO2 experienced a rapid decline in TSS shortly after aeration began, and stabilized on day 45. NO1 also stabilized on day 45, however the stabilized TSS was higher than in the aerated bioreactors. NO2 stabilized on day 66, and by day 115 was comparable to NO1. Both anaerobic bioreactors maintained higher TSS concentrations than the aerated bioreactors until the final study day. Aeration appears to have accelerated the time to stabilization, where on average the aerated bioreactors stabilized on the day 45 and the unaerated bioreactors with aged MSW stabilized on day 56. Furthermore, the aerated bioreactors had stabilized TSS concentrations lower than the values for the unaerated bioreactors.

5.2.3.3 Effect of Aeration on Suspended Solids in Fresh MSW

The aerated fresh waste bioreactors AF1 and AF2 both experienced little fluctuation in their TSS after day 73, though their prior conditions differed. The TSS of the leachate from AF1 peaked at 277mg/L on day 45, followed by a significant reduction in leachate TSS over the next 14 days to 48mg/L. From day 59 to day 129, the leachate TSS decreased steadily, with a final TSS of 7mg/L. After aeration on day 18, the TSS of bioreactor AF2 decreased throughout the remainder of the study. It reached a TSS of 21mg/L on day 66, and reacted similarly to AF1 afterwards.

Comparably, the TSS of NF1 acted more erratically, dropping to 40mg/L on day 31, and increasing to 100mg/L by day 52. Afterwards, the TSS of NF1 declined steadily to 40mg/L again by study end.
NF2 fluctuated the most, with a maximum TSS of 326mg/L on day 52. By day 129, the TSS of NF2 remained high at 133mg/L. The final TSS of NF2 changed little from day 87, however the magnitude of the TSS fluctuations had greatly diminished.

NF1 stabilized on day 94 with a TSS of 40mg/L. At the end of the study, the final TSS remained unchanged at 40mg/L. NF2 experienced little fluctuation in TSS after day 87, though the TSS remained high throughout the remainder of the experiment such that at the end of the study it had yet to reach a TSS comparable to the other bioreactors.

The aerated bioreactors AF1 and AF2 stabilized considerably earlier, on day 73 and day 66 respectively. The TSS for these samples on these dates was 21mg/L, which was already lower than the TSS of NF1 and NF2 at the end of the experiment. The final TSS for AF1 at the end of the study was 7mg/L, and AF2 had a final TSS of 14mg/L.

5.2.3.4 Overall Trends in Suspended Solids and Leachate Stabilization

Overall, a decrease in the leachate TSS concentration occurred, with the exception of bioreactor NF2. While initial TSS values fluctuated erratically, the magnitude of these fluctuations decreased over time. An absence of large spikes in a steady TSS concentration was synonymous with stabilized conditions (Figure 8).

The average time required for stabilization to occur was shorter in bioreactors implementing air injection. In the aged-waste bioreactors, aeration shortened the time to stabilization after day 18 from an average 38 days for NO1 and NO2 to 27 days for AO1 and AO2. All of the aged samples had final TSS values of 0mg/L, but this was only reached by the non-aerated bioreactors on the final analytical session. The cause of a low final TSS in both the aerated and unaerated bioreactors is likely due to the initial waste composition, in which most organic materials had already been utilized such that little remained for uptake into the leachate of the anaerobic bioreactors.

In the fresh-MSW bioreactors AF1 and AF2, stabilization occurred after an average 52 days of aeration. Their unaerated counterparts required longer: NF1 required 76 days to reach a comparable TSS concentration, and NF2 remained high throughout the study, and had a TSS on day 129 of 133mg/L.

Final and stabilized TSS values of the aerated bioreactors also differed from the anaerobic conditions. In the fresh-MSW bioreactors, final TSS concentrations in AF1 and AF2, at 7mg/L and 14mg/L, were considerably lower than NF1 and NF2, which were 40mg/L and 133mg/L at the end of the study. The final TSS for NF2 is indicative of the long potential for pollution from fresh municipal solid waste entombed without aeration.

Aeration both reduced the time to stabilization and the final stabilized TSS concentration in the fresh MSW bioreactors. Though similar advantages were observed in the aged samples, the initial TSS concentrations were low such that reductions in TSS were not noteworthy.
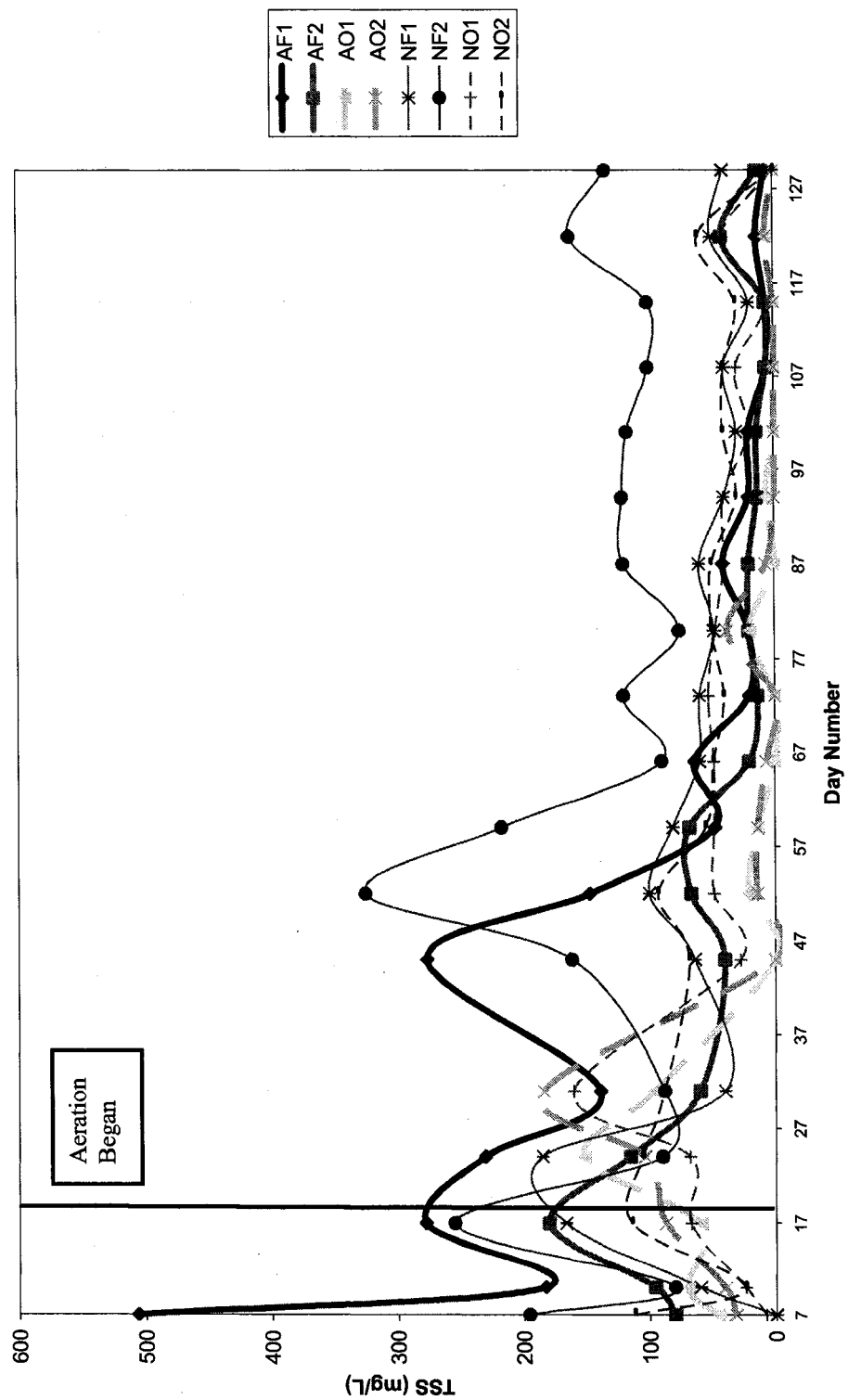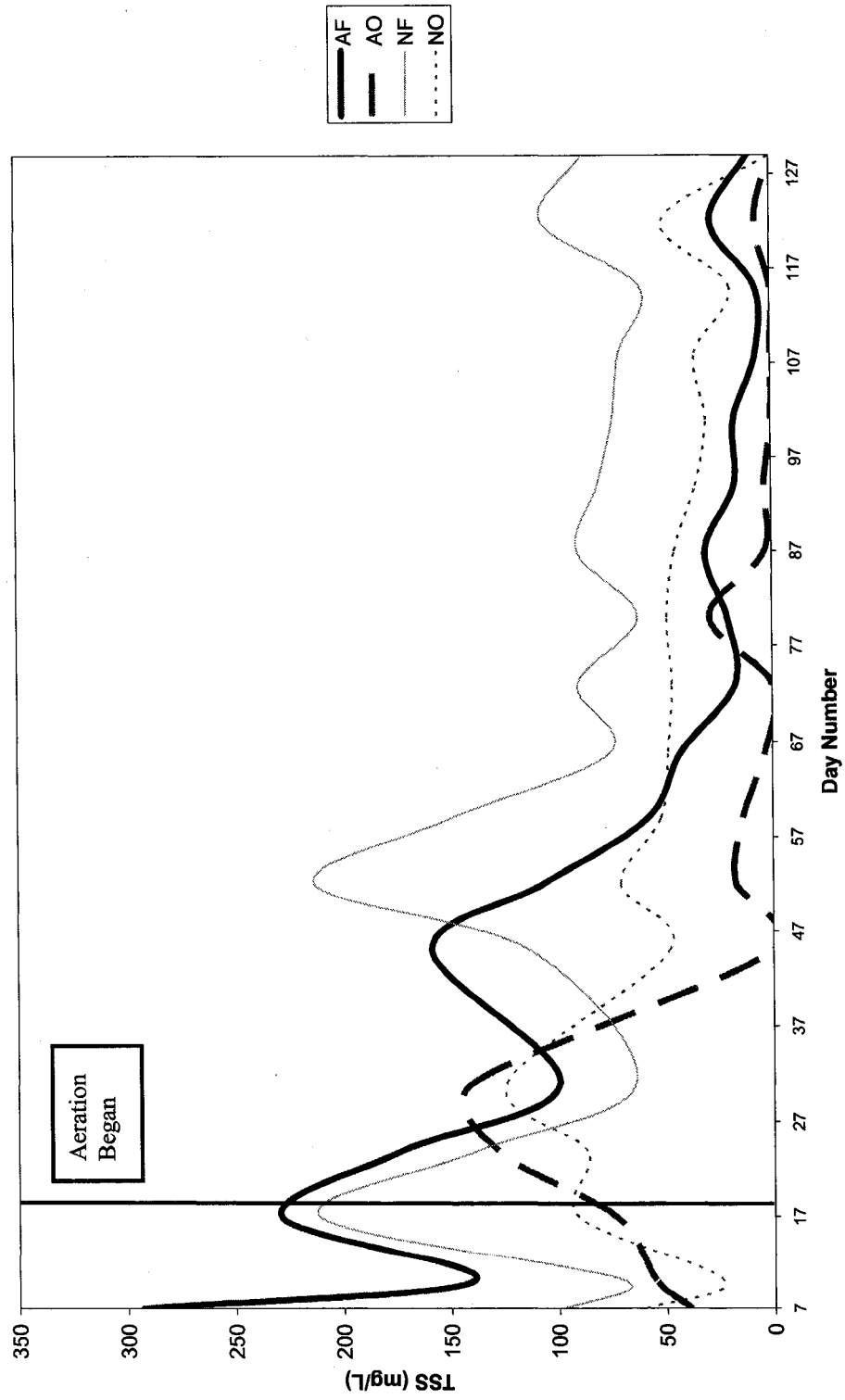
Figure 7. Change in TSS in the Individual Bioreactors.

Figure 8. Average Change in TSS for Bioreactor Replicates.

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5.2.4 Total Dissolved Solids (TDS)

5.2.4.1 Initial Dissolved Solids

The initial dissolved solids of the aged MSW bioreactors leachate ranged from 3000mg/L to 4450mg/L on day 18 of the study. In contrast, the leachate samples from fresh MSW bioreactors showed a greater variation in range for TDS and were higher in concentration, ranging from 6750mg/L to 11400mg/L (Figure 9).

5.2.4.2 Effect of Aeration on Dissolved Solids in Aged MSW

Aeration had little effect on the aged MSW. Both aerated and unaerated bioreactors experienced only a slight reduction in TDS concentration throughout the study duration. During the period of aeration, day 18 and day 129, the TDS dropped from 4450mg/L to 2250mg/L for NO1, from 3000mg/L to 2100mg/L for NO2, from 4050mg/L to 1650mg/L for AO1, and from 3550mg/L to 2800mg/L for AO2. This represents a 49% drop in TDS for NO1 and a 30% drop for NO2 over a period of 111 days. 59% and 21% drops in TDS for AO1 and AO2 over the same duration were also observed. Overall, the average reduction in leachate TDS for the aerated bioreactors, AO1 and AO2, and unaerated bioreactors, NO1 and NO2, were 40% and 39.5% respectively.

No differentiation in the rate of stabilization of aerated and unaerated aged samples could be determined. The four bioreactors containing aged waste appear to have been stable prior to aeration, though continuous leachate recycling may have aided the gradual drop in TDS demonstrated in all aged bioreactors over time, which averaged a reduction in TDS of approximately 2.5% per week for all aged samples.

5.2.4.3 Effect of Aeration on Dissolved Solids in Fresh MSW

Unlike the aged-waste bioreactors, the effect of aeration on leachate TDS is highly pronounced for the fresh MSW bioreactors. A rapid decline in the leachate TDS for AF1 reduced the TDS by 75% from 11400mg/L on day 17 to 2800mg/L, in just 21 days.

Similarly, the TDS of AF2 dropped 51% from 6750mg/L to 3300mg/L over the same duration ending on day 38.

In both of the unaerated fresh-MSW bioreactors NF1 and NF2, the TDS continued to rise after day 18, with peak TDS concentrations of 7950mg/L and 9250mg/L respectively on day 31. A gradual decline in TDS followed day 31, with possible stabilization of NF1 on day 122 and NF2 on day 108. Except for day 122, the average TDS for the unaerated bioreactors NF1 and NF2 remained higher than AF1 and AF2 for the duration of the study.

The TDS of leachate from AF1 stabilized on day 59 and did not change appreciably for the following 49 days. On day 108 the TDS for AF1 increased inexplicably, establishing a new stabilized TDS concentration. AF2 stabilized on day 52, with a TDS of 2400mg/L, which did not change appreciably by the end of the study 77 days later, with a final TDS of 2350mg/L.

The time to stabilization experienced with aeration was considerably shorter than for the unaerated bioreactors NF1 and NF2. NF1 may have stabilized on day 122, though there are insufficient data points afterwards to verify this possibility. Similarly, the TDS of NF2 varied little after day 108. The final TDS concentration of 4000mg/L for NF2 is higher than the other 7 test bioreactors, such that continued treatment would have likely resulted in a further reduction in TDS.

5.2.4.4 Overall Trends in Dissolved Solids and Leachate Stabilization

Leachate TDS concentrations from aged MSW samples gradually declined during the study, without any noticeable effect from aeration (Figure 10). In the fresh waste, anaerobic bioreactors also declined steadily, while aerated bioreactors experienced a rapid decline in TDS following aeration.

On average, the aerated bioreactors AF1 and AF2 stabilized 60 days earlier than the unaerated bioreactors NF1 and NF2. A rapid reduction in the TDS of AF1 and AF2

followed aeration, such that the average TDS from the aerated bioreactors remained lower than the non-aerated fresh-waste TDS concentrations for the duration of the study, excluding day 122. No appreciable difference was observed in the aged-waste bioreactors, likely due to a degree of stabilization that was initially high.
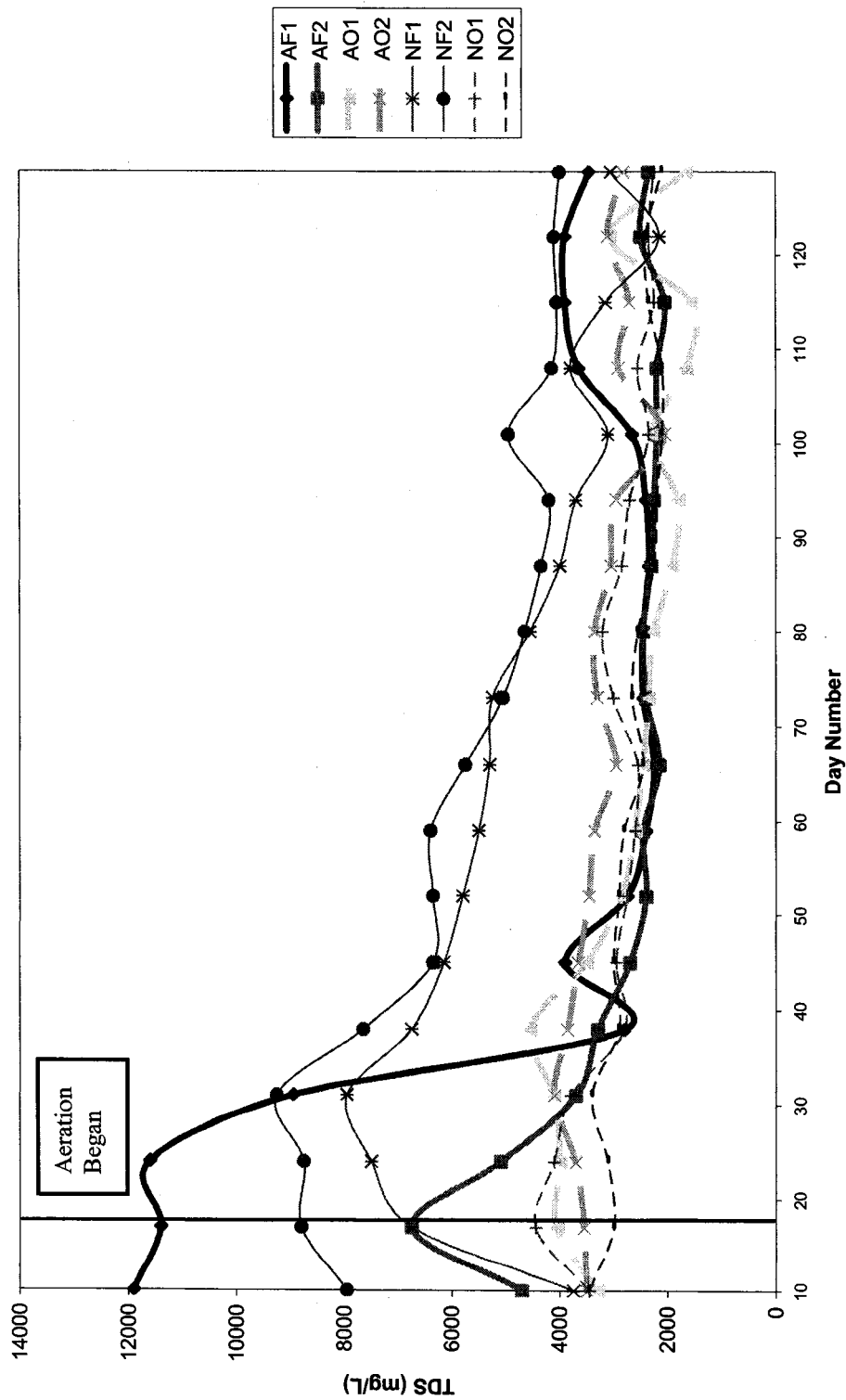
TDS versus Time

Figure 9. Change in TDS in the Individual Bioreactors.
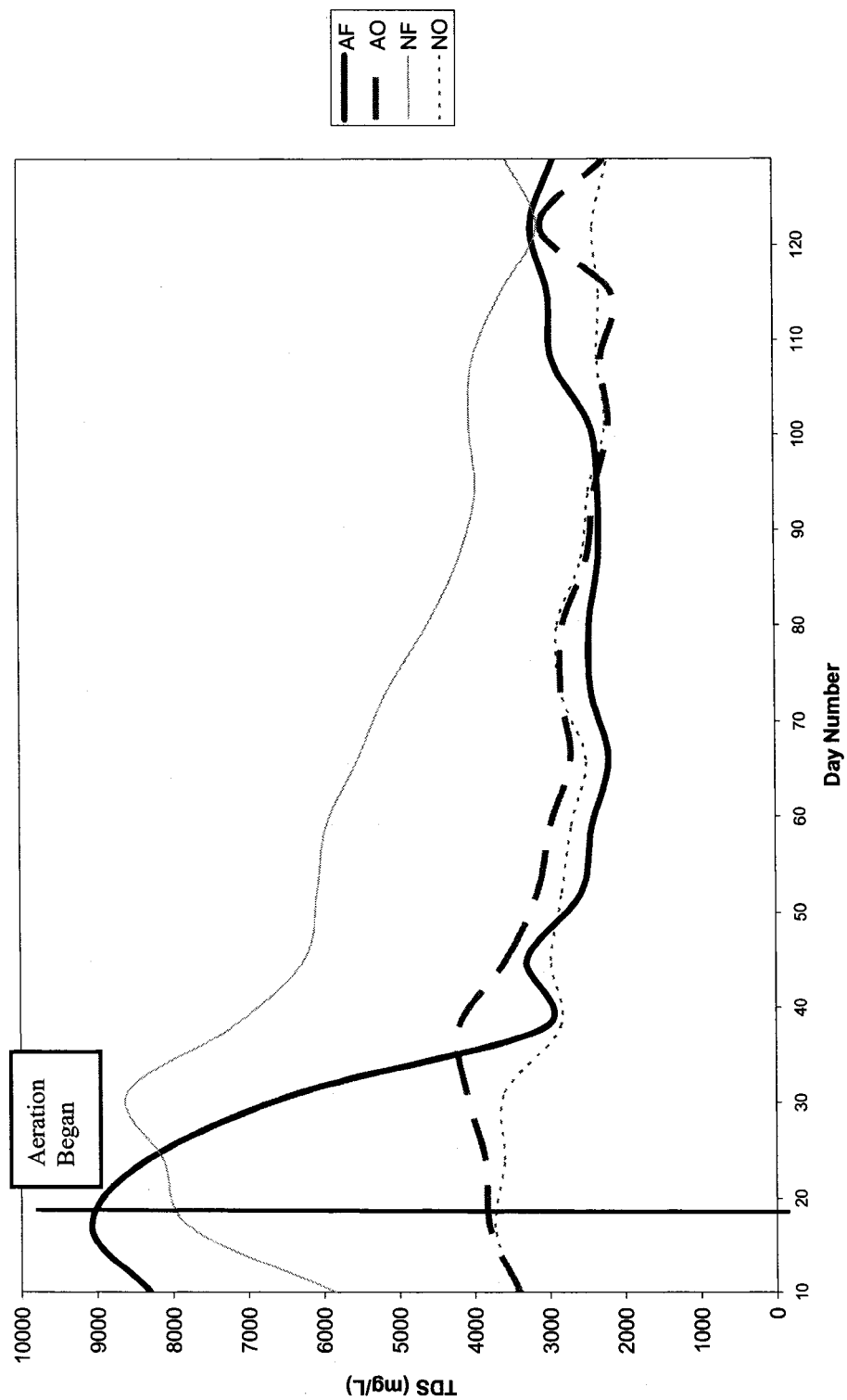
80

TDS versus Time for Averaged Replicates

Figure 10. Average Change in TDS for Bioreactor Replicates.

5.2.5 Ammonia-Nitrogen (NH$_3$)

5.2.5.1 Initial Ammonia Concentration

At the initiation of leachate sampling, the ammonia concentrations of all eight bioreactors ranged from 0.32mg/L to 3.48mg/L (Figure 11). Prior to aeration, the ammonia concentration of bioreactor AF2 increased to 13.54mg/L, while the concentrations on day 18 of all other bioreactors remained below 10mg/L. In this duration, the ammonia concentration increased in all bioreactors except AO2 and NO2, which experienced a slight drop in concentration.

5.2.5.2 Effect of Aeration on Ammonia in Aged MSW

Little reliable data were gained from the aged samples, as the ammonia concentrations in the aged-waste bioreactors remained very low throughout the study. The ammonia concentrations of NO1 and NO2 peaked on day 24 and declined shortly afterwards. Low ammonia concentrations dominated for the remainder of the study. During this period, the ammonia concentrations in NO1 and NO2 remained below 3mg/L and 2.78mg/L with average concentrations of 1.46mg/L and 1.81mg/L respectively. The aerated bioreactors AO1 and AO2 behaved similarly. The peak ammonia concentrations for AO1 and AO2 occurred on day 10, and maintained low concentrations for the remainder of the study period. These bioreactors had lower NH$_3$ concentrations, averaging 0.37mg/L and 0.13mg/L for AO1 and AO2 respectively over the final 91 days.

No difference in the time to stabilization can be extracted from this data, as the aged samples appear to be stable with respect to ammonia. A period of comparatively low NH$_3$ concentrations began on day 38 for NO1, AO1, and AO2 and on day 31 for NO2. These dates should not be mistaken for stabilization dates, as the ammonia-producing capacity of the waste was negligible at the inception of the study.

5.2.5.3 Effect of Aeration on Ammonia in Fresh MSW

Considerable variations in the ammonia concentrations for the fresh-waste bioreactors were observed, with NF2 and AF1 reaching values an order of magnitude higher than NF1 and AF2. This large variation is likely due to differences in MSW homogeneity, such that the bioreactors NF2 and AF1 may contain a larger degree of proteinaceous matter, such as meats and other putrescibles. For this reason, comparisons of the ammonia concentrations do not provide an accurate interpretation of the microbial processes occurring in the waste samples. Instead, focus should be directed to how ammonia concentrations changed over time in each bioreactor exclusively. The results of this analysis are presented in the following order: NF1, NF2, AF1, AF2.

Between day 17 and day 24, the ammonia concentration in bioreactor NF1 increased dramatically. The ammonia concentration then began to decline, such that on day 59 the ammonia concentration was at approximately half of its day 24 peak of 79.0mg/L. The ammonia concentration continued to decline at a slower rate until the end of the study. On day 129, the ammonia concentration in the leachate from NF1 was 4.01mg/L. Unexpectedly, the results for NF1 are somewhat parabolic and skewed to the left with trailing occurring over time. Under anaerobic conditions, the ammonia concentration in this bioreactor was expected to rise somewhat logarithmically, with a large initial increase in ammonia concentration giving way to a more constant ammonia concentration. Since removal of ammonia was supposed to be inhibited due to the lack of oxygen, the gradual reduction in ammonia concentration in this bioreactor was not anticipated.

Repeated sampling may have caused this reduction in ammonia concentration over time. After leachate was sampled from the LCR, the amount remaining in the LCR was diluted with the addition of fresh water, such that a reduction in the ammonia concentration occurs. If weekly ammonia production was not enough to replenish the LCR ammonia concentration, a gradual decline in ammonia concentration would be expected over time. However, if this was the case then a likewise drop in ammonia concentrations should have been observed in all bioreactors. This was not the case.

NF2 behaved as expected. The ammonia concentration in leachate sampled from NF2 began to rise rapidly in the same period as NF1, and did not reduce in concentration throughout the study. The ammonia concentrations after day 31 (525mg/L) remained high, such that at the end of the study 98 days later the ammonia concentration was reported to be 576mg/L. There was no indication of a gradual decrease in the concentration of ammonia, and it is likely this high ammonia concentration would continue for an extended period in a landfill application. In the United Kingdom, Burton and Watson-Craik (1998) reported that not one existing landfill was determined to be stabilized with respect to ammonia.

The changes in ammonia concentration in the aerated bioreactor AF1 yielded interesting results. On day 31, the ammonia concentration in the aerated bioreactor rose to 348mg/L, which was not expected since aerobic bacteria can readily utilize NH_3 as a substrate, and converting it to nitrite through nitrification (Burton and Watson-Craik 1998). Nitrite is easily biodegraded or transformed within the landfill. However, plotting the ammonia concentration over time describes a rounded curve beginning with a rapid increase in ammonia concentration (day 24 to 52) followed by a period where the concentration remained somewhat stable (day 52 to 101), and ending in a period of declining ammonia concentration (day 101 to 129). The 18-day period prior to this curve is considered a lag-phase before bacterially induced ammonia production could begin. This curve suggests that a second lag phase should be considered, which encompasses the time between the initial generation of ammonia and the beginning of its removal through bioconversion by aerobic bacteria. It is difficult to pinpoint when the utilization of ammonia began, though it was likely within the timeframe of day 32 and day 52. By day 32, the rate of ammonia removal equaled the rate of ammonia generation, as evidenced by a stable ammonia concentration for the following 49 days. As the rate of removal increased, it eventually surpassed the rate of ammonia generation, such that ammonia concentrations began to decline on day 101. By the end of the study, the ammonia concentration was 7.93mg/L. If the study continued past day 129, it is expected that the ammonia concentration would

remain stable at or below this low level, due to the greater microbial potential for ammonia utilization than generation.

AF2 performed similarly to AF1, with a significant increase of generation beginning on day 10 and continuing until day 24. After day 24 ammonia concentrations fluctuated, but overall experienced a gradual decline. The rate of ammonia removal in AF2 was slower than AF1, but resulted in a comparable final concentration. The final leachate concentration on day 129 for AF1 and AF2 were 7.93mg/L and 6.96mg/L respectively. Extending the study duration would have likely resulted in a further reduction in leachate ammonia concentrations.

AF1 and AF2 were near stabilization on the final day of the study with a significant reduction in the accumulated leachate ammonia concentrations. NF1 also experienced a drop in ammonia over time, with final ammonia concentrations comparable to the aerated bioreactors. This activity was unexpected. NF2 experienced a large increase in ammonia, and maintained a high ammonia concentration through the study. Graphing the concentration of ammonia over time did not suggest that the ammonia concentration would decline in the near future, and stabilization had not yet occurred by the study's end.

5.2.5.4 Overall Trends in Ammonia-Nitrogen and Leachate Stabilization

The aerated bioreactors AF1 and AF2 both experienced three phases concerning ammonia concentrations. These phases include a period of ammonia accumulation, a period of stable ammonia concentrations, and finally a period of ammonia reduction. The non-aerated bioreactor NF2 generated ammonia in large amounts, and was unable to provide for its subsequent removal. The performance of NF1 showed a somewhat erratic decline in ammonia over time.

As anticipated, aeration provided a means to reduce the pollution potential of ammonia in landfill leachate. Aeration altered the overall direction of the biological activity within the bioreactors from increasing ammonia production to ammonia utilization. Comparison

of AF1 and AF2 to NF2 indicated that aeration encouraged the biodegradation of previously accumulated ammonia by achieving a rate of utilization greater than ammonia was generated.

Since NF2 did not indicate a trend for future ammonia reduction, a comparison between aerobic and anaerobic degradation rates cannot be made; it can only be surmised that aeration allowed for the detoxification of the landfill leachate with respect to ammonia while the unaerated bioreactor NF2 did not.
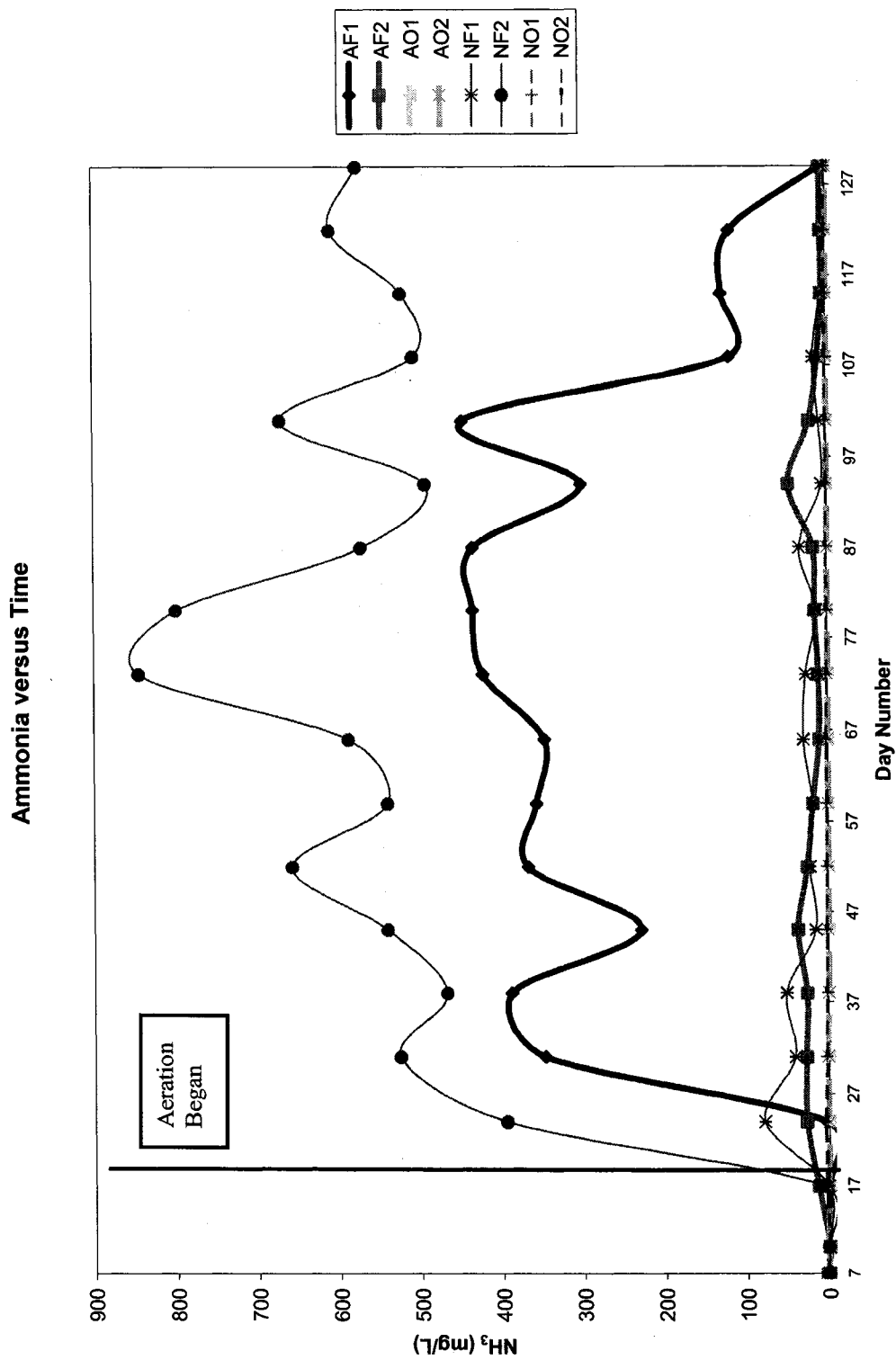
Figure 11. Change in Ammonia Concentration in the Individual Bioreactors.
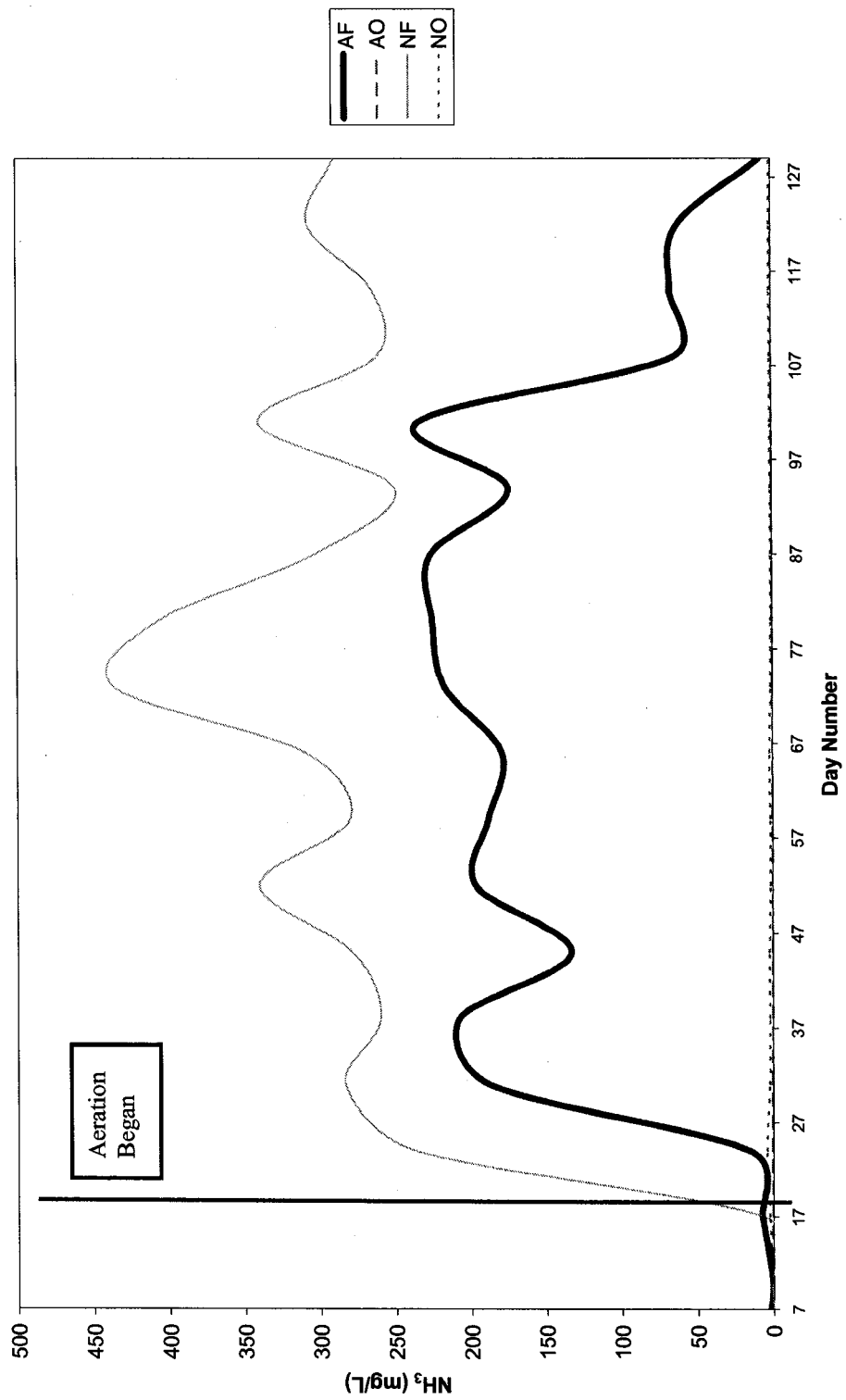
Figure 12. Average Change in Ammonia Concentration for Bioreactor Replicates.

5.2.6 5-Day Biological Oxygen Demand (BOD$_5$)

5.2.6.1 Initial BOD$_5$ Conditions

The initial BOD$_5$ increased continuously in the fresh-waste bioreactors, and to a small degree in the aged-waste samples (Figure 13). This rapid increase in the BOD$_5$ of the fresh bioreactors is due to their higher organic content, and is a result of mobilization of organic materials in the leachate. The aged samples did not increase as explicitly as the fresh waste, since the organic fraction of the aged samples was much smaller and more recalcitrant.

5.2.6.2 Effect of Aeration on BOD$_5$ in Aged MSW

The aerated bioreactors AO1 and AO2 had smaller peak BOD$_5$ concentrations, both of which occurred prior to the delivery of air to the system on day 18. The BOD$_5$ of the anaerobic reactors NO1 and NO2 reached higher concentrations on day 24 and 31. This indicates that aeration had an immediate effect on BOD$_5$, such that leachate from the aerated bioreactors did not continue to increase after aeration began with respect to biological oxygen demand. Furthermore, the period of elevated BOD$_5$ ended earlier in the aerated bioreactors, such that AO1 and AO2 returned to low levels on day 45, while NO1 and NO2 did not do so until day 59 and 73.

At the end of the study period, AO1 and AO2 had very low BOD$_5$ concentrations of 11mg/L and 9mg/L respectively. These values were approximately 18-fold and 6-fold lower than their peak values. NO1 and NO2 had higher final BOD$_5$ concentrations of 26mg/L and 53mg/L with 3-fold and 8-fold BOD$_5$ reductions respectively.

The BOD$_5$ of the four aged-MSW bioreactors stabilized on day 39 in the aerobic bioreactors AO1 and AO2, and on days 59 and 73 for NO1 and NO2 respectively. However, initial BOD$_5$ values were small. According to Pacey et al. (1999) leachate stabilization occurs when the BOD$_5$ is less than 100 mg/L. This occurred before aeration began in bioreactors AO1, AO2 and NO1. The leachate BOD$_5$ in bioreactor NO2 was stable after day 45 using this definition.

5.2.6.3 Effect of Aeration on BOD₅ in Fresh MSW

5.2.6.3 Effect of Aeration on BOD₅ in Fresh MSW

Maximum BOD_5 concentrations of 10050mg/L and 2265mg/L were reached on day 24 and day 17 for in the fresh aerated samples AF1 and AF2. Like the observations from the aged MSW samples, the unaerated bioreactors achieved maximum BOD_5 concentrations after the aerated series, on day 24 and day 31 for NF1 and NF2. These values were on average lower than the aerated bioreactors, with BOD_5 concentrations of 3050mg/L and 4950mg/L respectively.

BOD_5 declined rapidly in the aerated bioreactors. Between day 24 and 52, AF1 decreased from 10050mg/L to 210mg/L. AF2 decreased from 2265mg/L on day 18 to 150mg/L by day 38. In bioreactors NF1 and NF2, a more gradual reduction in BOD_5 was evident. NF1 decreased from 3050mg/L on day 24 to 137mg/L on day 94. NF2 dropped from 4950mg/L to 308mg/L between days 31 and 80. These two periods express an average BOD_5 reduction of 42mg/L and 95mg/L per day. Comparatively, the rapid declination phase in the aerated bioreactors had BOD_5 decreasing by 351mg/L and 101mg/L per day in bioreactors AF1 and AF2.

Final BOD_5 values in the aerated bioreactors were lower than in the anaerobic bioreactors. AF1 had a final BOD_5 of 92mg/L and AF2 was 45mg/L, while NF1 had a final BOD_5 of 127mg/L and NF2 was 303mg/L.

The time of stabilization was determined in the aerated bioreactors to be the point at which the leachate BOD_5 dropped below 100mg/L as defined by Pacey et al. (1999) with little fluctuation in concentration afterwards. This was determined to be day 59 and day 52 for AF1 and AF2.

The final BOD_5 concentrations in the unaerated bioreactors did not drop below 100mg/L. Stabilization was determined to be the point after which little fluctuation in the leachate BOD_5 occurred. In NF1, this took place on day 94, at which point the BOD_5 was

137mg/L. The final BOD$_5$ for NF1 on day 129 was 127mg/L. NF2 was stabilized on day 80, with a BOD$_5$ of 308mg/L. The final BOD$_5$ for NF2 was 303mg/L.

5.2.6.4 Overall Trends in BOD$_5$ and Leachate Stabilization

Rapid increases in leachate BOD$_5$ were observed in all fresh-waste bioreactors. Aeration resulted in an immediate steep decline in the leachate BOD$_5$ for the fresh-waste samples, while the BOD$_5$ reduced more steadily in the anaerobic bioreactors. The final BOD$_5$ of the aerated bioreactors were lower than their anaerobic counterparts. The aged samples were near or already stabilized at the beginning of the study, though similar BOD$_5$ trends to the fresh-MSW bioreactors were observed.

The MSW samples in the aged bioreactors were predominantly stable at the beginning of the study, such that the time to stabilization for aerated and unaerated aged waste could not be compared adequately. However, after a period of increased BOD$_5$ in these bioreactors, low and nearly constant BOD$_5$ values occurred. For AO1 and AO2, this stable period began on day 45, and occurred later on day 59 and day 73 for NO1 and NO2 respectively. If these days are used in place of stabilization, aeration allowed the aged MSW to stabilized more rapidly.

The influence of aeration is better expressed in the fresh-MSW samples. Stabilization occurred on day 60 and day 52 for the aerated bioreactors AF1 and AF2, which both achieved BOD$_5$ concentrations below 100mg/L.

In the unaerated bioreactors, the period of apparent stabilization occurred much later, beginning on day 94 and day 80 for NF1 and NF2 respectively. The BOD$_5$ values in these bioreactors were greater than 100mg/L, and remained so after the end of the study. According to Pacey et al. (1999), these samples are required to undergo further reductions in BOD$_5$ before being considered stable. However, the designation of 100mg/L BOD$_5$ as the criteria for stabilization by Pacey et al. (1999) is rather arbitrary.

Figure 13. Change in Average BOD_5 in the Individual Bioreactors.

Average BOD₅ versus Time for Averaged Replicates
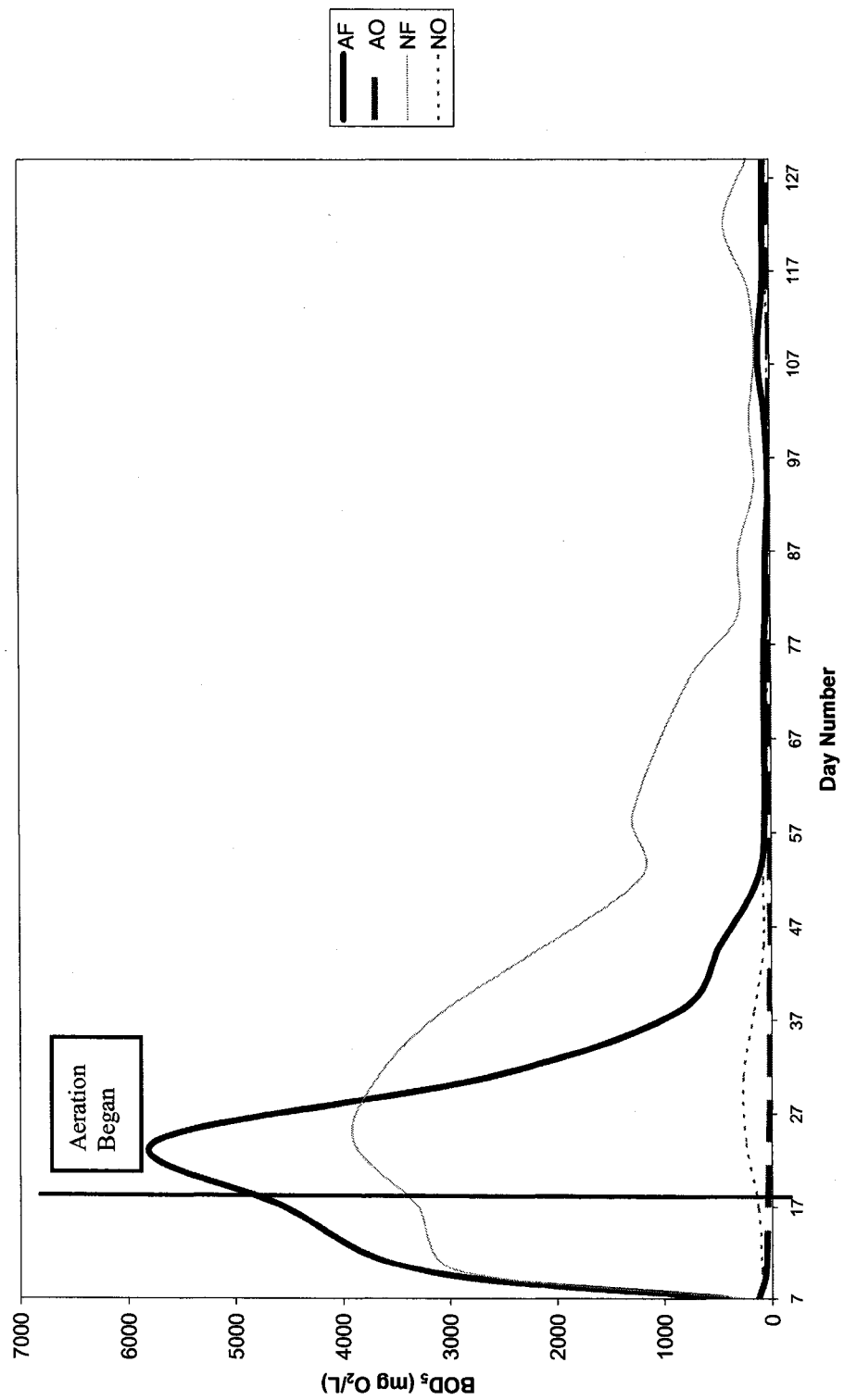
Figure 14. Average Change in BOD$_5$ for Bioreactor Replicates.

5.2.7 Chemical Oxygen Demand (COD)

5.2.7.1 Initial COD Conditions

The COD of the fresh-waste bioreactors AF1, AF2, NF1 and NF2 increased throughout the 11 days prior to aeration (Figure 15). In the aged waste bioreactors, the same trend occurred, though increases were lower in magnitude. With the exception of NF1, all of the bioreactors produced peak leachate COD concentrations on day 17. Aeration followed on day 18.

5.2.7.2 Effect of Aeration on COD in Aged MSW

Like in the BOD_5 results, the aged MSW had a significantly less potential for COD generation than the fresh waste bioreactors. However, the leachate from AO1, AO2, NO1 and NO2 all reached COD levels above 1000mg/L prior to aeration. According to Pacey et al. (1999), leachate is stabilized at COD concentrations below 1000mg/L.

The peak chemical oxygen demands of the aged leachate samples were all reached on day 17 of the study. The maximum COD in leachate from bioreactor AO1 was 1215mg/L, 1579mg/L in AO2, 2631mg/L in NO1 and 1579mg/L in NO2.

Stabilization of the leachate was defined as when the daily COD concentration continuously produced COD results within 100mg/L of the final COD on day 129. In the aerated bioreactors, AO1 and AO2 stabilized on day 52, with COD values of 196mg/L and 245mg/L respectively. The final COD values for these bioreactors were 131mg/L and 164mg/L. The unaerated bioreactors stabilized later than the aerated bioreactors, on day 94 for NO1 and 87 for NO2. The stabilized and final concentrations in these bioreactors were 311mg/L and 262mg/L for NO1 and 507mg/L and 450mg/L for NO2 respectively.

5.2.7.3 Effect of Aeration on COD in Fresh MSW

The COD of AF1 peaked at 22450mg/L before declining to 188mg/L at the end of the study. Similarly, AF2 experienced a rising trend in COD which was abruptly reversed after aeration began on day 18. The unaerated sample NF2 reached a COD of 22801mg/L before it eventually declined to 720mg/L. Like NF2, the COD of NF1 continued to tail, which prevented NF2 from reaching an acceptable stabilized value until day 80.

There was a large range in the magnitude of the COD for both aerated and non-aerated bioreactor leachate samples. The peak COD of 22450mg/L for AF1 was 13680mg/L above the peak COD in its replicate, AF2, which had a peak value of 8770mg/L. Similarly, in the unaerated bioreactors, NF2 reached a COD of 22801mg/L, which was greater than the peak COD of NF1 at 8770mg/L by 14031mg/L.

Defining COD stabilization as the point at which a predetermined COD concentration is insufficient for this analysis, since the amount of COD reduction required to reach a fixed value differs greatly depending on the sample. Instead, the leachate was considered stabilized when less than 10% of its peak COD remained, such that the COD at stabilization represents at least a 90% reduction in the overall COD of the leachate. Since this method incorporates the difference in magnitude of the bioreactors, it was preferred for this analysis.

Using 90% COD reduction as the criteria for stabilization, the stabilization dates for the bioreactors NF1, NF2, AF1 and AF2 were as follows: NF1 stabilized on day 108, NF2 stabilized on day 80, AF1 stabilized on day 52 and AF2 stabilized on day 31. It is evident from this data and visually from Figure 15 that the COD in the aerated bioreactors was reduced at a much faster rate.

By the end of the study, all of the fresh-MSW bioreactors had experienced a reduction in COD of 95% or more. For the unaerated bioreactors NF1 and NF2, a 21-fold and 32-fold reduction in leachate COD occurred. Comparatively, the final COD of AF1 was 119

times less than its peak COD, and AF2 was reduced by 45 times in the aerated bioreactors.

5.2.7.4 Overall Trends in COD and Leachate Stabilization

In both fresh and aged-MSW bioreactors, aeration accelerated the reduction in chemical oxygen demand (Figure 16). On average, the COD in the aerated fresh-waste bioreactors were reduced by more than 90% in 36 days. The average time required in the anaerobic bioreactors was 88 days, 52 days longer. Final COD values in the aerated bioreactors were also lower in COD than the anaerobic bioreactors. The final COD ranged from 131 mg/L to 196 mg/L in the four aerated bioreactors, while a range of 26mg/L to 720 mg/L spanned the anaerobic conditions. These results indicate that aeration not only accelerated the reduction of COD in both fresh and aged-waste bioreactors, it also reduced the final stabilization COD concentrations.
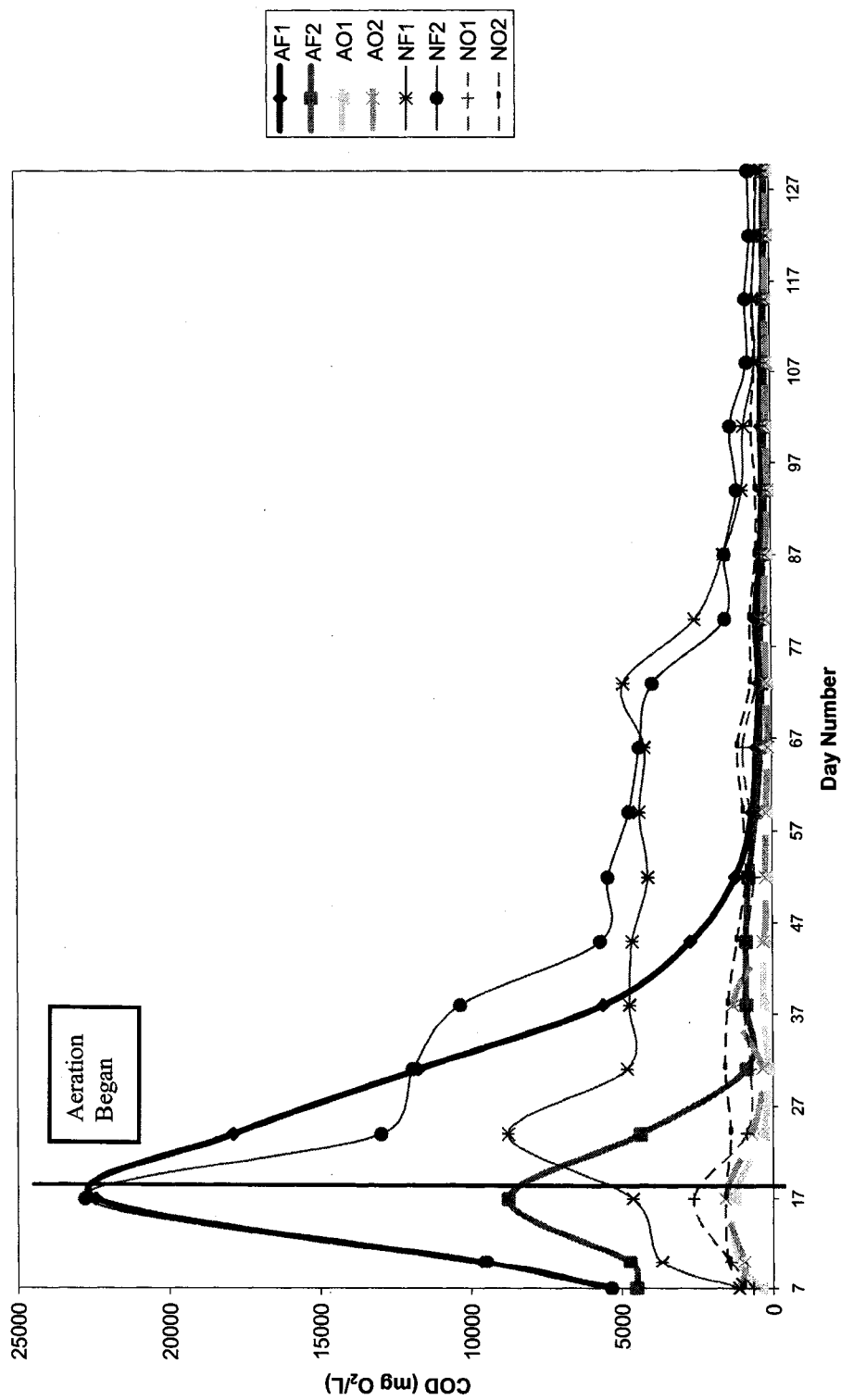
Figure 15. Change in COD in the Individual Bioreactors.
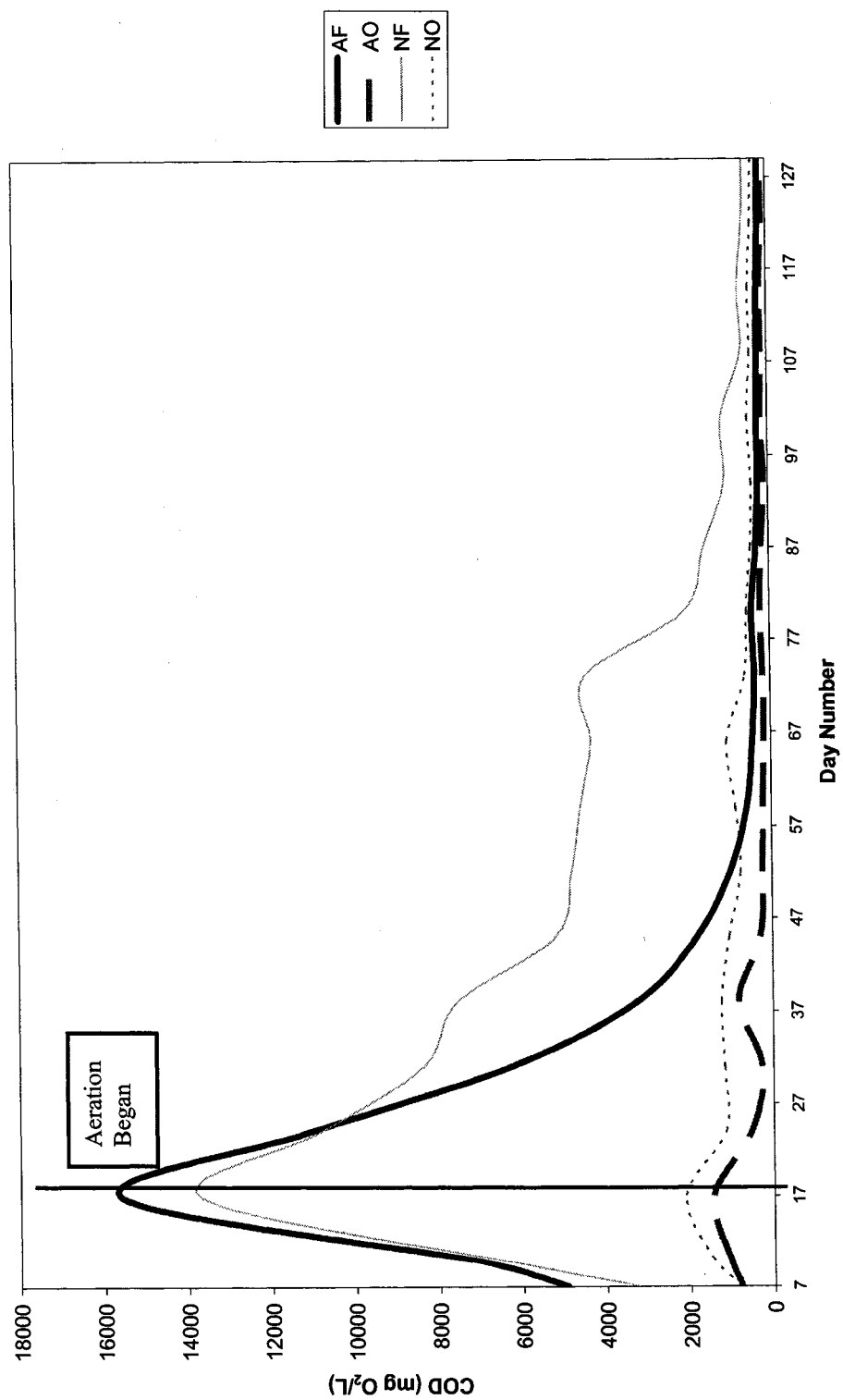
Figure 16. Average Change in COD for Bioreactor Replicates.

98

5.2.8 BOD$_5$/COD Ratio (BCR)

5.2.7.1 Initial BCR Conditions

The large initial uptake in both BOD$_5$ and COD in this study resulted in unstable BCR values, but generally showed BOD$_5$ /COD ratios between 0.3 and 0.6 for fresh waste and under 0.22 for the aged samples (Figure 17). Based on these ratios, fresh MSW has a moderate capacity for biodegradation and aged waste has a small capacity nearing stabilized values. However, because the BOD$_5$ and COD concentrations in the leachate samples were not relatively steady over time, the resulting initial BOD$_5$ /COD ratio is not very precise, and the true BCR was likely higher than reported. Typically fresh MSW can expect to have BOD$_5$ /COD ratios near 0.9 (Ehrig 1989). If given enough time, the organic load in the leachate would have continued to accumulate and likely resulted in a higher BDR. However, waiting for the BCR to stabilize also has the drawback of constant anaerobic biodegradation, such that a significant portion of the organic fraction could be consumed before aeration began.

5.2.7.2 Effect of Aeration on the BCR in Aged MSW

No notable influence on the BCR was observed in the aged-MSW bioreactors. These bioreactors maintained BCR values less than 0.1 for the majority of the study. The BCR increased to above 0.1 between day 24 and day 38 for NO1 and AO1 and to 0.28 for NO2 in the same duration. All four aged-MSW bioreactors appear to have been nearly stable prior to this study, with very little biodegradable organic matter to contribute to the leachate while a small fraction of recalcitrant organic compounds remained in the leachate.

Though all aged-waste bioreactors had initial BC ratios below 0.1 and are considered stable by Pacey et al.'s (1999) definition, a small increase in the BCR was noted in all bioreactors. The increase to the BCR experienced by bioreactor NO2 returned to its original BCR on day 45, while the other 3 aged-MSW bioreactors AO1, AO2, and NO1 returned to their initial BCRs by day 38.

5.2.7.3 Effect of Aeration on the BCR in Fresh MSW

BCR ratios in the aerated and unaerated bioreactors decreased over time. The aerated bioreactors decreased more rapidly, such that BCR values less than 0.1 were first obtained on day 59 and day 52 for AF1 and AF2. The unaerated bioreactors NF1 and NF2 did not result in BCRs less than 0.1 throughout the study, suggesting that they still have organic contents able to undergo further degradation.

AF1 and AF2 appear to have stabilized on day 59 and day 52 respectively, after which the BCR remained near to the value of 0.1, which Pacey et al. (1999) suggests as the criteria for stabilization. The BCR in the anaerobic bioreactors remained above 0.1, and did not stabilize under this definition.

5.2.7.4 Overall trends in the BCR and Leachate Stabilization

A more rapid decline in the BCR was noted in the fresh waste samples when subjected to aeration (Figure 18). Aged samples were near stabilization at the beginning of the study, such that no conclusions can be derived from the performance on the aged-waste bioreactors. After day 94, the BOD_5 and COD values in the fresh waste bioreactors were sufficiently low such that their error factors made the BOD_5 / COD Ratio data unusable, and data after day 94 was ignored in this regard.

BOD / COD Ratio versus Time
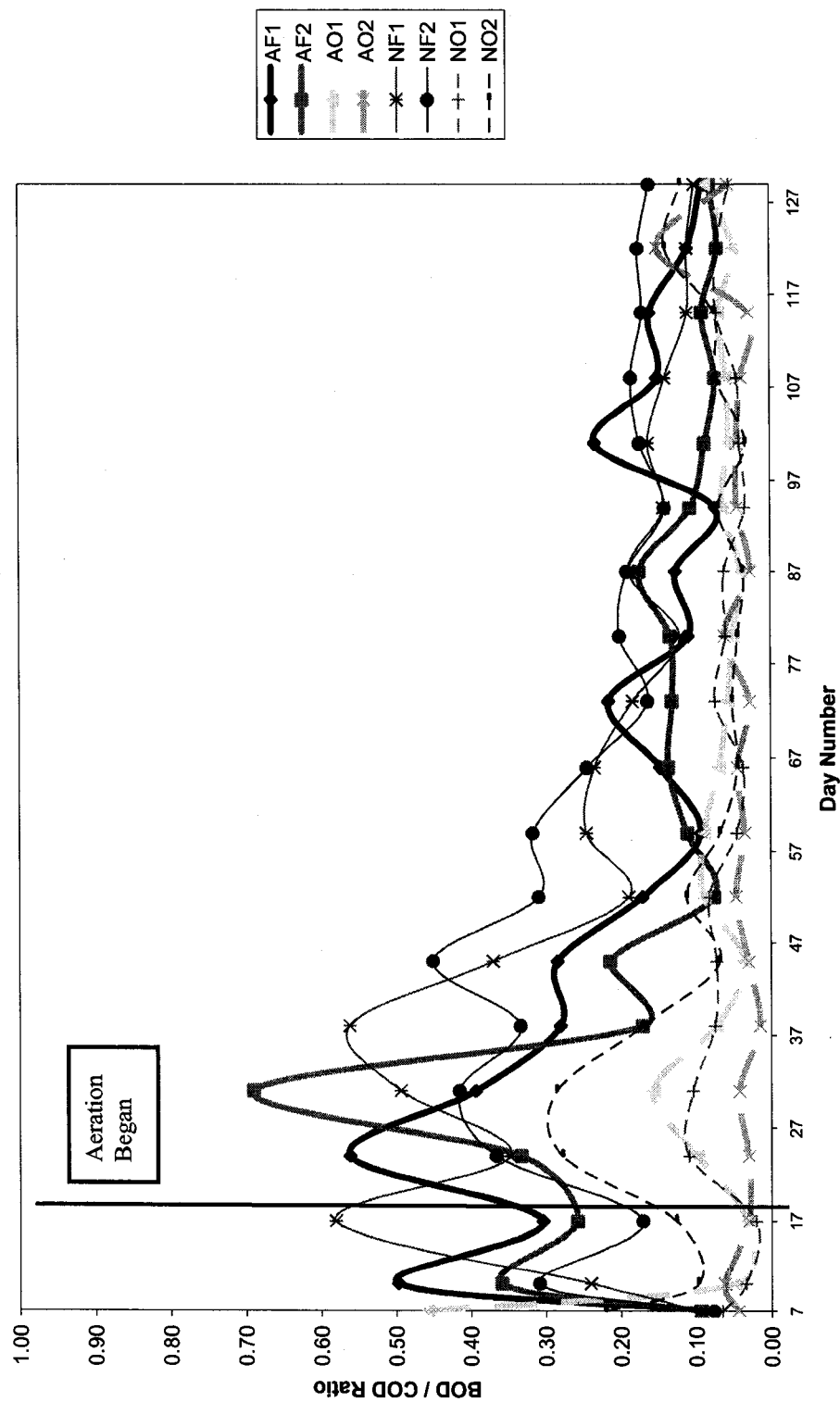
Figure 17. Change in BCR in the Individual Bioreactors.
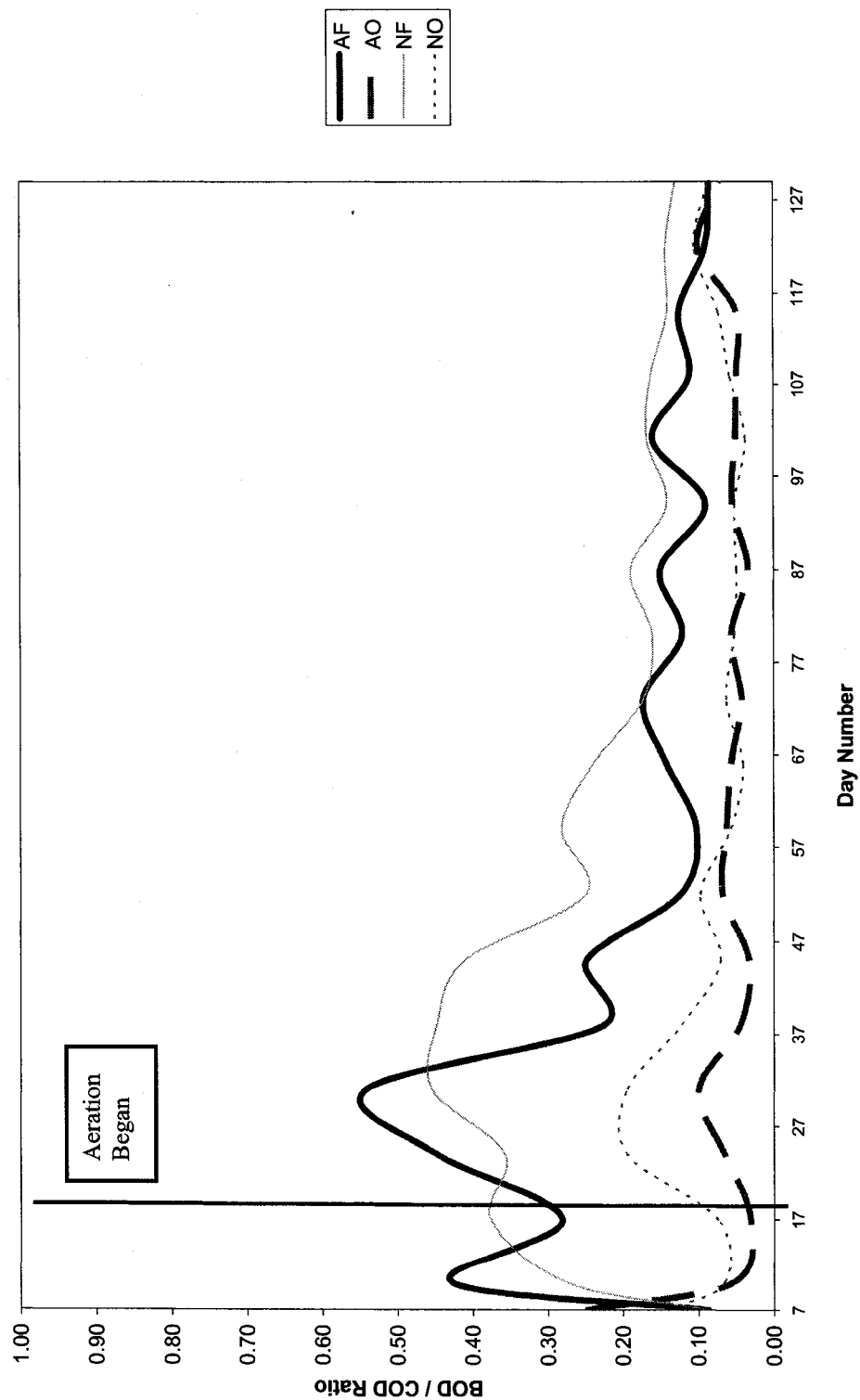
Figure 18. Average Change in the BCR for Bioreactor Replicates.

6. SUMMARY OF RESULTS

6.1 INFLUENCE OF AERATION ON LEACHATE STABILIZATION

Overall, the influence of aeration had very promising effects on MSW stabilization. In the fresh waste bioreactors, aeration accelerated the stabilization of each analytical parameter before the corresponding anaerobic counterpart by 25% to over 59%. In addition, final leachate concentrations of NH_3, BOD_5, and COD were consistently lower. Final values of TSS and TDS were more favourable in the Fresh Waste results, but no appreciable difference was apparent in the Aged Waste bioreactors. Table 14 summarizes the day of stabilization for each parameter and the corresponding concentrations at the time of stabilization and at the end of the study.

Prior to aeration, leachate BOD_5 and COD concentrations increased dramatically in the fresh waste bioreactors. An immediate reversal of this trend resulted after aeration began, such that the leachate was prevented from attaining the peak organic contents it may have expressed in a conventional or anaerobic bioreactor landfill. Furthermore, accumulated ammonia was gradually utilized and removed from the aerated systems. At the conclusion of the study, the ammonia concentration in the anaerobic fresh waste bioreactor NF2 was 576mg/L and showed no signs of further attenuation. Finally, leachate pH was neutralized more rapidly under aeration, and like the organic analyses, prevented the system from reaching excessive levels. With a more rapid neutralization of leachate pH, it is expected that the leachate would be less prone to the accumulation of soluble pollutants, such as heavy metals.

In the aged MSW samples, a similar trend was observed. However, since the aged waste was near stabilization at the beginning of the study, the overall potential and demonstration of accelerated waste stabilization through air injection was diminished.

Graphical representations of the overall leachate stability for the fresh and aged MSW are illustrated in Figure 19 and Figure 20. In Figures 19 and 20, the Ammonia, TDS, BOD_5

and COD concentrations are expressed as cumulative indicators of the overall leachate stability. The respective contributions to the overall pollution potential for each analyses drops to zero on its date of stabilization. In Figure 19, TDS, BOD_5 and COD are stabilized earlier in the aerated bioreactors than in the unaerated ones. Ammonia becomes stabilized on the last day for the aerated fresh waste bioreactors, but remains a continuing problem for the anaerobic fresh waste bioreactor at the end of the study. Similarly, Figure 20 illustrates that the aerated aged waste was stabilized earlier than in the corresponding anaerobic bioreactors.

6.2 INFLUENCE OF AERATION AND MSW AGE ON SETTLEMENT

Aeration resulted in a greater degree of settlement in both the fresh and aged wastes. An average 20.6% reduction in the MSW depth was observed in the aerated bioreactors and 17.5% in the anaerobic reactors. Fresh MSW settled more than aged MSW, with average reductions in MSW depth of 22.5% and 15.7% respectively. Although the bioreactors receiving air injection had settled to a greater degree, both conditions resulted in an excellent degree of settlement.

Table 13. Comparison of Aerobic and Anaerobic Treatments on the Day of Aerobic Stabilization and at the End of the Study for Fresh and Aged MSW based on Analytical Parameters.

Analysis	Unit	Criteria	AF	NF	AO	NO
pH		Day Number[1]	52	87	52	7
		Stabilization[2]	7.20	6.73	7.25	6.74
		Final[3]	7.14	7.00	7.33	6.77
TSS	mg/L	Day Number	70	91	45	56
		Stabilization	39	90	0	48
		Final	10	87	0	0
TDS	mg/L	Day Number	56	115	7	7
		Stabilization	2400	5925	3400	3450
		Final	2900	3525	2225	2175
Ammonia	mg/L	Day Number	129	X[4]	7	7
		Stabilization	7.45	4.01	0.67	2.39
		Final	7.45	290.04	0.03	1.35
BOD_5	mg/L	Day Number	56	87	45	66
		Stabilization	61	1378	9	72
		Final	68	215	10	39
COD	mg/L	Day Number	42	94	52	91
		Stabilization	1052	4779	221	745
		Final	192	565	147	356
BCR		Day Number	56	X	38	42
		Stabilization	0.085	0.277	0.044	0.123
		Final	0.358	0.365	0.071	0.108

[1] Day on which stabilization was achieved.
[2] Measure of the test parameter on the day stabilization was achieved in the aerated bioreactors.
[3] Measure of the test parameter on day 129.
[4] X = Did not stabilize

Overall Stability for AF

Overall Stability for NF
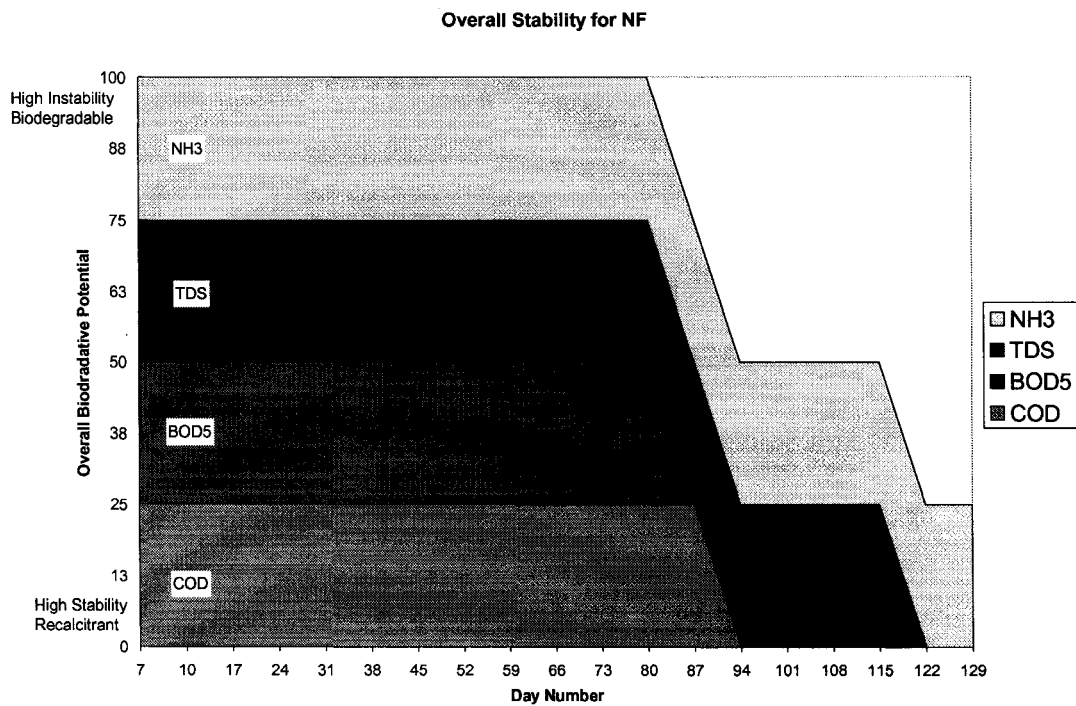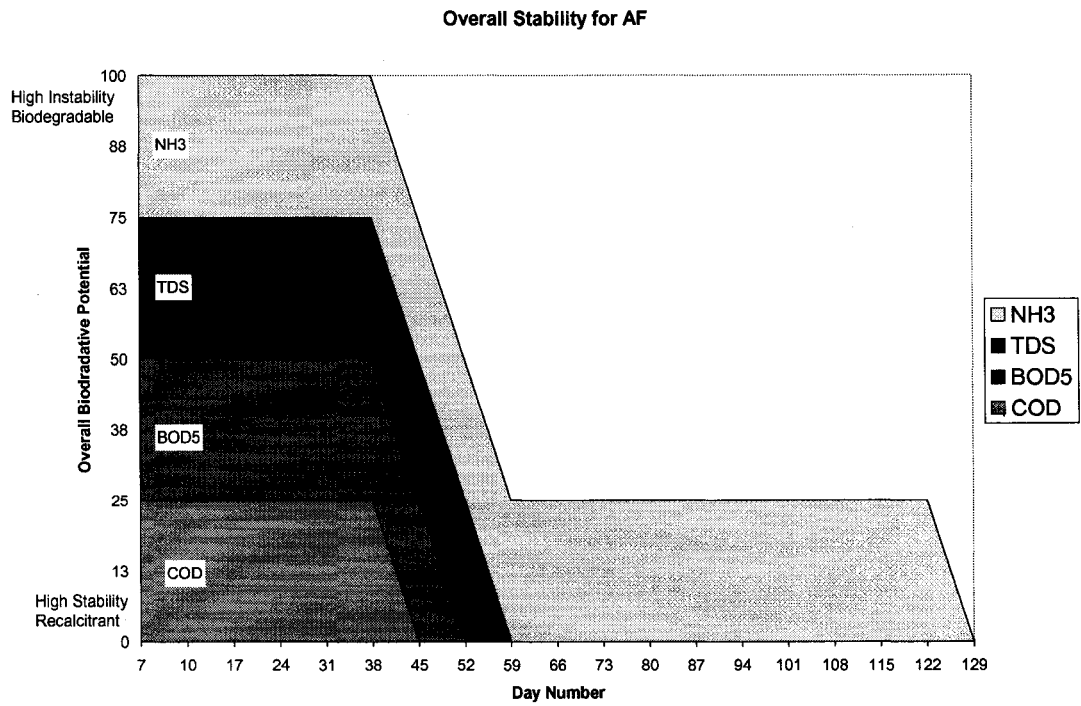
Figure 19. Cumulative Contribution of Ammonia, TDS, BOD$_5$ and COD to the Overall Biodegradative Potential of Fresh-Waste Bioreactor Leachate.

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Overall Stability for AO

Overall Stability for NO
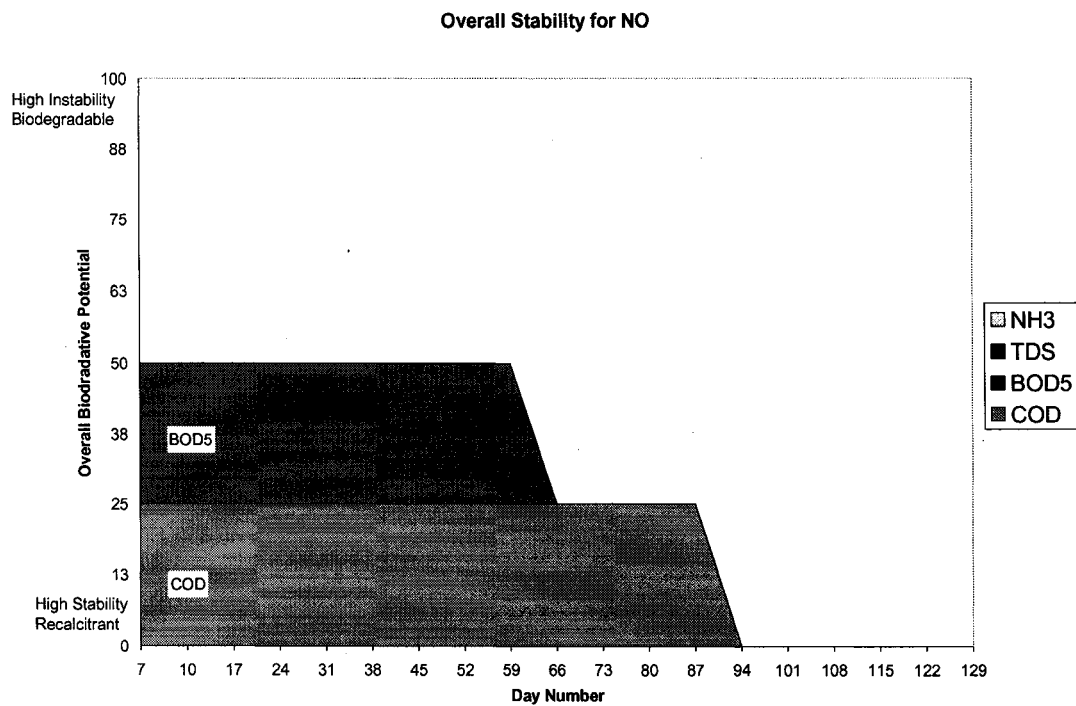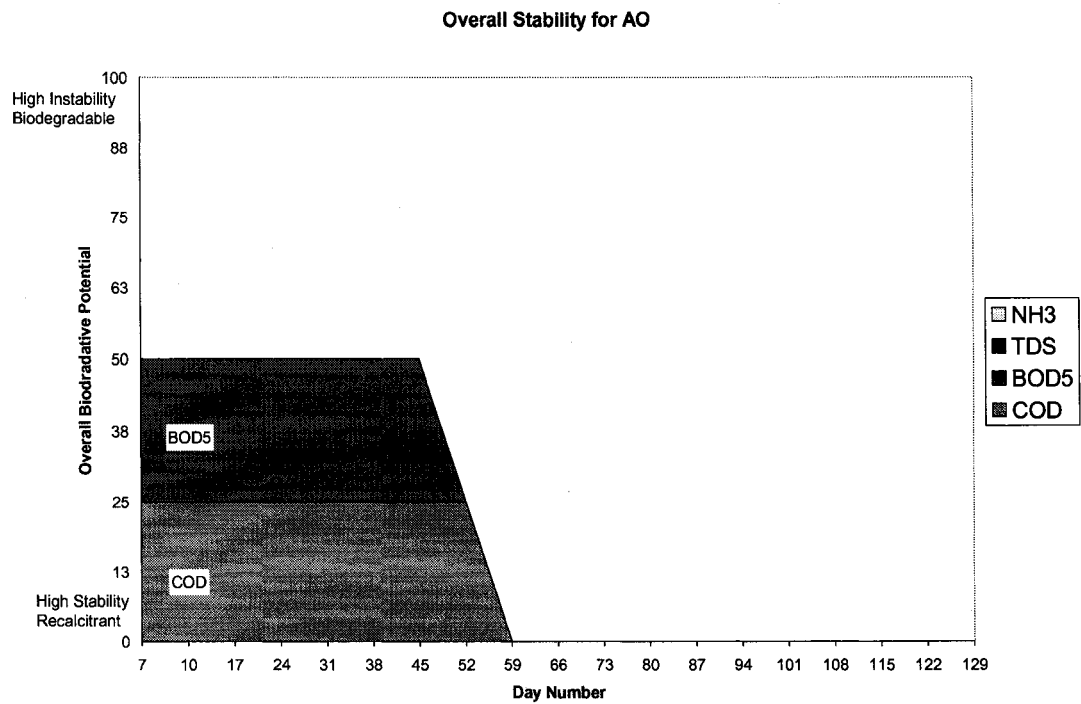
Figure 20. Cumulative Contribution of Ammonia, TDS, BOD5 and COD to the Overall Biodegradative Potential of Aged-Waste Bioreactor Leachate.

7. CONCLUSIONS AND RECOMMENDATIONS

7.1 CONCLUSIONS

Aeration was an effective method to rapidly reduce the pollution potential of leachate, and promote landfill settlement. The benefits of aeration were greater with the fresh waste since the fresh waste had a greater organic fraction. The aged waste was already significantly stabilized at the commencement of the study, such that the benefits provided through aeration were restricted; not by a lack of ability, but rather by the limited pollution potential of the aged wastes. Based on the results of this study, aeration is recommended for newly constructed bioreactor landfills as a method to rapidly stabilize municipal solid waste. The benefits of aeration for the aged waste bioreactors, though measurable, were not substantial enough to suggest aeration for landfills of this age. However, aeration of MSW of an intermediate age may still prove beneficial.

Not only did aeration decrease the time to leachate stabilization for all analytical parameters, but it also reduced the final stabilized leachate concentrations. The only parameter which provided some difficulty was ammonia. However, as the study proceeded, biotransformation of the ammonia led to its eventual removal from the system.

Air injection when combined with leachate recirculation resulted a 20.6% reduction in MSW depth in only 19 weeks. This settlement resulted in a 21.5% recovery of landfill space, such that an aerated bioreactor landfill could increase its final capacity significantly. Leachate stabilization of pH, TSS, TDS, BOD_5, and COD was accelerated by 44%, 25%, 54%, 38%, and 59% respectively for fresh wastes when subject to air injection. Ammonia concentrations stabilized under aeration, but remained problematic in the anaerobic bioreactors at the study conclusion. Likewise, the BOD/COD Ratio for fresh waste was stabilized under aeration while the fresh waste anaerobic bioreactor did not over the study period.

Overall, air injection was demonstrated to accelerate and improve both physical settlement of the MSW and leachate stabilization of TSS, TDS, NH_3, BOD_5, and COD. The degree of the improvement was related to the age of the waste, where fresh wastes with high organic fractions benefited more from aeration than the aged wastes.

7.2 RECOMMENDATIONS

Further study is recommended with MSW samples representing various different waste ages between the age limits of this study. Research in this area could help determine the maximum waste age at which the benefits of aeration are large enough to recommended aeration for implementation. Formulation of an economic model that combines economic principles with landfill parameters such as waste age, volume and composition for this purpose is also suggested. Finally, instead of obtaining aged wastes from actual landfills, aged wastes can be prepared in-lab in order to provide for more comparable and better-controlled aged waste samples.

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APPENDIX A – ACRONYMS

BCR   BOD / COD Ratio

BDOF   Biodegradable Organic Fraction

BOD_5 5-Day Biological Oxygen Demand

COD   Chemical Oxygen Demand

LCR   Leachate Collection Reservoir

LFG   Landfill Gas

MSW   Municipal Solid Waste

TDS   Total Dissolved Solids

TOC   Total Organic Carbon

TSS   Total Suspended Solids

APPENDIX B – SUPPLEMENTARY TABLES

Leachate Sample Volumes Diluted In 300ml And The Analytical BOD_5 Range.

Leachate Sample (mL in 300mL)	Analytical BOD_5 range
0.10	6000 – 21000
0.20	3000 – 10500
0.50	1200 – 4200
1.0	600 – 2100
2.0	300 – 1050
5.0	102 – 420
10.0	60 – 210
20.0	30 – 105
50.0	12 – 42

(Sawyer and McCarty 1978)

APPENDIX C – PHYSICAL MSW DATA

Percent Composition Data

Aged MSW:

	Including the Daily Cover	Excluding the Daily Cover	Typical Composition ca.1980[1]
Paper	22.2	36.4	31.7
Plastic	12.6	20.7	5.0
Yard Waste	2.0	3.3	20.1
Food Waste	5.1	8.4	9.5
Wood	1.9	3.1	5.1
Other Organics Including Textiles	8.0	13.1	6.3
Ferrous Metals	5.1	8.4	8.9
Aluminum	2.2	3.6	1.5
Glass	1.9	3.1	10.5
Other Inorganics Including Soil	39.0	0.0	1.5
Total (%)	100.0	100.0	100.0
Biodegradable Organic Fraction[2]	37.3	61.2	67.6

[1] (USEPA 2001).

[2] Biodegradable Organic Fraction = Paper, Yard, Food, and Other Organics assuming 100% biodegradability.

Fresh MSW:

	Including the Daily Cover	Excluding the Daily Cover[1]	Typical Composition ca. 1999[2]
Paper	18.4	29.7	30.6
Plastic	8.5	13.8	13.8
Yard Waste	0.9	1.5	9.1
Food Waste	17.6	28.4	14.8
Wood	0.0	0.0	6.9
Other Organics Including Textiles	6.3	10.2	10.1
Ferrous Metals	4.1	6.6	5.3
Aluminum	2.2	3.6	1.7
Glass	3.9	6.2	5.8
Other Inorganics Including Soil	38.0	0.0	1.9
Total (%)	100.0	100.0	100.0
Biodegradable Organic Fraction[3]	43.2	69.8	64.6

[1] As discarded.

[2] (USEPA 2001).

[3] Biodegradable Organic Fraction = Paper, Yard, Food, and Other Organics assuming 100% biodegradability.

APPENDIX C – PHYSICAL MSW DATA

Mass, Volume, Density, and Settlement Data

Initial Mass, Volume, and Density of MSW

	Unit	AF1	AF2	NF1	NF2	AO1	AO2	NO1	NO2
Mass of MSW Sample	kg	17.9	16.4	17.9	17.3	27.2	27.0	29.7	28.4
Initial Density	kg/m3	358.0	321.6	351.0	346.3	538.1	546.1	587.1	562.7
Mass of Soil Added	kg	10.9	10.2	10.9	10.7	0.0	0.0	0.0	0.0
Mass as Emplaced	kg	28.8	26.6	28.8	28.0	27.2	27.0	29.7	28.4
Volume as Emplaced	L	52.8	52.8	52.8	52.8	47.5	47.5	47.5	47.5
Density as Emplaced	kg/m3	545.3	503.7	545.3	530.2	572.3	568.1	624.9	597.5

Mass, Volume and Density of MSW at Field Capacity

	Unit	AF1	AF2	NF1	NF2	AO1	AO2	NO1	NO2
Mass as Emplaced	kg	28.8	26.6	28.8	28.0	27.2	27.0	29.7	28.4
Water Added to FC	kg	4.0	5.6	11.0	4.0	5.0	3.3	2.5	4.8
Mass at Field Capacity	kg	32.8	32.2	39.8	32.0	32.2	30.3	32.2	33.2
Volume at Field Capacity	L	52.8	52.8	52.8	52.8	47.5	47.5	47.5	47.5
Density at Field Capacity	kg/m3	621.1	608.8	753.6	605.9	677.5	636.5	676.4	698.5

Mass, Volume, and Density of MSW After Bioreactor Treatment

	Unit	AF1	AF2	NF1	NF2	AO1	AO2	NO1	NO2
Final Mass at FC	kg	29.3	32.2	37.6	30.6	31.0	30.4	33.1	31.4
Estimated Loss of Mass	kg	3.5	-0.1	2.2	1.4	1.2	-0.1	-1.0	1.8
Volume After Settlement	L	38.3	40.9	43.6	40.9	40.9	37.0	38.3	40.9
Final Density at FC	kg/m3	765.3	786.8	863.0	747.7	757.4	822.4	864.5	767.2
Change in Density	kg/m3	144.2	178.0	109.4	141.7	80.0	185.9	188.1	68.7

MSW Settlement

	Unit	AF1	AF2	NF1	NF2	AO1	AO2	NO1	NO2
Initial Headspace	cm	6.0	6.0	6.0	6.0	10.0	10.0	10.0	10.0
Final Headspace	cm	17.0	15.0	13.0	15.0	15.0	18.0	17.0	15.0
Settlement	cm	11.0	9.0	7.0	9.0	5.0	8.0	7.0	5.0

APPENDIX C – PHYSICAL MSW DATA

Moisture Content Data

Initial Moisture Content

Sample Replicate		Unit	AF1	AF2	NF1	NF2	AO1	AO2	NO1	NO2	Soil
Replicate 1	Initial Mass	g	122.7	117.7	119.1	113.1	112.1	121.4	120.0	116.9	105.0
	Dry Mass	g	73.2	76.0	83.4	67.2	82.3	89.1	87.9	85.2	60.1
	Initial Moisture Content	%	67.5	54.7	42.8	68.3	36.2	36.2	36.6	37.2	74.7
Replicate 2	Initial Mass	g	134.7	132.7	138.1	128.1	119.1	138.4	126.0	129.9	129.0
	Dry Mass	g	83.6	88.8	96.2	84.0	96.7	105.1	96.7	100.4	82.5
	Initial Moisture Content	%	61.0	49.3	43.5	52.5	23.2	31.7	30.4	29.4	56.4
Replicate 3	Initial Mass	g	135.7	127.7	137.1	142.1	139.1	133.4	143.0	132.9	120.0
	Dry Mass	g	79.0	77.4	87.3	70.0	86.6	95.3	90.4	88.9	64.2
	Initial Moisture Content	%	71.8	64.9	57.0	103.1	60.5	40.0	58.3	49.5	86.8
Average	Average Initial Mass	g	131.0	126.0	131.4	127.8	123.4	131.1	129.7	126.6	118.0
	Average Dry Mass	g	78.6	80.8	89.0	73.8	88.5	96.5	91.6	91.5	69.0
	Overall Initial MC	%	66.6	56.0	47.7	73.3	39.4	35.8	41.5	38.3	71.2

Moisture Content at Field Capacity

Sample Replicate		Unit	AF1	AF2	NF1	NF2	AO1	AO2	NO1	NO2	Soil
Replicate 1	Mass at FC	g	123.1	115.7	110.2	105.6	114.8	119.3	114.0	123.6	93.3
	Dry Mass	g	62.8	58.4	56.3	54.4	67.6	79.4	76.0	76.8	49.8
	Moisture Content at FC	%	95.9	98.2	95.9	94.0	69.9	50.3	50.0	60.8	87.3
Replicate 2	Mass at FC	g	135.1	130.7	129.2	120.6	121.8	136.3	120.0	136.6	117.3
	Dry Mass	g	77.2	74.4	65.1	69.6	90.0	89.8	88.8	89.6	66.6
	Moisture Content at FC	%	74.9	75.7	98.6	73.2	35.4	51.8	35.2	52.4	76.1
Replicate 3	Mass at FC	g	135.1	125.7	128.2	134.6	141.8	131.3	137.0	139.6	108.3
	Dry Mass	g	67.2	64.6	58.8	58.1	71.7	85.2	77.3	80.7	52.6
	Moisture Content at FC	%	101.1	94.6	118.1	131.6	97.7	54.2	77.1	72.8	106.0
Average	Ave. Mass at FC	g	131.1	124.0	122.6	120.3	126.2	129.0	123.6	133.2	106.3
	Ave. Dry Mass	g	69.1	65.8	60.1	60.7	76.4	84.8	80.7	82.4	56.3
	Overall MC at FC	%	89.8	88.5	104.1	98.0	65.0	52.2	53.2	61.7	88.7

Water Added to Reach Field Capacity

Sample Replicate	Unit	AF1	AF2	NF1	NF2	AO1	AO2	NO1	NO2
Uptake Volume	L	4.0	5.6	11.0	4.0	5.0	3.3	2.5	4.8

APPENDIX C – PHYSICAL MSW DATA

Water Additions

Water Additions   Unit: L

Day No.	AF1	AF2	NF1	NF2	AO1	AO2	NO1	NO2
1	2.000	2.000	2.000	2.000	0.000	2.000	2.000	2.000
2	3.000	3.000	3.000	3.000	2.000	3.000	3.000	3.000
3	2.000	0.500	0.000	1.000	2.000	0.500	0.500	0.000
5	0.250	0.250	0.250	0.250	0.250	0.250	0.250	0.250
7	2.000	0.250	0.000	0.000	0.250	0.000	0.000	0.000
9	2.000	0.000	0.000	0.300	0.100	0.200	1.500	0.250
10	2.000	0.250	1.000	1.250	0.100	0.100	0.000	0.000
12	1.000	0.000	1.000	0.500	0.000	0.000	0.250	0.750
16	0.150	0.400	0.200	0.550	0.350	0.400	0.400	0.200
18	0.300	0.500	1.275	1.000	0.200	0.250	0.800	0.750
29	0.500	0.400	0.900	0.700	0.550	0.380	0.800	0.800
36	0.275	0.300	0.900	1.175	0.200	0.250	0.675	0.725
43	0.400	0.150	0.800	0.950	0.250	0.250	0.900	0.900
50	0.000	0.000	2.200	0.000	0.000	0.000	0.000	0.000
53	0.400	0.400	0.500	1.000	0.450	0.375	0.600	0.550
57	0.550	0.450	1.200	1.200	0.400	0.250	0.900	0.900
64	0.250	0.250	0.650	0.550	0.400	0.400	0.600	0.700
71	0.000	0.000	0.000	0.000	0.550	0.000	0.350	0.300
78	0.350	0.250	0.700	0.600	0.550	0.550	0.800	0.700
85	0.450	0.450	0.800	0.550	0.450	0.300	0.700	0.800
92	0.500	0.400	0.850	0.800	0.350	0.500	0.900	0.900
99	0.250	0.250	0.000	0.550	0.000	0.250	0.450	0.450
106	0.850	0.200	0.450	0.450	0.400	0.150	0.350	0.750
113	0.000	0.000	0.250	0.000	0.150	0.400	0.400	0.600
120	0.500	0.400	0.600	0.550	0.000	0.400	0.350	0.550
127	0.350	0.250	0.300	0.300	0.400	0.400	0.300	0.550

APPENDIX C – PHYSICAL MSW DATA

Water Usage

Water Usage[1]   Unit: L

Day No.	AF1	AF2	NF1	NF2	AO1	AO2	NO1	NO2
18	0.156	0.301	0.308	0.044	0.513	0.825	0.171	0.159
29	1.181	1.051	0.358	0.294	1.063	1.325	0.121	0.159
36	1.831	1.501	0.608	0.444	1.613	1.875	0.421	0.289
43	2.481	2.426	0.633	0.494	2.038	2.350	0.371	0.289
50	3.031	3.126	0.783	0.394	2.688	3.000	0.371	0.289
53	5.231	3.126	0.783	0.394	2.688	3.000	0.371	0.289
57	5.481	3.876	0.933	0.544	3.038	3.300	0.571	0.414
64	6.431	4.826	1.233	0.744	3.688	3.950	0.721	0.414
71	6.831	5.126	1.233	0.744	4.038	4.400	0.871	0.564
78	6.581	4.876	0.983	0.494	4.138	4.450	1.171	0.314
85	7.031	5.226	1.083	0.494	4.688	4.900	1.471	0.614
92	7.581	5.526	1.283	0.694	5.138	5.450	1.671	0.664
99	8.181	6.076	1.533	0.844	5.788	6.100	1.771	0.914
106	7.931	6.376	1.533	0.844	5.988	6.300	1.521	0.914
113	8.131	6.576	2.133	0.794	6.088	6.800	1.671	0.814
120	8.131	6.326	1.883	0.544	6.238	7.150	1.571	0.964
127	8.481	6.626	2.133	0.694	6.338	7.450	1.321	1.114
134	8.781	6.926	2.483	0.944	6.638	8.000	1.721	1.514

[1] Water Usage is the cumulative amount of water added to the bioreactors less the amount removed for sampling (0.25L/wk).

Rate of Water Usage[2]   Unit: L/d

	AF1	AF2	NF1	NF2	AO1	AO2	NO1	NO2
Slope (L/d)	0.080	0.062	0.017	0.008	0.057	0.063	0.014	0.009

[2] The Rate of Water Usage was calculated as the slope from data appearing in Water Usage using linear trendline analysis Microsoft® EXCEL 2002™.

APPENDIX C – PHYSICAL MSW DATA

Internal Temperature

Unit: C

Day No.	Ambient	AF1	AF2	NF1	NF2	AO1	AO2	NO1	NO2
7	21.0	21.0	21.0	21.0	21.0	21.0	21.0	21.0	21.0
8	21.0	21.0	21.0	21.0	21.0	21.0	21.0	21.0	21.5
10	21.0	21.0	21.0	21.0	21.0	21.0	21.0	22.0	22.0
12	21.0	21.0	21.5	21.0	21.0	21.0	20.5	21.5	21.0
16	21.5	22.0	22.0	21.5	21.5	22.0	21.0	21.0	21.0
17	21.5	21.5	22.0	22.0	22.0	21.5	22.0	20.0	22.0
19	21.5	22.0	22.0	21.5	21.5	22.0	22.0	21.0	22.0
22	21.0	22.0	22.0	21.5	21.5	21.0	22.0	21.0	21.5
24	21.0	23.0	22.5	22.0	22.0	21.0	22.0	21.0	22.0
26	21.0	22.5	22.0	21.5	22.0	21.5	21.5	20.5	21.5
29	21.5	23.0	22.5	21.5	22.0	21.0	22.0	21.5	22.0
31	21.5	21.5	22.0	22.0	21.5	21.5	21.5	21.5	21.5
33	22.0	22.0	22.5	21.5	22.0	22.0	21.5	21.0	22.5
36	21.5	22.0	23.0	22.0	22.5	22.0	22.0	22.0	22.0
43	21.5	22.0	24.5	22.0	22.0	21.0	21.5	21.5	22.0
50	21.5	22.0	23.0	21.5	22.0	21.5	21.5	21.5	22.0
57	22.0	22.5	23.0	21.5	23.0	22.5	22.0	22.0	21.5
64	21.5	22.0	22.0	21.5	21.5	22.0	21.5	22.0	21.5
71	21.0	22.0	21.0	21.0	21.0	21.5	21.0	20.5	21.0
78	22.0	21.5	22.0	22.0	22.5	22.0	22.0	22.0	21.0
85	21.5	22.0	21.5	21.0	21.0	21.0	22.0	21.5	21.0
92	25.0	22.0	25.5	24.5	25.0	25.0	25.0	24.5	25.0
99	24.0	22.5	25.0	24.0	24.0	24.5	23.5	24.0	23.5
106	23.5	23.0	24.0	24.0	23.5	23.5	23.5	23.5	23.5
113	22.5	23.0	22.5	23.5	23.0	22.5	23.0	22.5	22.5
120	24.0	24.0	24.0	24.0	24.0	23.0	23.0	24.0	24.0
127	23.5	24.5	23.5	23.0	23.5	23.5	23.0	23.5	23.0
134	24.5	25.0	25.0	24.5	25.5	24.5	24.0	24.0	24.0

APPENDIX D – LEACHATE ANALYSIS DATA

Results of Leachate pH Analysis

Initial Data   Unit: pH

Day No.	AF1	AF2	AO1	AO2	NF1	NF2	NO1	NO2
7	5.24	6.05	6.67	6.55	6.32	5.98	6.38	6.50
10	5.54	5.76	6.63	6.55	6.04	5.77	6.55	6.54
17	5.64	6.18	6.64	6.64	5.76	5.87	6.64	6.73
24	6.05	6.69	6.79	6.74	5.84	6.13	6.66	6.65
31	6.44	6.73	6.56	6.77	6.07	6.32	6.58	6.49
38	6.95	6.97	7.07	7.11	6.29	6.53	6.68	6.55
45	6.95	6.80	6.81	6.86	6.55	6.72	6.67	6.55
52	7.26	7.13	7.22	7.28	6.68	6.78	6.80	6.68
59	7.56	7.12	7.20	7.33	6.72	6.82	6.83	6.71
66	7.25	7.02	7.13	7.15	6.68	6.86	6.78	6.66
73	7.30	6.99	7.11	7.22	6.69	6.89	6.63	6.67
80	7.25	7.07	7.10	7.23	6.86	7.08	6.70	6.69
87	7.42	7.10	7.17	7.29	6.95	7.08	6.79	6.72
94	7.42	7.12	7.24	7.36	6.97	7.13	6.85	6.72
101	7.36	7.10	7.23	7.35	6.95	7.09	6.80	6.69
108	7.15	7.10	7.30	7.38	6.98	7.08	6.84	6.77
115	7.11	7.13	7.28	7.41	7.08	7.07	6.85	6.69
122	7.04	7.06	7.14	7.21	7.02	7.10	6.91	6.77
129	7.20	7.07	7.27	7.38	6.93	7.06	6.84	6.69

Data for Averaged Replicates   Unit: pH

Day No.	AF	AO	NF	NO
7	5.65	6.61	6.15	6.44
10	5.65	6.59	5.91	6.55
17	5.91	6.64	5.82	6.69
24	6.37	6.77	5.99	6.66
31	6.59	6.67	6.20	6.54
38	6.96	7.09	6.41	6.62
45	6.88	6.84	6.64	6.61
52	7.20	7.25	6.73	6.74
59	7.34	7.27	6.77	6.77
66	7.14	7.14	6.77	6.72
73	7.15	7.17	6.79	6.65
80	7.16	7.17	6.97	6.70
87	7.26	7.23	7.02	6.76
94	7.27	7.30	7.05	6.79
101	7.23	7.29	7.02	6.75
108	7.13	7.34	7.03	6.81
115	7.12	7.35	7.08	6.77
122	7.05	7.18	7.06	6.84
129	7.14	7.33	7.00	6.77

APPENDIX D – LEACHATE ANALYSIS DATA

Results of Leachate TSS Analysis

Initial Data   Unit: mg/L

Day No.	AF1	AF2	AO1	AO2	NF1	NF2	NO1	NO2
7	506	80	45	32	0	196	8	112
10	183	96	70	40	60	80	24	24
17	278	180	60	87	167	255	67	114
24	231	115	152	104	185	90	68	105
31	139	60	96	184	40	88	160	88
38	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
45	277	40	0	0	64	161	28	68
52	147	67	21	14	100	326	48	93
59	48	68	14	14	81	217	48	55
66	64	21	0	7	60	90	48	48
73	21	14	0	0	60	120	53	40
80	21	21	20	38	48	76	47	50
87	41	21	0	7	60	120	40	50
94	21	14	7	0	40	121	40	30
101	21	14	0	0	30	117	20	40
108	7	7	0	0	40	100	30	40
115	7	7	0	0	20	100	7	30
122	14	41	7	7	50	162	40	60
129	7	14	0	0	40	133	0	0

Data for Averaged Replicates   Unit: mg/L

Day No.	AF	AO	NF	NO
7	293	39	98	60
10	139	55	70	24
17	229	73	211	91
24	173	128	138	87
31	99	140	64	124
38	n/a	n/a	n/a	n/a
45	158	0	113	48
52	107	17	213	71
59	58	14	149	52
66	42	3	75	48
73	17	0	90	46
80	21	29	62	49
87	31	3	90	45
94	17	3	80	35
101	17	0	73	30
108	7	0	70	35
115	7	0	60	18
122	28	7	106	50
129	10	0	87	0

APPENDIX D – LEACHATE ANALYSIS DATA

Results of Leachate TDS Analysis

Initial Data   Unit: mg/L

Day No.	AF1	AF2	AO1	AO2	NF1	NF2	NO1	NO2
10	11900	4700	3300	3500	3750	7950	3450	3450
17	11400	6750	4050	3550	6750	8800	4450	3000
24	11600	5100	4000	3700	7500	8750	4100	3100
31	8950	3700	4100	4100	7950	9250	3800	3400
38	2800	3300	4550	3850	6750	7650	2800	2900
45	3900	2700	3500	3650	6150	6350	2950	3000
52	2750	2400	2850	3450	5800	6350	2750	2900
59	2400	2450	2600	3350	5500	6400	2600	2800
66	2250	2150	2450	2950	5300	5750	2550	2450
73	2450	2400	2350	3300	5250	5050	3000	2650
80	2450	2450	2250	3350	4550	4650	3200	2550
87	2350	2300	1900	3050	4000	4350	2850	2250
94	2400	2250	1800	2950	3700	4200	2700	2200
101	2650	2200	2300	2050	3100	4950	2350	2100
108	3650	2200	1650	2900	3800	4150	2550	2100
115	3900	2050	1550	2700	3150	4050	2250	2350
122	3900	2500	3050	3100	2150	4100	2350	2400
129	3450	2350	1650	2800	3050	4000	2250	2100

Data for Averaged Replicates   Unit: mg/L

Day No.	AF	AO	NF	NO
10	8300	3400	5850	3450
17	9075	3800	7775	3725
24	8350	3850	8125	3600
31	6325	4100	8600	3600
38	3050	4200	7200	2850
45	3300	3575	6250	2975
52	2575	3150	6075	2825
59	2425	2975	5950	2700
66	2200	2700	5525	2500
73	2425	2825	5150	2825
80	2450	2800	4600	2875
87	2325	2475	4175	2550
94	2325	2375	3950	2450
101	2425	2175	4025	2225
108	2925	2275	3975	2325
115	2975	2125	3600	2300
122	3200	3075	3125	2375
129	2900	2225	3525	2175

APPENDIX D – LEACHATE ANALYSIS DATA

Results of Leachate Ammonia Analysis

Initial Data   Unit: mg/L

Day No.	AF1	AF2	AO1	AO2	NF1	NF2	NO1	NO2
7	0.97	3.11	0.54	0.80	0.50	0.32	3.48	1.29
10	1.18	0.62	1.43	1.07	1.15	1.51	1.51	1.94
17	1.56	13.54	0.82	0.43	2.20	9.10	4.11	1.06
24	3.38	27.52	0.87	0.87	79.01	395.03	5.41	3.67
31	347.99	27.52	0.77	0.86	41.59	525.89	3.00	1.98
38	389.19	26.62	0.40	0.20	51.98	468.54	1.90	2.78
45	229.45	37.48	0.45	0.25	16.02	541.06	2.68	2.18
52	368.01	25.73	0.34	0.13	22.65	658.38	1.81	1.98
59	357.82	19.06	0.45	0.12	19.27	541.06	2.38	1.42
66	347.87	10.75	0.22	0.09	30.15	588.82	2.60	2.15
73	423.08	11.32	0.62	0.18	27.12	845.22	1.37	1.50
80	434.99	15.51	0.66	0.16	14.27	799.55	0.59	1.60
87	434.99	17.42	0.37	0.15	34.12	572.51	0.50	1.81
94	301.50	47.27	0.39	0.12	6.53	493.02	1.43	1.02
101	447.23	21.50	0.50	0.13	9.61	670.85	1.61	1.19
108	119.00	10.75	0.31	0.07	16.14	507.24	0.75	2.01
115	128.45	5.45	0.26	0.13	6.96	521.81	0.64	2.37
122	118.11	5.94	0.14	0.09	5.94	608.99	0.86	1.81
129	7.93	6.96	0.06	0.00	4.01	576.08	1.34	1.37

Data for Averaged Replicates   Unit: mg/L

Day No.	AF	AO	NF	NO
7	2.04	0.67	0.41	2.39
10	0.90	1.25	1.33	1.73
17	7.55	0.63	5.65	2.58
24	15.45	0.87	237.02	4.54
31	187.76	0.82	283.74	2.49
38	207.90	0.30	260.26	2.34
45	133.46	0.35	278.54	2.43
52	196.87	0.24	340.51	1.90
59	188.44	0.29	280.16	1.90
66	179.31	0.15	309.49	2.37
73	217.20	0.40	436.17	1.43
80	225.25	0.41	406.91	1.10
87	226.21	0.26	303.32	1.15
94	174.39	0.25	249.78	1.22
101	234.37	0.32	340.23	1.40
108	64.88	0.19	261.69	1.38
115	66.95	0.20	264.38	1.50
122	62.02	0.11	307.47	1.33
129	7.45	0.03	290.04	1.35

APPENDIX D – LEACHATE ANALYSIS DATA

Results of Leachate BOD$_5$ Analysis

Initial Data   Unit: mg/L

Day No.	AF1	AF2	AO1	AO2	NF1	NF2	NO1	NO2
7	425	422	201	50	108	410	43	213
10	4800	1700	56	60	n/a	2925	48	145
17	6825	2265	42	48	2700	3900	54	200
24	10050	1460	35	22	3050	4750	96	390
31	4625	605	28	15	2380	4950	78	450
38	1575	150	23	20	2660	3450	72	254
45	773	188	9	9	n/a	n/a	64	80
52	210	60	17	11	780	1680	51	98
59	62	59	19	7	1075	1500	34	65
66	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
73	90	44	10	5	900	645	38	36
80	56	48	12	13	306	308	25	31
87	46	46	7	5	297	293	23	19
94	22	23	7	4	137	160	11	35
101	74	19	10	8	145	231	14	21
108	210	14	10	5	142	143	13	38
115	102	33	11	5	117	278	21	44
122	66	65	7	16	247	599	34	68
129	92	45	11	9	127	303	26	53

Data for Averaged Replicates   Unit: mg/L

Day No.	AF	AO	NF	NO
7	424	126	259	128
10	3250	58	2925	97
17	4545	45	3300	127
24	5755	28	3900	243
31	2615	21	3665	264
38	863	21	3055	163
45	480	9	n/a	72
52	135	14	1230	74
59	60	13	1288	50
66	n/a	n/a	n/a	n/a
73	67	8	773	37
80	52	13	307	28
87	46	6	295	21
94	22	6	149	23
101	46	9	188	17
108	112	7	142	26
115	68	8	197	33
122	66	12	423	51
129	68	10	215	39

APPENDIX D – LEACHATE ANALYSIS DATA

Results of Leachate COD Analysis

Initial Data   Unit: mg/L

Day No.	AF1	AF2	AO1	AO2	NF1	NF2	NO1	NO2
7	5349	4516	438	1140	1140	5349	658	965
10	9646	4736	1140	965	3683	9471	1403	1491
17	22450	8770	1315	1579	4648	22801	2631	1579
24	17890	4385	351	702	8770	12979	877	1403
31	11751	877	175	351	4823	11927	745	1579
38	5612	877	307	1315	4736	10348	965	1491
45	2719	877	263	307	4648	5700	877	1184
52	1228	789	196	245	4122	5437	614	877
59	658	526	213	196	4385	4736	745	965
66	515	376	213	131	4209	4385	965	1140
73	417	335	180	172	4911	3946	507	704
80	507	360	245	213	2543	1535	409	712
87	360	262	180	188	1579	1535	376	507
94	295	213	115	98	965	1140	311	499
101	317	216	183	167	894	1333	334	635
108	301	195	165	136	508	773	301	534
115	239	224	165	180	572	814	301	593
122	181	231	134	109	483	671	262	485
129	188	196	131	164	409	720	262	450

Data for Averaged Replicates   Unit: mg/L

Day No.	AF	AO	NF	NO
7	4933	789	3245	811
10	7191	1052	6577	1447
17	15610	1447	13724	2105
24	11137	526	10874	1140
31	6314	263	8375	1162
38	3245	811	7542	1228
45	1798	285	5174	1030
52	1008	221	4779	745
59	592	205	4560	855
66	446	172	4297	1052
73	376	176	4429	605
80	434	229	2039	560
87	311	184	1557	442
94	254	106	1052	405
101	267	175	1114	484
108	248	151	640	418
115	232	173	693	447
122	206	122	577	374
129	192	147	565	356

APPENDIX D – LEACHATE ANALYSIS DATA

Results of Leachate BOD / COD Ratio

**Initial Data**   Unit: n/a

| Day No.	AF1	AF2	AO1	AO2	NF1	NF2	NO1	NO2
7	0.079	0.093	0.458	0.044	0.095	0.077	0.065	0.221
10	0.498	0.359	0.049	0.062	n/a	0.309	0.034	0.097
17	0.304	0.258	0.032	0.030	0.581	0.171	0.021	0.126
24	0.562	0.333	0.098	0.031	0.348	0.366	0.109	0.278
31	0.394	0.690	0.157	0.043	0.493	0.415	0.104	0.285
38	0.281	0.171	0.073	0.015	0.562	0.333	0.075	0.170
45	0.284	0.214	0.035	0.030	n/a	n/a	0.073	0.068
52	0.171	0.076	0.089	0.046	0.189	0.309	0.082	0.112
59	0.094	0.111	0.089	0.035	0.245	0.317	0.046	0.067
66	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
73	0.216	0.131	0.057	0.028	0.183	0.163	0.074	0.051
80	0.109	0.133	0.050	0.062	0.120	0.201	0.060	0.044
87	0.126	0.175	0.039	0.028	0.188	0.191	0.062	0.036
94	0.073	0.107	0.064	0.044	0.142	0.140	0.034	0.069
101	0.233	0.087	0.056	0.045	0.162	0.173	0.041	0.033
108	0.150	0.073	0.058	0.039	0.140	0.184	0.044	0.071
115	0.160	0.090	0.068	0.029	0.110	0.170	0.071	0.074
122	0.110	0.070	0.049	0.150	0.110	0.175	0.070	0.139
129	0.090	0.080	0.087	0.056	0.100	0.160	0.054	0.118

Data for Averaged Replicates   Unit: n/a

| Day No.	AF	AO	NF	NO
7	0.086	0.251	0.086	0.143
10	0.428	0.055	0.309	0.066
17	0.281	0.031	0.376	0.073
24	0.447	0.064	0.357	0.194
31	0.542	0.100	0.454	0.195
38	0.226	0.044	0.448	0.123
45	0.249	0.032	n/a	0.070
52	0.124	0.068	0.249	0.097
59	0.102	0.062	0.281	0.056
66	n/a	n/a	n/a	n/a
73	0.173	0.042	0.173	0.063
80	0.121	0.056	0.161	0.052
87	0.151	0.034	0.190	0.049
94	0.090	0.054	0.141	0.051
101	0.160	0.050	0.168	0.037
108	0.112	0.049	0.162	0.058
115	0.125	0.048	0.140	0.072
122	0.090	0.100	0.143	0.105
129	0.085	0.071	0.130	0.086

APPENDIX E – SUPPLEMENTARY PHOTOGRAPHS
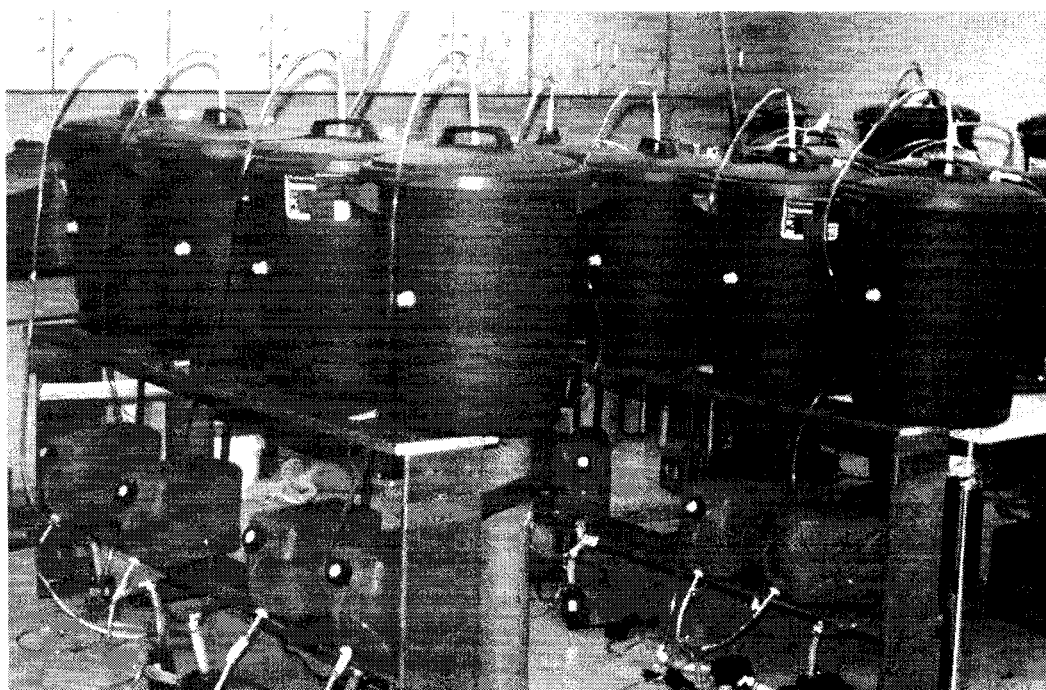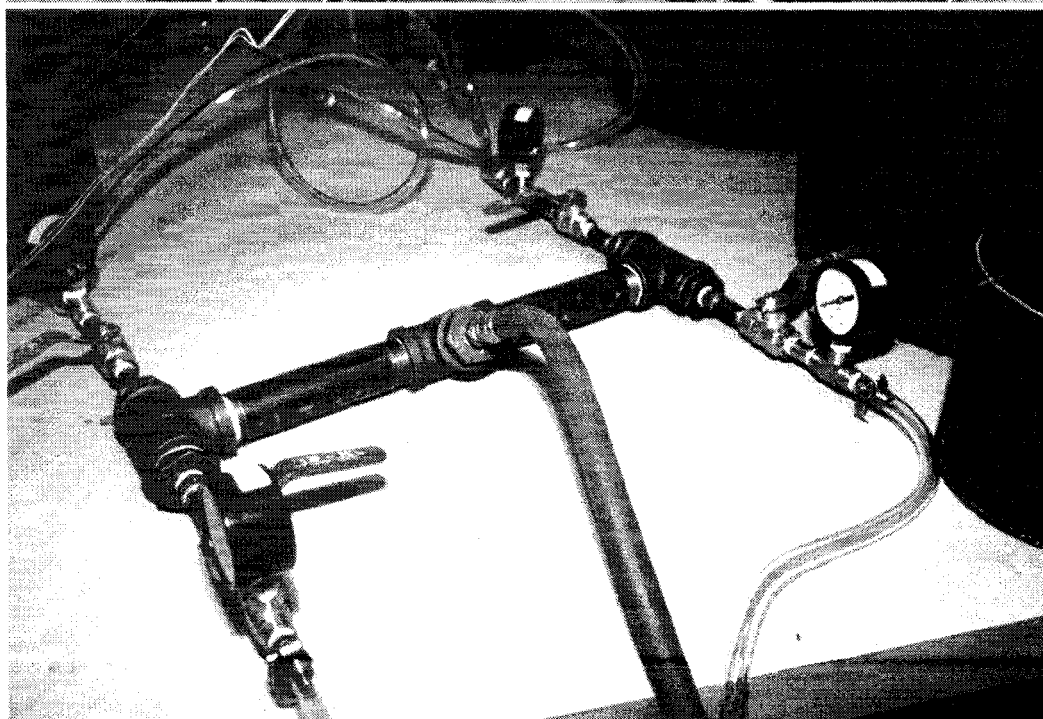
Photographs of the bioreactor apparatus (A) and air distribution arm (B).

APPENDIX E – SUPPLEMENTARY PHOTOGRAPHS
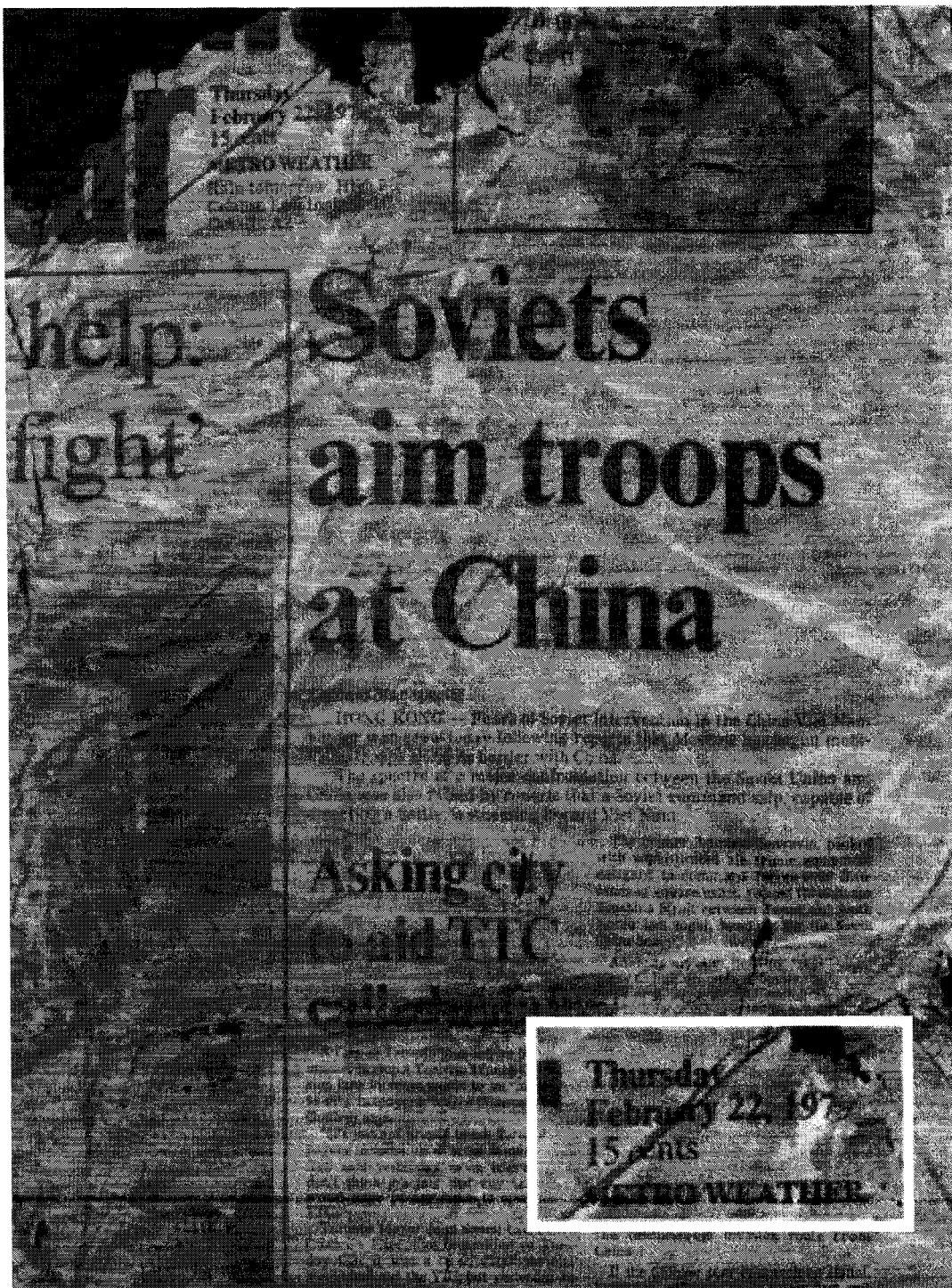
Front Page of a Newspaper Taken from the Aged Waste Sample Used to Determine the Waste Age.