IN-SITU PASSIVE TREATMENT OF MUNICIPAL SOLID WASTE (MSW) LEACHATE USING A MODIFIED DRAINAGE LEACHATE COLLECTION SYSTEM (LCS)

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Abstract

This thesis describes a laboratory investigation of in-situ treatment of synthetic leachate representative of that generated by a municipal solid waste (MSW) landfill. The overall objective is to evaluate alternative designs and operating procedures for effective leachate collection in conjunction with efforts to accelerate waste stabilization (i.e. leachate recirculation). In the investigation five 15 cm (6”) diameter PVC columns were packed with pea gravel and concrete of different sizes; geotextiles were also placed between the packed sections as filter-separators and promoters of bacterial growth. Synthetic leachate was continuously input to the top of the columns and circulated at rates representative of operating field conditions. For each column, effluent was discharged to a nitrification reactor before recirculation. The tests were conducted under anaerobic and unsaturated conditions in the columns. Results indicate about a 97% decrease in COD from the synthetic leachate concentration entering the top of the column, and about 98 % conversion of the ammonia to nitrogen gas. COD depletion and methane production were not significantly inhibited by the denitrification process. Optimum Hydraulic Retention Time (HRT) for the nitrification-denitrification system makes it economically viable for its development at a landfill site. Gas production shows low CO_2 values, decreasing the potential of clogging in the Leachate Collection System (LCS) and extending the Landfill Gas (LFG) network's life service by generating a less corrosive environment. The use of concrete as an alternative to the most commonly used natural gravel as leachate collection drains may not be a good option. During the experiment, the leachate that permeated the columns packed with crushed concrete, presented a higher pH than the leachate that permeated the natural stone. At the conclusion of the experiment noticeable weathering was observed when the columns where dismantled. Further studies are recommended until more conclusive evidence as to concrete performance is found. The overall results obtained from the experiment show that in situ passive treatment at landfills is viable.

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Chapter 1 Introduction

Safe and cost-effective management of Municipal Solid Waste (MSW) is a significant environmental challenge for modern society. This chapter explains how the challenge arises, what has been done in the past to deal with it, and how this thesis approaches the issue. Furthermore, the main objectives of the thesis are outlined, and a brief description of the content of each chapter is presented.

1.1 Subject

Despite intense government promotion of the 3R's concept (Reduce, Reuse, Recycle) as well as growing environmental awareness, in countries such as the United States, still disposed more than 55% of generated solid waste in landfills (USA EPA, 2002). Society continues to equate success with consumption and the result of this ideology is wide scale waste production and disposal. Thus, there is a growing urgency for environmental scientists and professionals to deal with the problem associated with the reality that waste will, in large measure, continue to be disposed of in landfill sites.

In Municipal Solid Waste (MSW) landfills, leachate is the contaminated liquor generated as a consequence of the expulsion of liquid from the refuse. The cause of this liquid expulsion can be compaction of the solid waste, degradation of organic material, as well as, irrigation, precipitation, and/or percolation of water through a landfill from sources such ground water or leachate recirculation (Quian et al. 2002). Leachate contact with either surface or groundwater resources could contaminate them. Leachate management is therefore an imperative to maintaining a healthy environment.

Recent improvements in sanitary landfill design and operation have been focused on barrier systems and the management of gas and/or liquid residues (Harris et al. 2000). Although improvements in lining systems have minimized the likelihood of ground water contamination, they also have resulted in higher leachate recovery. Leachate, which is typically high in organic content and dissolved solids, must be treated before discharge in order to protect the environment. Leachate treatment has a significant cost and has to be carried out during the active and post-closure period, which may last for decades after the landfill has ceased accepting wastes. Among the different schemes developed to deal with leachate treatment, those based on traditional sanitary wastewater treatment are often applied in leachate management (McBean et al. 1995). Several innovative and new applications of existing technologies are now being continuously developed (Reinhart et al. 2002).

A novel approach in leachate management is the bioreactor landfill. In this approach, refuse is not only contained but also rapidly decomposed. The method is based on the principle of accelerated waste degradation through leachate recirculation and wastewater anaerobic treatment. The bioreactor landfill was fist proposed by Pohland in the early 80's and has now emerged as an attractive solution for leachate treatment. It has several attractive features including:

i. Rapid stabilization of the organic fraction in the waste;
ii. Maximization of landfill gas capture over a shorter time frame, thus improving the economies of landfill gas (LFG) energy projects, and consequently reducing green house gas emissions;
iii. Significant reduction in off-site transport of leachate for treatment or disposal;
iv. Reduction of post-closure care and maintenance;
v. Increase of the volume of the landfill available for disposal as a result of rapid settlement

The utilization of landfills as bioreactors also generates challenges. Increasing the water content of the refuse by leachate recirculation can create engineering problems such as landslides, surface seeps and/or liner failure (Qian et al. 2002). Accelerating waste degradation can lead to clogging of the leachate collection system and increasing ammonia concentration (Fleming et al 1999, 2002; Rowe et al. 2002).

1.2 Need

The need for this thesis is based on two assertions:

1. A bioreactor landfill offers potential benefits for MSW management.

2. Fleming et al. (1999), (2002) demonstrated that the Leachate Collection System (LCS) of a landfill may play a central role in the composition and strength of the leachate. Their investigation suggested that novel designs and operating conditions for an efficient leachate collection and treatment, along with the understanding of the chemical, physical, and biological phenomena that take place in a landfill LCS, would greatly improve landfills operation and management.

The need for this thesis is therefore, to investigate the effects in leachate composition that a bioreactor landfill can generate not only by leachate recirculation but also by improving the design of LCS.

1.2.1 MSW Landfill Leachate Collection System (LCS) in Bioreactor Landfills

A landfill Leachate Collection System (LCS) is a drainage system that controls leachate mounding (minimizing the risk of either spills or liner failure) and transports the leachate to storage tanks for subsequent treatment. A LCS is composed of a drain (normally local natural stone), a filter/separator element (in most cases a geotextile) and a perforated pipe network.

In order to work as a bioreactor, the solid waste in a landfill has to reach field capacity which can be understood as the water content at which any additional water or liquid will drain out of the waste. Since many landfills in North America are located in non-humid locations, the addition of external liquids (e.g. water, wastewater, activated- sludge) is required to achieve field capacity. An increased water content of the solid waste improves the ability of the microorganisms to decompose organic material (Barlaz et al. 1989), and by this, increases the Chemical Oxygen Demand (COD) depletion and the likelihood of clogging (Armstrong et al. 1998; Fleming et al.1999, 2004; Rowe et al. 2002; Paksy et al. 1998; Cooke et al. 1999, and VanGulck et al. 2003).

The constant flow of leachate through the drainage system generates physical, chemical and biological processes. Particle encrustation (clogging) in the void space of the LCS is the most adverse consequence of these processes (Fleming et al 1999, 2004; Armstrong, 1998; Cooke et al. 1999). However, Rittman et al (1996), (2003); Fleming et al (1999), (2004); Rowe et al (2002) and VanGulck et al (2003), have demonstrated that the production of clogging material is closely related (by product) to the decrease in leachate organic matter concentration. Their laboratory investigations of clog formation have shown COD depletion of up to 85% of its original concentration in field and laboratory studies.

1.2.2 Ammonia in a Bioreactor Landfill

MSW has been estimated to contain about 4% protein (Barlaz et al. 1989) and therefore, ammonia (NH_3-N) is expected to be produced during the decomposition of organic nitrogen. Since ammonia is stable under anaerobic conditions, it will accumulate in leachate (Burton and Watson-Craik, 1998). Thus, high concentrations of ammonia will persist long after the COD has decreased to concentrations representative of well-decomposed refuse. Therefore, the treatment of leachate to remove ammonia becomes an important aspect of long-term landfill management.

1.3 Objectives

The general objective of this study is to evaluate alternative designs and operating procedures for effective leachate collection in conjunction with efforts to accelerate waste stabilization (e.g. leachate recirculation). Laboratory columns simulating a real landfill LCS were packed with different sizes and types of porous media and leachate was recirculated at different rates representative of those found in a typical North American landfill.

The specific objectives are as follows:

1. Investigate in-situ passive leachate treatment methods for the depletion of COD, nitrogen and some inorganic compounds.

2. Analyse the hydraulic effects of in-situ passive treatment on the drains and geotextile during the laboratory experiment.
3. Determine the feasibility of using concrete as a Leachate drain material.

In order to achieve these specific objectives, laboratory columns were operated as fixed biofilm reactors. Their functions were twofold: to work as an organic anaerobic digester and as a de-nitrifying reactor. Each one of the fixed biofilm reactors was coupled with an external aerated nitrification reactor as an intermediate oxidizing step in ammonia treatment and to provide further organic matter depletion.

The thesis is divided into six chapters. In Chapter 1, the objectives are introduced along with a brief description of the topic. In Chapter 2, the literature review and background information relevant to the investigation is presented. The nature and origin of leachate are described, along with established approaches for leachate characterization management and treatment. The principles of the bioreactor landfill are introduced, highlighting the advantages and potential problems, as well as the biological and geochemical aspects underlying the concept of the on-site passive leachate treatment.

The laboratory test program is described in Chapter 3. The experiment set up is shown and the methods for the physical and chemical analyses are described. In Chapter 4, the experimental results of relations to water quality and gas production are presented and analyzed. Chapter 5 is divided in two subchapters. In these subchapters, the columns autopsies are described as well as the physical changes in the different porous media used within the columns (natural gravel, concrete and geotextile). Overall conclusions and recommendation for further studies are provided in Chapter 6.

Chapter 2 Background Information and Literature Review

2.1 Introduction

Despite increasing environmental awareness, up to 55 % of the total Municipal Solid Waste (MSW) generated in nations such as the United Sates is still disposed in landfills (USA EPA 2002). The need for landfilling does not have a foreseeable end. Other solid waste management alternatives such as incineration and recycling face technical and social hurdles. Incineration is not a viable method of disposal for a wide variety of wastes such as those with high moisture content (Harris et al. 2000). Furthermore, incineration may lead to air pollution problems, and it creates an ash residue that still must be landfilled (McBean et al. 1995). Recycling efforts eventually encounter practical limits (since the concept relies heavily on voluntary participation) that make further reductions in the waste stream hard to achieve (Qian et al. 2002).

Among the existing solid waste management strategies, landfilling is the most cost-effective (Kiely, 1997). A properly engineered landfill can deliver satisfactory environmental and economic results. Municipal Solid Waste (MSW) leachate management strategies are continuously being developed and/or improved. Among them, on-site biological passive treatment is one of the most attractive leachate management strategies due to its economic advantages (Reinhart et al. 2002; Pholand and Kim, 1999).

This chapter outlines and explains the generation, characterization, management, and treatment of leachate. It also describes recent investigations that support the feasibility of passive treatment of leachate at the landfill site for removal of organic carbon, nitrogen and certain inorganic species. The role of geotextiles in landfills as filters and promoters of bacterial growth is also discussed.

2.2 MSW Leachate Generation

According to Farquhar (1989), leachate is created when moisture infiltrates the refuse in the landfill, dissolving electrolytes, nutrients, and contaminants, and producing moisture contents high enough to initiate liquid flow. The main source of moisture is the percolation of water through the waste by irrigation, precipitation, groundwater discharge, and/or leachate recirculation. Even if no water is allowed to infiltrate into the refuse, a small volume of contaminated liquid forms due to biological and chemical reactions (Quian et al. 2002).

Although it is unlikely that leachate will be generated at a constant rate throughout the life of the landfill, it will follow a pattern similar to that of precipitation in the region (Farquhar, 1989). Several water balance models such as the Water Balance Method (WBM) and the HELP model (Hydrologic Evaluation of Landfill Performance) provide insight into the trends of leachate production. However, due to the difficulty of determining some of the coefficients in such models, uncertainty may be associated with the result of such methodologies (McBean et al. 1995).

2.3 MSW Leachate Characterization

Most of the literature on leachate characterization claims that the main features of leachate are determined by the degradation of solid waste over time and the landfill mode of operation. This thesis argues that the design of the landfill and specifically the LCS can also play a central role in the composition of leachate as it is actually collected.

2.3.1 Solid Waste degradation

Approximately 80% of Municipal Solid Waste (MSW) is composed of anaerobically biodegradable organic matter (Barlaz et al. 1989). The organic matter found in solid wastes includes lignocellulosics (lignin, cellulose and hemicellulose), proteins, lipids and starch. (Palmisano and Barlaz, 1996). Among them, cellulose and hemicellulose are the major biodegradable constituents, comprising 45-60% of the MSW dry weight (Barlaz et al. 2002). Thus, anaerobic MSW decomposition can be explained largely by the transformation of cellulose and hemicellulose in simpler compounds such as methane and carbon dioxide.

The transformation of cellulose and hemicellulose takes place in four different phases (Barlaz, et al. 1989, Mcbean et al. 1995, Warith and Sharma, 1998). Phase I or aerobic phase, involves a short period of aerobic decomposition in which easily degradable organic matter is consumed and carbon dioxide is generated. During this phase some polymer hydrolysis occurs, converting the initial cellulose, hemicellulose and proteins in soluble sugar and aminoacids. This process is mediated by extracellular enzymes secreted by microorganisms (Palmisano and Barlaz, 1996).

In Phase two, the fermentation phase, sugars and proteins are fermented and converted to volatile fatty acids and ammonia. This phase is characterized by the presence of hydrogen producing and acetogenic bacteria as well as high levels of carbon dioxide. Phase three is called the acid phase. Volatile fatty acids (long chain carboxylic acids) are converted to short chain carboxylic acids (mainly acetate) by hydrogen producing, hydrogen consuming and acetogenic bacteria (Parkin and

Owen, 1986). Phase four is the methanogenic phase. Here, methanogenic bacteria convert the hydrogen, acetate and some of the carbon dioxide generated during the previous phases into methane gas. Figure 2.1 summarizes the decomposition phases described above.

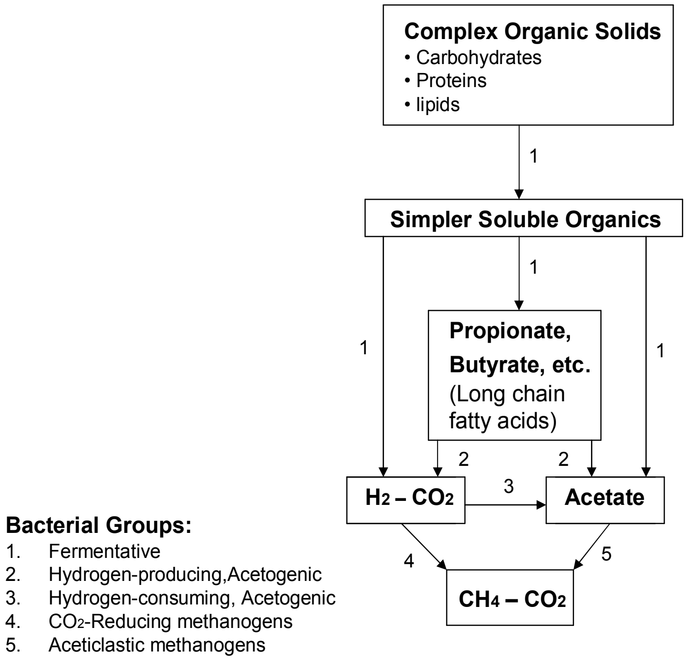

Figure 2.1 Solid waste decomposition (after Parkin and Owen, 1986)

2.3.2 Landfill mode of operation

Christensen et al. (1992) have described several individual management procedures and their effects on waste degradation (which is reflected in methane production):

Waste Composition: The composition of the waste will determine the rate at which the waste decomposes. For instance, an increased content of newspaper does not affect gas generation significantly, while an increased content of magazines increase the gas production (Stegmann and Spendlin, 1986). Large concentrations of sulphate increases the redox potential, and sulphate reduction may compete with methanogenic bacteria for organic carbon, thus decreasing methane gas production (Christensen and Kjeldsen, 1989).

Sewage Sludge: Positive results in methane generation due to the addition of sludge to the refuse, depends on the type of sludge added. A sludge with low pH (e.g. septic sludge) may have a negative effect in methane formation (Leckie et al. 1979), while neutral well-buffered sewage sludge may have positive effects (Leushner, 1989).

Buffer Addition: Despite some beneficial effects of buffering in waste degradation (Stegmann and Spendlin, 1989), it does not seem necessary in all landfill situations. However, if a landfill has failed to generate methane due to low pH values, the addition of a buffer is an obvious measure to help establish methanogenic conditions (Christensen et al.1992).

Shredding: Smaller particle size can increase the rate of the hydrolysis of the organic wastes as a result, the acid phase is intensified resulting in increasing production of carbon dioxide, low pH, and high content of organic carbon in the leachate with the consequences of no or reduced methane formation (Barlaz et al. 1989).

Compaction: Compaction of the refuse is necessary due to the need for optimum use of the landfill capacity and for obtaining geotechnical stability. Results obtained in filed and laboratory experiments may indicate that the time of compaction of the upper refuse layer may be seen as a possible control of the acid phase and hence it can delay the initiation of the methanogenic degradation (Christensen et al. 1992).

Recirculation of Leachate: This concept consists of removing the leachate from the base of the landfill and then to reintroduce it onto, or into, the waste mass by any one of a number of methods, such as: surface spraying, surface ponding, leach fields, shallow wells and/or deep wells (Quian et al. 2002). Pohland (1980) noted that the daily recirculation of the leachate provides microorganisms with sufficient nutrients and, as a result, overall conversion of the waste is enhanced.

2.3.3 Landfill design

Several researchers have found that leachate may undergo significant treatment after passing through a porous drainage medium. Fleming at al. (1999) demonstrated this phenomenon with forensic excavations and leachate analysis from the Keele Valley Landfill (KVL) site near Toronto (Canada). Peeling et al. (1998); Paksy et al. (1998); Fleming et al. (1999); Fleming and Rowe (2004); and VanGulck et al.

(2003) supported those findings using laboratory LCS drains simulating real landfill conditions.

Due to practical difficulties extracting the leachate directly from the base of the buried waste, leachate is taken from the end of the leachate collection pipes. Before reaching the end of the leachate collection pipes, during what may be a residence time of months or years (Fleming et al. 1999), biological stabilization of the leachate by biofilm (slime) will significantly change the leachate composition. Fleming et al (2002) hypothesised that the leachate collected at the end of the LCS system of landfills has undergone a degradation process within the “anaerobic-fixed biofilm reactor” represented by the slime-covered drainage blanket underlying the waste.

To support that statement, they compared two leachate samples coming from two landfills with similar characteristic of age and type of refuse in Ontario (Canada). One landfill has a leachate collection system (LCS), the other one does not. The landfill with a LCS presented an ongoing depletion of Chemical Oxygen Demand (COD) when compared to a stable contaminant such as chloride, whereas in the landfill without a LCS, COD and chloride appear to both increase and decrease in parallel.

It can be hypothesized then, that the leachate coming from the landfill with no LCS underwent waste decomposition dominated by the natural sequence of solid waste degradation phases as described in section2.3.1. In comparison, the landfill with a

LCS provided a more steady treatment of organic matter in the leachate due to the additional degree of degradation which occurred in the biofilm-rich LCS. The difference is important especially when trying to achieve rapid waste stabilization may bring substantial economic benefits and environmental regulatory compliance for the landfill's owner/operator.

Waste decomposition, and thus the leachate composition, is governed by the basic processes and the solid waste degradation phases described in section 2.3.1, but the LCS of landfills can play an important complementary enhancing role in dealing with leachate (due to the possibility of working as a fixed biofilm reactor) by creating a more controlled and efficient in-situ passive leachate treatment system.

2.4 MSW Leachate Management Strategies

Leachate management strategies can be classified as: natural attenuation, remove/treat/discharge, or leachate recycling (Quian et al. 2002).

Natural Attenuation: This was the most commonly adopted strategy for leachate management until the early eighties when regulatory agencies issued more stringent rules for leachate handling. This was due to uncertainties regarding the hydraulic conductivity and adsorption capability of the soil acting as a barrier (mostly clay soils). Today, there seems to be a resurgence in the utilization of natural attenuation in arid climates where the amount of leachate produced is low. Nonetheless, some factors remain difficult to manage under this strategy. Among the most important are: leachate strength and composition, and landfill site selection (it must be located in an isolated area with the consequence of increased hauling costs) (Barlaz et al. 2002).

Remove/Treat/Discharge: This is the most common strategy for liquid management in current landfills. The principle is to avoid any contact of the leachate with the surrounding environment. In order to do this, the waste is “encapsulated” by impermeable barriers which can be natural, synthetic or a combination of the two. Leachate is removed from the base of the landfill and conveyed to containment facility for later treatment and disposal. This approach of the containment system requires for 30 to more than 100 years post-closure.

Leachate Recycling: This concept is based on the finding that solid waste degradation is a dynamic process, primarily influenced by waste characteristics, availability of moisture and nutrients, and current operational circumstances. The last three can be assisted by controlled accumulation, containment, collection and recirculation of leachate back through the landfill. This management strategy offers opportunities to establish the necessary mutually advantageous relationship between the microbial population acting during the acid and methane production phases of landfill stabilization, thus enhancing reaction rates and conversion pathways in a shorter and more predictable manner (Pohland and al-Yousfi 1994).

A very attractive feature of leachate recycling from the economic perspective of owners and/or operators is that this will allow them to swiftly achieve "sustainability" which occurs when they are completely able to manage the outputs (liquids and gases), provide environmentally acceptable residues, avoid long post-closure care periods, and can potentially use the closed sites for beneficial purposes (Reinhart et al. 2002).

2.5 MSW Leachate Treatment

According to Kiely (1997), the selection of leachate treatment and disposal alternatives depends upon:

1. The estimated leachate generation rates.
2. Physical-chemical features of the leachate and variations in leachate characteristics/flow over time.
3. Identification of final disposal alternatives and evaluation of their feasibility in terms of cost effectiveness, environmental impact, technological constraints, regulatory requirements, and compatibility with other elements of the landfill design and operation.
4. Estimated capital, and operation and maintenance costs of treatment and disposal method with respect to reliability and flexibility, and
5. Age of landfill.

The most common alternative for leachate treatment is the off-site treatment at a Municipal Waste Water Treatment Plant (MWWTP) (Harris et al. 2000). Although this solution is typically economical, problems with biological treatment may be experienced at the MWWTP. Another disadvantage is the cost of hauling leachate.

There are also innovative technologies which are normally borrowed from different and bigger industries, such as food and petrochemical. Among them are reverse osmosis, thermal oxidation, air stripping. The use of such approaches is, in many cases, not affordable by the solid waste market (Harris et al. 2000).

On-site passive treatment, on which this thesis is focused, relies primarily on biological systems. This type of leachate treatment is becoming increasingly attractive due to lower costs of operation and maintenance. Among the on-site passive treatment options, the bioreactor landfill mode of operation is arising as one of the most investigated and promoted in North America by the landfill industry (Quian et al.2002). Although, a promising technology, it still has to overcome several challenges that arise due to operational issues, such as clogging of the LCS (Fleming et al. 1999), increased ammonia concentration in the leachate (Pohland, 1995) and the possibility of problems with the landfill stability and slope failure (Quian et al. 2002).

Table 2.1 illustrates the main features of different leachate treatment methods and their affordability compared to a sequencing batch reactor system which is a system

Table 2.1 Leachate treatment methods affordability (After Harris et al. 2000).

System Description	**Major Unit Process**	**Total Relative Cost***
Conventional	Equalization pH adjust-Chemical Precip-Sedimentation Biological, SBR Residual Management	5.0
Conventional	Equalization pH adjust-Chemical Precip-Sedimentation Biological, Fixed Film (Packed Towers, Trickling Filters, RBC) Residual Management	7-7.5
Conventional	Lagoon Residual Management	1.75-2.25
Membrane	Equalization pH Adjustment Pre-Filtration Reverse Osmosis Residual Management	4.75-5.55
Thermal	Equalization Evaporation Thermal Oxidation Residual Management	3.3-3.9
Thermal	Equalization pH Adjustment Distillation Residual Management	4.0-4.75
Biological –In Situ (Bioreactor)	Equalization Recirculation-Moisture Content Control LF Gas Control	2.25-2.75
Biological –In Situ (Facultative Bioreactor)	Equalization Nitrification Recirculation-Moisture Content Control LF Gas Control	2.9-3.5
Biological-Land Base	Equalization pH Adjustment Constructed Wetlands	2.0-2.6
Biological-Land Base	Equalization pH Adjustment Phytoremediation	4.4-5.0

*Scale 1-8, SBR (Sequencing Batch reactor) = 5, with 1 = easiest to operate and lowest capital cost and O&M.

that has been successfully used to treat municipal and industrial waste water as well as MSW leachate (US EPA, 1999)

It can be seen in table 2.1 that the biological in-situ based treatments are among the least expensive systems, which is expected since the treatment of the leachate take place on- site and is carried out by naturally ubiquitous bacteria.

2.6 Challenges of a Bioreactor landfill

The rapid stabilization of the waste in a landfill by leachate recirculation brings not only benefits but also challenges. Increasing ammonia levels and clogging of the LCS are two of the most important issues for a landfill that operates as a bioreactor. Rittman et al. (1996),(2003), Rowe et al. (1997), (2000), (2000a), (2000b), (2002), and VanGulck (2003), have found a direct relationship between anaerobic organic stabilization of the organic load (expressed as COD) and the precipitation of calcium carbonate and other minerals in the LCS. Furthermore, MSW has been estimated to contain about 4% protein (Barlaz et al. 1989), therefore, ammonia (NH_3-N) is expected to be produced during the decomposition of organic nitrogen and since ammonia is stable under anaerobic conditions, it will accumulate in leachate (Burton and Watson-Craik, 1998).

Similarly, in a landfill with leachate recirculation, with out any mechanism for nitrogen removal, ammonia concentration in the leachate will increase until the point at which it may interfere with the effectiveness of ongoing acetotrophic COD removal (Fleming et al. 2002).

Tchobanoglous et al. (1993) presents a sequence of reactions for solid waste decomposition that may shed additional light in understanding from where ammonia and carbonates come:

$$\text{Solid Waste} + H_2O + \text{Nutrients} \rightarrow \text{New cells} + \text{Resistant organic matter} + CO_2 + CH_4 + NH_3 + H_2S + \text{heat} \quad (2.1)$$

If acceleration of waste degradation is to be achieved through leachate recirculation, it becomes apparent that both issues (the clogging of the LCS and the accumulation of ammonia) must be addressed in order to achieve satisfactory leachate management results.

2.7 MSW LCS as a fixed biofilm reactor

The discovery of solid particles (mainly composed of calcite and iron sulphide) entrapped in a LCS in Germany by Brune (1991) prompted several researchers to investigate the phenomenon due to the potential implications for the long term serviceability of landfills. The presence of precipitates in the void spaces of the drain and /or the obstruction of the leachate collection pipes has the potential to create leachate mound, with the consequences of additional stress on the containing barrier (increased advection) and decreased landfill slope stability as a result of increased pore pressures.

Fleming et al. (1999) observed leachate monitoring, field excavation of the landfill drainage system, and analysis of the clogging material at the Keele Valley landfill site (KVL) (Ontario, Canada). It was found that a thick black slime layer built up on the 50 mm diameter clear stone drainage blanket underlying the KVL after 1 to 4 years of exposure to leachate. Its amount was visibly higher near the perforated drainage pipes where there is higher flow rate of leachate. Also within the pipes, significant precipitation of deposits resulted in the accumulation of large solid mineral clog structures (up to 30 cm diameter).

It was hypothesised that the physical, chemical and biological processes occurring during the anaerobic organic degradation of the waste have a strong relationship with biofilm and clog formation. Further laboratory studies simulating MSW leachate collection drains (Peeling et al. 1998; Paksy et al. 1998; Fleming et al. 1999; Fleming and Rowe 2004; Armstrong 1998 and Rowe et al. 2002) and theoretical models (Cooke et al. 2001 and Rittman et al. 2003) confirmed that hypothesis. The organic waste in a landfill will be reduced to simpler organic compounds such as carboxylic acids (mainly acetic acid). Methanogenic bacteria will consume those acids, oxidizing them into a carbonate form (mainly CO_2). Approximately 50% of the CO_2 will off-gas from the leachate, and the other 50 % will be hydrolyzed back into the leachate in the major form of $CO_{2(g)}$ or H_2CO_3*

(Rittman et al. 1996). The change of a weak acid (acetic acid) for an even weaker one (H_2CO_3*) will increase the pH, shift the total carbonate composition from H_2CO_3* to CO_3^{2-} and saturate (even supersaturate) the leachate with carbonate, which precipitates as calcium carbonate ($CaCO_3$). Much of the bacterial population that brings about the anaerobic leachate degradation, develops within the drain (gravel or washed quarry stone), where the biofilm grows on the surfaces and is in contact with the organic-rich leachate. The precipitated calcium carbonate, along with other minerals such as iron sulphide and residues of fine material from the crushed stone and/or daily cover soil (Bennet et al. 2000), will then be entrapped into the biofilm forming what is called a clog.

Based on the previous findings, it can be said that calcite precipitation in a LCS is intimately associated with organic matter stabilization (i.e. COD removal from the leachate). Thus, a beneficial process is associated with a significant operational challenge. The positive event of organic matter degradation (and in the case of leachate recirculation, an accelerated one), could lead to a negative side effect such as the obstruction of the LCS of the landfill. Fleming (1999); Fleming and Rowe (2004); Cooke et al. (2001), Rowe et al. (2002); Rittman et al. (2003) and VanGulck (2003), quantified the connection between organic matter stabilization and clog formation. This connection was called the Yield Coefficient (Yc) and relates the amount of calcium (or carbonates) precipitated with the depletion of Chemical Oxygen Demand (COD) which is a general indicator of the organic matter concentration in the leachate.

2.8 Nitrifying and Denitrifying Bioreactors

Increased levels of nitrogen in the form of ammonia as a result of acceleration of the waste degradation is expected when leachate is recirculated (Knox, 1985). The major concerns of untreated nitrogen discharges include eutrophication of receiving waters, toxicity to aquatic life, dissolved oxygen depletion in receiving waters, and possible contamination of ground water. Fleming et al. (2002) also suggested that ammonia can affect the efficiency of the LCS to work as a fixed biofilm reactor. Thereby, ammonia treatment becomes an imperative.

Ammonia naturally degrades by two well known phenomena: nitrification and denitrification. The former takes place in the presence of oxygen (aerobic) and the latter in its absence (anaerobic). Since a landfill can provide both environments, it seems plausible to attempt to remove ammonia at the landfill site. A LCS has very low levels of oxygen (it has been depleted by bacteria) and might work as a denitrification reactor. On the other hand, an external concrete tank for example, could also work as a nitrification reactor.

In order to determine the feasibility of in-situ ammonia treatment, it is important to understand what nitrification and denitrification are and how they can be implemented.

Nitrification is an autotrophic aerobic process which utilizes an inorganic carbon source (carbonates), an inorganic electron donor or energy source (NH_4^+ or NO_2^-), and elemental oxygen as a terminal electron acceptor. The complete oxidation of ammonium to nitrate occurs in two intermediary steps by two different genera of autotrophic bacteria (US EPA 1975):

Nitrosomonas:

$$55\ NH_4^+ + 76O_2 + 109HCO_3^- \rightarrow \underset{\text{(bacteria)}}{C_5H_7NO_2^-} + 54NO_2^- + 57H_2O + 104H_2CO_3$$

(2.2)

Nitrobacter:

$$400\ NO_2^- + NH_4^+ + 4H_2CO_3 + HCO_3^- + 195O_2 \rightarrow \underset{\text{(bacteria)}}{C_5H_7NO_2^-} + 3H_2O + 400NO_3^-$$

(2.3)

Heterotrophic denitrification is an anoxic process which utilizes an organic carbon source (such as methanol) for synthesis and as an electron donor, and nitrite or nitrate as the terminal electron acceptor. (Azevedo et al. 1995). Complete denitrification occurs in two steps by a broad range of facultative bacteria such as *Pseudomonas*, *Micrococcus*, *Archromobacter*, and *Bacillus*. Equations for nitrite and nitrate reduction can be represented as follows (US EPA 1975):

Nitrate reduction

$$NO_3^- + 0.33CH_3OH \rightarrow NO_2^- + 0.33H_2O + 0.33H_2CO_3 \quad (2.4)$$

(methanol)

Nitrite reduction

$$NO_2^- + 0.5CH_3OH + 0.5H_2CO_3 \rightarrow 0.5N_2 + HCO_3^- + H_2O \quad (2.5)$$

(methanol)

Biological treatment of leachate with high ammonia concentration has been investigated by a number of researchers. Azevedo et al. (1995) found that a biological reactor (Modified Ludzack Ettinger-MLE) can treat ammonia levels of up to 1500 mg/L and that temperature plays an important role. Nitrification and /or denitrification ceased below a temperature of 10°C due to inhibition of bacterial activity.

Shiskowsky and Mavinic (1998) and Price et al. (2003), showed the importance of a source of carbon for heterotrophic denitrification. Onay and Pohland (1998) investigated nitrogen management in bioreactor landfills concluding that leachate recirculation increased uniformity of moisture, substrate, and nutrient distribution creating an environment that promoted the rapid development of the desired microbial population of denitrifiers, nitrifiers and methanogens.

The denitrification potential of actively decomposing and well decomposed refuse was measured by Price et al. (2003). Results showed that nitrate did inhibit methane production (microorganism obtain more energy for growth under nitrate-reducing conditions relative to methane-producing conditions. $\Delta G^{\circ} = -1120.5$ kJ and -31 kJ, respectively). However, the reactors recovered their methane–producing activity with the termination of nitrate addition. It was also hypothesized that nitrate, because of the large scale of a landfill, and the tendency of the different processes to dominate in different zones of the landfill, will not likely have a significant effect on methane yields in full-scale landfills. Therefore, landfills have significant capacity to convert nitrate to nitrogen, which can be safely released to the atmosphere.

2.9 Geotextiles in landfills

A geotextile can be defined as any permeable textile used in a geotechnical engineering work as an integral part of a man made project, structure, or system. Geotextiles are made from synthetics fibres such as polyester, polyethylene, or polypropylene and are usually classified as knitted, woven, or non-woven products (Polarczyk 2000).

The main usage of geotextiles in landfills is as a filter over various drainage materials such as soil materials, geonets, leachate collection pipes, leachate collection trenches (Quian et al. 2002). The non-woven needled punched geotextiles are the most commonly used in landfills as a filtration layer (Koerner and Koerner, 1995). Geotextiles are also use in landfills to separate the different material that comprise a LCS. Since filtration is one the most important uses of geotextiles in landfills, clogging of the geotextile appears as a very important factor of consideration. Figure 2.4 shows a MSW LCS typical design and the placement of the geotextile.

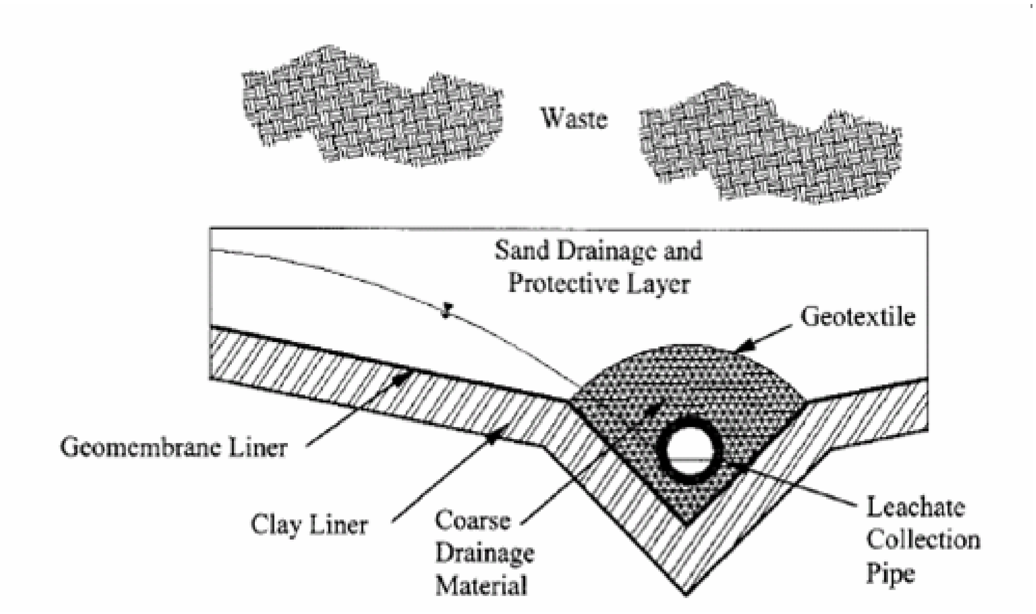

Figure 2.2 Leachate collection system in a MSW landfill. (After Craven et al. 1999)

Reinhart and Chopra (2000) described three types of clogging that can affect LCS of landfills and, therefore, geotextiles.

Chemical clogging

External phenomena such as the access of air and a drop in temperature can lead to changes in the solubility of compounds in leachate. For instance, leachate containing iron in contact with oxygen will oxidize the soluble Fe^{+2} into the insoluble Fe^{+3}. Also, a decrease in temperature may affect the solubility of certain salts (Ramke, 1989).

Another possible source of chemical clogging comes from changes in leachate chemistry which may lead to the saturation of carbonate in leachate and its precipitation in the form of calcite or hydroxides compounds (Rittman et al. 1996; Fleming et al. 1999, Rowe et al. 2002 and VanGulck 2003; Koerner et al. 1998, Halse et al. 1987 (part 1 and 2).

Particulate Clogging

According to Reddi (1997) particulate clogging can be divided in two main categories and both can be present in a landfill. The first one is a straining mechanism which occurs when the size of the filtration media is similar to that of the suspended particles resulting in a cake formation on the media.

The second one is the non straining mechanism where the driving forces of transport and removal between the particles and the media are physicochemical. This non straining mechanism can be of three types: Interception, which is the result of the collision of suspended particles with the fabric of the filtration media; Sedimentation which is the result of density differences between the suspended particles and water, and Brownian motion, which refers to the movement of micron and sub-micron particles due to diffusivity (Reinhart and Chopra, 2000)

Biological Clogging

Microbial biofilm consist of cells entrapped within a gelatinous matrix of extra cellular polymers (EP) produced directly from the surface associated microorganisms. It is believed that cells consume nutrients and redirect a portion of the substrate to exopolymer production, binding cells to the surface of the fibre. The EP acts as a cementing agent to reinforce cell binding to a surface (Polarczyk 2000).

Needle –punched non-woven geotextiles have been used successfully as bacterial support in aerobic and anaerobic up-flow reactors to treat domestic and industrial waste effluents. Because of their large porosity, their surface characteristic, the specific area of their fibres, the size of their pores and their permeability, these geotextiles offer many advantages for the development of bacterial biofilm growth (Rollin and Lombard 1988).

Based on the tendency of geotextiles to develop a bacterial biofilm growth, geotextiles are prone to clogging when permeated with a liquid with high organic matter content such as leachate. Knowing this, Koerner and Koerner (1992) and Koerner (1994) investigated several geotextile filters and soil filters, with different leachate strength, and compared their responses to water as a permeant. Their general finding is that geotextiles permeated by leachate with more than 2,500 mg/L of TSS or BOD required laboratory simulation to assess the severity of the clogging before placing them in the landfill.

Another source of biological clogging is through the degradation of the geotextile due to attachment and attack by microorganisms such as mould, mildew and fungi. They attack some types of finishes applied to textile fibres without attacking the fibres themselves. Among the materials used in geotextile manufacture, polyesters and polyolefins provide good biological resistance, while polyamides are known to be mildly attacked by mildew and bacteria (Gallagher, 1998).

Biological growth by itself, as discussed above, certainly creates a decrease in the ability of geotextiles to be permeated but, it is the mineral solid entrapment in the biofilm created by bacteria that aggravate the circulation problem. As Mlynarek and Vermeersch (1999) clearly explain:

> "Solid retention phenomenon is an additional factor in biological activities. At the filter interface, the solids, together with the microorganisms, are part of a dynamic system where all components interact with each other. This biofilm can combine with other biofilms around agglomerated particles. In well-aerated systems, aerobic organisms are active in breaking down the organic compounds while

> anaerobic conditions, such as those found in a LCS, favour encrustation. The result is the formation of a biofilm that may proceed to impede flow so much as to clog the system”.

2.10 Conclusions

Significant improvements have been made in leachate management during the past 20 years. The natural development in landfill research seems to be on issues related to post-closure time and minimisation of long term liabilities. Through proper operating conditions and innovative designs, bioreactors can act efficiently and shorten the time that waste needs to decompose. The advantages of reaching quicker stabilization are numerous including: maximization of landfill gas capture for energy projects, lower cost of leachate treatment and disposal, landfill space capacity reuse as a result of rapid settlement, and a reduction in landfill post-closure care and maintenance.

Nitrification, denitrification and organic matter degradation are phenomena well understood. They can play a central role in the objectives of reaching faster stabilization in landfills. The laboratory experiment described in the following chapters, demonstrates that these phenomena can be fully developed in the leachate collection system of a landfill and using an external aerated nitrification tank of an economically and technically feasible scale.

With this in mind, this thesis attempts to find a new role for LCS in landfills. Drains and geotextiles can be utilized as an engineered bacterial growth promoter with beneficial decreases in the resulting leachate strength. A combination of an effective drain-filter system with high surface area could play two important roles: to treat leachate while avoiding or mitigating the negative consequences of excessive clogging.

Chapter 3 Experimental Methods

3.1 Introduction

This chapter describes the experimental set up and the laboratory testing program. It presents the rationale for the different designs and operating conditions, a description of the initial hydraulic properties, and the methods used for the testing programs.

3.2 Column and Nitrification Reactor Set Up

The laboratory testing program described in this thesis used five 15cm (6”) diameter columns labelled C1, CC1, C2, CC2, C3 constructed from modular sections. Each column was made of five PVC bolted flanged "spool" sections 40 cm high. (See Figures 3.1). The uppermost “spool” section was used to contain a 15 cm layer of refuse. The refuse was dug up from an old closed landfill in Ontario (Canada). The main function of the refuse, was to inoculate the column with microorganisms and particulate matter since they are absent in the synthetic leachate (See Table 3.3). The material was cut, uniformly mixed, and placed on top of the uppermost geotextile in each column.

Since the size of the drain material has been shown to be an important controlling factor in the process of clog formation within the LCS (Armstrong 1998, Rowe et al, 2000), the lower four sections of the columns were each packed with a different size and type of granular drainage media as listed in Table 3.1.

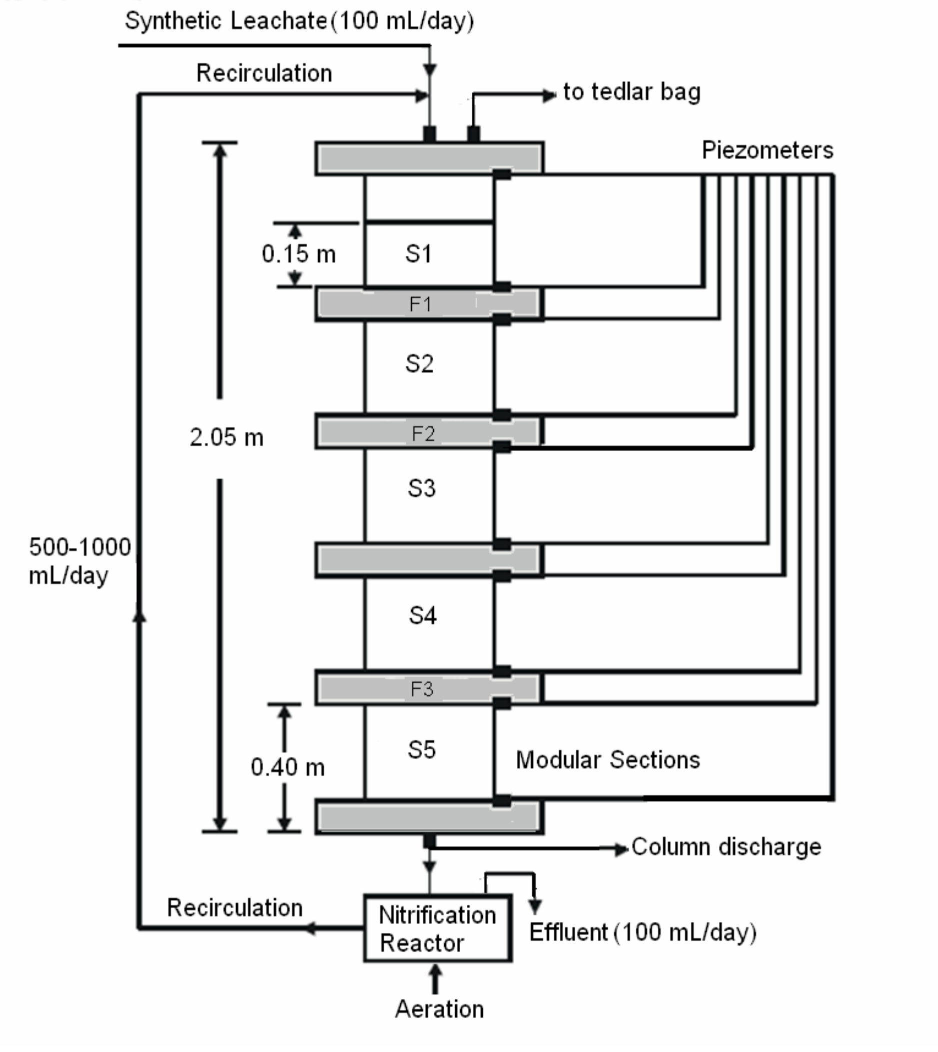

Figure 3.1 Column schematic

A large surface area increases the ability of microorganisms to attach and to grow, and increases the efficiency of the "passive treatment" COD removal (Fleming and Rowe, 2004). However, the likelihood of clog formation is greater for finer and more well graded material. Thus, each column represents a multilayer design in which each layer of finer material (for more treatment) are underlain by coarser layers (less susceptible to clogging).

Table 3.1 Anaerobic column reactors content

Section	**Column C1**	**Column CC1**	**Column C2**	**Column CC2**	**Column C3**
S1	Refuse	Refuse	Refuse	Refuse	Refuse
F1	Geotextile (1200R)	Geotextile (1200R)	Geotextile (1200R)	Geotextile (1200R)	Geotextile (1200R)
S2	37-50 mm gravel	20-25 mm WSC concrete	37-50 mm gravel	20-25 mm WSC concrete	37-50 mm gravel
F2	Geotextile (1200R)	Geotextile (1200R)	Geotextile (1200R)	Geotextile (1200R)	Geotextile (1200R)
S3	16-19 mm gravel	16-19 mm gravel	9-12.5 mm gravel	9-12.5 mm gravel	9-12.5 mm gravel
S4	9-12.5 mm gravel	9-12.5 mm gravel	5-9 mm gravel	5-9 mm gravel	5-9 mm gravel
F3	Geotextile (1200R)	Geotextile (1200R)	Geotextile (1200R)	Geotextile (1200R)	Geotextile (1200R)
S5	37-50 mm gravel	20-25 mm WSC concrete	37-50 mm gravel	20-25 mm WSC concrete	37-50 mm gravel

WSC concrete = washed-screen-crushed concrete

Recycled WSC concrete was also used in two columns (CC1 and CC2) in order to determine its suitability as a replacement for natural gravel as the drainage media. This was investigated since some landfills may have the possibility of using recycled concrete as a cheaper alternative.

The concrete was supplied by the City of Regina (Canada). The nominal diameter of the crushed concrete was somewhat smaller than that of the natural stone used in the top drainage layer. The concrete material contained fines in significant quantity which made it necessary to wash it thoroughly in order to avoid a rapid plugging of the geotextiles.

Geotextiles (Terrafix 1200R ^{TM}) were placed as indicated in Table 3.1. Their main functions were to work as filter, separator and promoter of bacterial growth. Koerner and Koerner (1992) demonstrated that geotextiles are suitable environments for the growth of microorganisms. Its main features are listed in Table 3.2

Table 3.2 Geotextile characteristics

Property	**MARV (minimum average roll value)**
GrabTensile Newtons (N) CAN/CGSB-148.1 No 7.3-92	1,550
Mullen Burst – Megapascals (Mpa) CAN/CGSB-4.2 No.11.1-94	3.80
Tear Propagation Newtons (N) CAN/CGSB-4.2 No.12.2-95	700
F.O.S. (um) Micrometres	50 to 150
Permeability K (cm/sec) CAN/148.1 No. 14-94	1.5 X 10^{-1}
Elongation % at Break	45 to 105
Standard Roll Sizes Metres (m)	3.5 X 50
Standard Roll Weight (lbs)	285

Manometers were installed along the length of the columns to monitor piezometric head and to allow for leachate sampling, if desired. Tedlar^{TM} bags with a volume capacity of 1L were placed at the top of every column for gas sampling.

Each column was also equipped with an external nitrification reactor that received leachate collected from the bottom of the column and which was completely aerated before it is recirculated to the top of the column. (See Figure 3.1).

Plastic 4 L pails were used as nitrification reactors vessels. Initially the column discharge was directly connected to the nitrification reactor. This initial system did not work out as expected because leachate was released to the nitrification reactor in batches and not continuously, resulting in upsets to the nitrification process. Therefore, 250 ml Erlenmeyers were used as column discharge collectors and hydraulic seals, allowing the column leachate discharge to be kept anaerobic and to be delivered steadily into the aerated nitrification reactor. (See figure 3.3)

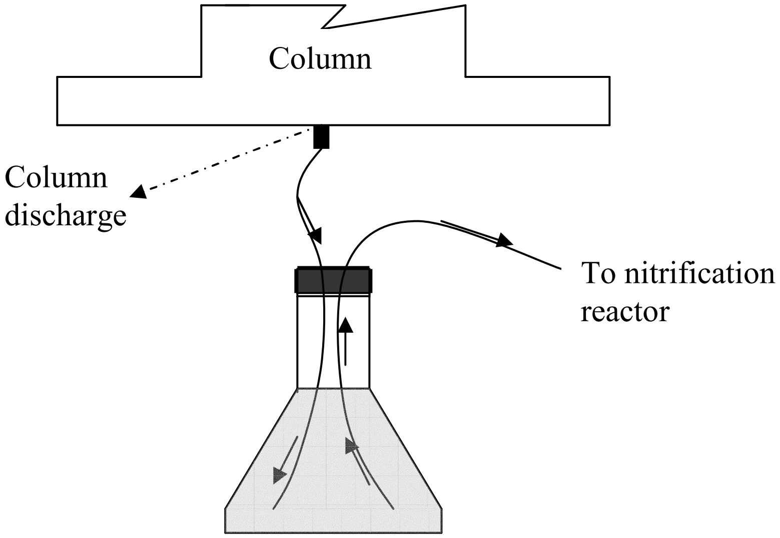

Figure 3.3 Column discharge apparatus.

The columns and the aerated nitrification reactors were operated in an environmental control chamber maintained at a constant temperature of 35°C which is within the temperature range that can be found during the anaerobic decomposition of solid waste at the LCS (Barone et al. 2000).

3.3 Experiment Mode of Operation

The experiment was divided into two phases. An initial phase 1 or preparation phase and a final phase 2 or the actual experiment.

3.3.1 Phase 1

The objective of the start-up phase 1 of the experiment was to inoculate bacteria inside the columns. In order to do so, a mixture was prepared with a microbial consortium representative of landfill conditions.

The mixture consisted of synthetic leachate mixed with leachate-saturated auger cuttings collected from boreholes drilled at a closed landfill (Brock west landfill, Ontario, Canada). The composition of the synthetic leachate was based on the work of Rowe et al (2002). The main characteristics of the synthetic leachate are given in Table 3.3. The mixture was incubated at 35°C for 3 days, then filtered with a geotextile (Terrafix 1200R) and circulated through the columns at a rate of 1 to 2 L/day for 120 days under unsaturated conditions. After this initial inoculation treatment the COD removal from the leachate was approximately 60%, and BARTs^{TM} (Biological Activity Reaction Test) showed that a stable and large bacterial population (approximately 5,000,000 cfu/mL) was present.

Concurrently, nitrifying bacteria were grown in the laboratory for use as an inoculating culture for each column's external nitrification reactor. In order to start up these nitrification reactors, sludge coming from the City of Saskatoon waste-waster treatment plant was used as a source of nitrifying bacteria.

Table 3.3 Synthetic leachate main characteristics

Characteristic	**Content or Level**
COD	17,345 mg/L
Hardness	4,400 mg/L as $CaCO_3$
Alkalinity	5,430 mg/L as $CaCO_3$
TDS	15,100 mg/L
Eh	-150 mV
pH	6
Total Nitrogen	766 mg/L as N
Calcium	2,900 mg/L as $CaCO_3$
Chloride	3,791 mg/L
Sodium	4,429 mg/L
Volatile Fatty Acids	13,000 mg/L
TOC	5,800 mg/L as C

Approximately 3 weeks after the sludge was introduced into the aerated vessel, and the reactor was fed with ammonia, nitrifying bacteria were detected indirectly through the presence of nitrite, and later nitrate, which confirmed the complete oxidation of ammonia.

3.3.2 Phase 2

Once the column inoculation and start-up phase was completed, synthetic leachate (see Table 3.4) which has been kept under anoxic conditions, and at a temperature of 4°C, was pumped to the top of each column at a rate of 100 mL/day. The concentration of the nitrification vessel was diluted from 3L to 15 L with distilled water and then evenly split amongst the five external 3 L nitrification reactors receiving leachate from the columns. The recirculation rate from the nitrification reactors to the column headspace was initially set to 1 L/day. As in phase one, phase 2 was conducted under unsaturated conditions.

3.4 Physical Parameter Testing

3.4.1 Hydraulic Conductivity Testing

The hydraulic conductivity and the permeability of the porous media packed into the five columns were measured before and after the experiment was run. To carry out these tests, water was used at the beginning and leachate at the end. Leachate was used at the end because of the possibility, if water was used, of perturbing or killing the bacterial community already established in the columns. The flow discharge used to calculate the initial and final hydraulic conductivity ranged from 130 to 320 mL/s. The differences in energy (ΔH) of every section (only represented by differences in pressure head) were measured using the piezometers installed on the different ports of the columns. Tables 3.4 to 3.8 show the piezometric line of each column obtained during the tests carried out in order to determine the original hydraulic conductivity.

Table 3.4 Composition of synthetic leachate (After Rowe et al, 2002)

Component	**Per 1L**
Acetic (ethanoic) acid	7 mL
Propionic (propanoic) acid	5 mL
Butyric(butanoic) acid	1 mL
K_2HPO_4	30 mg
$KHCO_3$	312mg
K_2CO_3	324 mg
NaCl	1440 mg
$NaNO_3$	50 mg
$NaHCO_3$	3012 mg
$CaCl_2$	2882 mg
$MgCl_2.6H_2O$	3114 mg
NH_4HCO_3	2439 mg
$CO(NH_2)_2$	695 mg
Trace metal solutions (TMS)(se below)	1 mL
$Na_2S.9H_2O$	Titrate to an Eh -120-180 mV
NaOH	Titrate to a pH 5.8-6.0
Distilled water	To make 1L
Component	Per 1L
Composition of trace metal solutions (TMS)	
$FeSO_4$	2000 mg
H_3BO_3	50 mg
$ZnSO_4.7H_2O$	50 mg
$CuSO_4.5H_2O$	40 mg
$MnSO_4$.H2O	500 mg
$(NH_4)_6Mo_7O_{24}$.4H2O	50 mg
$Al_2(SO_4)_3.16H_2O$	30 mg
$CoSO_4.7H_2O$	150 mg
$NiSO_4$.6H2O	500 mg
96 % concentration H_2SO_4 (Anal R)	1 mL
Distilled water	To make 1L

Tables 3.5-3.9 list the hydraulic conductivity values obtained for every section of the column. (See Figure 3.1 Column schematic) The initial and final hydraulic conductivity were measured for each section of the column under constant flow rate conditions, except for the final hydraulic conductivity of the uppermost geotextile in each column. For the tests, the columns were saturated and subjected to an upward flow of water (for the initial hydraulic conductivity) or leachate (for the final hydraulic conductivity) and the piezometers were read. With all this information, the hydraulic conductivity of the different sections of the columns were calculated. For the uppermost geotextiles at the end of the tests, the hydraulic conductivity was too low to enable this method to be used; accordingly the hydraulic conductivity was measured using falling head tests carried out on geotextile samples removed from the columns.

3.4.2 Porosity Testing

The porosity of all the porous media packed into the five columns was measured before and after the laboratory program. The volume of each PVC column section was determined and then packed with the porous media and filled with water. The liquid was drained and its volume determined. The differences between the volume of the PVC section and the drained liquid plus the additional water volume determined by drying the porous media, is the volume of solids of the sample. With this information, the porosity can be calculated by dividing the volume of voids by the total volume. (See From Figures 3.7 to 3.11 and Tables 3.10-3.14).

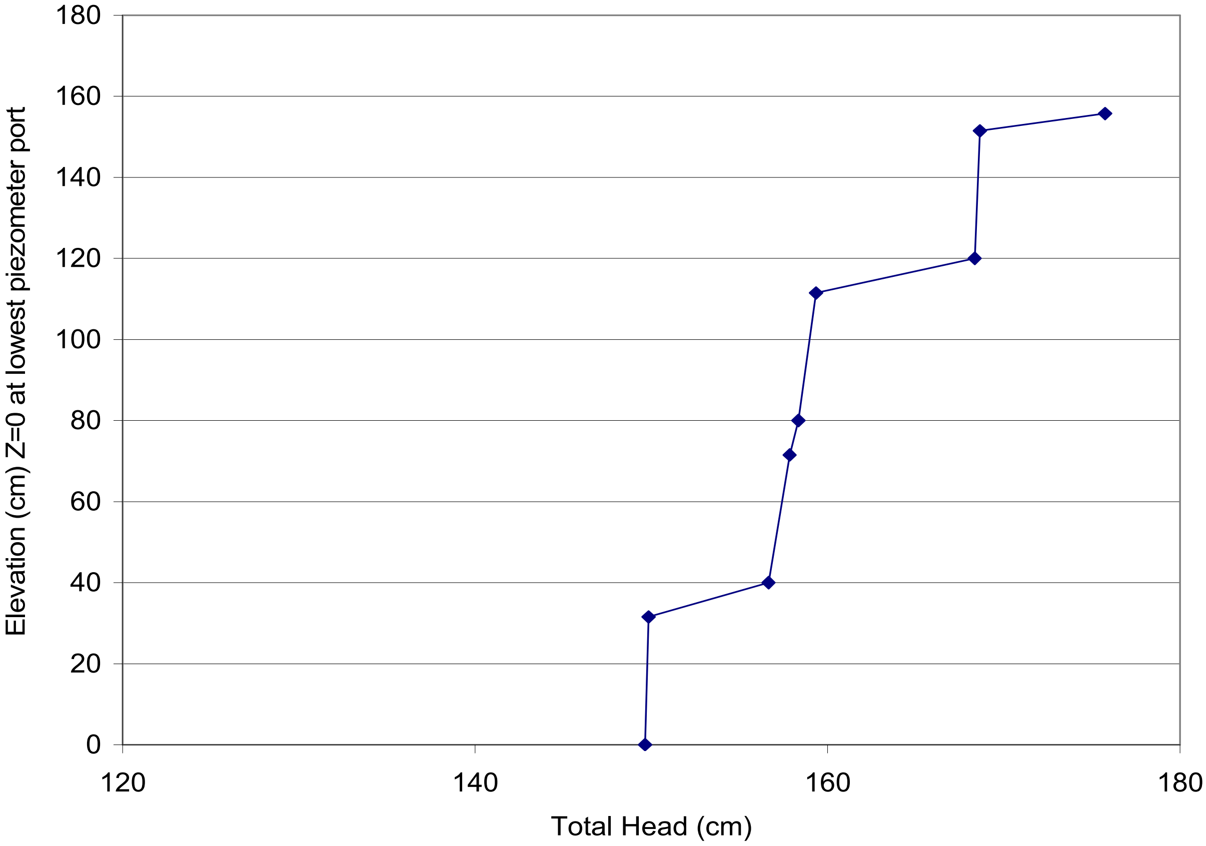

Figure 3.4 Column C1 initial piezometric line

Table 3.5 Hydraulic conductivity of different materials in column C1

Material	Hydraulic Conductivity (cm/sec)
Geotextile	0.076
Stone 37.5-50mm	179
Geotextile	0.056
Stone 16-20mm	54
Stone 9-12.5mm + Stone 16-20 mm	21
Stone 9-12.5 mm	45
Geotextile	0.075
Stone 37.5-50mm	269
Q = 312 mL/sec	
Area = 182.4 cm2	

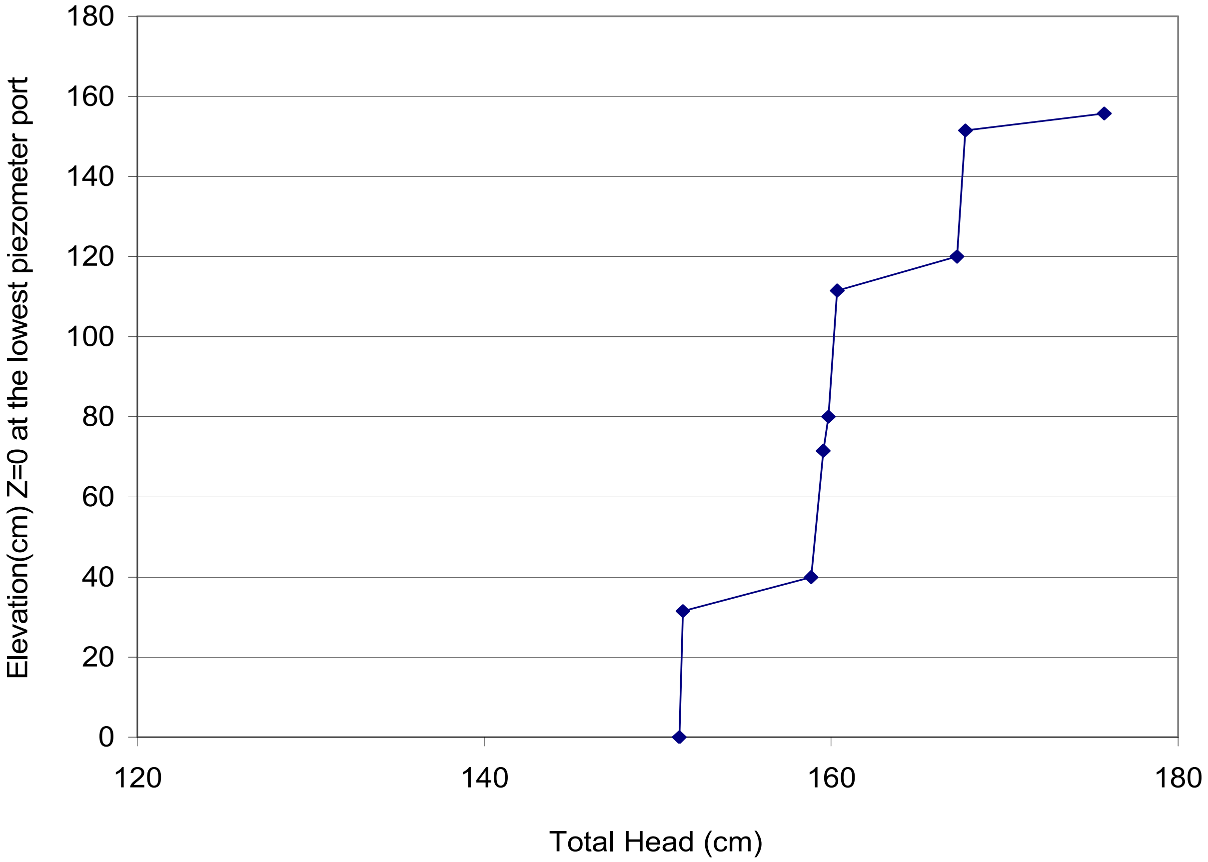

Figure 3.5 Column CC1 initial piezometric line

Table 3.6 Hydraulic conductivity of different materials in column CC1

Material	**Hydraulic Conductivity (cm/sec)**
Geotextile	0.04
Concrete 20-25mm	65
Geotextile	0.06
Stone 16-20mm	64
Stone 9-12.5mm + Stone 16-20 mm	22
Stone 9-12.5 mm	45
Geotextile	0.04
Concrete 20-25mm	161
Q = 186 mL/sec	
Area = 182.4 cm2	

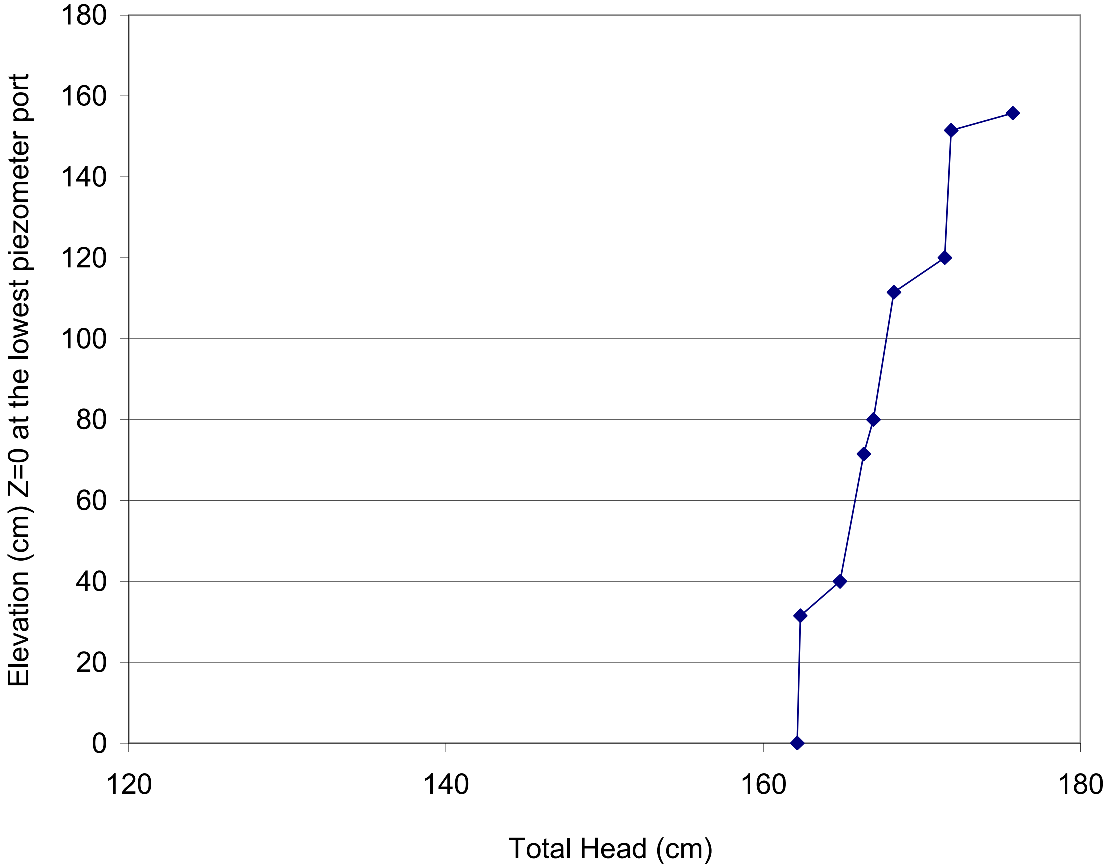

Figure 3.6 Column C2 initial piezometric line

Table 3.7 Hydraulic conductivity of different materials in column C2

Material	**Hydraulic Conductivity (cm/sec)**
Geotextile	0.12
Stone 37.5-50mm	123
Geotextile	0.14
Stone 9-12.5mm	38
Stone 5-9mm + Stone 9-12.5 mm	17
Stone 5-9 mm	33
Geotextile	0.08
Stone 37.5-50mm	242
Q = 280 mL/sec	
Area = 182.4 cm2	

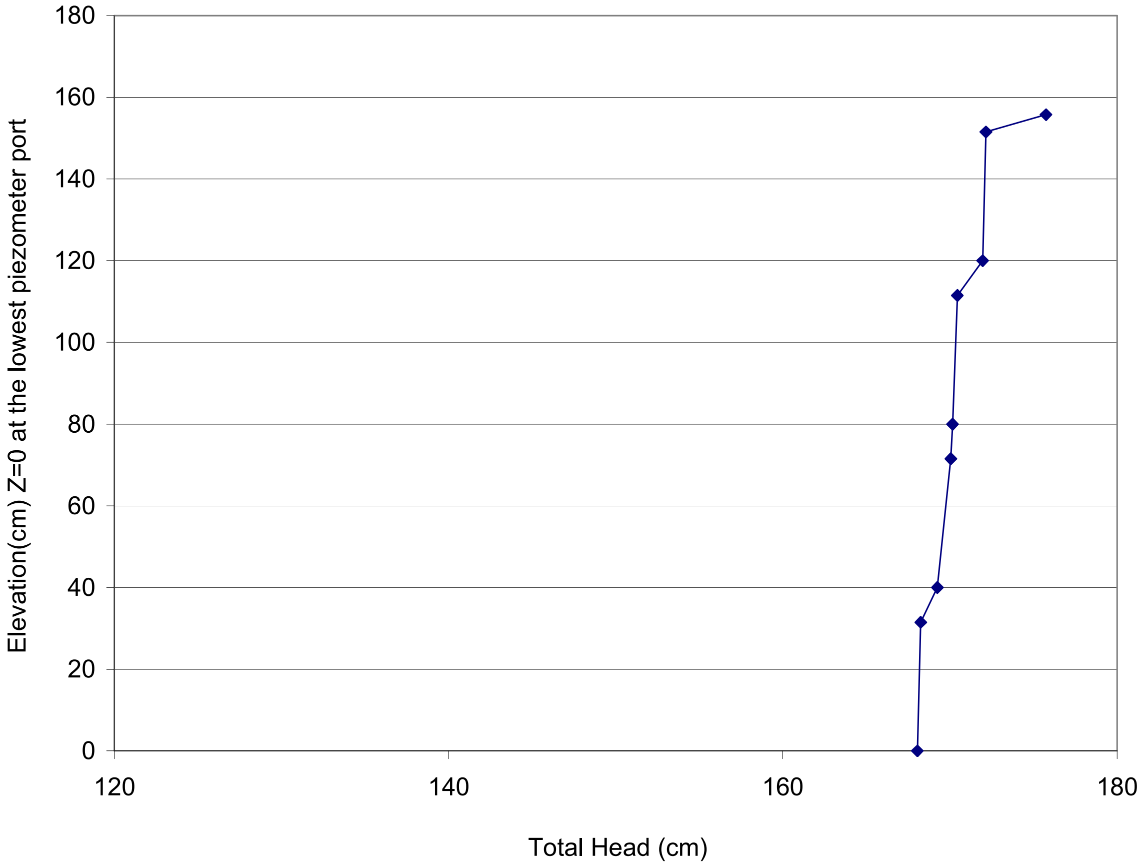

Figure 3.7 Column CC2 initial piezometric line

Table 3.8 Hydraulic conductivity of different materials in column CC2

Material	**Hydraulic Conductivity (cm/sec)**
Geotextile	0.056
Concrete 20-25mm	112
Geotextile	0.14
Stone 9-12.5mm	75
Stone 5- 9mm + Stone 9-12.5 mm	32
Stone 5-9 mm	28
Geotextile	0.23
Concrete 20-25mm	112
Q =130 mL/sec	
Area = 182.4 cm2	

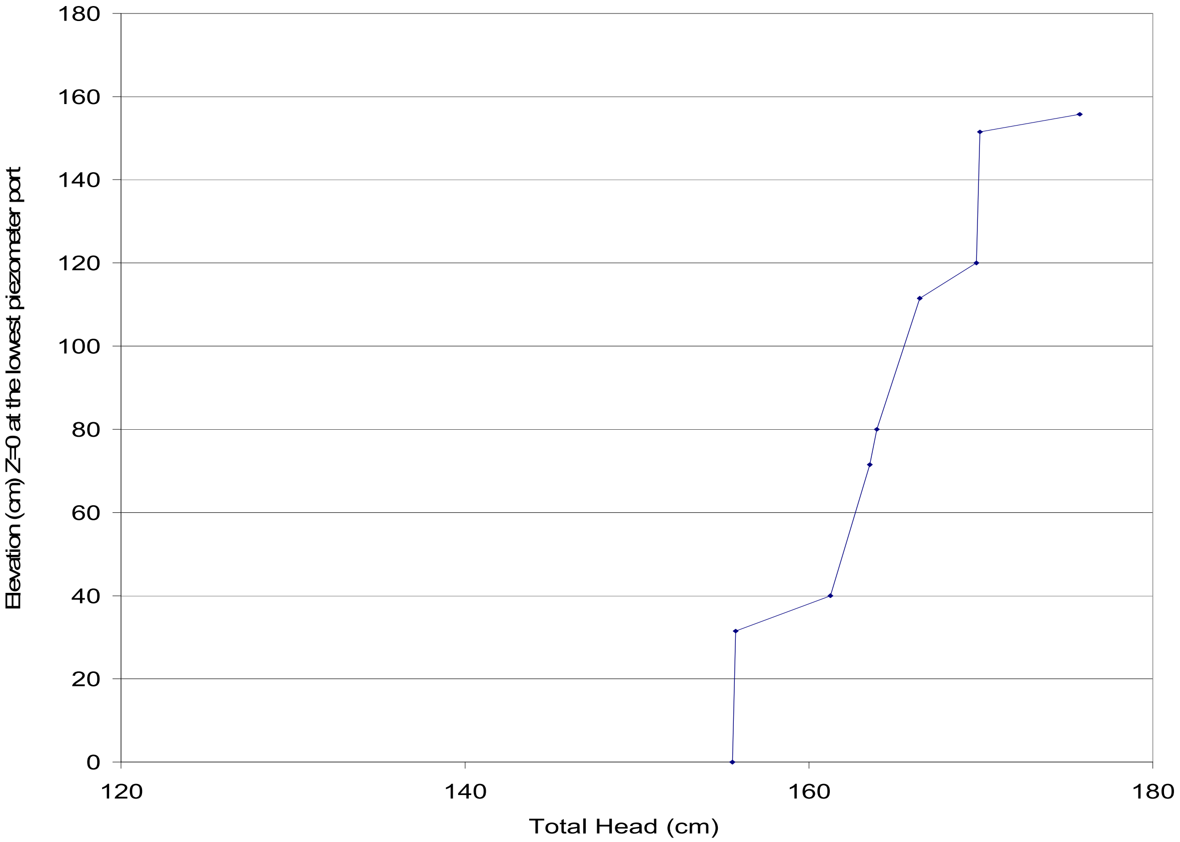

Figure 3.8 Column C3 initial piezometric line

Table 3.9 Hydraulic conductivity of different materials in column C3

Material	Hydraulic Conductivity (cm/sec)
Geotextile	0.08
Stone 37.5-50mm	241
Geotextile	0.15
Stone 9-12.5mm	20
Stone 5-9mm + Stone 9-12.5 mm	16
Stone 5-9 mm	21
Geotextile	0.085
Stone 37.5-50mm	242
Q = 280 mL/sec	
Area = 182.4 cm2	

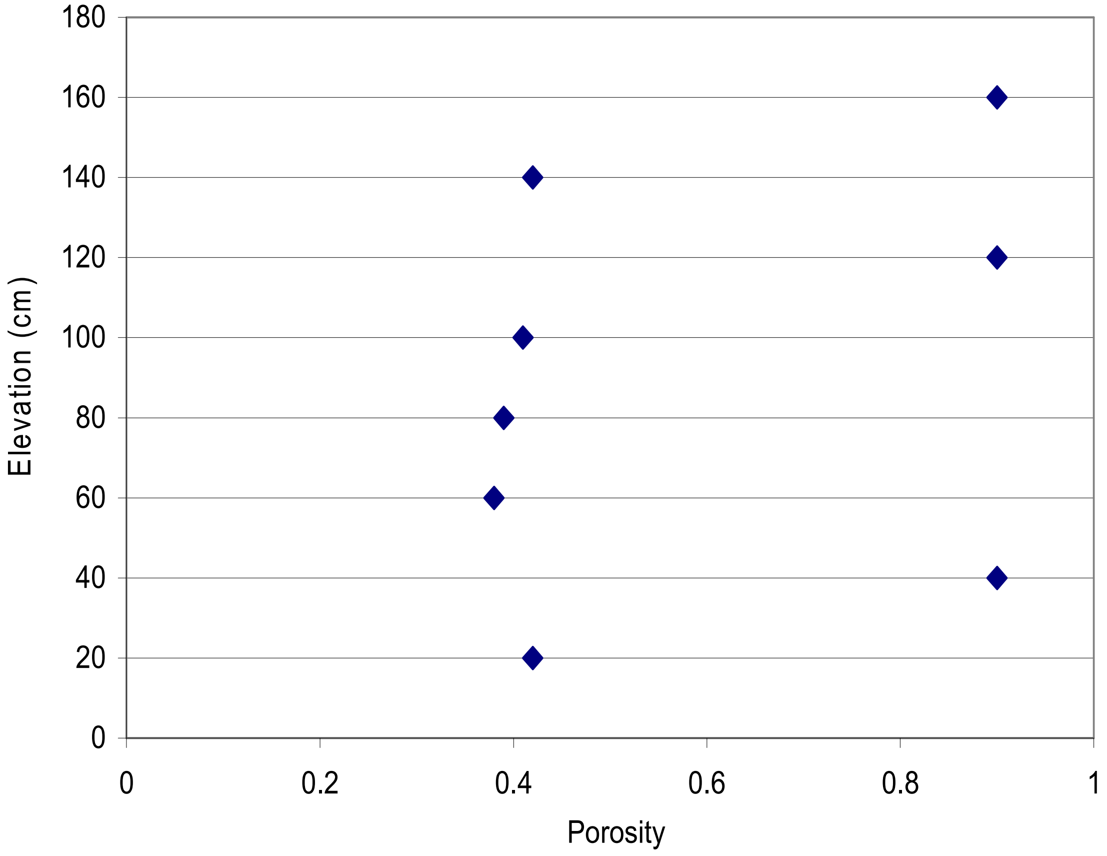

Figure3.9 Initial porosity profile of column C1

Table 3.10 Column C1 initial porosity

Material	Porosity
Geotextile	0.90
Stone 37.5-50mm	0.42
Geotextile	0.90
Stone 16-20mm	0.41
Stone 9-12.5mm + Stone 16-20 mm	0.39
Stone 9-12.5 mm	0.38
Geotextile	0.90
Stone 37.5-50mm	0.42

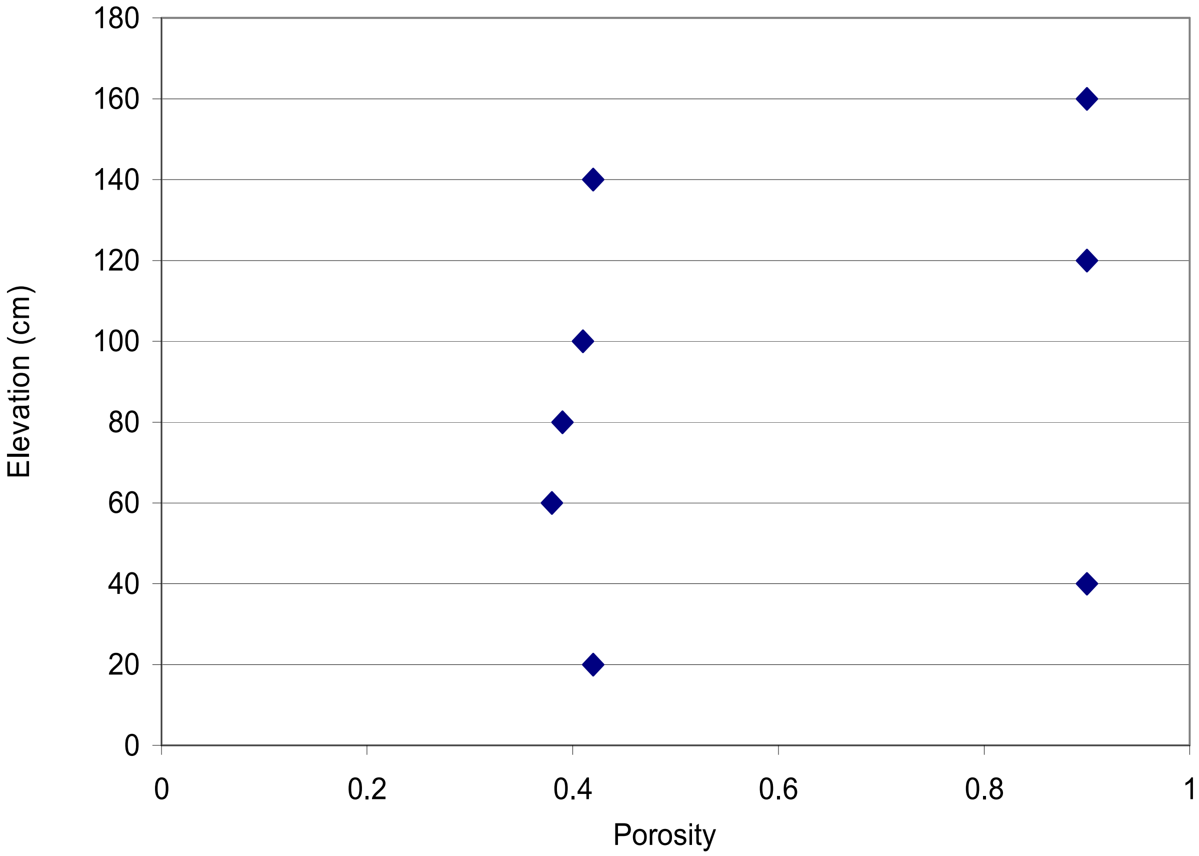

Figure 3.10 Initial porosity profile of column CC1

Table 3.11 Column CC1 initial porosity

Material	Porosity
Geotextile	0.90
Concrete 20-25mm	0.44
Geotextile	0.90
Stone 16-20mm	0.41
Stone 9-12.5mm + Stone 16-20 mm	0.39
Stone 9-12.5 mm	0.38
Geotextile	0.90
Concrete 20-25mm	0.44

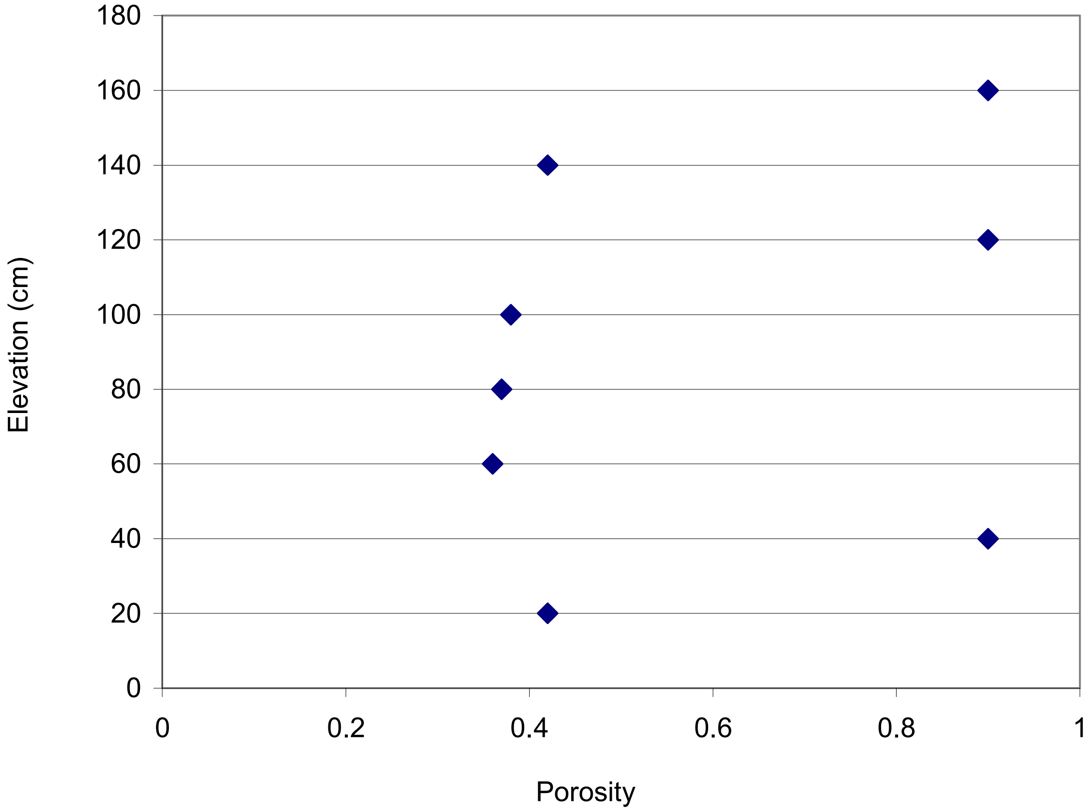

Figure 3.11 Initial porosity profile of column C2

Table 3.12 Column C2 initial porosity

Material	**Porosity**
Geotextile	0.90
Stone. 37.5-50mm	0.42
Geotextile	0.90
Stone. 9-12.5mm	0.38
Stone. 5-9mm + Stone. 9-12.5 mm	0.37
Stone. 5-9 mm	0.36
Geotextile	0.90
Stone.37.5-50mm	0.42

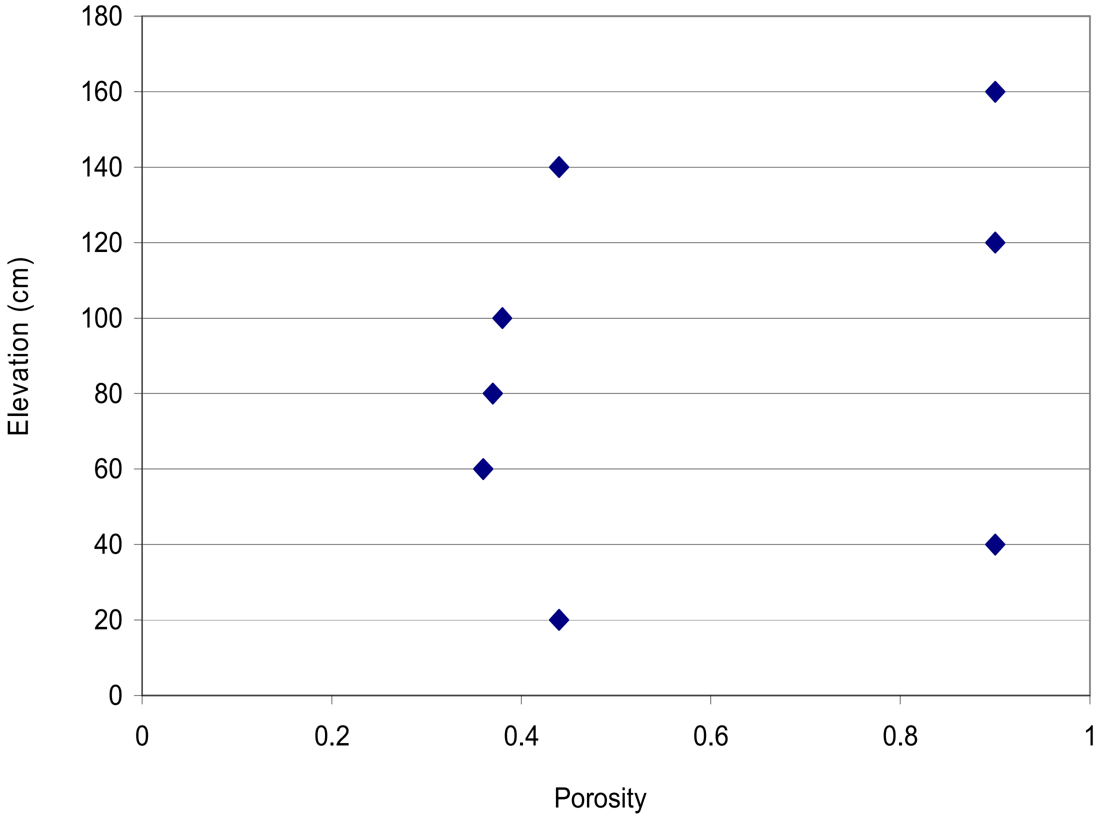

Figure 3.12 Initial porosity profile of column CC2

Table 3.13 Column CC2 initial porosity

Material	**Porosity**
Geotextile	0.90
Concrete 20-25mm	0.44
Geotextile	0.90
Stone 9-12.5mm	0.38
Stone 5- 9mm + Stone.9-12.5 mm	0.37
Stone 5-9 mm	0.36
Geotextile	0.90
Concrete 20-25mm	0.44

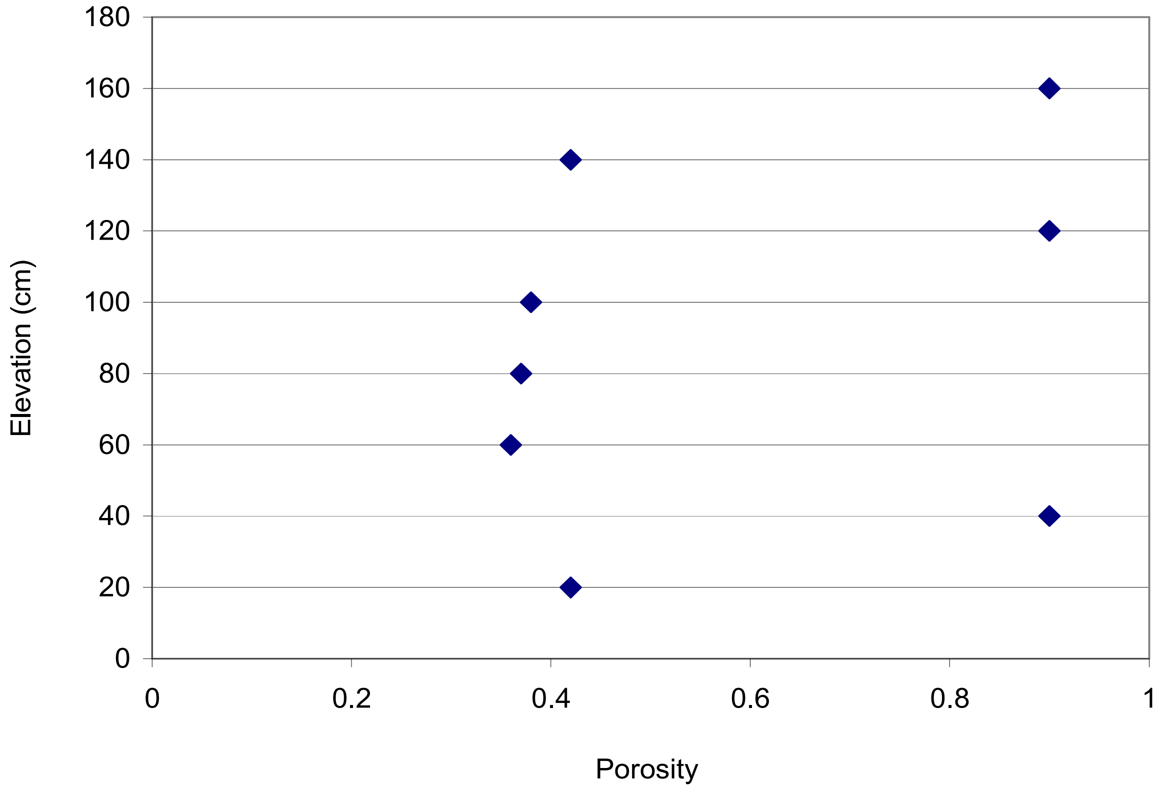

Figure 3.13 Initial porosity profile of column C3

Table 3.14 Column C3 initial porosity

Material	Porosity
Geotextile	0.90
Stone 37.5-50mm	0.42
Geotextile	0.90
Stone 9-12.5mm	0.38
Stone 5-9mm + Stone 9-12.5 mm	0.37
Stone 5-9 mm	0.36
Geotextile	0.90
Stone 37.5-50mm	0.42

3.5 Water Quality Testing Program

Water quality testing was performed on samples of influent and effluent leachate at a different frequency according to the experiment phase (see Table 3.15). Tests were conducted to obtain: pH, Electrical Conductivity (Ec), Oxidation Reduction Potential (Eh), Ammonia, Nitrite, Nitrate, Total Nitrogen, Alkalinity, Total Organic Carbon (TOC), Volatile Fatty Acids (VFA), Calcium Hardness, Chemical Oxygen Demand (COD), Biological Activity Reaction Test (BART^{TM}), Sodium, Magnesium, Sulphate, Iron, Chloride. The frequency of sampling conducted in each phase of the experiment is listed in Table 3.15.

Table 3.15 Frequency of the water quality testing program

Parameter	**Frequency 1st phase**	**Frequency 2nd phase**
pH	Three times/week	Three times/week
Ec	Three times/week	Three times/week
Eh	Twice/week	Twice/week
Ammonia	Twice/week	Twice/week
Nitrite	Not performed	Twice/week
Nitrate	Not performed	Twice/week
Total Nitrogen	Infrequently	Once/week
Alkalinity	Infrequently	Twice/week
Calcium Hardness	Once/week	Once/week
Chemical Oxygen Demand	Once/week	Once/week
Biological Activity Reaction Tests (BART^{TM}),	Once/week	Infrequently
Total organic Carbon (TOC)	Infrequently	Infrequently
Total Inorganic Carbon (TIC)	Infrequently	Infrequently
Volatile Fatty Acids (VFA),	Infrequently	Infrequently
Sodium	Infrequently	Infrequently
Magnesium	Infrequently	Infrequently
Sulphate	Infrequently	Infrequently
Iron	Infrequently	Infrequently
Chloride	Infrequently	Infrequently

pH, Electrical conductivity (Ec) and Oxido Reduction Potential (Eh) were measured with a HACH 44600^{TM} probe. The calibration of the probe was checked from time to time using standard solutions.

Ammonia was measured with TECATOR ^{TM} distillation apparatus . Nitrite and Nitrate with an auto-analyser by Technicon^{TM}, Total nitrogen and COD were monitored using the HACH ^{TM} Reactor Digestion Method. This method uses vials that are analysed with the HACH ^{TM} DR 4000U Spectrophotometer.

Alkalinity was measured using an autotitrator and bringing the pH down to 8.3 for carbonate alkalinity and down to 4.5 for bicarbonate alkalinity. The machine used was pH titrator E512 Metrohm Herisau. Sodium and Potassium were measured with a Corning flamephotometer 430. Iron with a Peekinelmer 5000 atomic absorption spectrophotometer. Sulphates were measured using the gravimetric method with ignition of residue.

TOC (Total Organic Carbon) and TIC (Total Inorganic Carbon) analyses were performed with a Telemar Dorhman^{TM} , Phoenix 800t uv-persulfate analyser. Volatile Fatty Acids (VFA) was carried out by distillation under standard methods.

Biological Activity Reaction Tests (BART^{TM}) were used to monitor the presence and changes to the microbial community within the columns and it is performed with prepared vials designed to measured bacterial activity.

3.6 Gas Production

The volume of gas generation during phase one was measured occasionally using a GEM 2000 device which indicates the percent of volume CO_2, CH_4, O_2 and Balance (which represent any element different to the three previously mentioned). For phase 2, gas production was collected in 1L Tedlar bags connected to the gas headspace at the top of each column. The Tedlar bags were almost emptied (some gas was left inside the bag in order to avoid any vacuum being created by sampling. This ensured that the gas pressure inside the columns would always be known, i.e. atmospheric. Similarly, the bags were emptied before they could fill completely and cause a build-up of pressure inside the columns. Cumulative gas production was measured once the experiment stabilized. Results are presented in Chapter Four. Analysis of the gas concentration was performed at the Saskatchewan Research Center Analytical Laboratory.

3.7 Simulation of Leachate Generation and Recirculation

One of the objectives of this thesis was to create operating conditions similar to those that occur in real landfills in order to evaluate the efficiency in leachate collection.

Infiltration into North American landfills may range from 0.05 $m^3/m^2/yr$ in dry regions to 0.2 $m^3/m^2/yr$ in humid regions (Quian et al, 2002). Assuming an average rate of infiltration into eastern Canadian landfills of 0.175 $m^3/m^2/yr$, a medium size landfill of 25 ha would produce approximately 120 m^3/day of leachate.

Furthermore, assuming a typical recirculation rate of 5 times the leachate production rate (in this case 600 m^3/day) and a typical hydraulic retention time (HRT) of 2.7 days for the nitrification reactor; then the volume of the nitrification reactor needed would be approximately 1,600 m^3. This detention requirement could easily be met using concrete tanks or a small lined lagoon at the landfill site. These field estimates were taken into account in scaling the laboratory tests; however if those field parameters were used in the experiment, the leachate production rate would be rather low for practical parameter testing (approximately 9 mL/day) due to the small cross sectional area of the columns (182.4 cm^2).

As a result of the previous considerations, the synthetic leachate input or infiltration rate was set at 100 mL/day (roughly 10 times larger than the midsize Canadian landfill previously discussed). The recirculation rate was also initially set 10 times larger that the infiltration rate. The nitrification reactor HRT was maintained at 2.7 days. These operating conditions resulted in a nitrification reactor volume of 3 L and a recirculation rate of 1 L/day for the experiment.

All the experiment operating parameters were subject to adjustments depending on the results obtained upon implementation of the feed of synthetic leachate into the columns. However, the experiment attempted as much as possible to maintain operating conditions representative of typical landfill conditions.

Chapter 4 Evaluation of Leachate Quality Using a Modified Drainage LCS

4.1 Introduction

The concept of a bioreactor landfill, with leachate recirculation as its main feature, has the advantage of being effective and economically viable mainly because it relies entirely upon in-situ biological treatment. However, the utilization of landfills for treatment in addition to accumulation of refuse has the potential to create additional geotechnical and environmental challenges. These include slope failure due to decreases in internal and/or interface shear strength (Quian et al., 2002), clogging of the leachate collection systems(LCS) (Brune et al., 1991; Fleming et al., 1999, 2004 and VanGulck et al. 2003) and increased levels of ammonia (Pohland, 1995).

This chapter describes the results obtained in synthetic leachate COD, nitrogen and calcium concentration as well as the composition of the biogas generated, after synthetic leachate underwent anaerobic and aerobic biodegradation through the use of columns simulating leachate collection drains and external aerated reactors, respectively. A mass balance for carbonaceous and nitrogenous compounds is also performed in order to confirm the leachate treatment effectiveness.

4.2 Laboratory Testing Results

4.2.1 pH and Alkalinity.

Figure 4.1 and 4.2 show the pH of the anaerobic column discharge and the aerated nitrification reactor effluent (termed the aerated effluents). Considering that the synthetic leachate has an initial pH of 6 (see Table 4.1), the much higher values of 8.3-8.6 in the anaerobic column discharge, suggest a high bacterial population that rapidly decreases the volatile organic acid content (mainly represented by acetic acid) of the leachate. Figure 4.1 also shows that the pH in the columns packed with crushed concrete is slightly higher than for those columns packed with natural gravel (this subject is deepened in section 5.3). This effect is not apparent in the aerated effluent as shown in Figure 4.2. Figures 4.1 and 4.2 also show that the pH of the anaerobic column discharge is about one-half of a pH unit lower than the aerated effluent, reflecting the evolution of dissolved carbonates from the open vessel of the nitrification reactor. Thus the anaerobic degradation of the leachate in the column replaces the weak carboxylic acids with even weaker carbonic acid (Fleming et al., 1999). When the leachate is discharged to the aerated nitrification reactor, off gassing of dissolved CO_2 occurs readily.

This CO_2 off gassing also explains the difference between alkalinities as seen in Figures 4.3 and 4.4. The biogas produced anaerobically in the columns was measured to be 1.6%

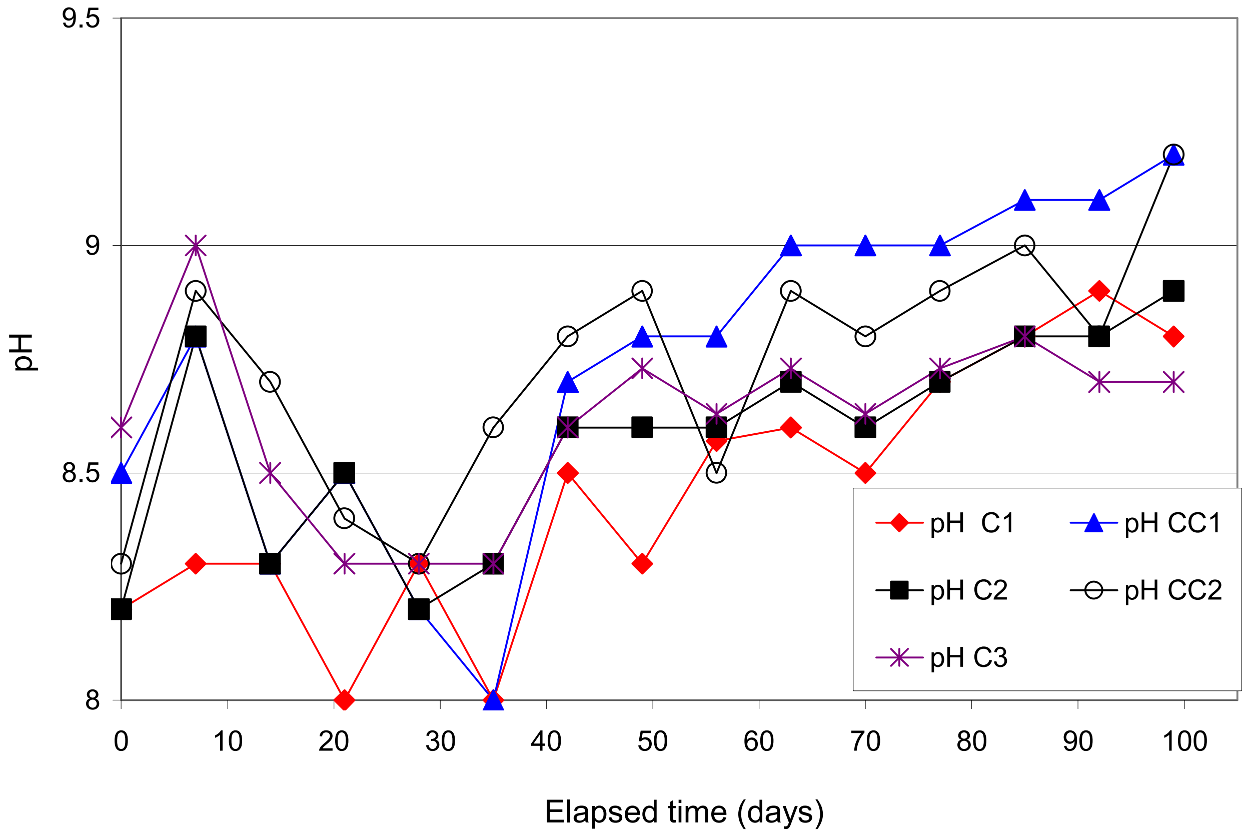

Figure 4.1 pH in anaerobic column discharge

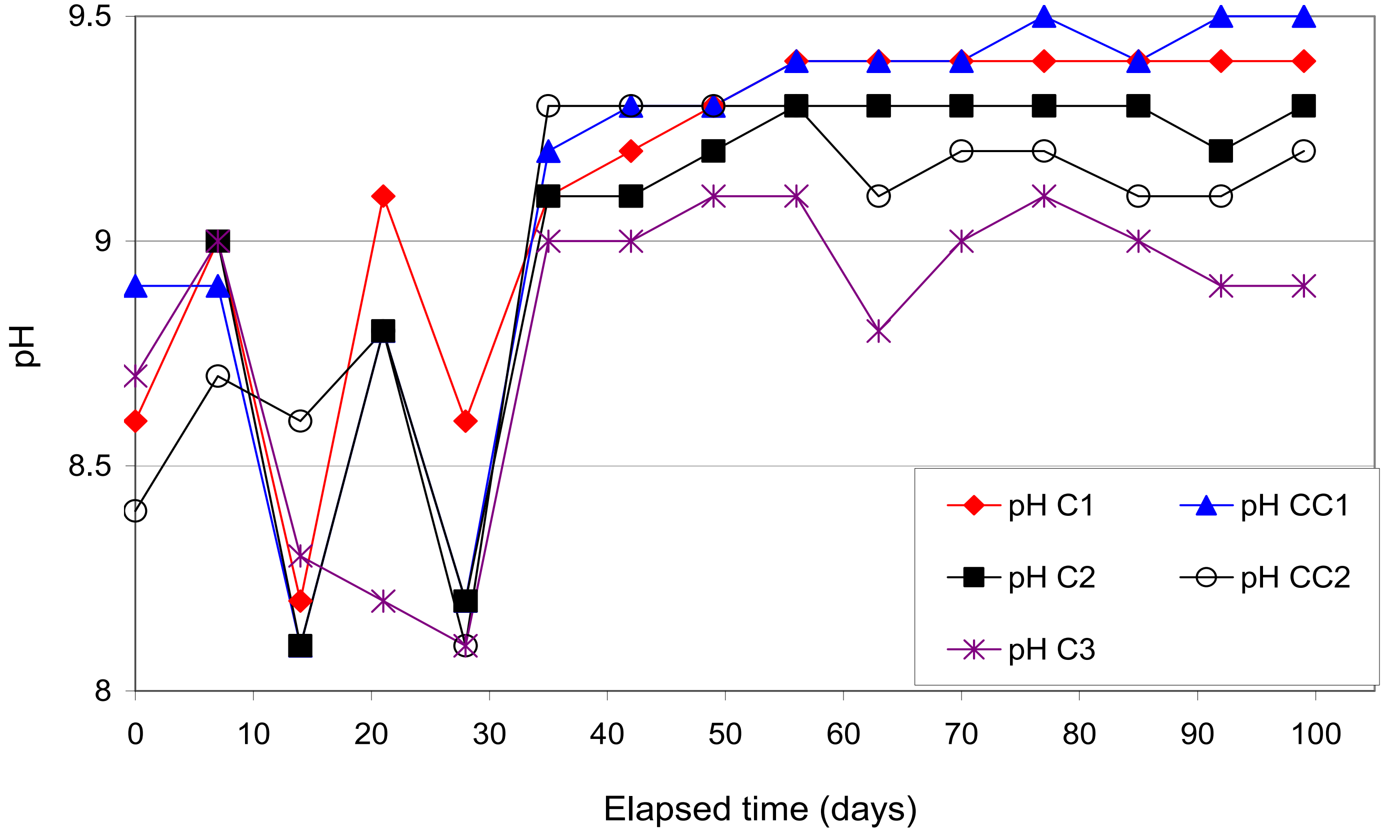

Figure 4.2 pH in aerated effluent

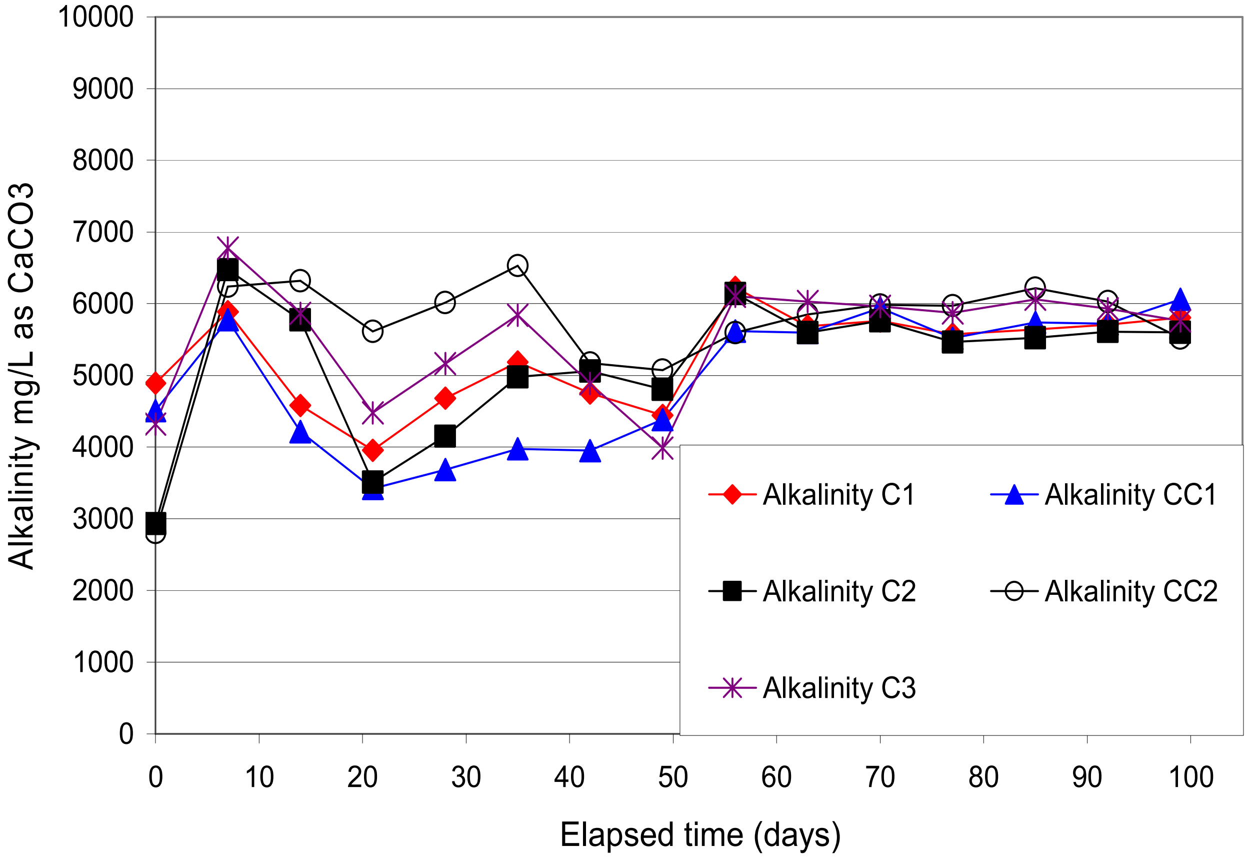

Figure 4.3 Alkalinity in anaerobic column discharge

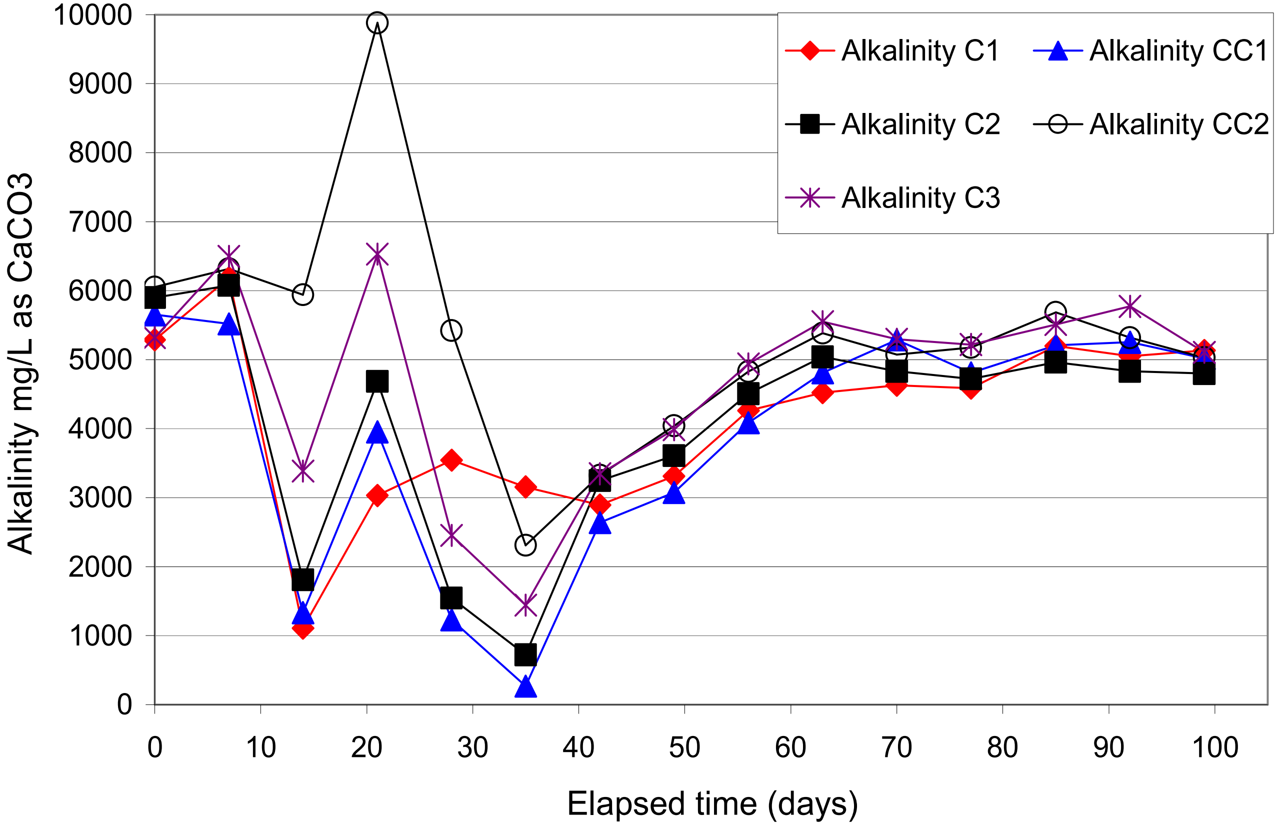

Figure 4.4 Alkalinity in aerated effluent

CO_2, thus within the columns reactors, leachate is assured to be at equilibrium with a CO_2 partial pressure (0.016 atm) higher than in the aerated nitrification reactor (0.0003 atm). This condition makes leachate in the column reactor more acidic (by carbon dioxide hydrolysis) and at the same time increases the alkalinity due to carbonates augmentation.

4.2.3 Ammonia Conversion to Nitrate

Figure 4.5 presents test results for ammonia concentration in the outflow of the aerated nitrification reactors. The ammonia removal efficiency is referenced to the same source concentration, that of the synthetic leachate. During the first 35 days some operational difficulties resulted in high levels of ammonia. This was solved with minor changes to the system that enabled a more continuous flow. Figures 4.5 and 4.6 show that after 50 days a stable nitrification process was established and all ammonia was converted to NO_3^-.

4.2.4 Nitrate Conversion to Nitrogen Gas

Data from only a single column (C2) will be presented in this section given that all the columns have followed very similar trends for each of the parameters tested. The other columns show essentially the same behaviour.

Figure 4.7 shows that after day 50, stable conditions are evident, ammonia is completely oxidized and only NO_3^- dominates the nitrogenous compounds. Further, Figure 4.8 does

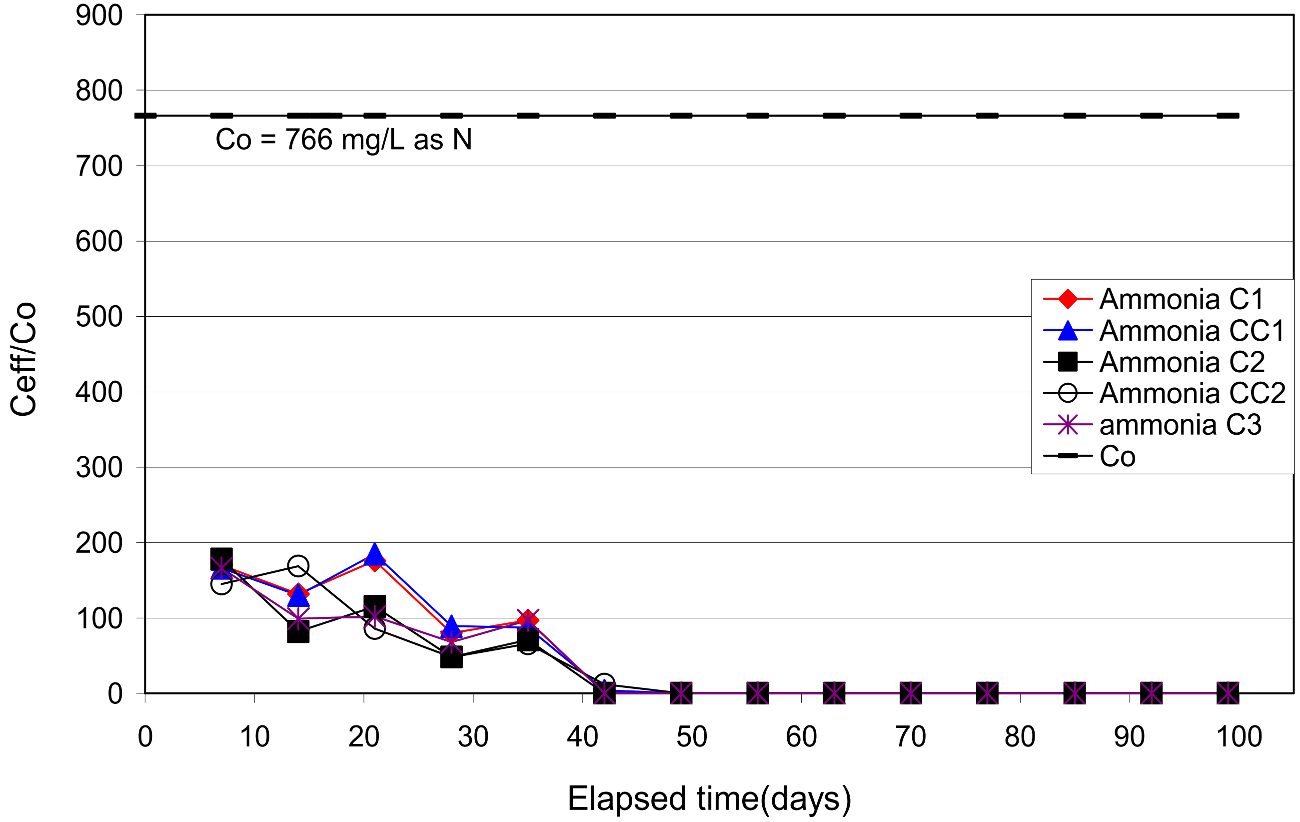

Figure 4.5 Ammonia concentration in aerated effluent

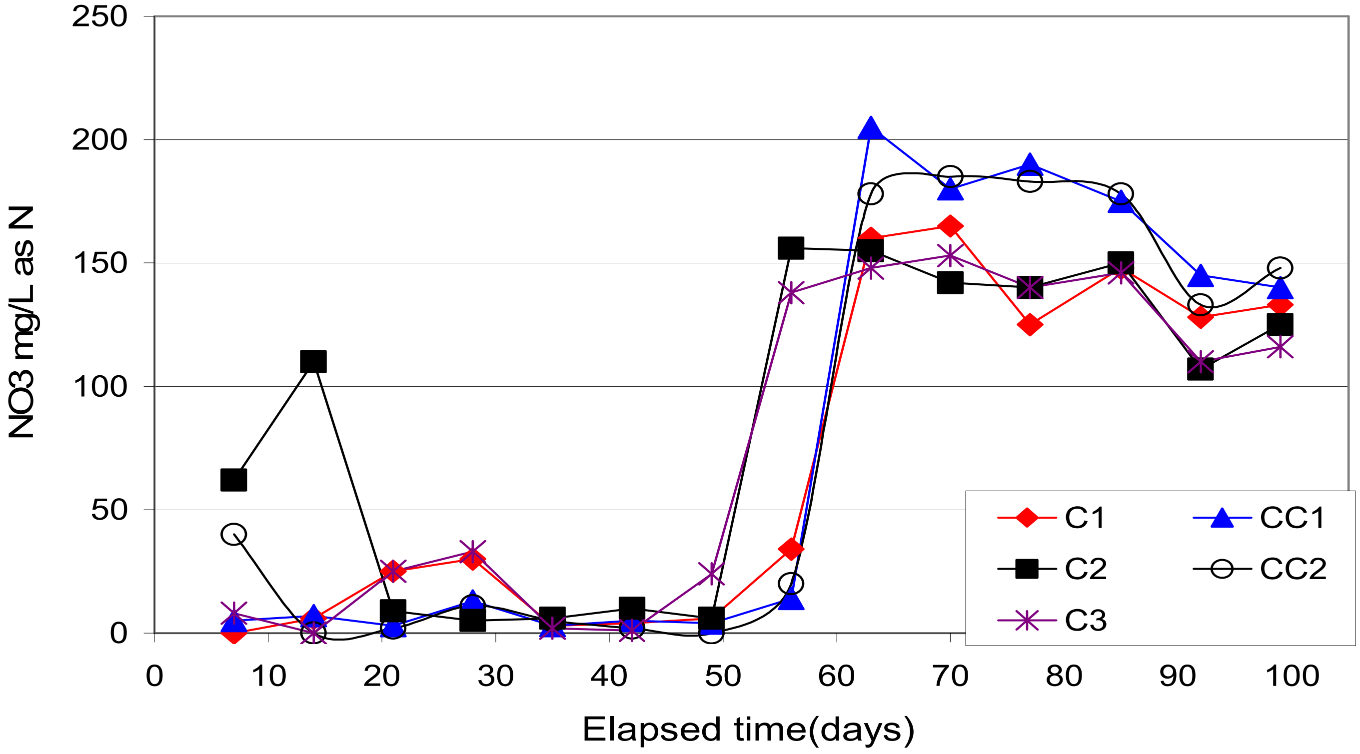

Figure 4.6 NO_3^- concentration in the aerated effluent

not show any sizable amount of either nitrite or nitrate in the anaerobic column discharge. This suggests nitrate conversion to $N_{2(gas)}$. Nitrogen gas production was confirmed by collection analysis of gas produced by the column. Approximately 37% of the daily gas production was $N_{2(gas)}$.

It is also interesting to see that between days 37 and 50, NO_2^- was the main nitrogenous compound produced in the aerated nitrification reactor (See Figure 4.7). However figure 4.8 shows almost no presence of it in the column reactor. This could be explained by anaerobic ammonium oxidation (Anammox) where ammonium is oxidized under anaerobic conditions, NO_2^- acts as a final electron acceptor and CO_2 is used as the main carbon source for growth (Jetten et al ,2001). The stoichiometry of this reaction is:

{1} $NH^{4+} + NO_2^- \rightarrow N_2 + 2H_2O$

The loss of ammonia by this process requires approximately equimolar concentrations of ammonia and nitrite (Price et al., 2004) and normally occurs at the interface of an anaerobic/aerobic system (Schmidt et al., 2002). Both conditions are met in this experiment. Nonetheless, further investigation, beyond the scope of this study, is necessary to determine the validity of this hypothesis.

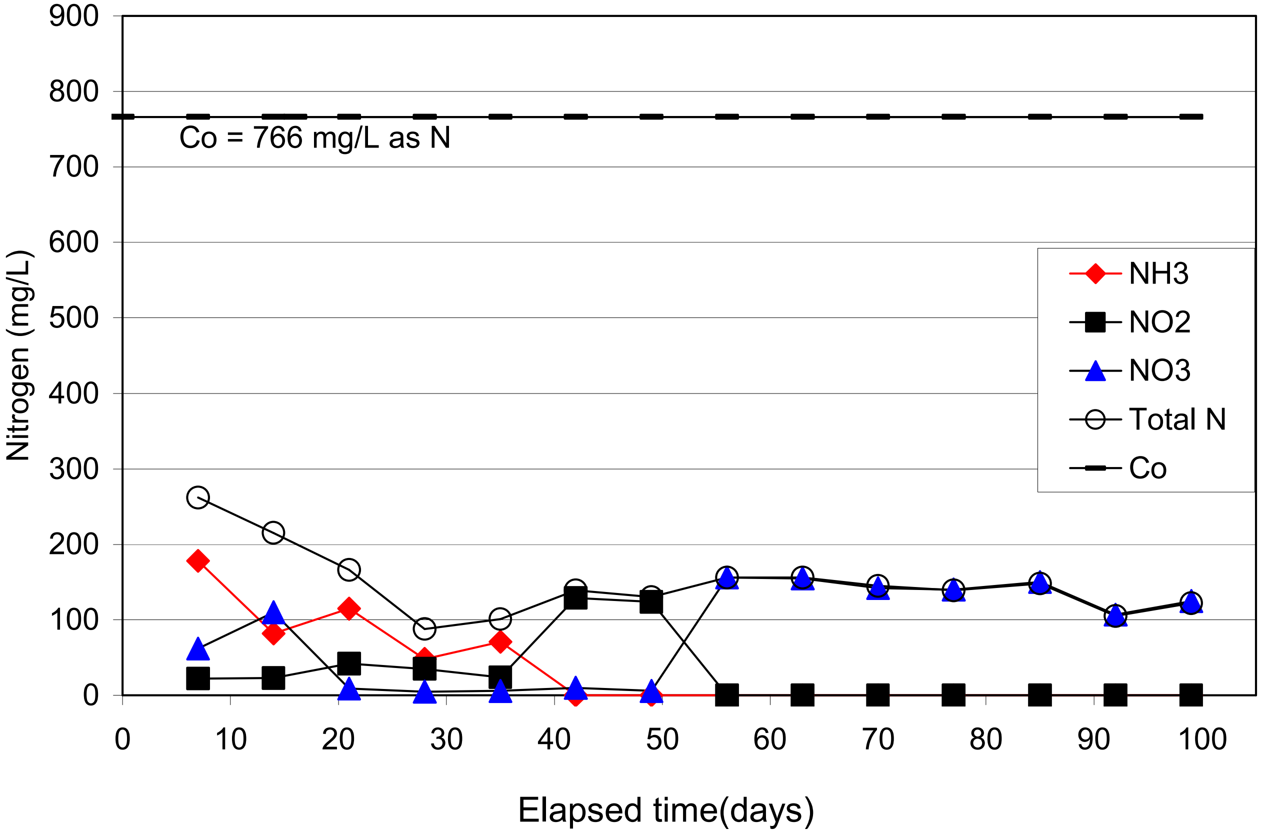

Figure 4.7 Nitrogen species in aerated effluent (column C2)

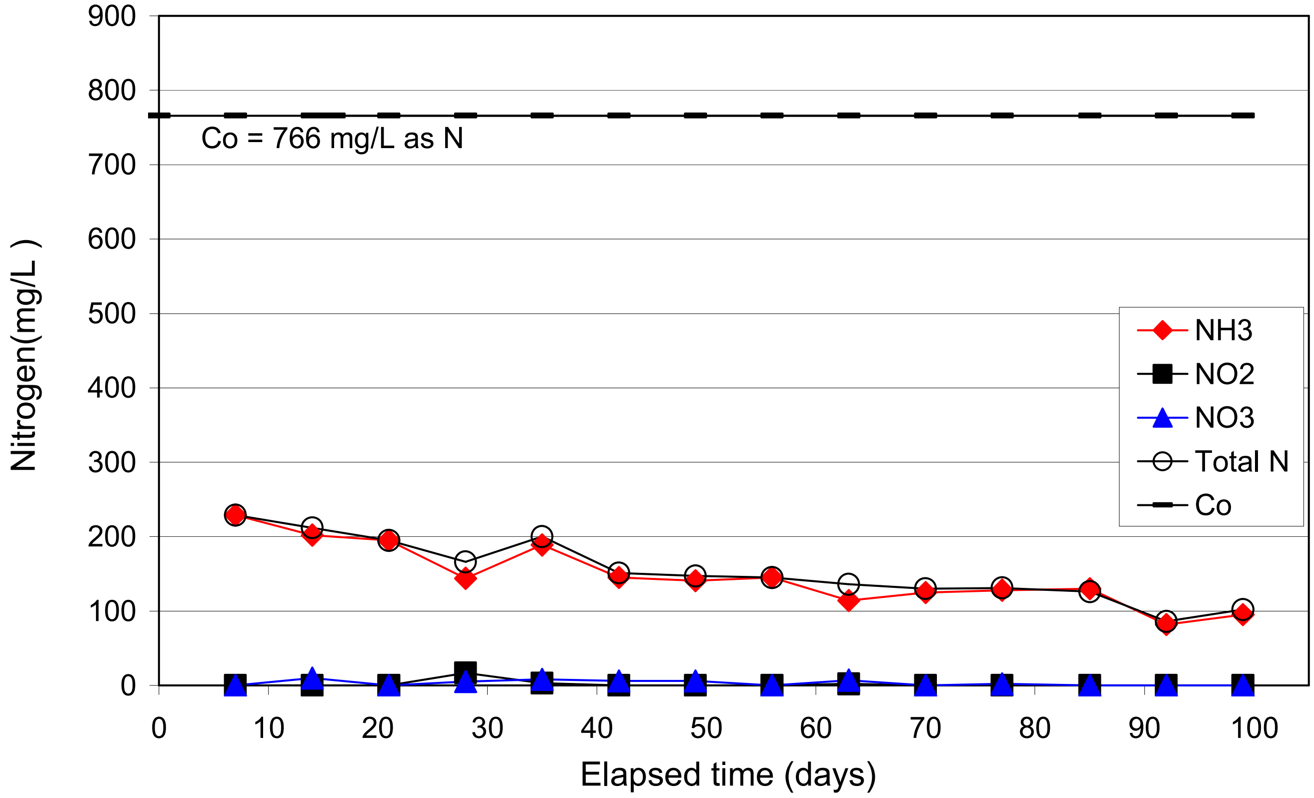

Figure 4.8 Nitrogen species in anaerobic column discharge (column C2)

4.2.5 Total Nitrogen and Total Nitrogen Removal

Figure 4.9 and 4.10 clearly show that nitrogen is removed from the system. The total nitrogen present in both the aerated effluent and the anaerobic column discharge are significantly reduced compare to the leachate input. Total nitrogen decreased from 766 mg/l to 130 mg/l as N (as an average), resulting in approximately 84% removal.

During the first 40 days of the experiment, there is some apparent loss of nitrogen between the column discharge and the nitrified effluent likely reflecting volatilization of ammonia from the aerated nitrification tank. This effect seems to have diminished after about 45 days as the main consequence of the improving conversion of ammonia in the reactor.

4.2.6 COD Removal

Chemical Oxygen Demand (COD) in this laboratory test is represented by the carboxylic acids present in the synthetic leachate (acetic, propionic and butyric). As shown in Figures 4.11 and 4.12, COD removal was stable and achieved approximately 97% removal with the exception of a brief upset during week 3. This clearly demonstrates and confirms that leachate collection drains can be utilized as an attached growth bioreactor for degradation of organic compounds in MSW leachate

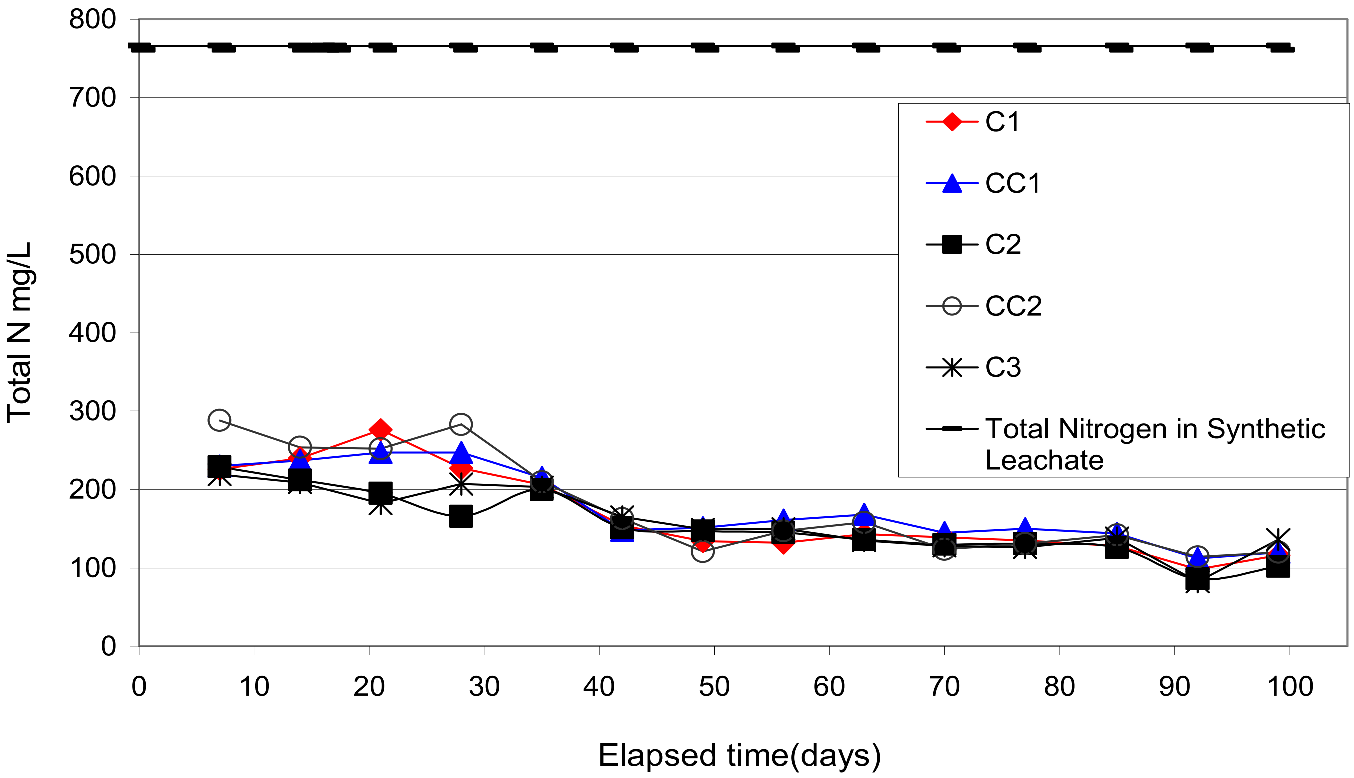

Figure 4.9 Total nitrogen in anaerobic column discharge

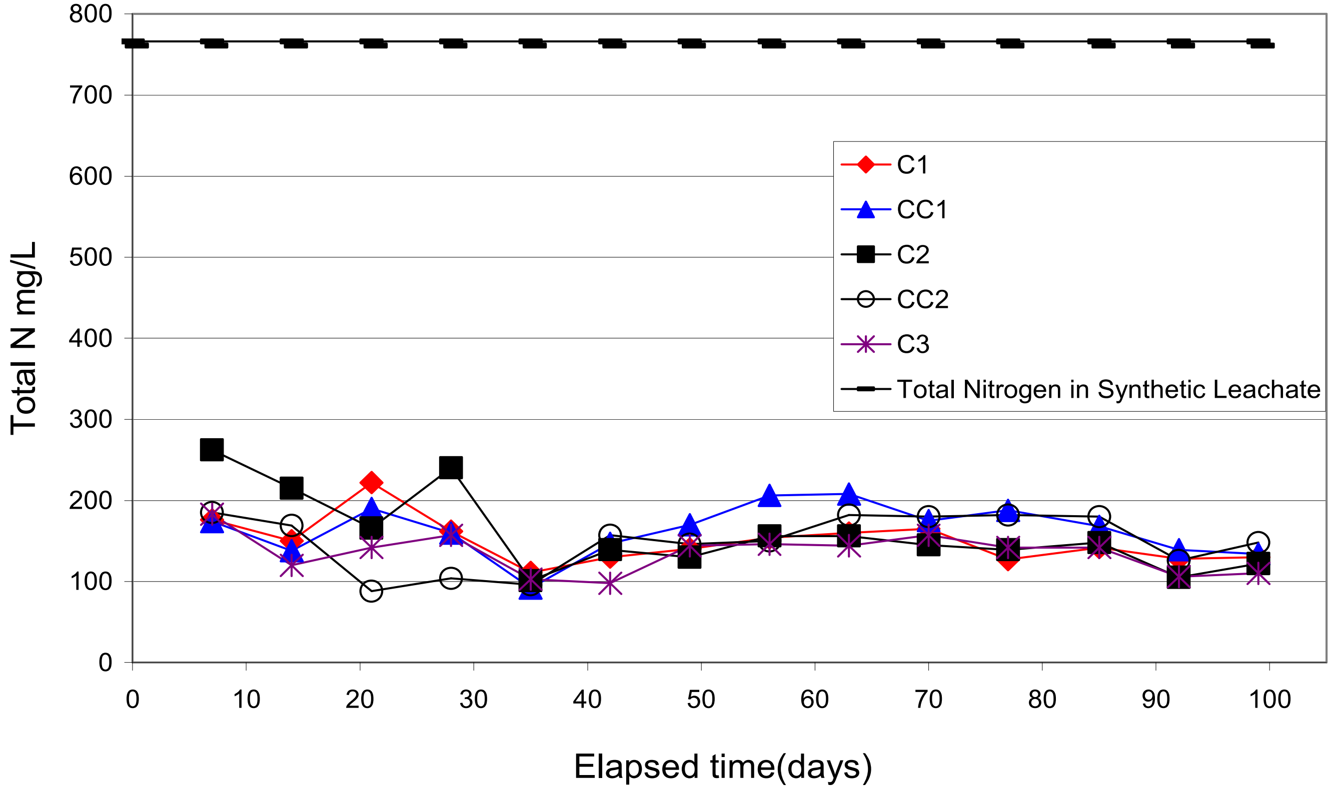

Figure 4.10 Total nitrogen in aerated effluent

Previous studies conducted with attached growth reactors simulating LCS showed a COD removal of approximately 14,000 mg/L as O_2 (from 16,000 mg/L to 2,000 mg/L as O_2) (Fleming et al., 1999; Fleming and Rowe 2004). In those studies the remaining COD was considered to be refractory materials that were difficult to biodegrade. Armstrong, (1998) and Rowe et al., (2002) also showed that one important factor in COD depletion is particle size of the porous medium. This finding is supported by the experiment. The geotextiles in the columns have a large a surface area and porosity and held a large bacterial population. Hence, the geotextile functions as an excellent fixed biofilm reactor that is largely responsible for the leachate treatment (see chapter 5.1).

The Hydraulic Retention Time of the aerated nitrification tank (HRT) also plays an important role in the removal of contaminants. The experiment started at a HRT of 2.7 days which produced a removal efficiency of approximately 90%. From day 14 on, the HRT was increased to 5 days. Figure 13 shows that the removal efficiency increased by 7% with a HRT of 5 days. Later on, around day 80, the HRT was cut back to 3.5 days for a week and an increase in COD and disruption of the aerated nitrification reactors were observed. Because of these results, the HRT was moved back to 5 days and the experiment returned to previous satisfactory performance.

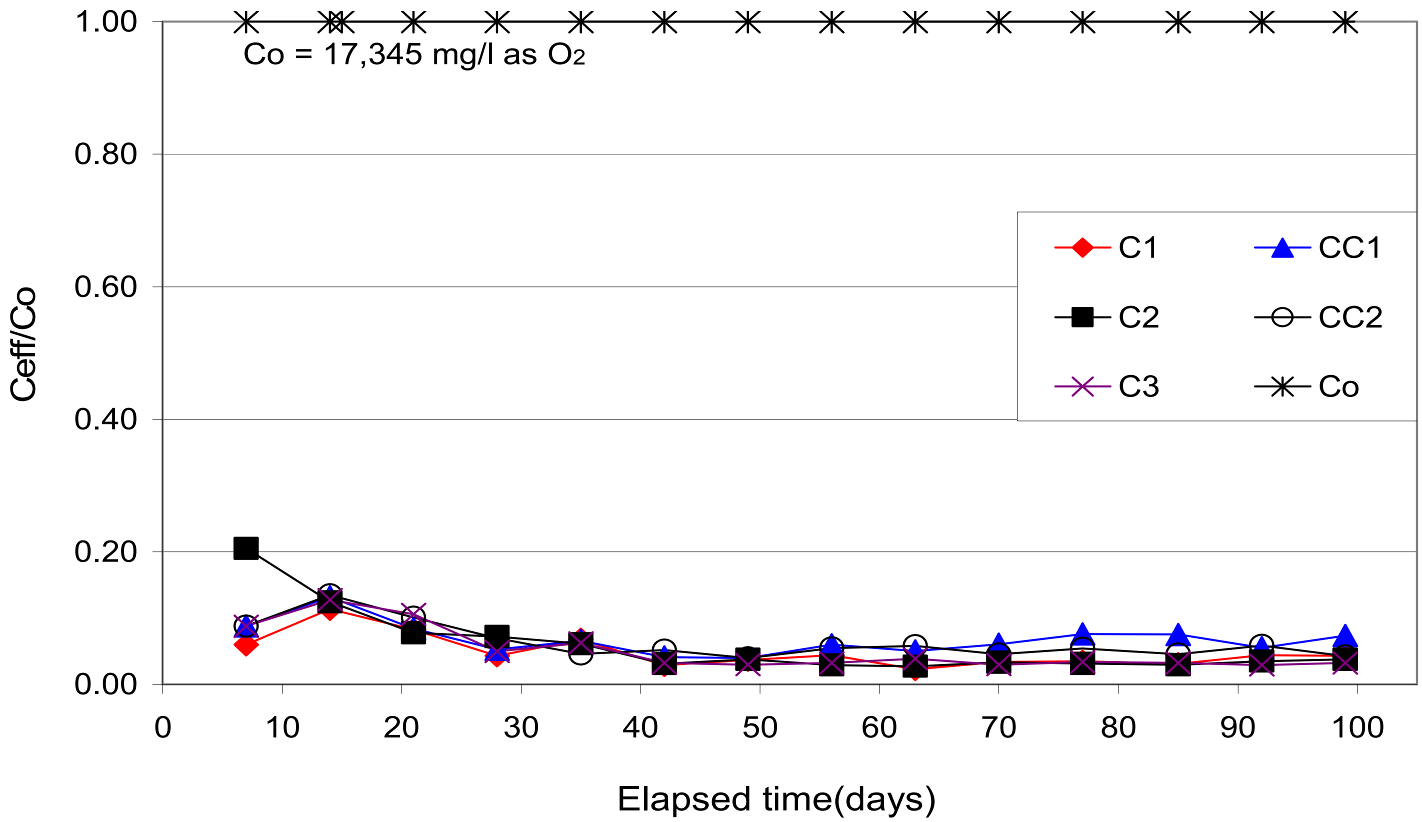

Figure 4.11

4.11 COD removal in anaerobic column discharge

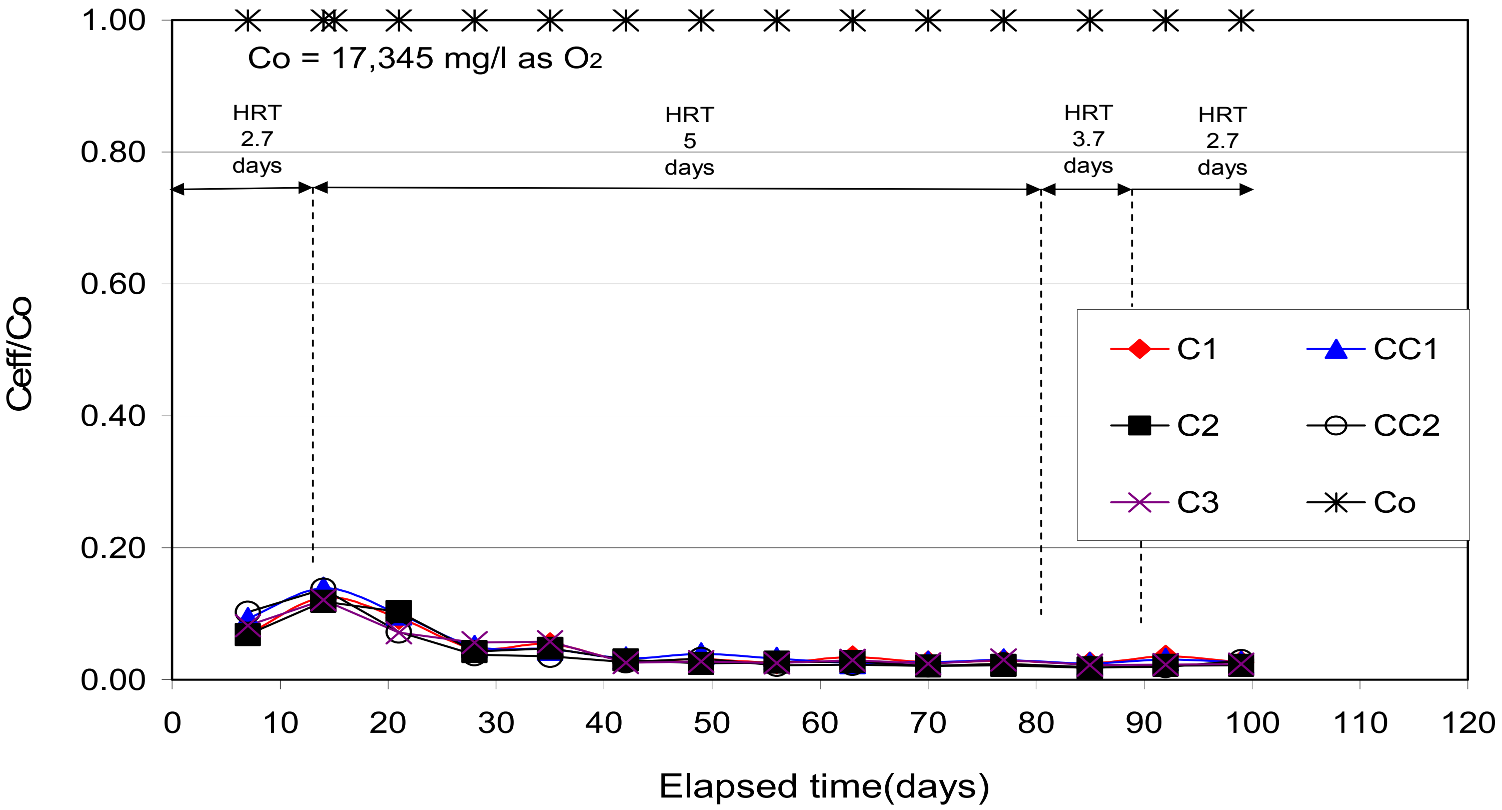

Figure 4.12 COD removal in aerated effluent (the HRT values refer to the aerated nitrification tank only)

The most likely reason for this behaviour lies in the nature of the experiment. Higher recirculation flow rates will transport more oxygen from the aerated reactor into the anaerobic column reactor, thus inhibiting the mainly methanogenic bacteria that are strictly anaerobes.

Overall, the synthetic leachate COD was reduced from approximately 17,000 mg/L to 600 mg/L. The following may have contributed to increased COD removal in this system compared to Fleming et al.(1999) and Fleming and Rowe (2004):

1. The presence of nitrifying and de-nitrifying bacteria in addition to the heterotrophic population of bacteria that utilize organic matter as substrate, and

2. The coupled nitrification reactor may result in additional degradation of organic compounds. Close examination of the data presented in Figure 4.11 and 4.12 suggests that there is not significant COD removal in the nitrification reactor itself, but that the nitrified effluent may undergo additional conversion when recycled through the anaerobic column. It is not known whether this effect reflects lessening the degree of inhibition by ammonia as the ammonia concentration is decreased within the column or some other synergistic effect.

4.2.7 Gas Production

Gas was collected in Tedlar bags at the top of the columns. Figure 4.13 shows a steady gas production at 180 mL/day for three of the column reactors

The biogas produced in the columns contains a mixture of methane, nitrogen and CO_2. Samples taken for analysis have confirmed approximately 60% methane, 1.6% CO_2 and 37% nitrogen. This gas composition is substantially different than that typically found in landfill gas. If it were possible to sustain this gas composition in the field, there could be beneficial consequences such as less acidic condensate.

Table 4.1 Gas composition

Gas	CH_4	N_2	CO_2	O_2
% by volume	59-61	36-39	1.0-1.6	< 1.0

Although several studies (Price et al. 2003, Onay and Pohland, 1998 and Lin and Chen, 1995) found that methane yields decrease when denitrification takes place. It can be argued that those findings do not necessarily fully apply for this experiment and even for a real landfill. The reason for decreased methane production lies in the fact that bacteria obtain much more energy converting nitrite to nitrogen gas than converting carboxylic acids to methane (ΔG° = -1120 KJ and ΔG° = -31 KJ respectively). Price et al (2003) used up to 5 moles of nitrate (310,000 mg/L as NO_3^-) and a BOD of 10,000 mg/L as O_2.

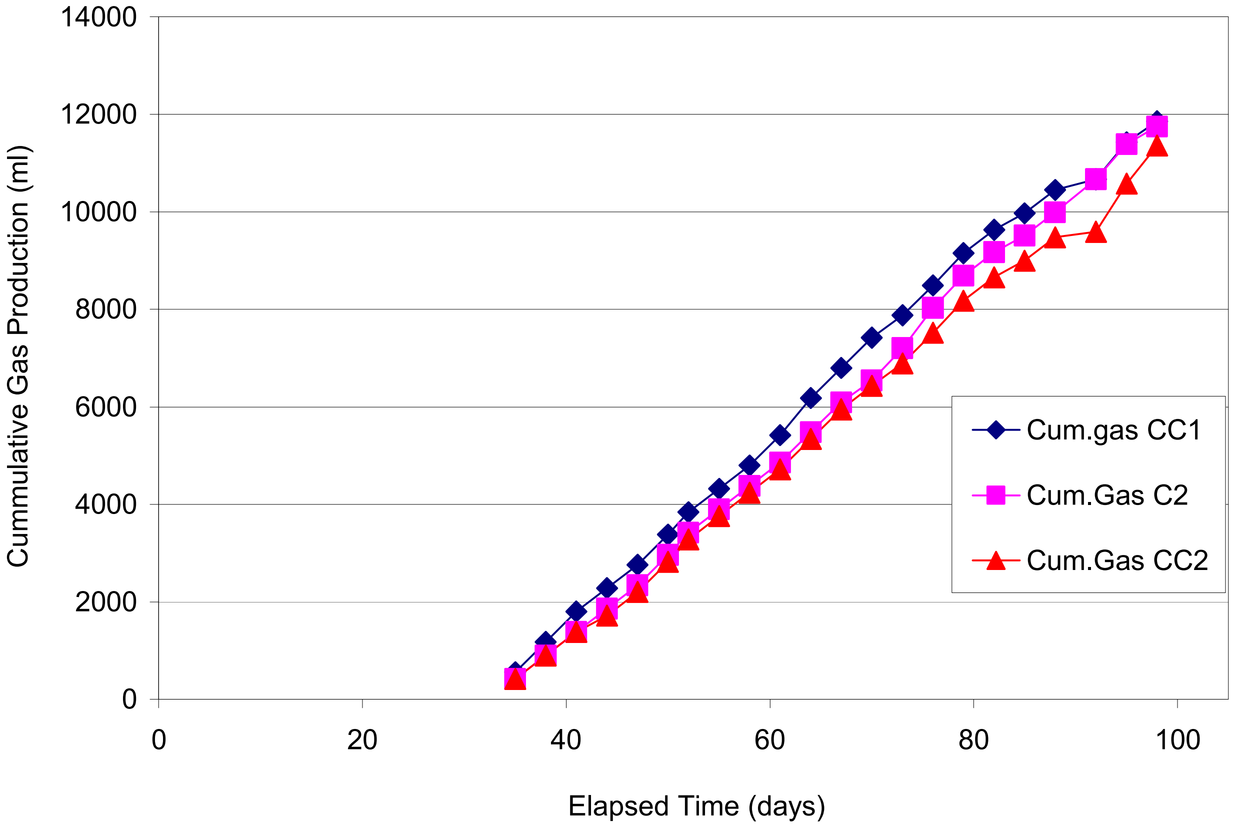

Figure 4.13 Anaerobic column reactors gas production

At such a large nitrate/BOD ratio, denitrifying bacteria will out compete methanogenic bacteria. However, in a relatively new landfill (as this experiment simulates) that is not the case.

The supply of organic carbon represented by a COD of 17,000 mg/L as O_2 proves to be enough to treat nitrogen values around 760 mg/L as N with out either affecting methane production or creating dissimilatory nitrate reduction to ammonia (DNRA) which may occur at low N/COD ratio and reduces nitrate back to ammonia (Tiedje, 1988).

4.3 Mass Balance Analysis

Mass balance calculations were conducted in an attempt to account for all nitrogen and carbon in the treatment system. The mass balance calculations provide quantification of the conversion processes involving nitrogen and carbon compounds and confirm the reported treatment efficiencies.

4.3.1 Nitrogen Mass Balance

The nitrogen mass balance calculations are summarized in Table 2. Nitrogen enters the system in the synthetic leachate (source) mainly as ammonium bicarbonate and urea. The nitrogen mass input is 77 mg/day based upon an average measured leachate concentration of 766 mg/L as N and an input flow of 0.1 L/day.

Table 4.2 Nitrogen mass balance

Component	Phase	Nitrogen Form	Mass/day (mg/day)
Source			
Synthetic leachate inflow	aqueous	ammonia and urea	77
		Σ Source =	77
Sinks			
Reactor effluent outflow	aqueous	nitrate	14
Column off gassing	gaseous	nitrogen gas	74
		Σ Sinks =	88
(Σ Sinks - Σ Source)/ Σ Source = 14%			

Ammonia nitrogen is converted to nitrate in the nitrification reactor, and recycled nitrate is converted to nitrogen gas by denitrification in the column. Therefore, nitrogen leaves the system (sinks) as nitrogen gas from the column, and as nitrate in the effluent flow from the nitrification reactor. The measured nitrogen content of the off-gas collected from the column was 37% by volume. The nitrogen gas nitrogen mass outflow (74 mg/day) was estimated using nitrogen mass density at 35°C, an off-gas pressure of one atmosphere, and the measured total off-gas flow (180 mL/day). The nitrate nitrogen mass outflow (14 mg/day) was estimated using the measured nitrate concentration in the nitrification reactor (140 mg/L) and the effluent flow (0.1 L/day). Therefore, the total estimated nitrogen outflow (sink) is 88 mg/day.

The nitrogen mass balance source and sinks estimates agree within approximately 14%. This small discrepancy is likely due to sampling and analysis errors. Despite this small error, the calculations clearly illustrate the efficient conversion and removal of ammonia nitrogen from the leachate entering the treatment system.

4.3.2 Carbon Mass Balance

Carbon enters the system mainly as volatile fatty acids (VFA) in the synthetic leachate (source). The VFA undergo anaerobic biodegradation in the column and are converted to methane (CH_4), carbon dioxide (CO_2) and other minor by-products. The CH_4 and CO_2 are both off-gassed from the column. The conversion of VFA results in the removal of aqueous phase COD and a significant increase in pH due to a dramatic increase in carbonate alkalinity resulting from CO_2 production and dissolution. The increased carbonate alkalinity and pH cause precipitation of carbonate solids (primarily $CaCO_3$) within in the column. The leachate leaving the column and entering nitrification reactor is supersaturated with CO_2 for open to atmosphere conditions. Therefore, off gassing of CO_2 from the nitrification reactor is an additional loss of carbon from the system. Each of the carbon sinks is estimated in the carbon mass calculations.

The carbon mass balance calculations are summarized in Table 3. The carbon input (source) is 600 mg/day based upon an average measured leachate concentration of

6000 mg/L as C and an input flow of 0.1 L/day. The input carbon concentration was measured using a total carbon analyser.

Table 4.3 Carbon mass balance

Component	**Phase**	**Carbon Form**	**Mass/day (mg/day)**
Source			
Synthetic leachate inflow	Aqueous	Carboxilic acids, Carbonates	600
		Σ Source =	600
Sinks			
Reactor effluent outflow	Aqueous	Organic carbon	6
	Aqueous	Inorganic carbon	116
Deposition of Carbonate precipitates	Solid	Calcite precipitate	33
Column off gassing			
	Gaseous	Carbon dioxide	1
	Gaseous	Methane	51
Nitrification Reactor off gassing	Gaseous	Carbon dioxide	102
		Σ Sinks =	309
(Σ Sinks - Σ Source)/ Σ Source = 49%			

Carbon mass flow to each of the carbon sinks was estimated as follows:

- The carbon outflow in the reactor effluent (122 mg/day) was estimated using the measured inorganic and organic carbon content of the effluent (1,160 and 60 mg/L respectively) and the effluent flow (0.1 L/day).

- The measured CH_4 and CO_2 content of the off-gas collected from the column was 60% and 1.6% respectively by volume. The carbon mass outflow rates for CH_4 and CO_2 (51 and 1 mg/day respectively) were estimated using CH_4 and CO_2 mass density at 35°C, an off-gas pressure of one atmosphere, and the measured total off-gas flow (180 mL/day).

- The loss of carbon to deposition of carbonate precipitates in the column was estimated using the drop in calcium concentration between the leachate and effluent outflow assuming the vast majority of the carbonate solids are $CaCO_3$, and that the dominate carbonate species in the effluent is bicarbonate. Using these simplifying assumptions, the measured drop in calcium concentration from 2900 to 150 mg/L through the system, and an effluent flow rate of 0.1 L/day produced an estimate of 33 mg/day carbon loss to precipitates in the column.

- The loss of carbon due to off gassing of CO_2 from the nitrification reactor can be estimated using the change in total alkalinity between the inlet and outlet of the reactor (a drop from 5,700 mg/L to 5,000 mg/L as $CaCO_3$). Assuming the dominate alkalinity species is bicarbonate (based upon pH) and the alkalinity lost is converted to CO_2, then the carbon loss from the system can be calculated using the reactor flow of 0.6 L/day (sum of the leachate and re-circulation flow). The estimated carbon loss using this approach is 102 mg/day.

The total carbon loss and deposition represented by the sinks described above is 309 mg/day.

The carbon mass balance calculations indicate a large proportion of the carbon entering the system (~ 50%) is unaccounted for. However, at least two additional carbon sinks exist that have not been quantified. First, a significant amount of CO_2 may be off gassed from the column discharge sample collection apparatus. This CO_2 loss is not accounted for in the calculation approach described above. Second, there is likely a build up of biomass within the column as the experiment progresses. Whereas the active biomass may approach an equilibrium condition, inactive biomass may continue to accumulate as biofilm thickens and cells die-off due to restricted access to substrate.

Despite the lack of closure of the carbon mass balance the calculations provide very encouraging results with respect to organic carbon discharged from the system in the aqueous phase. Of the 600 mg/day organic carbon entering the system in the leachate, only 60 mg/day organic carbon leaves the system in the nitrification reactor effluent. These calculations confirm the high level of treatment efficiency provided by the system that was indicated by the COD measurements presented earlier.

4.4 Conclusions

The experimental set-up simulated a leachate collection system (LCS) with recirculation coupled with a nitrification reactor. The results of the experiment showed that in-situ passive treatment of the leachate COD and ammonia is feasible. Furthermore, the tests shown that the LCS can be used as an efficient fixed biofilm reactor. The large surface area of the geotextile allows microbes to attach and grow. The biofilm that develops under anaerobic conditions, biodegrades carboxylic acids, producing mainly carbonates and methane gas as by products.

The generation of carbonates precipitate can be a source of incrustation and obstruction of the drain and/or leachate collection pipes. Saturation of the leachate with CO_3^{2-} causes increased pH, and precipitates Ca^{2+} as calcium carbonate that has the potential to clog the system. Clogging then, may result due to CO_2 released as a product of organic matter degradation and carbonate system equilibrium with the surrounding environment.

Nitrification and denitrification are processes that can be achieved reliably in the coupled LCS and aerated reactor system. The treatment of 776 mg/L of nitrogen as N (in the form of urea and ammonium bicarbonate) was carried out with an 84% average efficiency. Hydraulic retention time (HRT) plays a fundamental role in the system. The optimum HRT for the experiment was 5 days. Changes in this parameter led to inefficiencies in both ammonia and COD depletion. The pH values measured in the experiment were higher than values for optimal microbial growth (7 to 8 for denitrification and 7.5 to 8.6 for nitrification (Shiskowsky and Mavinik, 1995)). This shows the denitrifying and nitrifying bacteria are resilient.

COD removal (compared to the leachate input concentration) reached levels of up to 95% in the column reactor effluent and 97% in the nitrification reactor. Previous studies by Fleming and Rowe (2004) showed the difficulty of treating organic matter using suspended and porous medium beyond the 2000 mg/L concentration. This was thought to be due to the presence of recalcitrant organic compounds that are hard for bacteria to utilize. In this study COD levels were reduced to below 600 mg/L. Synergistic effects within the column may explain the lower effluent concentrations observed. The most likely effect is denitrifying bacteria and methanogenic bacteria competing for organic carbon as an energy source. As a result, a higher proportion of the available organic carbon is utilized, even the recalcitrant compounds.

The biogas produced in the columns was comprised of methane (60%), nitrogen (37%) and CO_2 (1.6%). The nitrogen and carbon dioxide values are substantially different than those typically produced in a landfill. The CO_2 content of the biogas in particular, is much lower than typically produced in a landfill. The consequences of a reduced CO_2 level are beneficial. A low carbon dioxide value decreases the amount of carbonates in the leachate, reducing the potential risk of clogging. It also means a less acidic (corrosive) environment for a landfill gas collection network, thus extending its serviceable life.

The overall results obtained from the experiment show that in-situ treatment at landfills site is viable. The required HRT for the aerated nitrification reactor is small enough to make the system physically and economically feasible. Furthermore, despite the fact that methanogenic and denitrifying bacteria compete for carbon availability, it is reasonable to think that the amount of refuse disposed of in a landfill will be able to provide enough organic carbon to feed both microbial populations allowing COD to be depleted and nitrate to be off-gassed. As a result, methane will be produced in amounts that would be attractive for use as a source of energy.

Chapter 5 Hydraulic Performance of the Modified Drainage LCS

5.1 Introduction

In order for a landfill LCS to work as a fixed biofilm reactor, it has to develop a microbial community within its porous media. Since geotextiles and drains are integral parts of a landfill LCS, their appropriateness for biofilm growth has to be assessed. Rollin and Lombard (1988), Koerner and Koerner (1992), and Mlynarek and Vermeersch (1999) have demonstrated the suitability of geotextiles for supporting media growth of bacterial biofilm. This is mainly due to their large surface area and porosity, which facilitates microbial attachment and growth. Correspondingly, Armstrong (1998) permeated leachate through a porous media packed in several columns resembling landfill LCS drains, and found a direct relationship between bacterial growth (leachate treatment) and the surface area of the drain material.

This chapter analyzes the impact on the hydraulic properties of the laboratory columns after being operated as fixed biofilm reactors with all the physical, chemical and biological consequences that such a mode of operation entails. It also projects the hydraulic performance findings of the laboratory experiment to a full scale landfill, represented by a typical North American landfill.

Lastly, the performance of the two columns packed with recycled crushed concrete is analyzed, and compare to the three columns packed with natural gravel. The purpose of this analysis is to investigate the potential (and benefits) of using a recycled material, in this case concrete, as a drainage material in a LCS.

5.2 Effects of leachate treatment in Geotextiles

Table 5.1 shows the coefficient on hydraulic conductivity of the different sections of the columns at the beginning and at the end of the experiment. The hydraulic conductivity results were obtained using a constant head method with flow rates ranging from 130 to 310 mL/day. For the uppermost geotextile a falling head method was used.

Except for the uppermost geotextiles (section F1), the sections showed no meaningful change in hydraulic conductivity before and after the experiment. As an average, the decrease in the hydraulic conductivity of the uppermost geotextiles was approximately 1.2 orders of magnitude. It went from an average of 0.05 cm/s to an average of 0.0028 cm/s

Figure 5.1 shows the piezometric profiles of the columns at the beginning and at the end of the experiment. As can be seen, no significant difference was found before and after the experiment, except for the uppermost geotextile, which showed a considerable increase in pressure head across the uppermost geotextile filter.

Table 5.1 Hydraulic conductivity of the different sections of the columns C1, C2, C3, CC1 and CC2

	Column C1		**Column C2**		**Column C3**		**Column CC1**		**Column CC2**	
Section	**K Ini.** **(cm/sec)**	**K Fin.** **(cm/sec)**	**K Ini.** **(cm/sec)**	**K Fin.** **(cm/sec)**	**K Ini.** **(cm/sec)**	**K Fin.** **(cm/sec)**	**K Fin.** **(cm/sec)**	**K Fin.** **(cm/sec)**	**K Ini.** **(cm/sec)**	**K Fin.** **(cm/sec)**
F1	7.6E-2	2.5E-3	1.2E-1	2.1E-3	8E-2	5.7E-3	4E-2	2.6E-3	5.6E-2	1.7E-3
S2	179	180	123	123	241	193	65	62	112	112
F2	5.6E-2	5.3E-2	1.4E-1	1.3E-1	1.5E-1	1.4E-1	6E-2	6E-2	1.4E-1	1.5E-1
S3	54	56	38	38	20	20	64	54	75	75
S4	45	45	17	17	21	21	45	45	28	28
F3	7.5E-2	7E-2	0.8E-2	8.4E-2	8.5E-2	8.6E-2	4E-2	4E-2	2.3E-1	2.7E-1
S5	270	270	241	121	242	194	161	161	112	112

The decrease in hydraulic conductivity of the uppermost geotextiles was a result of several factors including; particles washed out of the overlying wet refuse, bacterial slime formation, and calcite precipitation. All the uppermost geotextiles presented a “muddy” appearance like the one (column C2) shown in Figure 5.2.

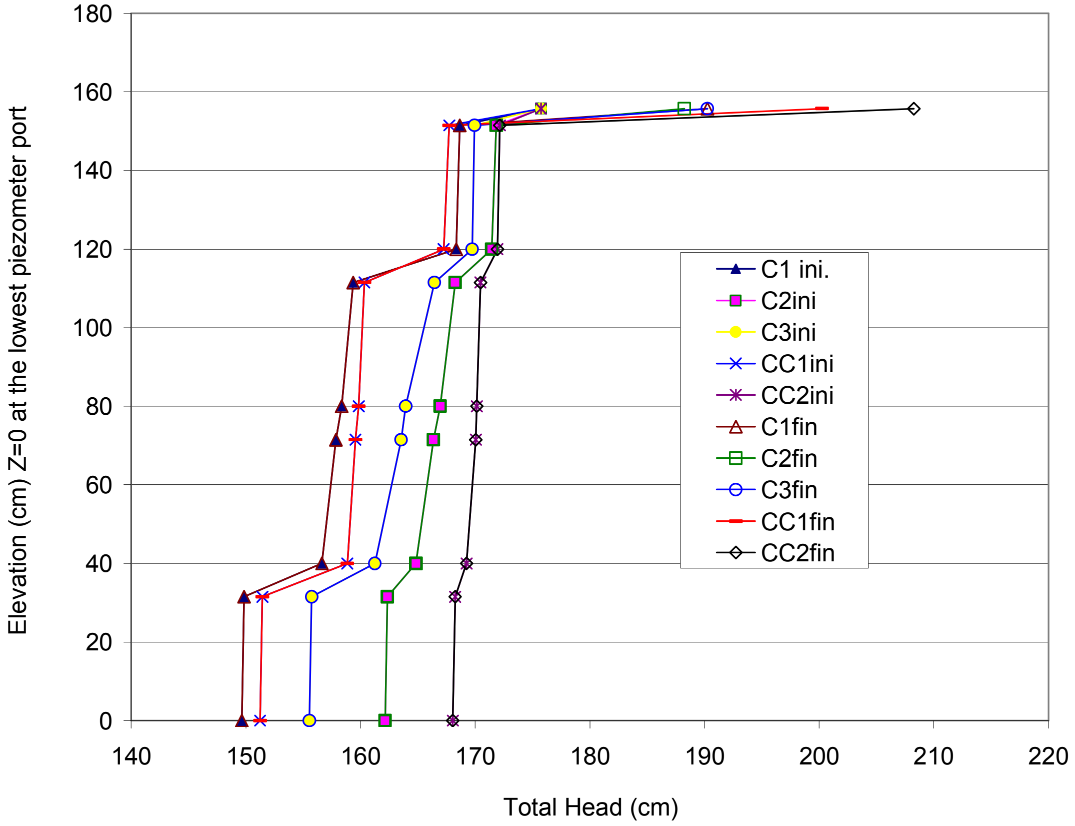

Figure 5.1 Initial and final piezometric line of the columns

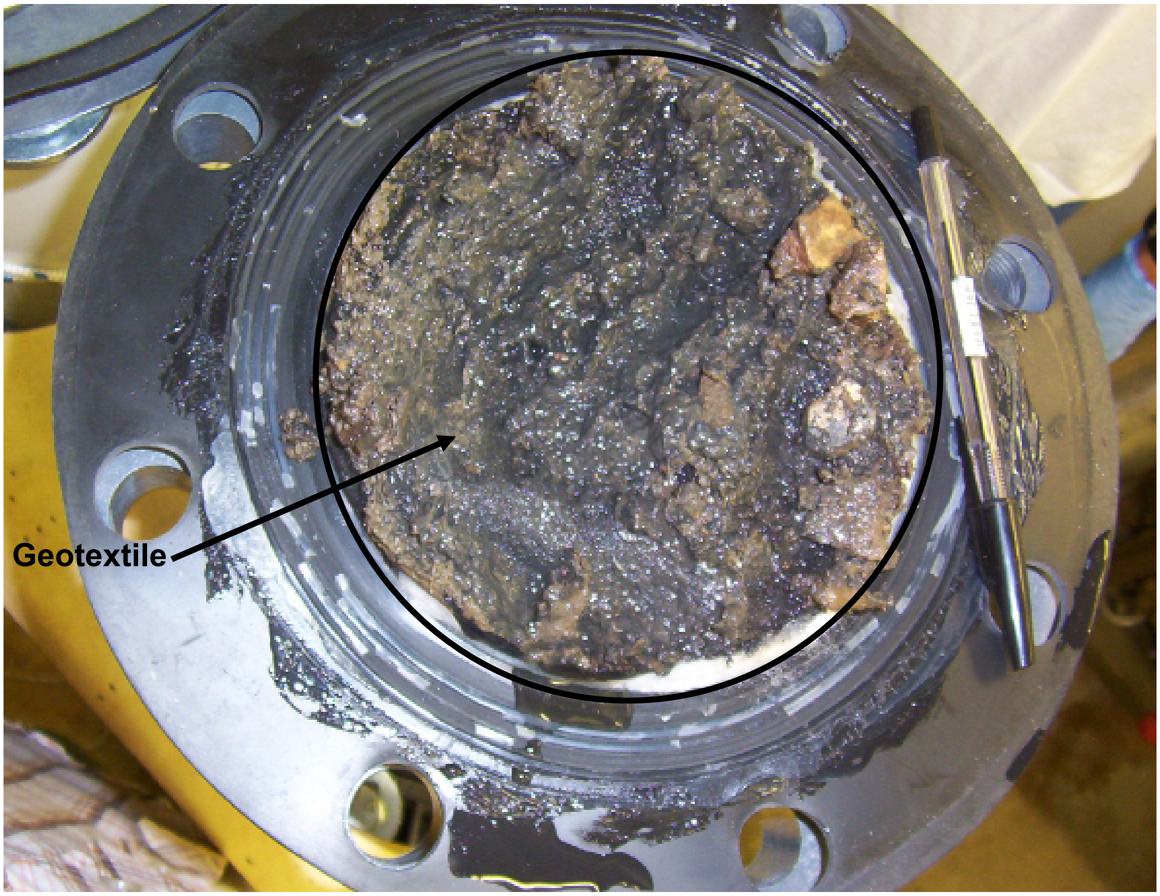

Figure 5.2 Uppermost geotextile in column C2 after 100 days of operation

Figures 5.3 and 5.4 show vertical thin section photomicrographs of the uppermost needle punched non-woven geotextile before and after the experiment. Mineral deposits, partially responsible for the lower hydraulic conductivity, can be seen as white blotches in Figure 5.4.

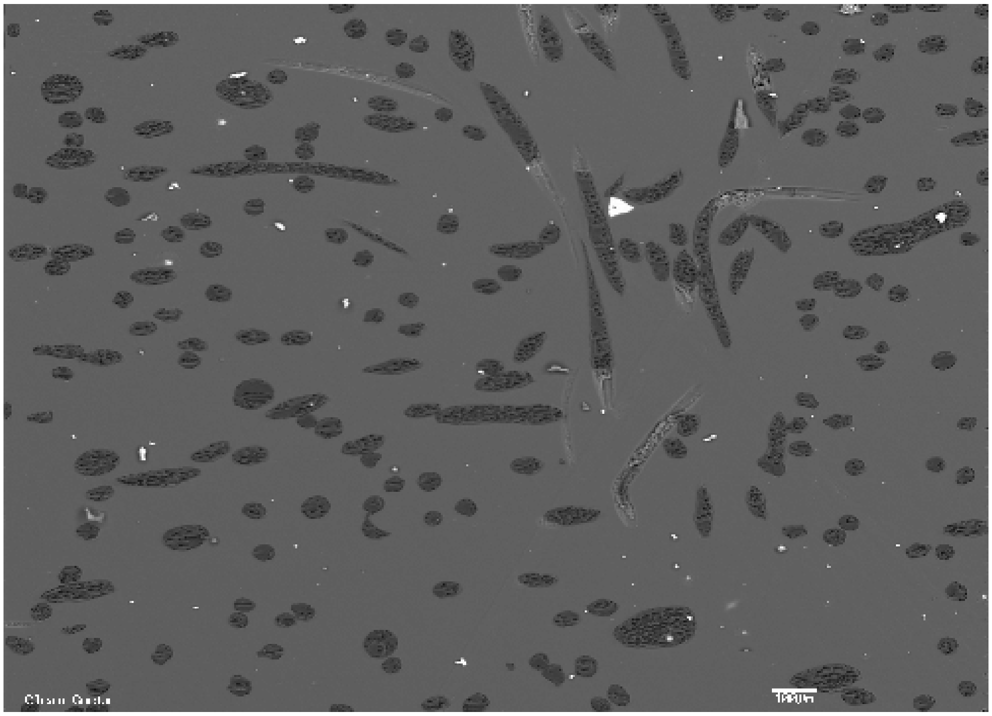

Figure 5.3 Geotextile before the experiment

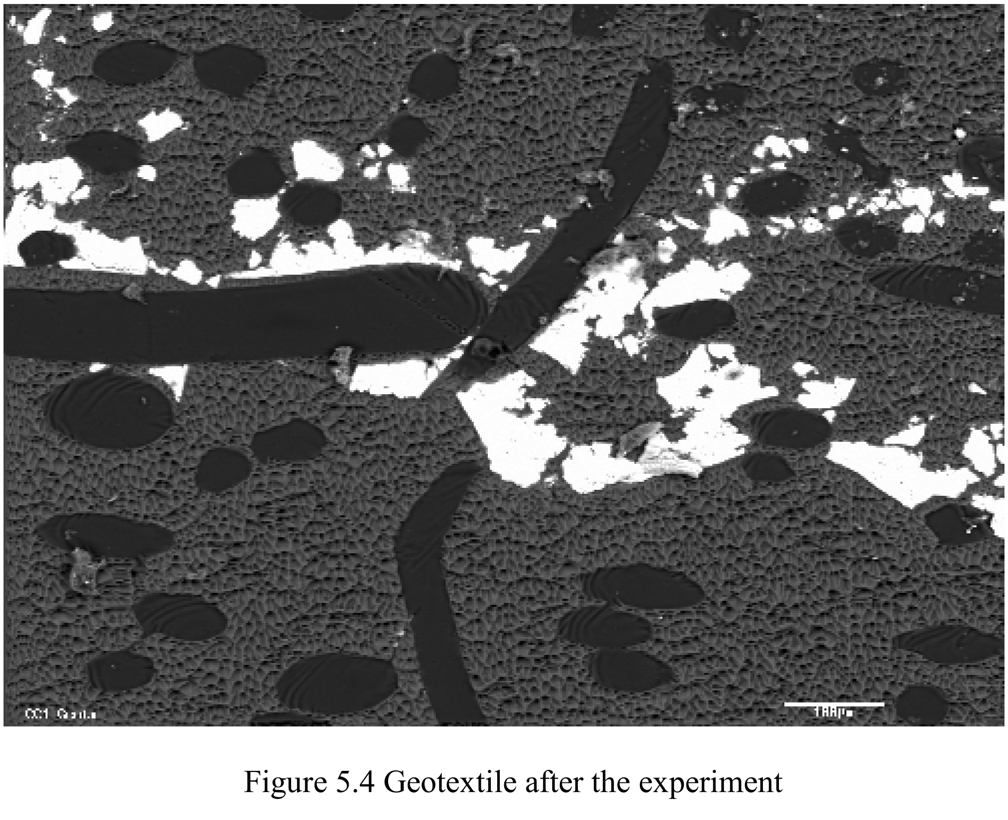

Figure 5.4 Geotextile after the experiment

5.2.2 Clog Material Analysis of Geotextiles

A gravimetric analysis was carried out in order to determine the composition of the material within the uppermost geotextiles (i.e. the relative proportion of volatile solids, calcite, and other minerals or "refractory material"). Two samples with an area of 32 cm^2 were cut from the uppermost geotextile of each column and were weighed in an oven-dried condition; following ignition at 550 ˚C and finally after combustion at 900˚C. The average mass of a clean geotextile was determined by weighting 30 pieces of clean geotextile of equal area (32 cm^2). The average mean mass and standard deviation were found to be 1.19 g and 0.14 respectively. The geotextiles samples were subsequently weighed after ignition in a laboratory furnace at 550˚C and it was found that the polypropylene material was almost completely burned off; LOI (loss on ignition) was 99.6 % at 550˚C. Therefore, if the area of the pieces of the uppermost geotextiles is known, its mass can be reasonably estimated within a range of $(^{+}_{-})$ 0.69 units. This estimate of mass is required when determining the mass of organic and inorganic material attached to or retained within the fabric. The results of this analysis are listed in Table 5.2.

Fleming et al. (1999) performed elemental analysis of slime and solid samples removed from Canadian landfill drains. Calcite was found to comprise between 40 and 50 % of the dry weight of the clog material that was deposited within the drains.

Table 5.2 Classification of clog material within the uppermost geotextile

	Measured			Calculated				Dry Wt basis		
Column	**M 105˚C**	**M 550˚C**	**M 900˚C**	**Mvs(g)**	**Mcalc(g)**	**Mrefr(g)**	**total mass(g)**	**% VS**	**% Calc**	**% Refr**
C1	1.7558	0.55	0.4209	0.01	0.29	0.26	0.56	2.10	52.23	45.67
C1	1.8674	0.565	0.3772	0.11	0.43	0.14	0.67	16.10	63.38	20.52
C2	1.9823	0.8081	0.6768	0.00	0.30	0.51	0.81	0.00	36.93	63.07
C2	2.1108	0.8179	0.7142	0.10	0.24	0.58	0.92	10.79	25.71	63.51
C3	3.1521	1.8019	1.5781	0.16	0.51	1.29	1.96	7.98	25.98	66.05
C3	2.8504	1.6117	1.3169	0.04	0.67	0.94	1.66	2.70	40.45	56.85
CC1	2.9839	0.8368	0.7042	0.95	0.30	0.54	1.79	53.25	16.84	29.91
CC1	2.5979	0.7707	0.6566	0.63	0.26	0.51	1.40	45.10	18.47	36.43
CC2	3.821	2.1826	1.7499	0.44	0.98	1.20	2.63	16.92	37.43	45.65
CC2	5.2828	3.274	2.6318	0.81	1.46	1.81	4.09	19.93	35.70	44.38

Mean	0.36	0.57	0.84	1.77	19.20	33.43	47.37
STDEV	0.36	0.40	0.52	1.08			

Area of Sample = 32 cm2
Area of Geotextile = 201 cm2
Mvs = Mass of volatile solids
Mcalc = Mass of calcite
Mrefr = Mass of refractory material

For the five laboratory columns , the average mass of calcite precipitate within the uppermost geotextiles during the life of the experiment can be calculated using Table 5.2, the area of the geotextile samples (32 cm^2), and the area of the whole geotextile (201 cm^2). This average mass of calcite was 3.4g, which in turn corresponds to 33% of the total dry weight of the material deposited within the geotextile. Similarly, the mass and percentage of dry weight of volatile solids, and “refractory material’, can be calculated. They correspond to 2.1g or 19% and 4.8g or 47% respectively.

Fleming et al.(1999) also found that the amount of organic matter as a percentage of clog weight varies from 2 up to 8.5%.Thus between 42 and 58% by weight would be composed of inorganic material inherent to the refuse. Table 5.3 summarizes the laboratory findings and the ones found by Fleming et al (1999).

It is important to note that in this study, the mineral deposits that were analyzed were taken from the geotextile filter overlying and protecting the drainage media. The higher proportion within the geotextile of non-calcite “refractory” mineral deposit tends to support conclusions (i.e. Fleming and Rowe, 2004) that a geotextile filter overlying the drainage media is effective at protecting the underlying drain from clogging.

Table 5.3 Clogging composition in the laboratory experiment vs. clogging composition in the findings of Fleming et al. (1999)

	Volatile Solids	**Calcite ($CaCO_3$)**	**Refractory material**
% by dry weight in lab experiment	19	33	47
% by dry weight in Fleming et al.	3.8	58	38.2

Although clog composition is a time varying property (VanGulck and Rowe, 2004), some comparisons can be drawn form the results listed in Table 5.3.

Firstly, the percentage by weight of volatile solids in the uppermost geotextiles in this laboratory experiment was greater than that found in field excavations by Fleming et al. (1999). This likely reflects the higher porosity and surface area of the geotextiles compared to the landfills drains, from where the clog was extracted.

Studies done by Koerner and Koerner (1992) showed that the hydraulic conductivity of geotextiles (non-woven needle punched) will decrease by 65 % after being permeated with leachate with different strength from different landfills in the United States over 6 months. Koerner (1995) concluded that geotextiles used in landfills that produce leachate with higher values than 2,500 mg/L of BOD_5 should follow field and laboratory analysis prior placement. Reinhart and Chopra (2000) also recognized the likelihood of geotextile clogging due to bacterial growth.

Geotextiles present an interesting design conflict. Due to their intrinsic characteristics, they could be useful for leachate treatment working as fixed biofilm reactors (i.e. high surface area, high porosity), but at the same time bacterial growth and mineral precipitation within the fabric could hinder leachate flow.

Secondly, the calcite fraction reported by Fleming et al. (1999) is almost double that found in the geotextile in this laboratory experiment. It is hypothesized that the external aerobic nitrification reactor may play a role in this phenomenon. The pH of the aerated effluent, as it was seen in chapter 4, is higher than that of the anaerobic column discharge. This higher pH would lead to a greater calcite precipitation since the solubility of calcite is pH dependant. Therefore, calcite may precipitate in a substantial amount in the aerated reactor.

Thirdly, what is called refractory material (inorganic material other than calcite) comprises a larger percentage of the total deposited material in the laboratory experiment than in the exhumations done by Fleming et al. (1999). Reinhart and Chopra, 2000, concluded that due to fines and particles coming from the waste, a layer of sand should be placed on top of the geotextile (acting as a filter) and geotextiles should be used to separate that filter media from the drainage media. However, Fleming and Rowe (2004) see the amount of non-calcite mineral deposits as a proof of geotextiles effectiveness at protecting the underlying drains from clogging.

5.2.3 Calculating the Time for the Geotextiles to Clog

Rittman et al (1996), (2003), Fleming et al (1999); Fleming and Rowe (2004); Rowe et al., (2002) and VanGulck et al. (2003) have shown a direct relationship between organic matter stabilization, measured as chemical oxygen demand depletion, and the amount of calcite precipitate. This relationship, termed the "Yield Coefficient"(Yc), was found to range from 0.17 to 0.2 mg of calcite precipitate per mg of COD depleted.

During the present laboratory experiment, the yield coefficient was found to be 0.16 mg of calcite precipitate per mg of COD depleted. The calcite concentration decreased from for 2,900 mg/L to 150 mg/L as $CaCO_3$ (Ruiz et al. 2004). For a flow rate of 0.1 L/day the correspondingly rate of calcite precipitation may be taken to be 0.275 g/day per column.

In order to determine how long it will take for a geotextile to clog under the rates of calcite precipitation experienced during the laboratory testing, the following calculations were performed.

The density of calcite at 25 ˚C is 2.7 g/cm^3. The experiment showed a calcite production per column of 0.275 g/day. Thus, the total volume of calcite deposited in each column per unit area per day (V_c) can be estimated in the next fashion:

V_c (Volume of calcite deposited in each column / day*cm^2) = (0.275 g/day)/ (2.7 g/cm^3*182.4 cm^2) = $\underline{5.6*10^{-4}\ \text{cm}^3/\ \text{cm}^2*\text{day}}$

The analysis of the clog material deposited within the uppermost geotextiles (presented in section 5.2.2) shows that the mass of calcite precipitated over 100 days was 3.3 g or 0.033 g/day. This allows the quantification of the total amount of calcite deposited within the uppermost geotextiles per unit area per day:

V_{cg} (Total calcite deposited in geotextiles/ day*cm^2) = (0.0331g/day) /(2.7g/ cm^3 *182.4 cm^2)= $\underline{6.7*10^{-5}\ \text{cm}^3/\ \text{cm}^2*\text{day}.}$ Thus, it may be seen that (6.7/54) = 12% of the calcite was precipitated in the uppermost geotextiles.

Further, the initial porosity and thickness of the geotextile are 90 % and 0.3 cm, respectively. Thus, the volume of the geotextile available to be filled per area

$V_{geotextile}$ (Volume of geotextile available to be filled/cm^2) = 0.3 cm*0.9 = $\underline{0.27\ \text{cm}^3/\text{cm}^2}$

With all this information, and based on the amount of calcite deposited in the geotextiles, the time for the void space in the uppermost geotextile to be completely filled by calcite would be:

$\mathbf{t_g}$ =(0.27 cm^3/ cm^2)/(6.7*10^{-5} cm^3/ cm^2*day) = 4,030 days = <u>11 years (approx.)</u>

This result shows that it would take up to 11 years for an underlying geotextile to clog according to the amount of calcite deposited within the uppermost geotextiles used in the laboratory experiment. The clogging time would be dramatically shortened if all the calcite produced by organic matter stabilization were to be trapped in the geotextile fabric.

It is important to notice that the previous “time to clog” calculation accounts for neither biological nor “refractory” clogging. The calculations were done in this way to reflect the fact that the biofilm should reach a near steady state, and as the overall pore volume decreases, there will be a time in the LCS when the leachate mounding above the geotextile will create a sufficiently high advective flow and leachate will be able to flow through this biological mass by shearing it. Similarly, for the refractory material, it is hypothesized that since such material largely consists of fines from overlying refuse etc, there will be an initial supply of refractory mineral particles and the supply of such material would decrease over time. Further, it is difficult (and beyond the scope of the experiment) to develop a model that accounts for the fines and the solid material coming out of the refuse.

Clearly, there is a connection between geotextile clogging and the amount of organic matter stabilization. However, other factors could influence the location of mineral precipitation. In the experiment for instance, the amount of calcite precipitate within the geotextiles was only on average 12 % of the total amount of calcite precipitate. The balance of the mineral was deposited in the refuse and/ or the gravel-sized porous media and/or in the aerated nitrification tank.

Calcite precipitation and its accumulation can occur at different places since they not only depend on the amount of calcium-carbonate available to precipitate (solubility), but also, rely strongly on the carbonate equilibrium with the surrounding environment and the pH conditions (Jefferis and Bath 1999; Fleming et al. 1999; Fleming and Rowe 2004; Rittman et al.1996; 2003). These factors may vary over time and not be the same all everywhere in the system. For instance, in chapter 4, it was shown that pH was substantially higher in the aerated effluent than that of the column discharge. This situation could lead to higher levels of calcite precipitation in the aerated nitrification reactor. Furthermore, a recirculation mode of operation could also play a role in buffering the fresh leachate that is being produced by the ongoing refuse degradation (or as it was the case in the experiment, buffering the synthetic leachate being input at the top of the columns). All these mechanisms were involved in the laboratory experiment, and, it is hypothesised, they explain the difference between the amount of calcite deposited within the geotextiles and the total amount of calcite precipitated.

5.2.4 Consequences of Geotextile Clogging

Although a decrease of more than one order of magnitude (as experienced by the uppermost geotextiles in the laboratory columns) seems to be a considerable drop in hydraulic conductivity, the critical issue, is how well the LCS can work under such circumstances (lower geotextile hydraulic conductivity). A few calculations are presented below in regard to landfill serviceability.

Based on Darcy's equation, geotextile hydraulic conductivity can be determined by:

$$K = -(q_o * \Delta z) / \Delta h \tag{5.1}$$

Using Darcy's equation for a typical landfill infiltration rate (q_o) of 0.15 m^3/m^2/yr, a geotextile thickness (Δz) of 0.3 cm, and geotextile hydraulic conductivity (K) of 0.0021cm/sec, leachate mounding, and (Δh) will account for $6.8*10^{-5}$ cm. This value is much smaller than the standard industry requirement of a maximum of 30 cm for leachate build-up above a landfill LCS (Subtitle D, USA EPA). In order to have a build-up of head of concern (e.g. 150 cm), for a 0.3 cm thick geotextile, its hydraulic conductivity would have to be lower than 10^{-9} cm/sec which would require a decrease of six orders of magnitude.

Although, the hydraulic conductivity results obtained in the laboratory experiment admittedly are not enough in order to forecast the leachate mounding in a real landfill, they do highlight the issue of a likely decrease in the permeability of the geotextiles overtime, and the possibility that such a decreased permeability can be controlled and ultimately beneficial for the purpose of leachate treatment.

5.2.5 Effects of Leachate Treatment on the Hydraulic Performance of the Drains.

Figure 5.5 shows a view of one section of the column C2. The autopsy of the columns showed blackened spots, most likely reflecting the path that the leachate

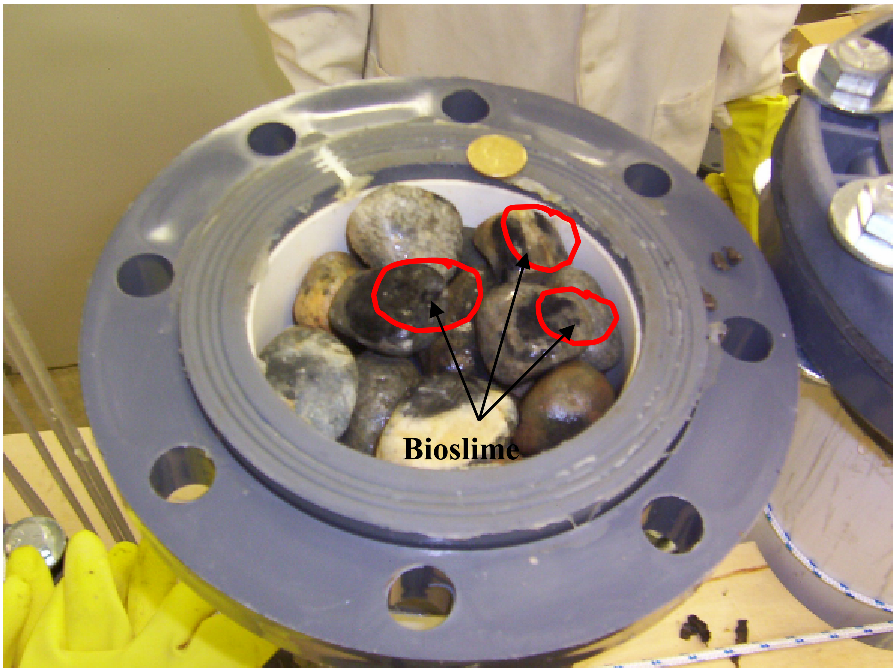

Figure 5.5 Black slime on drains

took while going from top to bottom of the columns. However, most of the biological activity was developed in the uppermost geotextiles due to its higher surface area and porosity.

Figure 5.6 depicts the gravel material shown in figure 5.5 after being hardened with a non calcium based epoxy. The objective of this was to be able to take cross-sections (such as the one shown in Figure 5.7) and examine the pore spaces, as well as to enable microscopic observations and mineral analyses of the precipitate coating the granular drainage media.

Figure 5.6 Hardened gravel material

Figure 5.7 Cross sectional view of the gravel material

The overlying geotextiles acted as a filter and promoter of bacterial growth, generating most of the leachate stabilization as well as trapping most of the by-products, leaving the hydraulic properties of the drains intact.

The implications of these findings are important for the operation bioreactor landfills. Leachate stabilization is linked to bacterial growth and mineral precipitation. Both factors, as it was seen before, can affect the permeability of the LCS and especially of the geotextiles. The potential for significantly decreased permeability of the geotextiles filters has to be addressed in order for a landfill to take advantage of the benefits of working as a bioreactor.

5.3 Potential of Recycled Concrete as a Drainage Material in a Landfill.

A major part of any LCS is the granular drainage medium. Natural gravel deposits are the primary source of such drainage material in North America. However, in the interest of preserving these deposits and in recycling used material, the use of crushed concrete as a drainage medium was investigated.

The purpose of using two different drainage mediums in this laboratory experiment, was to compare the performance differences (if any) between natural stone and crushed concrete. The crushed concrete had a somewhat smaller nominal diameter than the natural stone used in the top drainage layer (See chapter 3, section 3.2).

5.3.1 Fines Accumulation

Before the crushed concrete could be used it was necessary to screen the fines from the concrete. The screened concrete particles were then placed in the columns and the experiment began. It was immediately evident that a significant quantity of concrete fines had migrated in and partially clogged the geotextile immediately below the crushed concrete, as shown in Figure 5.8. It was then necessary to wash the concrete prior to use as drainage media to ensure sufficient removal of fines. Clear stone, on the other hand, did not exhibit such problems.

Figure 5.8 Geotextile clogged with fines originating from sieved, but not washed, crushed concrete.

This finding has two implications when using crushed concrete as a drainage medium. The first one is that crushed concrete may need costly preparation before it can be used in the LCS, and the second one is that fines coming into the drainage system from the crushed concrete, may provide nucleation sites for the precipitation of calcite ($CaCO_3$) which would be expected to contribute to more severe clogging of the LCS (Fleming et al, 1999).

These fines could accumulate and cause precipitation in three locations: the drainage medium (the crushed concrete), adjacent to or near the drainage pipes, or within a geotextile filter placed to separate two layers of drainage material.

5.3.2 Leachate Mounding

The hydraulic head above the first geotextile (directly beneath the refuse) was monitored within all the columns throughout the entire experiment duration. It was found that these head measurements were consistently higher in the two columns filled with crushed concrete, relative to the head measured in the columns packed with natural gravel (Figure 5.9). While the precise cause is not known, it seems likely that the crushed concrete, through a physical or chemical process is associated with more severe clogging of the overlying geotextile.

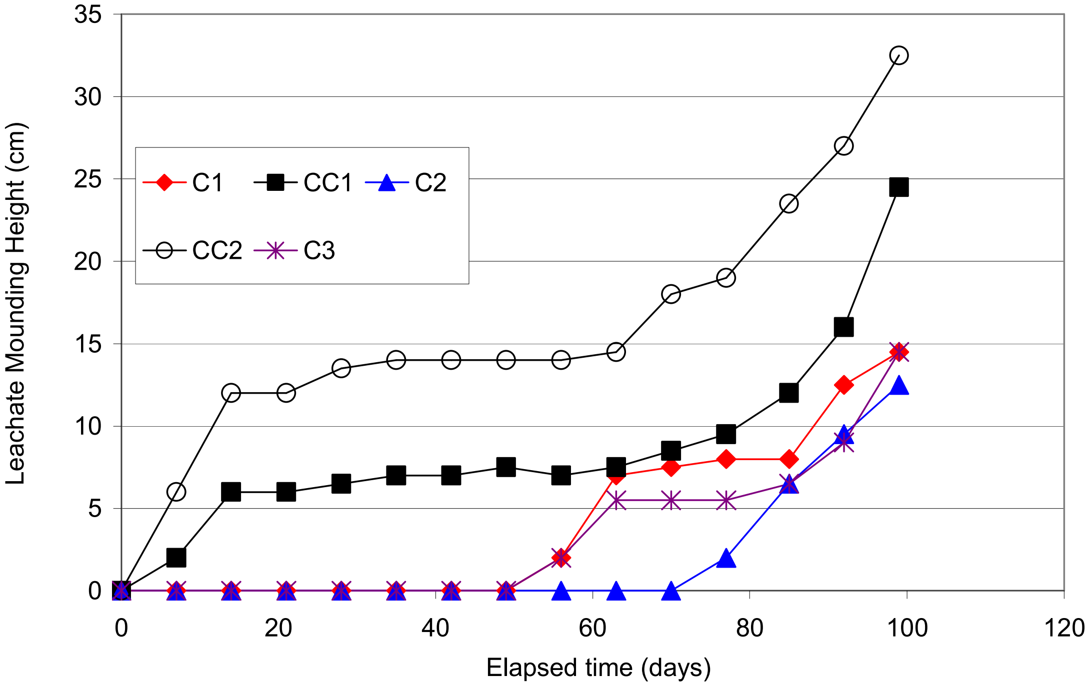

Figure 5.9 Graph representing leachate mounding above the top geotextile

5.3.3 Alteration of Leachate pH

Throughout the experiment, the discharge pH in the columns filled with crushed concrete was higher than the columns which used natural stone. This is likely due to interaction between the naturally alkaline concrete and the leachate. This effect is a concern as the solubility of calcite is lowered as pH is raised. The resulting consequence is potentially greater precipitation of calcite. Such calcite precipitation would further contribute to the clogging of the LCS, since LCS clog material has been shown to be composed largely of calcite (Fleming et al, 1999, Bennet et al, 2000, Fleming and Rowe, 2004). Figure 5.10 illustrates the history of the leachate pH for each column.

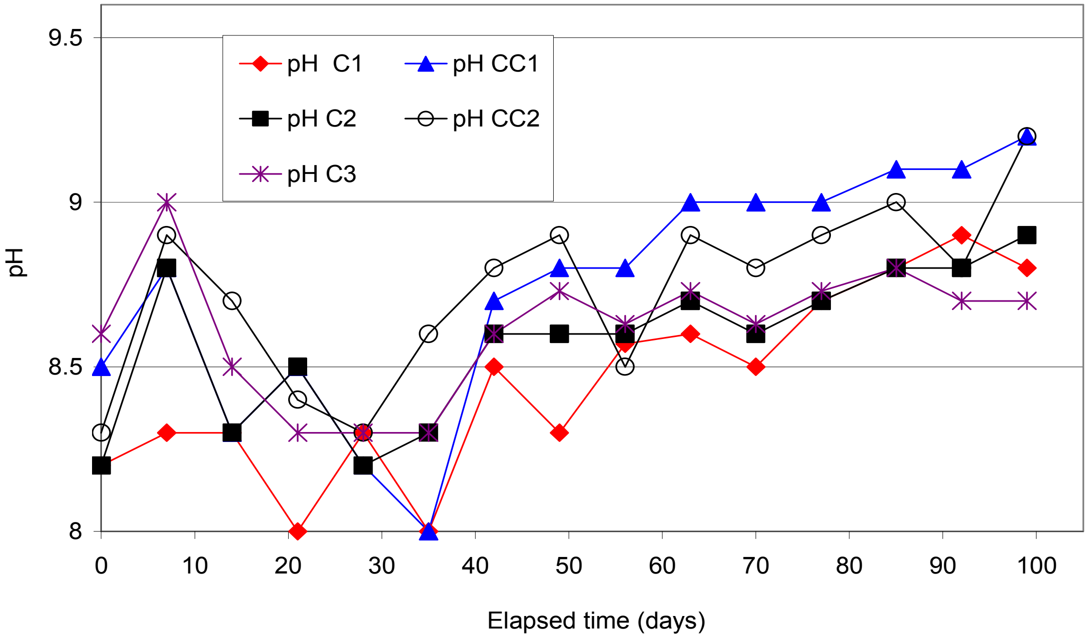

Figure 5.10 pH history of leachate. (See chapter 4: pH in anaerobic column discharge)

5.3.4 Durability of Aggregate

When the experiment was completed, one stone column and one crushed concrete column were impregnated with epoxy and upon hardening, cut into slices for examination of the cross-sections.

Inspection of the sections revealed a band of black discolouration permeating into both the crushed concrete and the clear stone. Figure 5.11 and 5.12 display photographs of weathered stone and weathered crushed concrete respectively. This discolouration penetrated more deeply into the crushed concrete and limestones, and to a lesser degree, sandstones and igneous rocks.

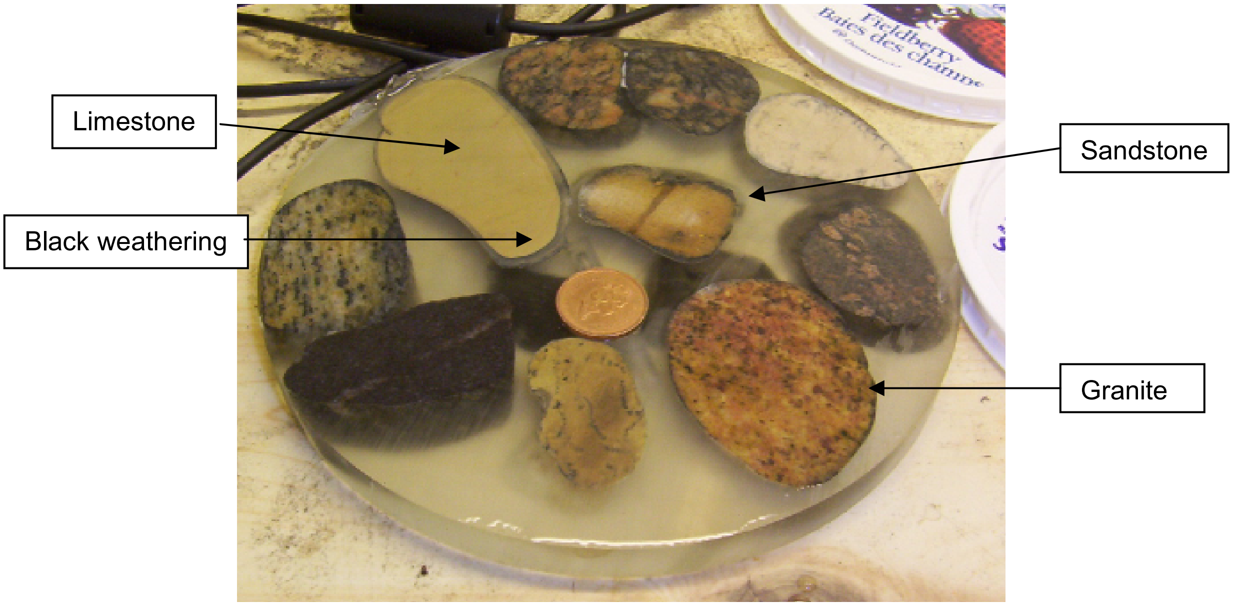

Figure 5.11 Slice of weathered stone with rock types labelled

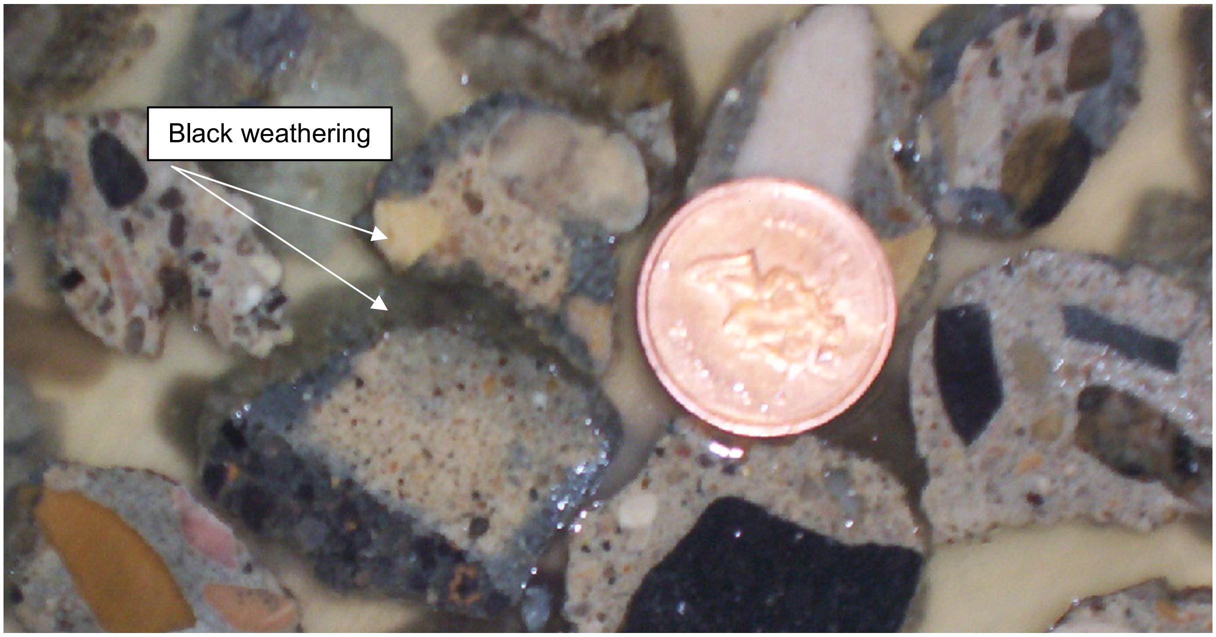

Figure 5.12 Slice of weathered crushed concrete.

The discoloured concrete exhibited signs of reduced durability. Upon visual inspection of the discoloured regions, the concrete was crumbly and small grains could be dislodged by running a finger across the surface. Though the natural stone also had black discolouration, it did not exhibit any obvious signs of weakening as a result of such alteration. The discoloration in the concrete can also be seen by comparison of the photomicrographs of non-weathered and weathered concrete represented by Figures 5.13 and 5.14 respectively.

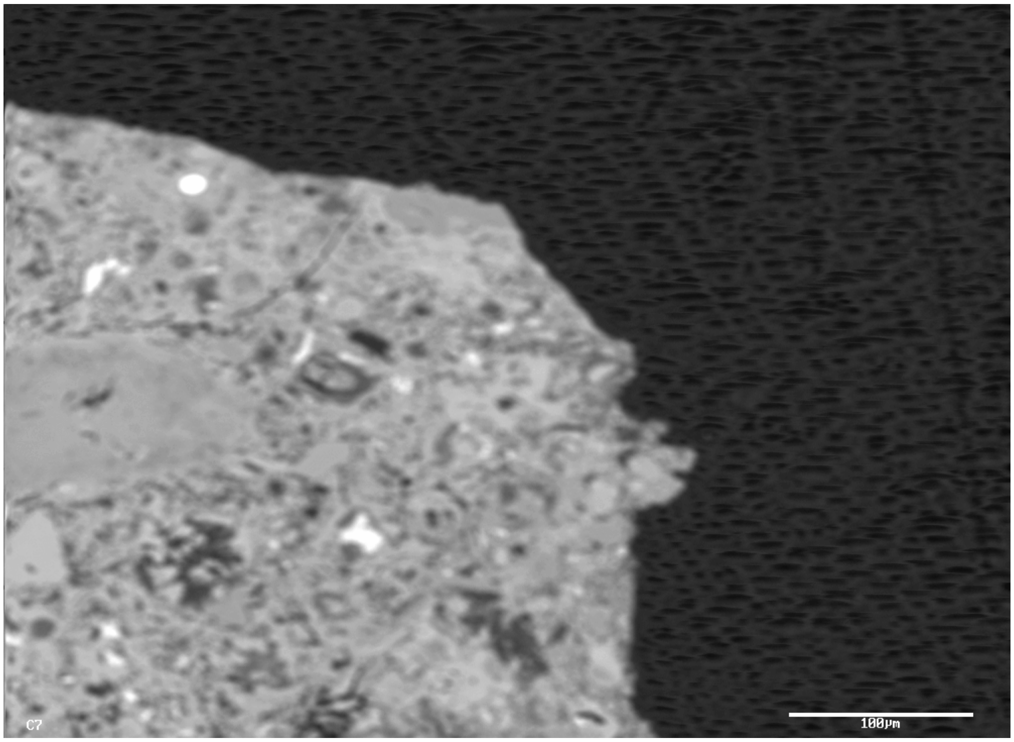

Figure 5.13 non-weathered concrete (100 μm)

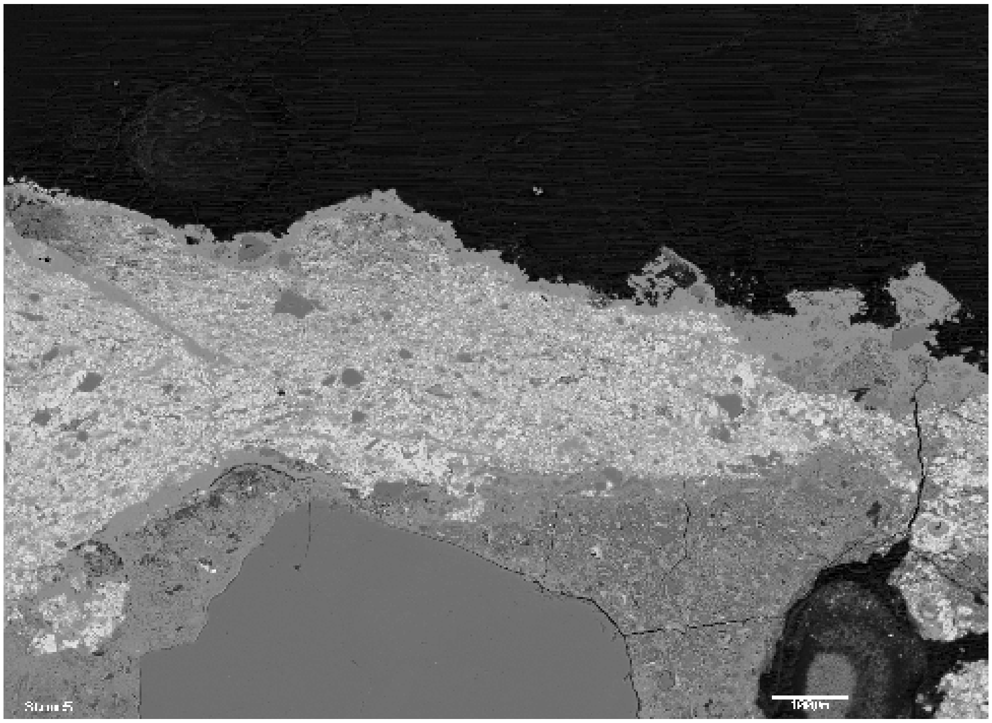

Figure 5.14 weathered concrete(100 μm)

In order to quantify the seeming crumbly appearance of the weathered concrete, a slake test was carried out using the non-weathered and weathered concrete. Table 5.4 shows the results of the test.

Table 5.4 Slake test for non-weathered and weathered concrete

Type of Concrete	Weight before slake test (g)	Weight after slake test (g)	Variation (%)
Non-weathered	498.46	493.3	1
Non-weathered	464.20	459.81	0.9
CC1 weathered	471.17	469.16	0.4
CC1 weathered	483.46	480.12	0.7
CC2 weathered	480.95	477.06	0.8
CC2 weathered	465.8	462.92	0.6

No significant variation was observed among the different concretes. The explanation for these results may lie in the low speed at which the test is performed. The visual weathered-crumbly appearance was not reflected in a test that is designed for softer materials in order to show a sizable difference in weight.

5.5 Conclusions

After 100 days of percolation by high-strength synthetic leachate, the uppermost geotextiles experienced a decrease in hydraulic conductivity of around 1.2 orders of magnitude as an average for the laboratory columns. The reason for this permeability reduction was clogging by overlying wet refuse, calcite precipitation and biological growth.

Based on the amount of refractory material found within the geotextile, it is likely useful to separate the geotextile from the refuse. A layer of sand in between can be a viable option in order to avoid having particles from the waste getting trapped in the geotextile fabric.

Based on extrapolating the data from the laboratory columns, it is hypothesized that an increased rate of recirculation in a landfill will more quickly decrease the hydraulic conductivity of the geotextile, thus potentially increasing leachate mounding over the geotextile filter. Leachate recirculation has been proven to accelerate waste stabilization which in turn is closely related to calcite precipitation and bacterial growth (therefore to geotextile permeability). This presents a potential design and operational challenge for landfills operated as bioreactors.

Even though a faster waste decomposition means a greater amount of carbonates released into leachate, calcite deposition also depends upon other factors such as pH, CO_2 partial pressure, temperature and calcium availability. Since all these components are not always constant, the place and intensity of such deposition can vary widely throughout the landfill.

The crushed concrete contained a substantial content of fines, even after screening to a uniform size of approximately 30 mm. Unless these fines are removed, the greater fines content would potentially result in accelerated clogging within filters, within the drainage media itself, or within the perforated leachate collection pipes, which would increase the cost of preparation of the material for use.

The higher leachate pH within the crushed concrete drainage media may contribute to more severe clogging of the LCS, due to the fact that with increasing pH, there is a decrease in the solubility of calcium carbonate (the main mineral component of clog material).

The columns containing the crushed concrete exhibited greater mounding of leachate above the primary geotextile than the natural stone counterparts. This may have been associated with either or both of the above-mentioned factors.

When considering the lifespan of the landfill, the questionable durability of the concrete may be a serious limitation. After only approximately one hundred days of operation, the crushed concrete appeared to have undergone noticeable weathering when compared to natural stone. The performance of the crushed concrete over a lifetime of 50 to 100 years or greater cannot be assured. Further study is required with respect to this phenomenon, however until this issue has been resolved, it is recommended that substantial caution be exercised in using crushed concrete as an alternative granular drainage media in municipal landfill leachate collection systems.

Chapter 6 Discussion and Recommendations

6.1 Introduction

This chapter discusses and summarizes the important conclusions reached in previous chapters. It also provides recommendations that will guide future investigations in the field of solid waste management.

6.2 Discussion

The management and operation of Municipal Solid Waste (MSW) landfills has evolved greatly over the past 20 years. Landfill settlements, landfill gas (LFG) management, and ground water contamination are some of the challenges facing landfill operators and/or owners in meeting increasingly stringent environmental regulations. Landfills have to be monitored until their outputs have stabilized (i.e. leachate, landfill gas, settlements), and no longer represents a threat to the environment. If solid waste is left to degrade without any intervention, the time span to reach stabilization would be much longer than the landfill can be used for waste collection.

Thus, every effort aimed at shortening the time necessary to reach stabilization is worthy. As was explained in chapter two, the benefits for landfill owners and operators are numerous.

The bioreactor landfill is a novel approach design to reach rapid stabilization of the solid waste. The main feature of this system is leachate recirculation; however, leachate recirculation also brings challenges to landfills such as potential slope instability, increased ammonia loads in the leachate, and the possibility of clogging of the Leachate Collection System (LCS). Despite these challenges, the central argument of this thesis is that the LCS can play a fundamental role in achieving rapid solid waste stabilization.

Several researchers have conducted field and laboratory studies on the clogging of LCS under a number of conditions. The main conclusion of these studies is that clogging is a process that occurs naturally, as a by-product of organic matter degradation. Based on this conclusion and on the works of Fleming et al. (1999) and Fleming et al. (2002), this thesis argues that a LCS can act as a fixed biofilm reactor. A landfill LCS possesses surface area and porosity that makes it suitable for being colonized by microorganisms, which in turn will decompose contaminants (nutrients for them), converting complex potentially harmful compounds into more simple, valuable and/or harmless forms, such as carbon dioxide, nitrogen gas, and methane gas.

A fine equilibrium is the key for the workability of this proposed argument. Biological clogging means that leachate is being treated but also means that the free drain, and therefore subsequent collection of the leachate, can be hampered. Thus, a LCS design that promotes bacterial settlement and allows free flow of the leachate will meet this fine equilibrium.

In this regard, five PVC columns were packed with porous media (granular material and geotextile), resembling those found in a landfill LCS. The size and location of these porous media, as well as the operating parameters such as leachate recirculation rates were intended to be similar to those in Municipal Solid Waste (MSW) leachate collection and treatment. An external aerated reactor connected to each column, was also set up with the primary function of achieving nitrification of the ammonia present in the leachate, an intermediate step in the conversion of this ammonia to harmless nitrogen gas.

The specific objectives of the laboratory experiment were to evaluate the removal of; Chemical Oxygen Demand (COD), nitrogen in the form of ammonia, and some inorganic material (Calcium), as well as to evaluate the hydraulic performance of the modified drainage LCS after the physical, chemical, and biological process, associated with that removal, took place.

The results regarding leachate quality were very satisfactory. Approximately 97% of the initial COD was removed. 98% of the ammonia was converted to nitrogen gas and 95% of the original calcium concentration was precipitated in the form of calcium carbonate. The hydraulic properties of the upper most geotextiles were affected by bio-slime formation along with fine material and mineral precipitate (mainly calcite) that were entrapped within the geotextiles. As a result, the uppermost geotextiles experienced a decrease in hydraulic conductivity of approximately 1.2 orders of magnitude on average. No clogging was observed in the natural gravel, in the crushed concrete, or in the other geotextiles. There are two reasons for this. First, the surface area of the natural gravel and the crushed concrete is smaller than that of the geotextiles, making it more difficult for the bacteria to attach and grow. Second, leachate strength is the highest when it is passing through the uppermost geotextile, thus most of the bacterial activity is developed at that location.

The use of concrete as a drain material for a landfill is not recommended. Noticeable weathering appearance was observed on the material, as well as a higher pH of the column discharge when compared to that of the columns containing natural gravel. When considering the lifespan of the landfill, the questionable durability of the concrete may be a serious limitation.

The biogas produced during the laboratory experiment was in constant amounts and its composition presented positive implications from the perspective of landfill gas management and operation. Firstly, the levels of methane gas were large enough to be considered as a possible source of energy. Secondly, the level of carbon dioxide was very low compared to that found in typical landfills. Carbon dioxide is a corrosive agent. Its low concentration might extend the life of the landfill gas (LFG) collection system.

6.3 Recommendations

Future researchers who continue with the line of experimentation described in this thesis should find the following recommendations useful:

1. A separator between the filter and the refuse should be incorporated in the design of landfill LCS. In the laboratory experiment, almost 50 % of the material entrapped in the geotextile fabric is thought to come from the waste. The type of material used for the separator will be determined based largely on the operating conditions of the landfill (i.e. rates of recirculation).

2. The use of an external aerated nitrification reactor may be useful not only for nitrogen treatment but also for decreasing the amount of carbonate present in leachate and therefore the amount of calcite precipitation. The aeration of the tank, coupled with the unbalanced carbonate system (supersaturated with carbonates) may decrease the amount of carbonates present in the reactor by releasing it to the atmosphere.

3. Given the rather substantial cost savings that might be achieved through the use of crushed concrete in lieu of natural gravel, a limited extension of this study

should be conducted in order to determine to what extent crushed concrete should be used in landfill drainage only with extreme caution or not at all.

4. The overall results of the laboratory experiment were positive, especially for nitrogen and COD removal rates. It is recommended, that the knowledge acquired during this laboratory experiment be applied to a pilot experiment under real landfill conditions.

References

Armstrong, M.D. 1998. Laboratory program to study clogging in a leachate collection systems. ME.Sc thesis, The University of Western Ontario, London, Ont.

Azevedo, B.D., Mavinic, D.S., and Robinson, H.D. 1995. The effect of ammonia loading and operating temperature on nitrification and denitrification of a high ammonia landfill leachate. Canadian Journal of Civil Engineering.**22** : 524-534.

Barlaz, M.A., Schaefar, D.M., and Ham, R.K. 1989. Bacterial Population Development and Chemical Characteristics of Refuse Decomposition in a Simulated Sanitary Landfill. Appl.Env. Microbiol. **55**(1): 55-56.

Barlaz, M.A., Rooker, A.P., Kjeldsen, P., Gabr, M.A., and Borden, R.C. 2002. Critical evaluation of factors required to determine the post-closure monitoring period at solid waste landfills. Environmental Science and Technology. **36** (16): 3457-3464.

Bennet, P.J., Longstaffe, F.J., and Rowe R.K., 2000. The stability of dolomite in landfill leachate collection systems. Canadian Geotechnical Journal. **37**: 371-378.

Brune, M., Ramke, H.G., Collins, H., and Hanert, H.H. 1991. Incrustation processes in drainage systems of sanitary landfills. Proc. Sardinia 91, 3rd International Symposium on Sanitary Landfills, CISA, Cagliary, Italy, pp.99-1035

Burton, S.A.Q., and Watson-Craick, I.A. 1998. Ammonia and nitrogen fluxes in landfill sites: applicability to sustainable landfilling. Waste Management and Research. **16** (1): 41-53.

Cooke, A.J., Rowe, R. K., Rittmann, B. E., VanGulck, J., and Millward S.C. 2001. Biofilm growth and mineral precipitation in synthetic leachate columns. ASCE Journal of Geotechnical and Geoenvironmental Engineering. **127**(10):849-856.

Christensen, T.H., and Kjeldsen, P. 1989. Basic Biochemical processes in landfills. In: Sanitary Landfilling: Process, Technology, and Environmental Impact, ed. T.H. Christensen, R. Cossu and R. Stegmann. Elsevier Applied Science, London, UK.

Christensen, T.H., Kjeldsen, P. and Stegmann, R. 1992. Effects of landfill management procedures on landfill stabilization and leachate and gas quality. Ed T.H. Christensen, R. Cossu and R. Stegmann. Academic Press, London, UK.

Farquhar, G.J. Leachate: Production and Characterization. 1989. Canadian Geotechnical of Civil Engineering. **16**: 317-325.

Fleming, I.R., Rowe, R.K.,and Cullimore, D.R. 1999. Field observations of clogging in a landfill leachate collection system. Canadian Geotechnical Journal.**36**(4):685-707.

Fleming, I.R., and R.K. Rowe. 2000. Toward Sustainable Leachate Treatment in Landfills. CSCE/CGS Specialty Environmental Conference, London, Ont., June, p15-20.

Fleming, I.R., Rowe, R.K., Dewacle, P., and Benda, E. 2002. Evolution of leachate strength in municipal landfills; can leachate be treated in-situ? In: Proceedings form the Solid Waste Association of North America. 7th Annual landfill symposium. Louisville, Kentucky.

Fleming, I.R., and Rowe, R.K. 2004. Laboratory studies of clogging of landfill leachate collection and drainage systems. Canadian Geotechnical Journal. **41**(1):134-153.

Gallagher, E.M. 1998. Biological clogging of geocomposites exposed to raw landfill leachate over an extended period. Waste Management Research **16**(5):421-429.

Halse, Y., Koerner, R.M., and Lord, A.E. 1987. Effect of High Levels Alkalinity on Geotextiles. Part 1: Ca (OH) Solutions. Geotextiles and Geomembranes. **5**:261-282.

Halse, Y., Koerner, R.M., Lord, A.E., 1987. Effect of High Alkalinity Levels on Geotextiles. Part 2: NaOH Solution. Geotextiles and Geomembranes. (6):295-305.

Harris, J.M., Purschwitz, and D.E., Goldsmith, C.D. 2000. Leachate treatment options for sanitary landfills. Intercontinental Landfill Research Symposium. Lulea, Sweden.

Jetten, S.M., Wagner, M., Fuerst, J., Van Loosdrecht, M., Kuenen, and G., Strous, M. 2001. Microbiology and application of the anaerobic ammonium oxidation (‘anammox’) process. Current opinion in biotechnology. **12**: 283-288.

Jefferis, S.A., and A. Bath., 1999. Rationalising the debate on calcium carbonate clogging and dissolution in landfill drainage materials. Proceedings Sardinia 99, Seventh International Waste Management and Landfill Symposium*,* S. Margherita di Pula, Cagliari, Italy.

Kiely, G. 1997. Environmental Engineering. McGraw-Hill.

Knox, K. 1985. Leachate treatment with nitrification of ammonia. Water Research. **19**:895-904.

Koerner, R.M., Lord, A.E., and Halse, Y. 1988. Long-term durability and aging of geotextiles. Geotextiles and Geomembranes. **7**:147-158.

Koerner, G.R., and Koerner, R.M. 1992. Leachate flow rate behaviour through geotextile and soil filters and possible remediation methods. Geotextiles and Geomembranes. **11**:401-430.

Koerner, R.M. 1994. Performance evaluation of geotextiles filters used in leachate collection of solid waste landfills. PhD dissertation, Drexel University, Philadelphia, PA.

Koerner, G.R. and Koerner, R.M. 1995. Leachate clogging assessment of geotextile (and soil) landfill filters. US EPA Report, CR-819371.

Leckie, J.O., Pacey, J.G. and Halvadakis, C. 1979. Landfill management with moisture control. ASCE, Journal of Environmental Engineering Division.**105:** 337-355.

Leushner, A.P. 1989. Enhancement of degradation: laboratory scale experiment. In Sanitary Landfilling: Process, Technology, and Environmental Impact, ed. T.H. Christensen, R. Cossu and R. Stegmann. Academic Press, London, UK.

Lin, Yin-Feng, and Chen, Kuo-Cheng. 1995. Denitrification and methanogenesis in a co- immobilized mixed culture system. Water Research **29**(1):35-43.

McBean, E.A., F.A. Rovers, and G.J. Farquhar. 1995. Solid Waste Landfill Engineering and Design. Prentice Hall. Englewood Cliffs, N.J.

Mlynarek, J., and Vermeersch, O. Filter criteria for geotextile materials in soil filtration and leachate collection systems. 1999. Proceedings Sardinia 99, Seventh International Waste Management and Landfill Symposium, Cagliari, Italy, vol. 3, pp.233-240.

Onay, T. and Pholand, F. 1998. In Situ Nitrogen Management in Controlled Bioreactor Landfills. Water Research. **32** (5):1383-1392.

Paksy, A., Powrie, W., Robinson, J.P., and Peeling. 1998. A Laboratory investigation of anaerobic microbial clogging in granular landfill drainage media. L. Geotechnique **48**(3):389-401.

Palmisano, A.C. Barlaz, M. 1996. Microbiology of solid waste. CRC press. New York, N.Y.

Parkin G.F., and Owen, W.F. 1986. Fundamental of anaerobic digestion of waste water sludges. ASCE Journal of Environmental Engineering. **112** (5):867-920.

Peeling, L., Paksy, A., Robinson, J.P. and Powrie, W. 1999. Removal of volatile acids from synthetic landfill leachate by anaerobic biofilms on drainage aggregates: A laboratory study. Waste Management and Research. **17**(2):141-149.

Pohland, F.G. 1980. Leachate recycle as a management option. J.Environ.Eng.Div. ASCE **106**(6): 1057-1069.

Pohland, F.G. and Yousfi, B.A. 1994. Design and operation of landfills for optimum stabilization and biogas production. Water Science and Technology. **30** (12):117-124.

Pohland F.G. 1995. Landfill bioreactors; historical perspective, fundamental principles, and new horizons in design and operation. In Landfill Bioreactor Design and Operation Sem. Proc., EPA/600/R-95/146,pp.9-24.

Pohland F.G., and Kim, J.C. 1999. In situ anaerobic treatment of leachate in landfill bioreactors. Water Science and Technology. **40** (8):203-210.

Polarczyk, K.W. 2000. Geosynthetics and Geosystems in Hydraulic and Coastal Engineering. A:A. Barkena/Rotterdam/Brookfield.

Price, G.A., Barlaz, M.A., and Hater, G.R. 2003. Nitrogen management in bioreactor landfills. Waste Management. **23**: 675-688.

Quian, X., Koerner, R.M., and Gray, D.H. 2002. Geotechnical aspects of landfill design and construction. Prentice Hall.Upper Saddle River, N.J.

Ramke, H.G. 1989. Leachate Collection Systems, Published in Sanitary Landfilling: Process, Technology and Environmental Impact, Academic Press.

Reddi, L.N. 1997. Particle transport in Soils: Review of Significant Processes in Infrastructure Systems. Journal of Infrastructure Systems **3**(2): 78-75.

Reinhart, D.R, and Chopra, M.B. 2000. MSW Landfill Leachate Collection Systems for the New Millennium. State University System of Florida. Florida Center for Solid and Hazardous Waste Management. University of Florida. Report #00-13

Reinhart, D.R., McCreanor, P.T. and Towsend, T. 2002. The bioreactor landfill: Its status and future. Waste Management and Research. 20 (2):172-186.

Rittmann B. E, Fleming, I.R. and Rowe, K.R. 1996. Leachate chemistry: its implication for clogging. North American Water and Environmental Congress ’96 , Anaheim, CA. paper 4(CD ROM), session GW-1, Biological processes in ground water quality.

Rittmann B. E., Banaszak J.E., Cooke A., and Rowe, R. K. 2003. Biogeochemical evaluation of mechanisms controlling $CaCO_3$. precipitation in landfill leachate-collection systems. ASCE Journal of Environmental Engineering.**129** (8): 723-730.

Rollin, A.L., Lombard, G. 1988. Mechanisms affecting long-term filtration behaviour of geotextiles. Geotextiles and Geomembranes. **7**:119-145.

Rowe, R.K., I.R. Fleming, M.D. Armstrong D.R. Cullimore, B.E. Rittmann, P. Bennett, and F.J. Longstaff, 1997b. Recent Advances in Understanding the Clogging of Leachate Collection Systems. Proceedings Sardinia '97, Sixth International Landfill Symposium, Cagliari, Italy, pp. 383-390.

Rowe, R.K., and I.R. Fleming. 1998. Estimating the Time for Clogging of Leachate Collection Systems. Proc. 3rd International Congress on Environmental Geotechnics. Lisbon, Portugal. pp 23-28.

Rowe, R.K, Armstrong, M.D, and Cullimor C.D. 2000. Particle size and clogging of granular media permeated with leachate. Journal of Geotechnical and Geoenvironmental Engineering. **126**(9):775-786.

Rowe, R.K., VanGulck, J.F., and Millward, S.C. 2002. Biologically induced clogging of a granular medium permeated with synthetic leachate. Journal of Environmental Engineering and Science. **1:**135-156.

Rowe, R.K., Armstrong, M.D., and Cullimore, D.R. (2000a). Particle Size and Clogging of granular Media Permeated with Leachate. ASCE Journal of Geotechnical and Geoenvironmental Engineering. **126**(9):775-786.

Rowe, R.K., Armstrong, M.D. and Cullimore, D.R. (2000b). Mass Loading and the Rate of Clogging due to Municipal Solid Waste Leachate, Canadian Geotechnical Journal. **37**(2):355-370.

Ruiz, E.F, Fleming, I.R., Putz, G. Passive treatment of municipal landfill leachate in a granular drainage layer. 2005. Submitted to the Journal of Environmental Engineering and Science. NRC.

Shiskowsky D.M., and Mavinic, D.S. 1998. Biological treatment of a high ammonia leachate influence of external carbon during initial startup. Water Research. 32(8):2533-2541.

Schmidt, I., Sliekers, O., Scmid, M., Cirpus, I., Strous, M., Bock, E., Kuenen, G., and Jetten, M.S.M. 2002. Aerobic and anaerobic ammonia oxidizing bacteria-competitors or natural partners. FEMS Microbiol.Ecol. **39:** 175-181.

Stegmann, R., and Spendlin, H.H. 1986. Research activities on enhancement of biochemical process in sanitary landfills. Waste Management and Research. **1**:201-211.

Stegmann, R., and Spendlin, H.H. 1989. Enhancement of degradation: German experiences. In: Sanitary Landfilling: Process, Technology, and Environmental Impact, ed. T.H. Christensen, R. Cossu and R. Stegmann. Academic Press, London, UK.

Tchobanoglous, G., Theisen H., and Vigil, S. 1993. Integrated solid waste management: Engineering principles and management issues. McGraw-Hill Inc.

Tiedje, J.M. 1988. Ecology of denitrification and dissimilatory nitrate reduction to ammonium. In: Zehnder, J.B.A. (Ed), Biology of Anaerobic Microorganism. John Wiley and Sons. NY, pp. 179-244.

US EPA Process design manual for nitrogen control. 1975. Technology Transfer. Washington, DC United States.

US EPA Waste water technology fact sheets: Sequencing batch reactors. 1999. Office Water, Washington, DC United States.

US EPA Municipal solid waste in the United States: 2000 facts and figures. 2002. Office of Solid Waste and Energy Response, Washington, DC. United States.

VanGulck, J.F., Rowe, R.K., Rittmann B. E, and Cooke, A.J., 2003. Predicting biogeochemical calcium precipitation in landfill leachate collection systems. Biodegradation **14**(5): 331-346.

VanGulck, J.F. and Rowe, R.K. 2004 Evolution of clog formation with time in columns permeated with synthetic landfill leachate. Journal of Contaminant Hydrology **75:** 115– 139

Warith M.A. and Sharma, R. 1998. Technical review of methods to enhance biological degradation in sanitary landfills. Water Quality Research Journal. **33** (3): 417-437.