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Facultative Bioreactor Landfill: An Environmental and Geotechnical Study

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FACULTATIVE BIOREACTOR LANDFILL: AN ENVIRONMENTAL AND GEOTECHNICAL STUDY

A Dissertation

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ABSTRACT

A relatively new concept of Municipal Solid Waste treatment is known as bioreactor landfill technology. Bioreactor landfills are sanitary landfills that use microbiological processes purposefully to transform and stabilize the biodegradable organic waste constituents in a shorter period of time. One of the most popular types of bioreactor landfills is the landfill with leachate recirculation. However, it is observed that ammonia rapidly accumulates in landfills that recirculate leachate and may be the component that limits the potential to discharge excess leachate to the environment. In the facultative landfill, leachate is nitrified biologically using an on-site treatment plant and converted by denitrifying bacteria to nitrogen gas, a harmless end-product. In this research, three pilot-plant scale lysimeters are used in a comparative evaluation of the effect of recirculating treated and untreated leachate on waste stabilization rates. The three lysimeters are filled with waste prepared with identical composition. One is being operated as a facultative bioreactor landfill with external leachate pre-treatment prior to recirculation, the second is being operated as an anaerobic bioreactor landfill with straight raw leachate recirculation, and the third one is the control unit and operated as a conventional landfill.

Apart from environmental restrictions, geotechnical constraints are also imposed on new sanitary landfills. The scarcity of new potential disposal areas imposes higher and higher landfills, in order to utilize the maximum capacity of

those areas. In this context, the knowledge of the compressibility of waste landfills represents a powerful tool to search for alternatives for optimization of disposal areas and new solid waste disposal technologies.

This dissertation deals with and discusses the environmental and geotechnical aspects of municipal solid waste landfills. In the Environmental Engineering area, it compares the quality of the leachate and gas generated in the three lysimeters and discusses the transfer of the technology studied through lysimeters to procedures for full-scale operation.

In the geotechnical area, this dissertation discusses the compressibility properties of the waste and provides a state-of-the-art review of MSW compressibility studies. It also evaluates the compressibility of MSW landfills for immediate and long-term settlements and proposes a new model for compressibility of waste landfills.

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CHAPTER 1

INTRODUCTION

1.1. Initial Considerations

Land disposal of solid wastes has been the most common destination for solid wastes throughout the world, particularly for Municipal Solid Wastes (MSW). According to the United States Environmental Protection Agency (US EPA 2000), approximately 55% of the 220 million tons of MSW generated in the United States in 1998 were disposed in landfills. However, the concept and requirements for waste disposal have changed with time. While primitive men buried their wastes in order to destroy the traces of their presence to avoid pillage from other tribes, historically wastes of urban agglomerations have been disposed in desert areas or areas with low economic value. At the end of the nineteenth century, sanitary issues and the necessity of avoiding the transmission of diseases became the basic concerns for the disposal of solid wastes. From the middle to the end of the last century, environmental protection was incorporated in the list of requirements for a MSW disposal area (Sowers, 1968). However, new areas for MSW disposal are becoming more and more scarce near most populated areas and the necessity for new systems that are able to treat the waste, not only to store it, must be addressed.

A relatively new concept of waste treatment is known as bioreactor landfill technology. Bioreactor landfills are sanitary landfills that use microbiological processes by design to transform and stabilize biodegradable organic waste constituents in a shorter period of time by enhancing microbial decomposition (Pacey et al., 1999). Although several enhancement techniques are discussed in the literature, such as pH adjustment, waste shredding, sewage sludge addition, pre-composting, and enzyme addition, the most common and investigated process-based management option utilized is leachate recirculation. It promotes the active microbial degradation by providing optimum moisture, induces a liquid flux to provide a mechanism for the effective transfer of microbes, substrates, provides nutrients throughout the refuse mass, and dilutes local high concentrations of inhibitors (Yuen, 2001).

At landfills where raw leachate recirculation is practiced, the concentrations of ammonia are typically high, since there is no mechanism for ammonia removal (Clabaugh, 2001). Thus, ammonia may be the component of leachate that limits the potential to safely discharge excess leachate into the environment. In the facultative landfill, leachate is nitrified biologically using an on-site treatment plant before returning to the landfill bioreactor as nitrates. Nitrates may be converted to nitrogen gas, a harmless end-product, by denitrifying bacteria. The carbon required for this process is obtained from the organic matter in the waste mass. This combined process promotes rapid degradation of the waste when compared to anaerobic metabolism.

Apart from environmental restrictions, geotechnical constraints are also imposed on new sanitary landfills. The scarcity of new potential disposal areas produces higher and higher landfills, in order to utilize the maximum capacity of those areas. From this point of view, it is also necessary to understand the importance of knowing the geotechnical characteristics of solid wastes. In this context, the knowledge of the compressibility of waste landfills represents a powerful tool to search for alternatives for optimization of old, present, and future disposal areas, as well as for the development of new solid waste technologies.

Although the concept of improving waste stabilization rates with leachate recirculation has been developed over the last 30 years and is increasingly being promoted as an environmentally sound municipal solid waste disposal method, there is no long-term full experience. Some key issues need further research, and several practical questions still need to be answered. One of these questions is how efficiently can the use of externally treated leachate benefit and treat the waste in order to obtain cleaner, and, therefore, become less harmful to the environment?

Similarly, although the mechanisms that generate settlements in MSW landfills are well known, there is no available model at present to adequately represent the settlements that occur in waste landfills due to the high complexity of the phenomenon. Although classical soil mechanics compressibility theories are commonly used for waste settlement prediction throughout the world, they have several deficiencies when translated to MSW. Other formulations that take into consideration several of the waste settlement mechanisms, such as Zimmerman’s Mathematical Model (Zimmerman et al., 1977), are extremely complex and require as input an extensive number of parameters very difficult to determine; therefore, it is very difficult to use these models in practice.

1.2. Objectives of this Dissertation

This dissertation is part of a larger project being carried out at the University of New Orleans by the Civil and Environmental Engineering Department called “Facultative Landfill Bioreactor Project”. The main objective of this project is to evaluate the effect of external leachate pre-treatment prior to recirculation on waste stabilization rates, and leachate and gas production. To achieve this objective, three lysimeters are operating under different conditions:

1. Recirculation with treated leachate (facultative landfill)

2. Recirculation with untreated leachate (landfill with raw leachate recirculation)

3. No-recirculation (conventional landfill).

Additional research objectives of the project are to:

- Establish differences in leachate and gas macro constituents among the three lysimeter operation modes;
- Compare the quantity and quality of the leachate produced in terms of Chemical Oxygen Demand (COD), ammonia, nitrates, metals, chlorides, phosphorus, Total Suspended Solids (TSS), and Volatile Suspended Solids (VSS);
- Compare the MSW geotechnical characteristics such as compressibility and hydraulic properties;
- Evaluate the effectiveness of an aluminum-plate electro-coagulation unit for metal removal from leachate;
- Evaluate the effectiveness of biological fluidized bed-reactors used for anaerobic removal of organic matter;
- Evaluate the effectiveness of aerobic biological fluidized bed-reactor for nitrification of ammonia contained in leachate; and
- Evaluate the rate of denitrification reached in the facultative landfill lysimeter.

1.2.1. Environmental Study

This dissertation deals with and discusses some significant environmental and geotechnical aspects of municipal solid waste landfills. In the Environmental Engineering area, the objectives are to:

- Review and update the study conducted by Cadenas (2002), comparing the quality of the leachate generated in terms of COD, ammonia, nitrates, metals, chlorides, phosphorus, TSS, and VSS, and establish differences in gas macro constituents among the three lysimeter operation modes;

- Discuss the denitrification process inside the facultative landfill lysimeter;

- Compare the quantities of leachate produced by the three lysimeters; and

- Discuss the transference of the technology studied through lysimeters to procedures for full-scale operation regarding the environmental issues.

1.2.2. Geotechnical Study

In the geotechnical area, this dissertation compares the MSW geotechnical characteristics, mainly the compressibility properties of waste in the three lysimeters. The dissertation objectives in the geotechnical are to:

- Provide a state-of-the-art review of MSW compressibility studies;

- Evaluate the compressibility of MSW landfills for immediate and long-term settlements by analyzing the data obtained from this research, comparing and incorporating it to the data obtained about MSW compressibility studies previously done at the University of New Orleans (Debnath, 2000) and in the general literature (some of them presented by: de Abreu, 2000, and Carvalho, 1999) with the objective of proposing a new model for compressibility of waste landfills; and

- Discuss the transference of the technology studied through lysimeters to procedures for full-scale operation regarding the geotechnical issues.

1.2.3. Organization of this Dissertation

Chapter 2 presents the literature review for bioreactor landfills, including the basics of MSW degradation, the types of bioreactor landfills and the main studies at small and full-scale landfills regarding the use of bioreactors. Chapter 3 presents the literature review regarding the compressibility of MSW landfills and points out the present knowledge about this issue and most of the theoretical and empirical models used to predict settlements in sanitary landfills. In Chapter 4, the methodology used to study the environmental and geotechnical issues of waste landfills, according to the stated objectives, is presented. Chapters 5 and 6 present, respectively, the results and the analyses of the results of this study, relating them to the data obtained by other researchers. Finally, Chapter 7 concludes and presents the main results obtained and furnishes guidance to other researchers interested in continuing the study on MSW compressibility and/or bioreactor landfills.

CHAPTER 2

BIOREACTOR LANDFILLS

2.1. Introduction

Bioreactor landfills are sanitary landfills that explicitly use microbiological processes to transform and stabilize the biodegradable organic waste constituents in a shorter period of time by enhancing microbial decomposition (Pacey et al., 1999). Although the general concept is well understood, there are nuances in the definitions presented by some authors, and to come up with a final definition that everyone can agree on has been nearly impossible, as pointed out by Campman and Yates (2002). The EPA (2000) also points out the differences: some authors view bioreactor landfills as large-scale, in-ground composting operations, while others view them as providing solid waste treatment and draw similarities to a wastewater treatment facility with the intention of using the landfill space for treatment rather than indefinite future storage.

For the purpose of this dissertation, the most recent definition proposed by the Solid Waste Association of North America (SWANA) (2001) will be used: "A bioreactor landfill is any permitted Subtitle D landfill or landfill cell, subject to New Source Performance Standards/Emissions Guidelines, where liquid or air, in addition to leachate and landfill gas condensate, is injected in a controlled fashion into the waste mass in order to accelerate or enhance biostabilization of the waste".

Although several enhancement techniques are discussed in the literature, such as pH adjustment, waste shredding, sewage sludge addition, pre-composting, and enzyme addition, the most common and frequently investigated process-based management option utilized is leachate recirculation.

Still there is some controversy over the advantages and disadvantages of bioreactor landfills. Regarding bioreactor landfills with leachate recirculation, besides reducing the time required for landfill stabilization, the advantages are (Reinhart and Townsend, 1998; Yuen, 2001):

- Distribution of nutrients and enzymes throughout the waste mass;
- pH buffering;
- Dilution of inhibitory compounds;
- Recycling and distribution of methanogens;
- Improvement of landfill gas production rate;
- Liquid storage; and
- Reduction in time and cost of post-closure monitoring.

It has been also suggested that leachate recirculation can reduce the time required for landfill stabilization from several decades to two to three years (Pohland, 1975, as reported by Reinhart and Townsend, 1998).

Regarding the limitations, the primary criticism of leachate recirculation is that it increases the hydraulic loading on the landfill liner and, therefore, could increase the rate of groundwater pollution by leachate (Lee and Lee, 1994). Other problems associated with leachate recirculation are the increasing of pore pressures and reducing of the materials' strength affecting the geotechnical stability of the landfill, and also a potential for leachate seepage through the slopes of the landfill. The practice is prohibited in some countries and in some U.S. states, for example, New Jersey, according to Lee and Lee (2000)

Bioreactor landfills can be classified based on the environmental conditions for the microorganisms (EPA, 2002). They can be aerobic, anaerobic or facultative (hybrid):

- Aerobic: In an aerobic bioreactor landfill, oxygen (or air containing oxygen) is injected into the waste mass, using vertical or horizontal wells, to promote aerobic activity and accelerate waste stabilization. Leachate is removed from the bottom layer and can be piped to liquid storage tanks and re-circulated to the landfill in a controlled manner.

- Anaerobic: In an anaerobic bioreactor landfill, moisture is added to the waste mass in the form of re-circulated leachate and other sources to obtain optimal moisture levels. Biodegradation occurs in the absence of oxygen (anaerobically) and produces landfill gas. Landfill gas, primarily methane, can be captured to minimize greenhouse gas emissions and for energy projects.

- Facultative (or Hybrid): The facultative bioreactor landfill accelerates waste stabilization by using nitrogen management to remove the ammonia in the leachate generated, using an aerobic treatment to convert ammonia into nitrate (nitrification process), followed by a denitrification process (the conversion of nitrate to nitrogen gas - N_2, a harmless product), reinjecting the treated nitrified leachate into the waste mass, and using the carbon present in a large quantities inside the landfill for the conversion of nitrate to nitrogen gas .

2.2. Municipal Solid Waste Decomposition

A comprehensive review of the process of decomposition of MSW is necessary for a better understanding of how bioreactor landfills work. Basically, MSW decomposition can occur under either aerobic (using free oxygen) or anaerobic (in the absence of free oxygen) conditions.

The basic unbalanced equation for aerobic decomposition for complex organic compounds of the form $C_xH_yN_z$ is (Peirce et al., 1997):

$$O_2 + C_xH_yN_z \rightarrow CO_2 + H_2O + NO + \text{products} \quad \text{……………………………..………..(2.1)}$$

As seen in Equation 2.1 carbon dioxide and water are two of the end products of aerobic decomposition. Both are stable, low in energy, and used by plants in photosynthesis. Organic nitrogen is oxidized through a series of compounds ending in nitrates, as shown in Equation 2.2:

Organic nitrogen → NH_3 (ammonia) → NO_2^- (nitrite) → NO_3^- (nitrate)
…....……………..(2.2)

Because of this progression, nitrogen can be used as an indicator of water pollution, for example. Figure 2.1 presents the aerobic nitrogen, carbon and sulfur cycles.

Anaerobic decomposition is performed by a different type of microorganism, to which oxygen is toxic. The basic unbalanced equation for anaerobic biodegradation is:

$$C_xH_yN_z \rightarrow CO_2 + CH_4 + NH_3 + \text{partly stable compounds} \quad \text{………......……..(2.3)}$$

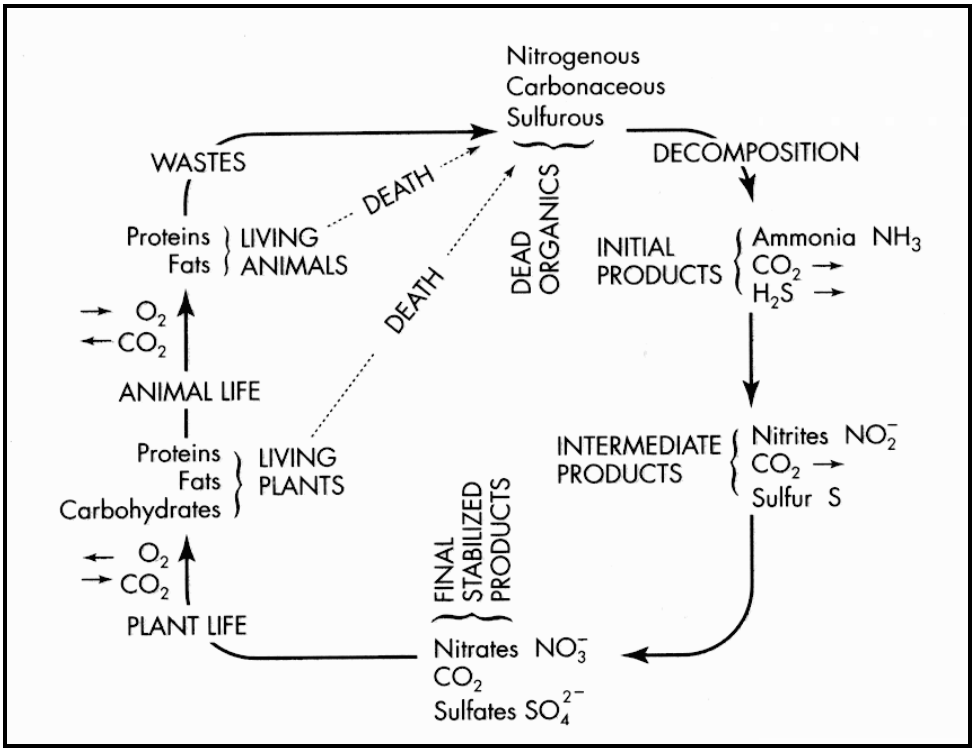

Figure 2.1 – Aerobic Nitrogen, Carbon and Sulfur Cycles (Peirce et al., 1997)

Many of the end products of the anaerobic reaction are biologically unstable. For example, Methane (CH_4), a high energy gas, is physically stable but can be decomposed biologically. Ammonia (NH_3) can be oxidized, and sulfur is anaerobically biodegraded to sulfhydryl compounds like hydrogen sulfide (H_2S). Figure 2.2 presents a schematic representation of anaerobic decomposition. Note that the left part of the cycle, photosynthesis by plants, is identical to the aerobic cycle.

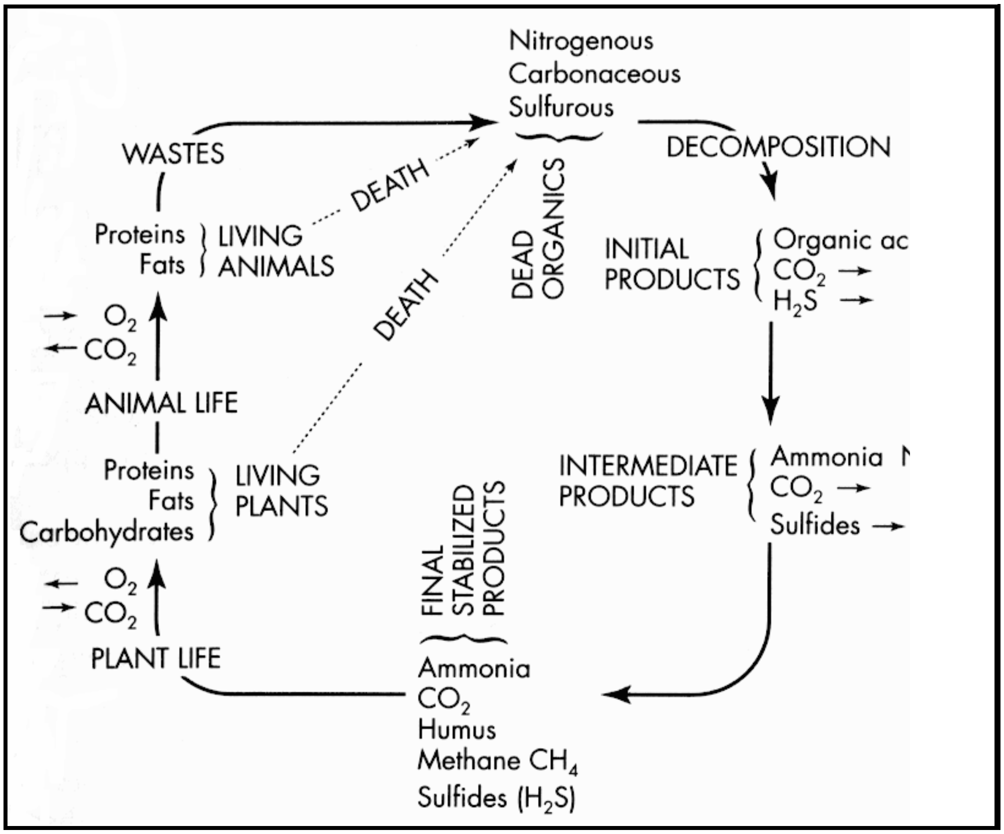

Figure 2.2 – Anaerobic Nitrogen, Carbon and Sulfur Cycles (Peirce et al., 1997)

2.2.1. Decomposition Phases

It is well known that in conventional landfills, MSW decomposition or biodegradation occurs in 5 chronological phases (Palma, 1995; Carvalho, 1999; EPA, 2000; Waste Management, 2002):

- <u>Phase I (phase of aerobic decomposition)</u>: it starts immediately after waste disposal and in the presence of oxygen, where microorganisms decompose the organic matter into carbon dioxide, water, and partially degraded organic materials. The temperature of the waste mass increases due to biological activity (heat generation) and high CO_2 levels during this phase. The aerobic decomposition phase can last from a few hours to one week for conventional landfills.

- Phase II (transition phase or hydrolytic phase): it corresponds to the transition of the aerobic to the anaerobic phase. As oxygen is depleted, decomposition caused by facultative anaerobic microorganisms predominates (Qasim and Chiang, 1994). The reaction between water and carbon dioxide generated in the previous phase produces large amounts of volatile fatty acids such as acetic acid. These acids reduce the pH, and the low pH helps to solubilize inorganic materials, which along with the high concentrations of volatile acids produce a high ionic strength. The high concentrations of volatile acids also contribute to the high level of the chemical oxygen demand (COD) often found during this phase. In this phase, nitrogen gas is depleted, cellulose begins to be broken down, methane gas (CH_4) begins to be formed, and CO_2 levels decline. High strength leachate is generally generated in this phase.

- Phase III (acid anaerobic phase or acetogenic phase): this phase is processed under the action of microorganisms that, with oxygen consumption, become dominant in the landfill. In this phase, high concentrations of organic acids, ammonia, hydrogen, and carbon dioxide are produced, as well as heat generation. The pH decreases due to the multiplication of acetogenic microorganisms. In some cases, the low pH of the environment mobilizes metal species from the waste into the leachate. The second and third phases last about one to six months in conventional landfills.

- Phase IV (methanogenic anaerobic phase): in this phase the population of methane producing bacteria builds up and becomes dominant. With the increase in methanogenic bacteria, more acetic acid is converted into methane, increasing the pH (methanogenic bacteria are strictly anaerobes and require neutral pH (6.6 to 7.3) - Qasim and Chiang, 1994). The redox potential decreases and nitrates are reduced to ammonia. The methane concentration generated is approximately 50 to 60 percent by volume, and carbon dioxide is also produced in large quantities. This phase lasts 8 to 40 or more years for conventional landfills.

- Phase V (post-methanogenic phase): this phase presents the decline in methane generation and other landfill gases production in general, stable concentrations of leachate constituents and the very slow degradation of materials remaining in the landfill.

Figure 2.3 presents the typical landfill evolution sequence in terms of leachate and gas composition, according to Yuen (1999). Figure 2.4 shows generalized evolution curves for pH, gas composition, volatile acids, conductivity, and oxidation/reduction potentials, according to Qasim and Chiang (1994).

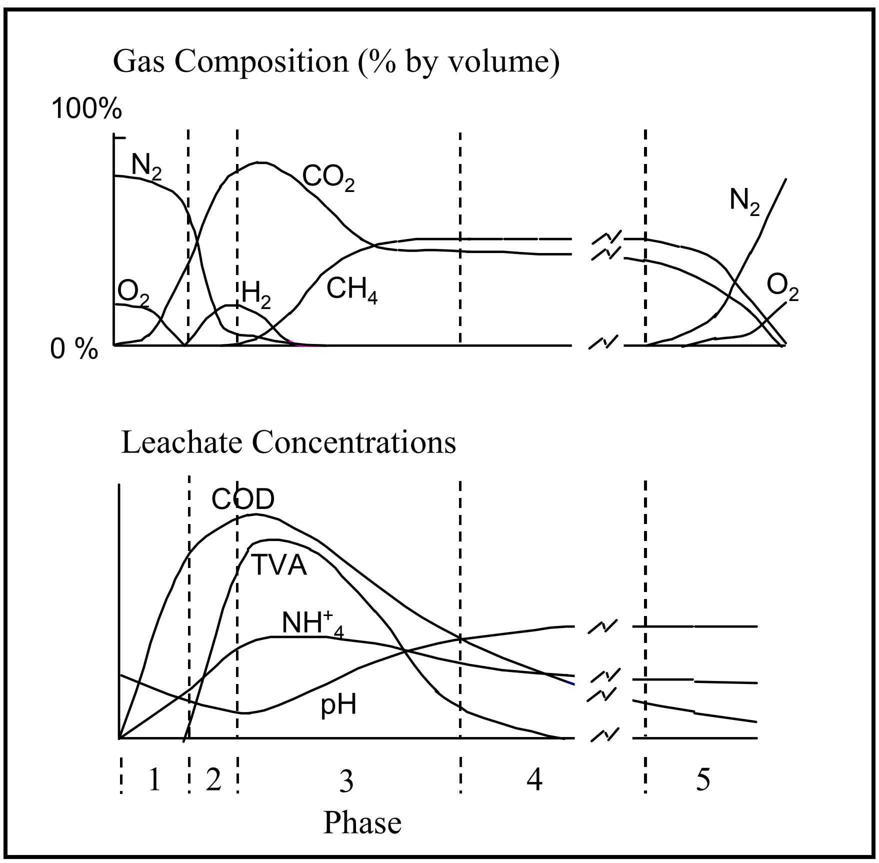

Figure 2.3 – Schematic of Theoretical Curves for Evolution of Leachate and Gas Components Concentrations (Yuen, 1999)

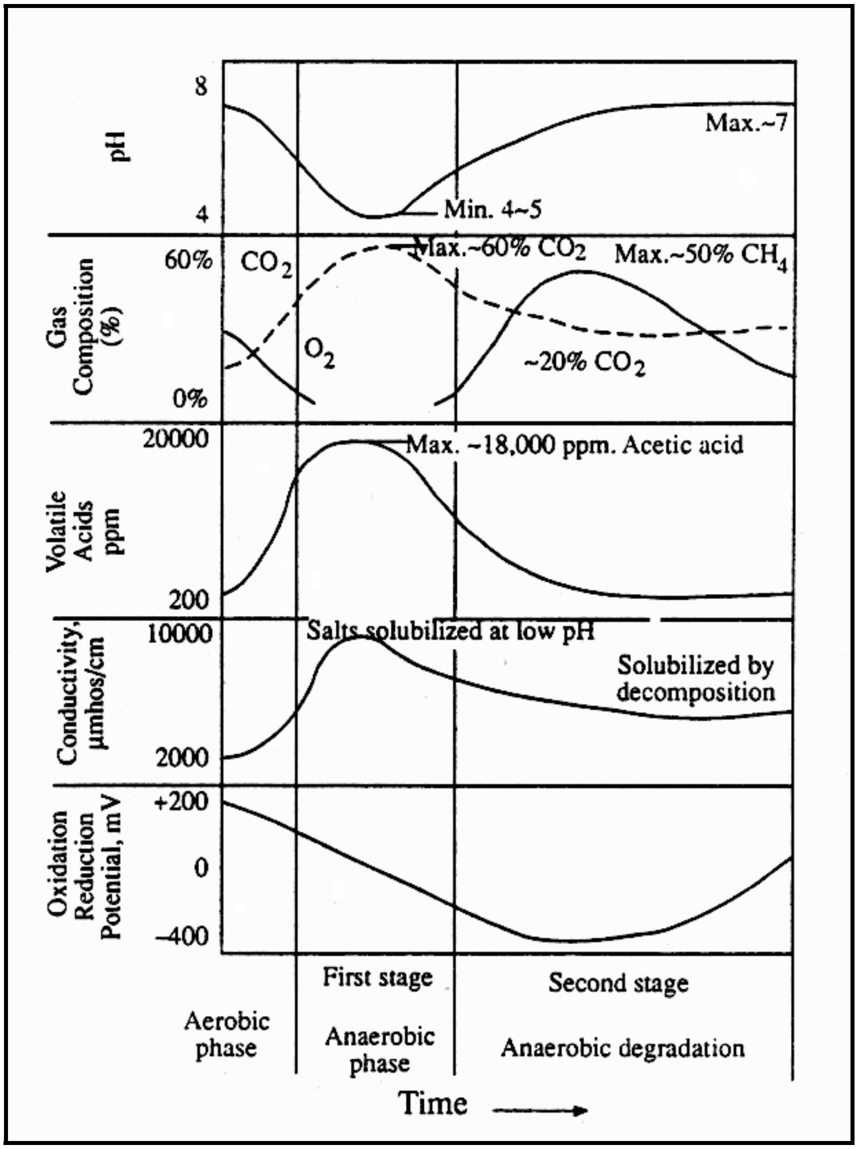

Figure 2.4 - Generalized Evolution Curves for pH, Gas Composition, Volatile Acids, Conductivity, and Oxidation/Reduction Potentials (Qasim and Chiang, 1994)

2.2.2. Influencing Factors

Regarding the factors that can affect MSW degradation, several studies present the optimum conditions and possible inhibitors in landfills. Table 2.1 presents a summary of influencing factors on landfill degradation, based on Yuen (2001).

Table 2.1 - Summary of Influencing Factors on Landfill Degradation (Yuen, 2001; modified)

Influencing factors	Criteria / Comments
Moisture	Optimum moisture content : 60% and above (by wet mass)
Oxygen	Presence of oxygen reduces the activities of anaerobic bacteria and enhances the activities of aerobic bacteria
pH	Optimum pH for methanogenesis: 6 to 8
Alkalinity	Optimum alkalinity for methanogenesis : 2000mg/l Maximum organic acids concentration for methanogenesis : 3000mg/l Maximum acetic acid/alkalinity ratio for methanogenesis : 0.8
Temperature	Optimum temperature for methanogenesis : 34-41^{o}C
Hydrogen	Partial hydrogen pressure for acetogenesis: Below 10^{-6} atm.
Nutrients	Generally adequate in most landfill except local systems due to heterogeneity
Sulphate	Increase in sulphate decreases methanogenesis
Inhibitors	Cation concentrations producing moderate inhibition (mg/ l) : Sodium 3500-5500 Potassium 2500-4500 Calcium 2500-4500 Magnesium 1000-1500 Ammonia (total) 1500-3000 Heavy metals : No significant influence Organic compounds: Inhibitory only in significant amount

2.2.3. Waste Stabilization Techniques

Another type of classification used is based on the process technique used for stabilization (Yuen, 2001). The techniques can be operationally grouped under the following:

- <u>Shredding of Waste</u>: The shredding of waste theoretically may help to homogenize by size reduction and mixing, increase the specific surface area of the waste components for biodegradation, and increase the permeability by reducing impermeable materials and making easier the distribution of water. On the other hand, shredding of waste can have a negative effect on degradation by promoting excessive initial hydrolysis and acid formation, inhibiting methanogenesis. Yuen (1999) points out that shredding can be beneficial only if the excessive initial hydrolysis can be controlled by pH adjustment. Tittlebaum (1982), in his study, did not observe any effect on biological stabilization of waste by shredding the waste as well.

- pH Adjustment: Whenever the pH environment of a landfill becomes low, caused by the production of acids, the growth of methanogenic bacteria is inhibited. The results of some experiments (Christensen et al., 1992 as reported by Yuen, 2001; Tittlebaum, 1982; Reinhart and Townsend, 1998) suggest that the addition of a buffer has a positive influence for waste stabilization in those cases.

- Sewage Sludge Addition: In theory, the addition of sewage sludge may promote waste decomposition by increasing the availability of moisture, nutrients, and microorganisms in the waste mass. However, studies showed that when methanogenic conditions are already established, the addition of sewage sludge does not bring any beneficial effect, and in some cases presents a negative effect if the sludge has a low pH (Leckie and Pacey, 1979).

- Enzyme Addition: The addition of enzymes produced by fermentative bacteria can control the hydrolysis process in landfills operating under anaerobic conditions. Lagerkvist and Chen (1993), as reported by Yuen (2001), suggested that it is viable to intensify acidogenic and methanogenic conditions by enzyme addition. In their study, laboratory-scale lysimeters are used to investigate the effect of adding industrial cellulolytic enzymes to MSW during the two phases.

- Air/oxygen Injection: This process corresponds to the "controlled injection of moisture and air into the waste mass through a network of horizontal and/or vertical pipes" (Campman and Yates, 2002). The process is analogous to wet composting operations in which biodegradable materials are rapidly decomposed using air, moisture, and increased temperatures created by biodegradation.

- Leachate Recirculation: This is by far the most utilized process for waste stabilization. Recirculation can be done with raw leachate or treated leachate. As stated before, it promotes the active microbial degradation by providing optimum moisture, provides a mechanism for the effective transfer of microbes and substrates, supplies nutrients throughout the refuse mass, and dilutes local high concentrations of inhibitors (Yuen, 2001). This process will be detailed further in subsequent paragraphs.

2.2.4. Leachate Characterization

MSW landfills contain biological materials and chemical compounds in solution with highly variable qualities. The leachate generated by those facilities is characterized by larger concentrations of pollutants when compared to raw sewage or many industrial wastes. However, it is interesting to note that prior to 1965 very few people were aware that water passing through a solid waste in a sanitary landfill would become highly contaminated (Qasim and Chiang, 1994). This polluted water, the leachate, was generally not a matter of concern because few cases of water pollution were noted where leachate had caused harm. Now it is known that MSW landfills may be an important source of groundwater pollution.

Since the release of constituents from solid wastes is associated with the decomposition processes, leachate can be an indicator of the state of MSW stabilization. The characteristics of MSW leachate are highly variable and the factors that generally affect its quality are: MSW composition, depth, degree of compaction, age of landfill, stage of landfill stabilization, rate of infiltration, moisture content, temperature, and landfill design and operation. However, the results can be affected due to sampling procedures, sample preservation, sample handling and storage, and the analytical methods used.

Tables 2.2, 2.3, and 2.4 present the general composition ranges of several parameters for leachate generated by MSW conventional landfills according to different authors. Table 2.5 presents the general composition ranges for MSW landfill leachate as a function of the degree of landfill stabilization. Tables 2.6 and 2.7 compare the MSW leachate composition between conventional landfills and landfills with leachate recirculation.

Table 2.2 – Composition of MSW Landfill Leachate According to Tchobanoglous et al., 1993

Parameter	Value (mg/L) [(a)]		
	New landfill (less than 2 years)		Mature landfill (greater than 10 years)
	Range[(b)]	Typical[(c)]	Range[(b)]
BOD_5 (5-day biochemical oxygen demand)	2,000 – 30,000	10,000	100-200
TOC (total organic carbon)	1,500 – 20,000	6,000	80 – 160
COD (chemical oxygen demand)	3,000 – 60,000	18,000	100 – 500
TSS (total suspended solids)	200 – 2,000	500	100 – 400
Organic nitrogen	10 – 800	200	80 – 120
Ammonia nitrogen	10 – 800	200	20 – 40
Nitrate	5 – 40	25	5 – 10
Total phosphorus	5 – 100	30	5 – 10
Ortho phosphorus	4 – 80	20	4 – 8
Alkalinity as $CaCO_3$	1,000 – 10,000	3,000	200 – 1,000
pH	4.5 – 7.5	6	6.6 – 7.5
Total hardness as $CaCO_3$	300 – 10,000	3,500	200 – 500
Calcium	200 – 3,000	1,000	100 – 400
Magnesium	50 – 1,500	250	50 – 200
Potassium	200 – 1,000	300	50 – 400
Sodium	200 – 2,500	500	100 – 200
Chloride	200 – 3,000	500	100 – 400
Sulfate	50 – 1,000	300	20 – 50
Total iron	50 – 1,200	60	20 – 200

(a) Except pH, which has no units.
(b) Representative range of values. Higher maximum values have been reported in the literature for some of the constituents.
(c) Typical values fro new landfills will vary with the metabolic state of the landfill

Table 2.3 – Composition of MSW Landfill Leachate According to Chian and DeWalle (1976, 1977) as reported by Qasim and Chiang (1994)

Parameter	Age of Landfill		
	1 year	5 years	16 years
BOD	7,500 – 28,000	4,000	80
COD	10,000 – 40,000	8,000	400
pH	5.2 – 6.4	6.3	
TDS	10,000 – 14,000	6,794	1,200
TSS	100 – 700		
Specific conductance	600 – 9,000	-	
Alkalinity ($CaCO_3$)	800 – 4,000	5,810	2,250
Hardness ($CaCO_3$)	3,500 – 5,000	2,200	540
Total phosphorus	25 – 35	12	8
Ortho phosphorus	23 – 33	-	
Ammonia nitrogen	56 – 482		
Nitrate	0.2 – 0.8	0.5	1.6
Calcium	900 – 1,700	308	109
Chloride	600 – 800	1,330	70
Sodium	450 – 500	810	34
Potassium	295 – 310	610	39
Sulfate	400 – 650	2	2
Manganese	75 – 125	0.06	0.06
Magnesium	160 – 250	450	90
Iron	210 – 325	6.3	0.6
Zinc	10 – 30	0.4	0.1
Cooper	-	<0.5	<0.5
Cadmium	-	<0.05	<0.05
Lead	-	0.5	1

Note: All values in mg/L, except specific conductance measured as $\mu\Omega$/cm and pH (no units)

Table 2.4 – Composition of MSW Landfill Leachate According to Keenan et al. (1983) as reported by Qasim and Chiang (1994)

Parameter	Year 1	Year 2	Year 3	Overall
BOD_5	4,460	13,000	11,359	10,907
COD	11,210	20,032	21,836	18,533
TSS	1,994	549	1,730	1,044
TDS	11,190	14,154	13,181	13,029
pH	7.1	6.6	7.3	6.9
Alkalinity ($CaCO_3$)	5,685	5,620	4,830	5,404
Hardness ($CaCO_3$)	5,116	4,986	3,135	4,652
Calcium	651	894	725	818
Magnesium	652	454	250	453
Phosphate	2.8	2.6	3.0	2.7
Ammonia nitrogen	1,966	724	883	1,001
TKN	1,660	760	611	984
Sulfate	114	683	428	462
Chloride	4,816	4,395	3,101	4,240
Sodium	1,177	1,386	1,457	1,354
Potassium	969	950	968	961
Cadmium	0.04	0.09	0.10	0.09
Chromium	0.16	0.43	0.22	0.28
Copper	0.44	0.39	0.32	0.39

Table 2.4 – Composition of MSW Landfill Leachate According to Keenan et al. (1983) as reported by Qasim and Chiang (1994) – cont.

Parameter	Year 1	Year 2	Year 3	Overall
Iron	245	378	176	312
Nickel	0.53	1.98	1.27	1.55
Lead	0.52	0.81	0.45	0.67
Zinc	8.7	31	11	21
Mercury	0.007	0.005	0.011	0.007

Note: All values in mg/L, except pH (no units)

Table 2.5 – Composition of MSW Landfill Leachate as a Function of the Degree of Landfill Stabilization (Pohland, 1986 as reported by Reinhart and Townsend, 1998)

Parameter	Phase II Transition	Phase III Acid Anaerobic	Phase IV Methanogenic Anaerobic	Phase V Post-Methanogenic
BOD (mg/L)	100 – 10,000	1,000 – 57,000	600 – 3,400	4 – 120
COD (mg/L)	480 – 18,000	1,500 – 71,000	580 – 9,760	31 – 900
TVA (mg/L)	100 – 3,000	3,000 – 18,800	250 – 4,000	0
BOD/COD	0.23 – 0.87	0.4 – 0.8	0.17 – 0.64	0.02 – 0.13
Ammonia Nitrogen (mg/L)	120 – 125	2 – 1,030	6 – 430	6 – 430
pH	6.7	4.7 – 7.7	6.3 – 8.8	7.1 – 8.8
Conductivity ($\mu\Omega$/cm)	2,450 – 3,310	1,600 -17,100	2,900 – 7,700	1,400 – 4,500

Table 2.6 – Composition of MSW Landfill Leachate for Conventional and Bioreactor Landfills with Leachate Recirculation (Reinhart and Townsend, 1998)

Parameter	Conventional Landfill	Landfill with Leachate Recirculation
BOD (mg/L)	20 – 40,000	12 – 28,000
COD (mg/L)	500 – 60,000	20 – 34,560
Iron (mg/L)	20 – 2,100	4 – 1,095
Ammonia (mg/L)	30 – 3,000	6 – 1,850
Chloride (mg/L)	100 – 5,000	9 – 1,884
Zinc (mg/L)	6 – 370	0.1 – 66

Table 2.7 – Composition of MSW Landfill Leachate as a Function of the Degree of Stabilization for Conventional and Bioreactor Landfills with Leachate Recirculation (Reinhart and Townsend, 1998)

Parameter	Phase II Transition		Phase III Acid Anaerobic		Phase IV Methanogenic Anaerobic		Phase V Post-Methanogenic	
	Conventional	With leachate recirculation	Conventional	With leachate recirculation	Conventional	With leachate recirculation	Conventional	With leachate recirculation
BOD (mg/L)	100-10,000	0-6,893	1,000-57,000	0-28,000	600-3,400	100-10,000	4-120	100
COD (mg/L)	480-18,000	20-20,000	1,500-71,000	11,600-34,550	580-9,760	1,800-17,000	31-900	770-1,000
TVA (mg/L)	100-3,000	200-2,700	3,000-18,800	1,-30,730	250-4,000	0-3,900	0	-
BOD/COD	0.23-0.87	0.1-0.98	0.4-0.8	0.45-0.95	0.17-0.64	0.05-0.8	0.02-0.13	0.05-0.08
Ammonia Nitrogen (mg/L)	120-125	76-125	2-1,030	0-1,800	6-430	32-1,850	6-430	420-580
pH	6.7	5.4-8.1	4.7-7.7	5.7-7.4	6.3-8.8	5.9-8.6	7.1-8.8	7.4-8.3
Conductivity ($\mu\Omega$/cm)	2,450-3,310	2,200-8,000	1,600-17,100	10,000-18,000	2,900-7,700	4,200-16,000	1,400-4,500	-

Reinhart and Grosh (1998) made a comprehensive study of the leachate generated by 55 landfills that receive MSW and non-hazardous industrial wastes in Florida, trying to characterize Florida landfill leachate. From the study the authors conclude that:

- In general, the Florida climate (with heavy rainfalls and warm temperatures) appeared to produce dilute leachate, with concentrations at relatively low levels compared to literature values.

- Biochemical Oxygen Demand (BOD) and COD concentrations remained low (less than 1500 mg/L) throughout the life of the landfills, most likely due to dilution and stimulation of methanogenesis. No clearly determined chronological pattern in BOD and COD concentrations was observed.

- Leachate from shredded waste fill had a significantly higher concentration of organic pollutants (evidenced in the high COD and BOD levels) than leachate from unshredded waste landfills.

- A wide variety of toxic and organic compounds could be found in the Florida leachate, however, with concentrations on the order of micrograms per liter.

- Codisposal of ash with MSW did not appear to adversely impact leachate quality.

2.3. Small-Scale Bioreactor Landfill Studies

Several small-scale studies investigating the effects on leachate quality, waste stabilization, waste settlement, and gas production using bioreactor landfills are reported in the literature. As pointed out previously, moisture content, pH, temperature, availability of nutrients, and the presence of microorganisms are some of the parameters that influence MSW stabilization, and can be manipulated easier in the laboratory.

The following are some studies conducted to study bioreactor landfills in the laboratory, as presented in chronological order.

2.3.1. Pohland (1975)

Pohland (1975) studied the effect of raw leachate recirculation on waste stabilization by using four test columns, each 0.9 m (3 ft) in diameter filled with 3 m (10 ft) of compacted MSW. The first column was the control cell (no leachate recirculation). The second column was subjected to simple leachate recirculation. The third column was operated with leachate recirculation and using NaOH for pH control (to maintain the pH close to 7). The fourth column was operated with leachate recirculation, NaOH addition, and initial seeding utilizing wastewater sludge. Water was added to the four columns to produce leachate immediately. Samples were collected at regular intervals for 1100 days, the duration of the experiment.

For control column, the COD increased very quickly to 19,000 mg/L in 200 days and gradually decreased to a level of 4,000 mg/L after 1000 days. Total volatile acids (TVA) concentrations also had a peak after 200 days with a concentration of approximately 10,000 mg/L and decreased after that to a 2,000 mg/L level. The pH ranged from 5.0 to 6.5 throughout the duration of the experiment.

For the second column (with leachate recirculation only), the COD had an earlier peak than the control cell of 11,000 mg/L in about 100 days, then a gradual decrease to 250 mg/L at day 500, where it remained. TVA peaked after 200 days with a value of 6,000 mg/L, then declined to a value of 200 mg/L at day

700. The pH was around 5 for the first 200 days, increasing to 7 approximately after 500 days.

The third column (with leachate recirculation and pH control) presented peaks of COD and TVA of 10,000 mg/L and 5,000 mg/L, respectively, after 150 days, and then declined more quickly than the second column to values of less than 500 mg/L for COD and 250 mg/L for TVA after 200 days. The pH remained neutral due to buffer addition during all times. The fourth column (with wastewater sludge addition did not show an acceleration of the MSW degradation process when compared to the third column.

This was the first experiment showing that the recirculation of leachate can accelerate the MSW degradation process, by demonstrating a more rapid decrease in the COD and TVA concentrations when compared to the dry cell.

2.3.2. Leckie and Pacey (1975)

Leckie and Pacey (1975) investigated the quality of leachate generated under three different conditions: 1) with raw leachate recirculation, 2) under a continuous flow of water, and 3) adding sewage sludge.

The study showed that by using the leachate recirculation technique and by continuously flushing the waste with water, the MSW decomposition process was accelerated. However, the authors discarded this technique for full-scale landfills, pointing out that by flushing the waste continuously with water, large quantities of leachate were generated.

The addition of sewage sludge did not accelerate the MSW decomposition and had a negative effect on the development of the methanogenic phase. The added sewage sludge had a low pH, suppressing the buffer capacity of the system due to the acceleration of the acidic phases.

2.3.3. Tittlebaum (1982)

Tittlebaum's study was conducted utilizing four 0.9 m (3 ft) diameter test cells filled with 2.4 m (8 ft) of compacted MSW. The first cell was the control cell with unshredded waste. The second cell was composed by shredded waste and operated with pH control, using NaOH. The third cell was operated similar to the second cell, however, by utilizing unshredded waste. The fourth cell was similar to the third cell, however by operating with nutrient control (with phosphorus and nitrogen added). Water was added to the second, third, and fourth cells to accelerate leachate generation, and also was added on a regular basis with a value equivalent to the total daily rainfall. The leachate was analyzed periodically for approximately 500 days.

The study showed that shredding and nutrient control processes did not accelerate the stabilization of waste. Also, the leachate recirculation with pH control significantly reduced the COD, the BOD and the total organic carbon (TOC) values, and accelerated the MSW biological stabilization process.

2.3.4. Robinson et al. (1982)

Yuen (1999) discusses a study conducted by Robinson et al. (1982) using leachate recirculation in two cells operated under different conditions. The first cell was operated with leachate recirculation only. The second one was operated with leachate recirculation, but prior to recirculation, the leachate collected was aerated and phosphate was added as nutrient.

The study showed that in the first cell the COD, ammonia, and chloride concentrations were high at the beginning, but stabilized to a "reasonably constant level in 12 months", according to Yuen (1999). In the second cell, the concentration values were generally lower, but fluctuated over a long period.

2.3.5. Hartz and Ham (1983)

Hartz and Ram (1983) investigated the rate of methane production as a function of moisture content. The study suggested that for values of moisture content lower than 10%, there is no methane production. Field capacity was reached at a 40% moisture content level, and for lower levels there is no available free moisture for recirculation. The study also concluded that a continuous moisture flow produces methane rates about 10 times higher than under a more inert condition.

2.3.6. Mata-Alvarez and Martinez-Viturtia (1986)

This study investigated the effects of temperature on MSW decomposition, also using leachate recirculation, shredded waste, and pH control. Five tests were made with temperature ranging between 30°C and 46°C. Some conclusions of the study were:

- The optimum temperatures that lead to maximum MSW degradation rates are in the 34°C - 38°C range.
- Maximum gas production rates were obtained at pH values of 7.5.
- The pH adjustment was very important in the start-up of the methanogenic process.
- By extrapolation, the authors suggested that biodegradable matter in a real landfill, operating under the same conditions simulated in the study, should be stabilized in about two years.

2.3.7. Stegmann and Spendlin (1986)

Stegmann and Spendlin (1986), as reported by Yuen (1999), presented the use of lysimeters tests to investigate the benefits of leachate recirculation, leachate recirculation with pH control, mixing MSW with pre-composted waste prior to final disposal, and addition of sewage sludge. The study revealed that mixing MSW with pre-composted waste prior to final disposal had a positive effect in reducing leachate strength. The findings of other tests were not conclusive.

2.3.8. Kinman et al. (1987)

In this study, sixteen cells running at different conditions were used to evaluate gas generation. The parameters studied by the authors are: moisture flow, elevated moisture content, leachate recirculation, buffer addition, nutrient addition, anaerobic digested sludge addition, and elevated temperature.

One of the interesting findings of the study is that leachate recirculation alone sometimes can overstimulate the acidic phases, which produces a negative effect on the subsequent methanogenesis phase. Adjusting the pH and bringing the cell to more neutral conditions enhance the production of methane and reduce the leachate strength according to the authors.

2.3.9. Barlaz et al. (1987)

Leachate recirculation operating under several different conditions was studied by the authors with 1) pH control, 2) the addition of acetate, 3) the addition of old degraded waste, 4) the addition of aerobically digested sludge, and 5) the use of a sterile soil cover.

The use of leachate recirculation with pH control promoted a large production of methane when compared to the other conditions of the experiment. No beneficial positive effect was observed for the cells operated either with the addition of acetate or with the addition of sludge.

The use of old degraded waste enhanced the generation of methane. According to the authors, old waste can be used as an effective seeding of anaerobic bacteria.

The use of a sterile soil cover did not inhibit the production of methane. This suggests that soil covers are not the only source of methanogenic bacteria, as previously believed.

2.3.10. Leuschner (1989)

Leachate recirculation benefits were studied using six lysimeters running at different conditions: 1) no recirculation; 2) recirculation only; 3) recirculation with pH control; 4) recirculation with pH control and addition of nutrients; 5) recirculation with pH control, addition of nutrients, and anaerobically digested sludge; and 6) recirculation with pH control, addition of nutrients, and septic tank sludge.

The study compared the lysimeters with no recirculation and with recirculation only, and showed that hydrolysis and acid fermentation were stimulated by recirculation, but methanogenesis was not. The author pointed out that the natural buffer capacity of the MSW was unable to overcome the drop in pH. A mathematical model developed in the study presented 19.3 years as the time necessary to reach 80% degradation for the control cell. The time to reach the same level of degradation for the cell with recirculation would be only 10.2 years.

The lysimeter with leachate recirculation and pH control presented a rapid production of methane. The time estimated to reach 80% degradation in this case would be 4.2 years.

Also, the lysimeters with the addition of nutrients and the addition of nutrients plus anaerobically digested sludge presented very good results in accelerating MSW decomposition, with 80% degradation times of 7.8 years and 3.2 years, respectively.

The lysimeter tested with the addition of septic tank sludge had a very poor performance, leading to the conclusion that this technique is not suitable to enhance MSW decomposition due to its low pH nature.

2.3.11. Doedens and Cord-Landwehr (1989)

Test cell experiments on leachate recirculation were conducted using four cylindrical test cells, each 1.5 m in diameter, filled with 1.35 m of compacted shredded waste. The initial moisture content was 24% to 31%, and all cells were air tight, temperature controlled (35^{o}C), and equipped with a leachate redistribution system.

The first cell received rainwater equivalent to 660 mm/yr, and all other cells received 330 mm/yr of simulated precipitation. The first cell had straight leachate recirculation, the second one was the control cell (no recirculation), the third one had straight leachate recirculation as the first one (but half of simulated precipitation), and the fourth one was initially brought to field capacity with leachate from a stabilized landfill, then received rainwater in addition to all of the leachate generated.

The first cell (with straight leachate recirculation and double the rainwater rate) presented a more rapid decline and lower values in COD concentration than any other cell, while the second cell (control cell) took the longest time to decline and had the highest COD concentration. Approximately the same COD concentration was obtained for the third and fourth cells, which were higher than the first cell and lower than the second cell.

2.3.12. Otieno (1989)

Reinhart and Townsend (1998) discussed Otieno's (1989) investigation of the effect of leachate recirculation for different types of waste under different conditions of operation. Four lysimeters, each 0.5 m in diameter, were filled with different types of waste with different densities.

The first lysimeter was filled with fresh MSW with a moisture content of 61% and a density of 383 kg/m^3. The second lysimeter was filled with the same type of waste and moisture content of the first lysimeter, but with a density of 418 kg/m^3 and operated under saturated conditions. The third lysimeter consisted of shredded MSW with moisture content of 44% and a density of 306 kg/m^3, while the fourth one consisted of aged MSW with a moisture content of 85% and a density of 550 kg/m^3.

The main conclusions obtained from the study are: 1) shredding increases the degradation rate; 2) lower density helps to increase the degradation rate; and 3) operation under saturation does not have any positive benefit to waste degradation and increases the strength of leachate.

2.3.13. Pohland et al. (1992)

Pohland et al. (1992), as reported by Reinhart and Townsend (1998), investigated the effect of leachate recirculation and single-pass operation on several inorganic and organic pollutants codisposed with shredded MSW, through the use of 10 columns, each 0.9 m (3ft) in diameter and 3 m (10 ft) in height. The characteristics of each column are presented on Table 2.8.

Table 2.8 – Column Loading Characteristics (Reinhart and Townsend, 1998)

Column	Operation	Initial Loading Height (cm)	Compact Density (kg/m^3)	Inorganic Pollutant Added [(a)]	Organic Pollutant Added [(b)]
1	Recirculation	29	313	None	None
2	Single pass	30	301	None	None
3	Single pass	29	309	None	Yes
4	Single pass	28	327	Low	Yes
5	Single pass	30	305	Medium	Yes
6	Recirculation	28	317	None	Yes
7	Recirculation	29	309	Low	Yes
8	Single pass	30	305	High	Yes
9	Recirculation	29	313	Medium	Yes
10	Recirculation	31	293	High	Yes

(a) Low: Cd=35 g; Cr=45 g; Hg=20 g; Ni=75 g; Pb=105 g; Zn=135 g
Medium: low doubled
High: medium doubled

(b) 120 g of 12 different organic compounds

The main conclusions obtained from the study are:

- Organic pollutants impacted negatively the stabilization process in the columns with recirculation, and totally inhibited it for single-pass columns.

- Both recirculating and single pass columns were capable of having decomposition under the effect of heavy metals, however, with recirculating columns showing a better capacity.

- Leachate recirculation resulted in an efficient conversion to gas of many organic constituents that otherwise would be washed out.

2.3.14. Woelders et al. (1993)

Woelders et al. (1993), as reported by Yuen (1999), investigated the effects of water infiltration and leachate recirculation rates in pre-treated (mechanically separate organic residue derived from MSW) waste decomposition through the use of three column cells. The first column was operated with water at a rate equivalent to natural infiltration. The second column was operated with leachate recirculation at twice the rate of the first column. The third column was also operated with leachate recirculation, but with a recirculation at 5 times the rate of the first column. The leachate recirculation in the second and third columns was initiated with methanogenic leachate from an old landfill.

The use of methanogenic leachate resulted in an early onset of methanogenesis for the second and third columns. The third column exhibited better results than the second column, suggesting that a higher recirculation rate promoted a more efficient system. The first column, without initial methanogenic bacteria seeding, had the production of gas initiated only at a later stage and after addition of a buffer.

2.3.15. Chugh (1996)

Chugh (1996), as reported by Yuen (1999), investigated the effect of a "Sequential Batch Anaerobic Reactor" (SBAR), involving the exchange process of leachate between a fresh cell and an anaerobically stabilized cell. The methanogenic leachate obtained from the stabilized cell is recirculated in the fresh cell, initiating the degradation process and flushing out organic acids that can inhibit it. Then the leachate generated by the fresh cell is injected back into the stabilized cell, where the established microbial population can convert the organic acids to methane. The leachate recirculation sequence continues until the cell is stabilized.

The study showed that the described process converted approximately 80% of the degradable organic portion of MSW to methane in 60 days. The study also demonstrated that low recirculation leachate rates could also begin the degradation of a fresh waste cell successfully. Yuen (1999) pointed out that lower leachate recirculation volume can be more adequate to full-scale bioreactor landfills.

2.3.16. Karnchanawong and Noythachang (1996)

The effects of leachate recirculation on leachate and gas characteristics produced from anaerobic and semi-aerobic landfill simulators were investigated in this study through the use of eight lysimeters, each 0.15 m in diameter and 1.8 m in height. Four different rates of leachate recirculation were applied. The wastes were prepared from selected materials and wastes to simulate MSW. Water was added daily to each lysimeter to simulate the rainfall.

The results showed that leachate recirculation enhanced the rate of the biodegradation process in both anaerobic and semi-aerobic lysimeters. The recirculation rate corresponded to 20% of daily rainfall and produced the lowest concentrations of pollutants for the anaerobic lysimeter. The recirculation rate corresponding to 10% of the daily rainfall produced the lowest concentration of pollutants for the semi-aerobic lysimeter.

The authors pointed out that leachate recirculation caused the higher amount of leachate when compared to the non-recirculation one.

2.4. Pilot Studies and Full-Scale Experiences with Bioreactor Landfills

A number of pilot and full-scale experiences with bioreactor landfills are described in the literature for investigating the effects of leachate recirculation on landfill stabilization, leachate quality, landfill gas production, and other parameters. Some of these studies are discussed below as presented by Reinhart and Townsend (1998), Yuen (1999), and US EPA (2002).

2.4.1. Pilot Scale Studies

2.4.1.1. Sonoma County Landfill, California

This pilot-study started in 1972 at Sonoma County using five large MSW cells (15 m by 15 m by 3 m deep) to investigate the effect of moisture on the rate of MSW stabilization. Cell A was the control cell (dry cell), Cell B was initially brought to field capacity using water, Cell C received water at a rate of 3.8 m^3/day, Cell D had leachate recirculation at rates that varied from 1.9 to 19 m^3/day, and Cell E initially had the addition of septic tank pumpings. Cells A, B, and E received additional moisture only from infiltrating rain.

Based on the performance of the five cells, the following conclusions could be drawn:

- Leachate recirculation in Cell D provided a more rapid decrease in BOD, COD, and TVA concentrations than any other cell. Also, the gas composition indicated an increased rate of biological stabilization.

- Larger settlements were obtained in Cell D when compared to the other cells. A 20% settlement was obtained for this cell, while an 8% settlement was obtained for the other cells.

- Although Cells C and D enhanced the MSW stabilization, the large volumes of leachate generated required *ex situ* treatment.

- The addition of septic tank pumpings increased the rate of acid fermentation, but was not beneficial without pH control and leachate recirculation.

2.4.1.2. Georgia Institute of Technology Study, Georgia

The objective of the study conducted at the Georgia Institute of Technology was to investigate the effects of leachate recirculation on MSW degradation. In August 1976, two simulated landfill concrete cells, each 3m (10 ft) by 3 m (10 ft) by 4.3 m (14 ft) deep were constructed and filled with 3 m of compacted shredded MSW. The first cell was left open to received rainfall directly, and the second one was sealed to permit gas collection and eliminate evaporation, but tap water was added equivalent to the rainfall received by the first cell.

After approximately 1 year, field capacity was achieved and leachate generation started in both cells. Leachate samples were collected from both cells at regular intervals and analysed for various parameters, including BOD, COD, TOC, TVA, pH, phosphorus, chloride, and some metals. Gas samples were analysed for methane, carbon dioxide, hydrogen, and nitrogen. The recirculation of leachate started for both cells 208 days after leachate production.

Some of the findings of the study were:

- Concentrations of COD, BOD, and TOC were generally lower in the sealed cell than in the open cell, with not much difference at the end of the test (520 days after leachate production started).

- The sealed cell provided a better environment for methanogenic bacteria growth since it excluded the presence of oxygen.

- Based on the difference of chloride concentrations between the two cells, moisture loss due to evaporation was estimated between 20 to 30% of the incident rainfall.

2.4.1.3. Seamer Carr Landfill, England

This study, started in 1980, had, as the objective, to investigate the benefits of leachate recirculation in shredded MSW when applied to a full-scale landfill.

A cell of 2 hectares was divided into two parts. The first part was operated as the control cell, while the second part was operated with leachate recirculation. The recirculation was done through the use of spray irrigation.

The results of the experiment showed that the benefits of leachate recirculation studied in the laboratory can be achieved also in full-scale landfills, but required a longer period of recirculation compared to small-scale studies. In the study, COD and ammonia concentrations remained high in the leachate generated until the end of the experiment, therefore, needing further treatment or dilution prior to direct disposal.

2.4.1.4. Mountain View Landfill, California

In 1981, six cells, each 30 m (100 ft) by 30 m (100 ft) by 14 m (47 ft) deep, were constructed and operated as part of this pilot study that investigated the effectiveness of the methods used to enhance methane gas generation by controlling the addition of moisture, buffer, inoculation, and leachate recirculation.

Cell F was the control cell, and Cell A was the only cell that received leachate recirculation. The other cells did not receive leachate recirculation and were operated under different conditions of moisture, buffer addition, and inoculation. The experiment was conducted for 1597 days.

The results show that leachate recirculation produced the lowest gas production rates, when compared to the other cells, in disagreement with other studies performed. In general, the cells with higher moisture content, sludge addition, less settlement, and lower internal temperatures had lower measured gas production rates. Since the cells with higher moisture infiltration had lower measured gas production rates, the study suggests that maybe moisture infiltration and gas escape might have the same pathway.

2.4.1.5. Brogborough Landfill, England

In the late 1980's, six test cells were operated under different conditions to investigate the quality and quantity of landfill gas production in this landfill located in Bedfordshire, England. The following techniques were investigated in this study: the use of low density waste, a mix of MSW with industrial/commercial waste, addition of sewage sludge, addition of water, and air injection.

Gas production, leachate production and composition, waste temperature, and waste settlement were the parameters investigated. The main conclusions of the study were:

- The use of waste with low density had little influence on the results.

- The addition of non-hazardous industrial/commercial waste to MSW accelerated the start of landfill gas production, maybe due to the natural pH buffer of the waste added, which might have contributed to make favorable conditions for methanogenesis.

- The addition of sewage sludge also promoted an early production of landfill gas, maybe due to the significant increase in moisture.

- The addition of water also increased gas production.

- Air injection also increased gas production rates. This contradicts the initial intention to create an aerobic composting activity by air injection. It was suggested that air injection forced some leachate movement within the waste, which improved the moisture distribution.

2.4.1.6. Nanticoke Landfill, New York

This pilot-scale study conducted for the New York Energy Research and Development Authority investigated the enhancement of landfill gas production with leachate recirculation by controlling parameters such as moisture content, pH (by buffer addition), temperature, and nutrients (using wastewater treatment plant sludge).

Nine pilot-scale cells, each 17 m (57 ft) by 23 m (75 ft) by 6.4 m (21 ft) deep, were constructed and equipped with leachate collection, leachate/moisture distribution, and gas collection and metering systems. The first cell was the control cell, the second cell received moisture only, the third cell received moisture and buffer (lime), the fourth cell received anaerobically digested sludge but no moisture or buffer, the fifth, sixth, and seventh cells received both sludge and buffer in varying quantities. Although nine cells were constructed, only seven were operated (the eighth and ninth cells were not operated). The cells were monitored for a period of two years.

According to the monitoring data, the cells that received the addition of sewage sludge had higher methane content and lower COD, TVA, and alkalinity concentrations in the leachate compared to the other cells. From the study, it was concluded that leachate recirculation with the addition of sewage sludge (at a rate of 0.45 kg per 115 to 160 kg of MSW) resulted in improved gas production, gas quality, and leachate quality.

2.4.1.7. Delaware Landfills Studies, Delaware

Several studies reported the use of leachate recirculation in test-cells in landfills located in Central and Southern Delaware. In the early 1980s, tests were done in three landfill cells to evaluate the use of different operational techniques on leachate recirculation. The results suggested that a combination of spray irrigation, surface infiltration, and deep injection wells was the more effective way to recirculate leachate.

In 1987, the Delaware Solid Waste Authority conducted a test cell program in association with the EPA to evaluate the efficiency of different liner and landfill systems. Two double-lined test cells were constructed and filled with MSW; four different liners were tested. The first cell was constructed and operated as a leachate recirculation cell; the drainage layer consisted of 2 feet of sand. The second cell was the dry cell; a geotextile was used for the drainage layer in association with two different collection system types (with and without piping).

Some of the aspects monitored and compared were:

- Leachate characteristics from both cells were very similar during the period of operation, with COD greater than 10,000 mg/L in the first year, decreasing to 500 to 700 mg/L in the wet cell and to 200 mg/L in the dry cell after approximately 5 to 6 years later.

- Results indicated that landfill gas generation in the wet cell was about an order of magnitude higher than in the dry cell. However, the gas characteristics were very similar between the two cells, with methane consisting of about 50% of the total gas.

- No significant deterioration of any kind was found in the four types of liners studied.

2.4.1.8. Breitenau Landfill, Austria

In 1986, three test cells 17 m deep, with 2929 m^2, 3798 m^2 and 4622 m^2 in area were constructed and filled with MSW at the Breitenau Research Landfill in Austria. The first cell was the control cell, the second cell received leachate recirculation, and the third one was filled with shredded MSW and received leachate recirculation.

From the study it was concluded that the anaerobic degradation process could be accelerated with a reduction in the methanogenic phase by leachate recirculation. However, several problems could be associated with the bioreactor operation such as: production and escape of gas before completion of landfill, leachate ponding, and leachate toxicity due to high ammonia content.

It is interesting to note that the pH varied from 6.2 to 8.3 for the second cell throughout the experiment. This upper limit of pH (8.3) is not usually noted in the literature.

2.4.1.9. Spillepeng Landfill, Sweden

Six test cells, each 35 m by 35 m by 2 to 9 m deep, filled with different waste compositions were used in the Spillepeng Landfill, Malmö, Sweden to investigate primarily the influence of waste composition in methane production. One of the cells included leachate recirculation.

The first cell was filled with 30% MSW and 70% non-hazardous industrial/commercial waste. The second cell was filled with the same composition of the first cell, but with 5% grease trap sludge also added. The third cell contained high organic composition, with selected MSW. The fourth cell was filled with 100% MSW. The fifth cell was filled with 95% MSW and 5% grease trap sludge. The sixth cell was filled with 100% MSW but included leachate recirculation.

All six cells had fractions of 50% or more in methane concentration within 6 months after completion of filling and peaked at 55% to 60% in two years. The two cells containing a large proportion of non-hazardous industrial/commercial waste produced the highest total gas quantities. No explanation was given for these observations.

Another important conclusion was that leachate recirculation did not present any benefit in the production of methane when compared to the dry cell.

2.4.1.10. Lower Spen Valley Landfill, England

The study, called “Landfill 2000 Project”, started in 1990/1991 and had as objectives an investigation into the practicability of accelerating the stabilization of MSW and verification of the possibility of re-mining the waste and re-using the engineered landfill cells.

Two cells, each 36 m by 23 m by 1.4 to 5 m deep, were filled with MSW mixed with sewage sludge (12% by wet weight). Sewage effluent (10% by volume) was added to one cell and the leachate produced was constantly recirculated into the waste mass. The other cell was the control cell, without recirculation.

Some of the findings of the study were:

- Methanogenesis was established within one year, with high gas generation rates achieved in both cells.

- The cell with leachate recirculation produced landfill gas at double the rate of the control cell.

- High methane product rates were observed even though acetogenic leachate was still detected in the unsaturated waste.

- The biochemical methane potential (BMP) measured in waste samples after 3 years showed that complete waste stabilization was not achieved for any of the two cells.

2.4.1.11. Yolo County Landfill, California

A demonstration bioreactor project was designed and started in 1993 inside a landfill opened in 1975 in Yolo County, California. Two cells, 30 m by 30 m, were constructed to investigate the effects of leachate recirculation on MSW stabilization. The research had as objectives: to demonstrate that water addition can accelerate waste decomposition and gas production, monitor biological conditions in the landfill, estimate the potential for landfill life extension, better understand moisture movement inside the landfill, assess performance of shredded tires as drainage material, and provide data to the EPA and the private sector. The two test cells (with and without leachate recirculation) had double the composite liners with leak detection, compacted clay sidewalls, manholes to collect leachate, and vertical gas collection system.

Some of the findings of this study were:

- The addition of water accelerated MSW decomposition and methane recovery.

- Significant settlement was observed after a relatively short time (within 6 months) for the cell with leachate recirculation.

- Shredded tires performed well in supporting landfill gas migration and leachate injection.

2.4.1.12. Florida Bioreactor Landfill Project, Florida

This full-scale study conducted by the University of Central Florida started in 2001 with an objective to demonstrate the full-scale use of bioreactor technology, evaluate aerobic bioreactor technology, and compare aerobic and anaerobic processes by controlling and measuring all inputs and outputs.

The landfill for this demonstration consists of three cells with composite liners. Two of the cells are operated with leachate recirculation and air injection. Leachate and air injection are conducted through well clusters on 50-foot spacings and drilled to various depths. Since 75 feet of waste were already in place at the beginning of the project, the wells were installed into the existing waste mass using direct push technology as well as air-driven rotary drill. The study evaluates the leachate quantity and quality, the landfill gas production and quality, the change of waste properties with time, and also settlement. In-place instrumentation measures the head on liner (using pressure transducers), leachate flow, landfill temperature, and moisture content (through measured resistance).

2.4.2. Full-Scale Experiences in the United States

2.4.2.1. Alachua County Southwest Landfill, Florida

The Alachua County Southwest Landfill, located in North Central Florida, receives approximately 900 metric tons per month in a 10.9-ha area lined with 1.5-mm HDPE geomembrane over 30 cm of clay. The maximum landfill depth is approximately 20 m.

Leachate recirculation started in 1990 through the use of infiltration ponds. Another system was installed in 1993 to provide direct injection into the landfill as cells were constructed. The landfill is permitted to recirculate up to 230 m^3/day of leachate.

A beneficial impact on leachate quality could be observed through the continuous pumping to trenches at high rates, not observed when infiltration ponds were being used.

2.4.2.2. Central Facility Landfill, Maryland

The Central Facility Landfill, located in Worcester County, Maryland, started to operate in 1990. The landfill is sited on a 6.9-ha area and is formed by four cells, with an estimated maximum height of 27 m. The landfill received approximately 180 metric tons/day of MSW around 1998. The cells are lined with a 1.5-mm HDPE geomebrane over a natural clayey soil.

Leachate recirculation is done using vertical discharge wells using 1.2-m diameter perforated concrete pipes. Excess leachate is transported to a local wastewater treatment facility by trucks. Since minimal off-site treatment has been required, the landfill operators have suggested that the wells have a limited area of influence and recommended modifications that would move leachate laterally away from the wells.

2.4.2.3. Winfield Landfill, Florida

The Winfield Landfill, located in Columbia County, Florida, started to operate in 1992. The double-liner system consists of a 46-cm drainage layer, 1.5-mm HDPE geomembrane, over a 46-cm clayey soil liner. Waste received is approximately 49 metric tons/day.

Leachate is collected and pumped to an aerated lagoon and then sent to a storage unit or recirculated back to the landfill. The leachate is recirculated through the use of spraying (sprinkler heads), but surface ponds were also used in the past as the recirculation method. Problems with ponding were reported as being associated with the low permeability intermediate cover used. Tire chips were incorporated to the intermediate cover to facilitate drainage; the low permeability cover is used on the slopes to reduce side seeps.

Leachate is recirculated primarily at the top of the landfill. This area is equipped with a system capable of avoiding contamination of the stormwater.

2.4.2.4. Pecan Row Landfill, Georgia

The Pecan Row Landfill, located in Lowndes County, Georgia, started to operate in 1992, and is located on a 39-ha area, with individual cells of 1.5 to 1.6 ha. The maximum waste depth planned is about 18 m. MSW is received at a rate of 540 metric-tons/day. The liner system is composed of a 1.5-mm HDPE geomembrane over a 0.9-m compacted clay layer. Geonet, geotextile, and a 0.6-m sand layer compose the drainage system.

Leachate recirculation is done using corrugated, perforated pipes placed in 0.9 to 1.2-m deep gravel-filled trenches dug into the waste. A separate recirculation system is provided at each waste lift. Leachate is normally pumped for one hour, and then discontinued for another hour.

Difficulty was encountered in recirculating at early stages due to the insufficient waste available to absorb moisture. Also, recirculation near the waste surface on the slope seeps.

2.4.2.5. Lower Mount Washington Valley Landfill, New Hampshire

The Lower Mount Washington Valley Landfill, located in Conway, New Hampshire, started to operate in 1992 and is composed of eight separate cells.

This landfill receives averages between 9,100 and 13,600 metric-tons/year (25 to 37 metric-tons/day) of waste. Leachate is collected and stored in a 38 m^3 collection tank.

Leachate recirculation began four months after the first cell started, by pre-wetting the waste prior to compaction. Recirculation is also done using a PVC pipe manifold placed in a shallow excavation of the daily cover. To reduce the lateral movement of leachate, lateral trenches were installed near the slopes to recirculate leachate in those areas. However, this last practice was discontinued due to operational problems. Also, leachate recirculation was temporarily discontinued in the entire landfill in November 1993 due to the high rates of leachate production in the spring. This was the consequence of the fill being saturated during the previous fall and winter, followed by freezing and then the spring thaw. This problem was minimized in the following seasons by using an alternative daily cover that would minimize the rainfall infiltration. Gas measurements suggested that leachate recirculation enhanced waste degradation.

2.4.2.6. Coastal Regional Solid Waste Management Authority Landfill, North Carolina

The Coastal Regional Solid Waste Management Authority Landfill, located in Craven County, North Carolina, receives approximately 320 metric-tons per day at a 8.9-ha area subdivided into three separate cells, 15 m deep. The composite liner is composed of 0.6 m drainage sand, a fabric filter, and a 1.5-mm HDPE geomembrane over a 0.6-m compacted clay layer.

Leachate is pumped back to the waste using a system of flexible hoses feeding a movable vertical injection system, with flow rates varying between 200 and 300 liters per minute to an area approximately 30 m by 30 m.

Leachate recirculation was temporarily discontinued due to waste saturation that impeded the movement of heavy vehicles on the landfill surface. Operators attributed this problem to heavy precipitation and the inaccessibility of off-site leachate management.

2.4.2.7. Lemons Landfill, Missouri

The Lemons Landfill, located in Stoddard County, Missouri, with a fill area of 30 ha and maximum depth of 30 m, started to operate in 1993, at a receiving rate of approximately 270 metric-tons/day of MSW. The landfill is lined with a 1.5-mm PVC geomembrane over a 0.6-m thick layer of compacted bentonite and soil mix. The leachate collection system is composed of a 30-cm layer of pea-gravel and perforated PVC pipes.

Leachate is collected in two ponds and recirculated through the use of vertical wells spaced 61 m within the fill area. Recirculation in this landfill began approximately one year after the initiation of waste reception, delayed until the area was filled and temporarily capped with 0.6 m of clayey soil. The leachate recirculation wells are 1.2-m diameter perforated concrete pipes filled with 10 cm diameter stones.

Leachate is collected in the first of the available two ponds and recirculated back to each well at a 370-liters-per-minute rate for approximately six hours, and continues until leachate strength is reduced significantly. After that, leachate is diverted to the second pond and used to irrigate the final area of the landfill.

2.4.2.8. Mill Seat Landfill, New York

The Mill Seat Landfill, located in Monroe County, New York, has a bioreactor research project that aims to investigate the benefits of leachate recirculation using three cells operated under different conditions: one is the control cell, and the other two are operated using two different recirculation techniques. The cells have areas between 2.2 and 3.0 ha each, and are part of a large project that would expand the landfill ultimately to a 38-ha area, using the results of the test-cells.

Leachate recirculation is done using two different horizontal injection systems. The first system is composed of three pressurized loops (constructed from 10-cm diameter perforated HDPE pipes) in trenches filled with crushed cullet, tire chips, or other highly permeable materials that distribute leachate at three elevations at the cell. The second system is composed of deep horizontal trenches filled with permeable materials that distribute leachate in the cell. As waste is placed on top of the trenches, vertical wells are constructed to allow continued feeding of leachate to the trenches.

Recirculation rates are between 20 and 100 m^3/day. Gas recovery will be accomplished from both the pressured loop system and from the vertical well/trenches system, in addition to vertical gas wells installed at closure. Gas would be either flared or used to generate electricity.

2.4.2.9. Summary of Full-Scale Experiences in the United States

Table 2.9 presents a summary of the characteristics of the full-scale landfills with leachate recirculation discussed in the previous paragraphs.

Table 2.9 – Characteristics of Full-Scale Landfills with Leachate Recirculation in United States (Reinhart and Townsend, 1998; modified)

Site	Design Area (ha)	Active Area (ha)	Leachate Production (m^3/ha/day)	Leachate Recirculation (m^3/ha/day)	External Storage (m^3/ha)	Off-Site Treatment (m^3/ha/day)
Alachua County Southwest Landfill, Florida	10.9	10.9	7.8	4.3	124	4.3
Central Facility Landfill, Maryland	6.9	6.9	2.6	2.1	220	0.64
Winfield Landfill, Florida	8.9	2.8	19	14	67	0.55
Pecan Row Landfill, Georgia	16	4.5	2.7	1.1	690	0
Lower Mt. Washington Valley Landfill, New Hampshire	3.2	0.45	15	9.5	12	4.2
CRSWMA Landfill, North Carolina	8.9	5.7	17	12	1600	0
Lemons Landfill, Missouri	30	NA	2.2	5.0	110	NA
Mill Seat Landfill, New York – Test Cell 2	2.8	NA	2.8	6.8	35	NA
Mill Seat Landfill, New York – Test Cell 3	2.2	NA	2.8	5.2	41	NA

2.4.3. Full-Scale Experiences in Germany

One of the countries that has used leachate recirculation practice in full-scale landfills for several years is Germany. Doedens and Cord-Landwehr (1989) reported that, in 1981, 13 large-scale landfills, mainly in the north of the country, were practising leachate recycling. In those landfills, BOD and COD concentrations were measured with time and compared to the concentrations measured in other German conventional landfills. A trend was observed showing that BOD and COD concentrations were greatly reduced in landfills that used leachate recirculation after a certain period of time.

Table 2.10 presents a summary of the characteristics of full-scale landfills with leachate recirculation in Germany.

Table 2.10 – Characteristics of Full-Scale Landfills with Leachate Recirculation in Germany (based on Doedens and Cord-Landwehr, 1989)

Site	Area of Landfill (ha)	Maximum Elevation of Waste (m)	Amount of Waste (metric-tons/year)	Leachate Production (m^3/ha/year)	Leachate in Excess since LR started (m^3)	External Storage Capacity (m^3)	Distribution System	Starting Landfill / Leachate Recirculation
Stapelfeld	10	6	28,000	NA	2,000	1,650	Irrigation	1973/1981
Flechum	2.4	8	NA	417	0	600	Tank Lorry	1975/NA
Dorpen	2.7	9	NA	211	0	800	Tank Lorry	1979/NA
Venneberg	6	9	NA	5,900	7,000	2,100	Irrigation	1976/1977
Morgenstern	4.8	35	124,000	NA	0	0	Distribution Pipes	1977/NA
Blankenhagen	12	12	188,000	NA	NA	50	NA	1976/1978
Nauroth	7	24	122,000	NA	0	2,250	Irrigation	1973/1975
Reinstetten	5.4	15	111,400	1,574	470	110	Irrigation	1975/1982
Beltesrot	3	12	85,000	NA	8,333	300	Distribution Pipes	1980/1980
Supplingen	5.0	11	98,400	3,600	1800	1,000	Irrigation	1975/NA
Watenbuttel	15	32	257,800	140	0	0	Irrigation	1967/NA
Bornhausen	2.9	13	24,600	21,400	7,800	0	Distribution Pipes	1974/NA

2.5. Facultative Bioreactor Landfills

There are very few studies regarding the benefits of facultative landfills in comparison with landfills with straight leachate recirculation. It is known, however, that ammonia-nitrogen can be kept in high concentrations in leachate long after organic and metal concentrations are stable. In this case, ammonia is the product that can limit the direct discharge of leachate to either land or receiving waters, and also can influence the end of the post-closure monitoring period (Barlaz, 2002). The conceptual model consists on removing the ammonia in the leachate generated using an aerobic treatment to convert ammonia into nitrate (nitrification process), followed by a denitrification process (the conversion of nitrate to nitrogen gas - N_2, a harmless product), reinjecting the treated nitrified leachate into the waste mass, and using the carbon present in a large quantity inside the landfill for the conversion of nitrate into nitrogen gas.

2.5.1. Nitrification and Denitrification Processes

Due to the toxic effects that high concentrations of ammonia create on aquatic organisms, causing, for example, fertilization of lakes and reservoirs which leads to algae growth and eutrophication, reduction of chlorine disinfection efficiency, dissolved oxygen depletion in receiving waters, and adverse public health effects, ammonia must be treated to an acceptable level (<10 mg/L) before it is discharged (Clabaugh, 2001). Ammonia present in high concentrations in MSW leachate is generated by the decomposition of organic materials by bacteria.

Biological nitrification is a very common process utilized in wastewater treatment to convert ammonia to nitrates by oxidation, and can be also applied to leachate treatment plants. Some of the processes used to remove ammonia nitrogen from landfill leachate are the use of trickling filters (Clabaugh, 2001), and fluidized bed reactors (Cadenas, 2002).

The overall biochemical process that converts ammonia into nitrates, under aerobic conditions performed by bacteria known as nitrifiers, is represented by:

$$NH_4^{+} + 2O_2 \rightarrow NO_3^{-} + 2H^{+} + H_2O \qquad \text{.......................................(2.4)}$$

However, the process of converting ammonia into nitrates follows two steps: first, aerobic *Nitrosomonas* bacteria oxidize ammonia to nitrite, then Nitrobacter bacteria complete the oxidation by converting nitrite to nitrate. This two-step process is represented by the following equations:

$$2NH_4^{+} + 3O_2 \rightarrow 2NO_2^{-} + 4H^{+} + 2H_2O, \quad \text{Nitrosomonas} \qquad \text{.....................(2.5)}$$

$$2NO_2^{-} + O_2 \rightarrow 2NO_3^{-}, \quad \text{Nitrobacter} \qquad \text{.......................................(2.6)}$$

Nitrifying bacteria are naturally present in the soil, freshwater, and saltwater, and are found wherever their required nutrients, ammonia, and oxygen exist. However, they can be difficult to maintain because of their specific environmental requirements, such as a pH range between 7.0 and 8.8 and liquid temperatures between 20 and 35^{o}C. COD must be at levels that do not use all the available oxygen or create inhibitory conditions.

As previously stated, denitrification is the biochemical process of converting nitrate to nitrogen gas. The process is accomplished by denitrifiers, which include *Pseudomonas, Micrococcus, Archromobacter, and Bacillus*, facultative and anaerobic bacteria. The denitrification also requires a source of carbon, plenty in the case of landfills.

However, it is interesting to note that depending on the pE-pH conditions, two types of reaction can occur: one is the nitrate being converted into nitrogen gas as described before, but also the nitrate being converted back to ammonia. The reactions are presented below:

$$2\,NO_3^{\;-} + 12\,H^{+} + 10\,e^{-} \rightarrow N_2 + 6\,H_2O \text{ .... (2.7)}$$

$$NO_3^{\;-} + 10\,H^{+} + 8\,e^{-} \rightarrow NH_4^{\;+} + 3\,H_2O \text{ .... (2.8)}$$

In the reaction presented in Equation 2.8, the nitrate is reduced to ammonia-nitrogen rather than nitrogen gas, and this process consumes 8 electrons per NO_3^-, while in Equation 2.7 the reaction consumes only 5 electrons per NO_3^-.

Since NO_3^- may have large harmful impacts to a receiving system, it must be removed before discharge. This is the reason denitrification is an important step when managing nitrogen conversion for ultimate nitrogen removal.

2.5.2. Some Studies on Facultative Bioreactor Landfills

2.5.2.1. Onay and Pohland (1995)

One of the first studies investigating the effects of *in situ* denitrification in landfill systems was presented by Onay and Poland (1995). The study consisted in detailing the potential for attenuation of nitrogenous compound by using a 56.5 L, PVC reactor filled with a mixture of 10 kg compost and 3kg of bulking chips to simulate a stabilized refuse matrix. (Compost has a relatively lower available carbon content than MSW, whereas its nitrogen content is higher, according to Onay and Pohland, 1998).

The reactor used in the study was made of PVC, 33 cm in diameter and 61 cm high, equipped with a leachate collection/recycle system and a volumetric gas meter to measure gas production. The reactor was designed to operate in three different conditions at the same time: anoxic at the top, aerobic at the bottom, and anaerobic in between. At the bottom of the reactor, a layer of gravel was placed before the waste matrix, which permitted the controlled introduction of oxygen to initiate nitrification at that location. The nitrified effluent was then reintroduced into the anoxic denitrification zone at the top of the reactor.

Seven liters of tap water were added to the reactor at the beginning of the experiment to allow immediate production of leachate. Two liters of primary sewage were also added to introduce an active microbial population.

The investigation consisted of two phases: Phase I corresponded to the initial anaerobic decomposition of the waste matrix (days 1-120) and Phase II corresponded to the *in situ* denitrification. After Phase I ended, a feed solution of potassium nitrate (KNO_3) in concentration equivalent to the ammonia produced during the anaerobic decomposition of the compost was prepared and introduced to simulate recycle sequences, in a total of 16 batches during an operational period of 600 days.

In the study, the leachate generated was analyzed for pH, COD, oxidation-reduction potential, ammonia-nitrogen, nitrate, and sulfate. Gas production was monitored daily and analyzed for carbon dioxide, methane, nitrogen, oxygen, hydrogen, and hydrogen sulfide.

The results indicated a decrease of COD from a high of 13,000 mg/L to about 6,000 mg/L in 50 days, then a decrease to 2,000 mg/L level after day 150. The denitrification phase (Phase II) was marked by an initial increase in gas production, predominantly nitrogen with smaller amounts of methane and carbon dioxide. Correspondingly, the increase in nitrogen gas was observed as a result of the conversion of nitrate to nitrogen gas. The pH of the leachate ranged between 7.0 and 7.5, with the compost and the residual ammonia serving to stabilize the pH at near neutral, however, with the pH tending to increase after day 200.

The authors also presented the nitrogen management process in bioreactor landfills in two other articles (Onay and Pohland, 1998, and Onay and Pohland, 2001), using that time three reactors (operating in three operational stages: separate reactor operation, combined reactor operation with internal leachate recycle, and combined reactor operation with single pass) with sequential *in situ* removal of nitrogenous substrates in dedicated nitrification and denitrification zones.

2.5.2.2. Outer Loop Landfill Bioreactor Studies, Louisville, Kentucky (EPA, 2002)

The study, a joint bioreactor landfill research between Waste Management, Inc. and the U.S. EPA, will evaluate the economic and operational issues of aerobic, anaerobic, and facultative full-scale bioreactor landfills as compared to conventional MSW landfills. The waste area at the Outer Loop landfill is about 160 hectares; each test and control cell involves about 2.5 hectares.

The facultative bioreactor landfill study intends to evaluate MSW stabilization resulting from nitrate-enriched leachate application and assess the commercial viability of the operation. Leachate containing ammonia will be treated external to the landfill by nitrification to convert ammonia to nitrate. The treated leachate will be introduced to a landfill cell where the nitrate will be used by facultative bacteria to convert nitrate in nitrogen gas. Trenches will be used for liquid infiltration and there will be separate leachate and gas collection systems.

2.5.2.3. Barlaz (2002)

Tests to evaluate long-term nitrogen management in bioreactor landfills have been conducted in 12 liter reactors filled with shredded MSW. To date, the author points out that decomposing refuse has significant capacity for nitrate reduction. When nitrate was added to refuse that was actively producing methane, it was rapidly depleted; however, methane production recovered once the nitrate addition was terminated. The research, in progress, suggests that the addition of nitrate rich leachate to actively decomposing refuse represents a viable alternative for nitrogen management in landfills.

CHAPTER 3

MUNICIPAL SOLID WASTE COMPRESSIBILITY

3.1. Introduction

Published information concerning the behavior of Municipal Solid Wastes (MSW) settlement and compression processes when compared to the compressibility study of soils is relatively new. One of the first articles, a progress report published in the Journal of the Sanitary Engineering Division of the American Society of Civil Engineers in 1959 (ASCE, 1959), provided insight about the MSW volume reduction after disposal in some California and other states waste landfills through topographic survey analysis. In this study, the volume reductions reached approximately 25% of the total initial volume disposed after 5 years.

Merz and Stone presented in 1962 the first typical curves for settlements as a function of time by observing landfills cells monitored during 12 months (Wall and Zeiss, 1995).

However, it was only after the publishing of Sowers' article in 1973 (Sowers, 1973) in the Eighth International Conference on Soil Mechanics and Foundation Engineering, presenting the use of classical soil mechanics formulation for MSW settlements prediction, that research about the subject began.

The prediction and monitoring of waste landfill settlements, as well as the associated settlement rates, are important to:

- <u>Estimation of the waste landfill lifetime</u>: the prediction of disposal time, considering the additional volume available due to settlements. This is important for operation logistics and predicting the need for new disposition areas. However, although the knowledge of MSW volume reduction is real, the lack of a universally accepted model prevents some environmental agencies from accepting projects that consider the use of this additional space (Edil et al., 1990);

- <u>Reuse of areas after landfill closure</u>: the scarcity of areas in the large urban centers makes the areas occupied by old MSW disposal sites desirable for use as roadways, parks, playgrounds, parking lots, and other light structures;

- <u>Design and implementation of hydraulic structures and drainage systems</u>: the occurrence of differential settlements makes possible the appearance of spots with negative slopes on the landfill, causing inappropriate accumulation of rainwater in the case of the superficial drainage system, or leachate, in the case of the leachate drainage system;

- <u>Geotechnical monitoring</u>: in sanitary landfills with large MSW thickness, the monitoring of settlements and settlement rates give fundamental and valuable information to the geotechnical slope stability analysis. For example, the adoption of monitoring limits-criteria directly depends on settlement estimations as well as on the systematic observation of landfills through the use of superficial marks (Kaimoto et al., 1999); and

- <u>Final cover performance</u>: the landfill final cover system, composed preferably of clayey soil and/or “impermeable” geomembrane can be damaged by differential settlements that can cause ruptures on the soil layer, increasing the pore pressures inside the landfill due to possible rainwater infiltration to the waste mass, and sometimes affecting the landfill global or local geotechnical stability.

Landfill settlements can be problematic; however, they can increase the landfill lifetime with the possibility of additional disposal (Edil et al., 1990).

3.2. Settlement Mechanisms

The compression of waste landfills is the result of loading and alteration of the characteristics and properties of the component materials inside the landfill. This includes mechanical compression by weight of the overburden, raveling, physical-chemical changes, biodegradation, and interaction among these mechanisms (Van Meerten et al., 1995). Another mechanism, not as acute in North American landfills, but certainly present in landfills where the quantities of organic matter are above 50% (in the case of Latin American countries presented by Carvalho, 1999), is the slow dissipation of pore pressures of liquids and gases from the interior of the landfill due to loading, causing settlements.

Each one of the settlement mechanisms is described below (Sowers, 1973):

- <u>Mechanical compression</u>: distortion, bending, crushing, rupture, and rearrangement of the materials;

- <u>Raveling</u>: erosion and migration of the small particles to the voids among larger particles;

- <u>Physical-chemical changes</u>: corrosion, oxidation, and combustion of the components;

- Biodegradation: biological degradation that corresponds to the fermentation and components degradation that causes the mass to change from the solid phase to liquid and gaseous phases;

- Interaction: interactions among the several mechanisms, for instance, the spontaneous combustion of methane generated by biodegradation reactions due to the increase of heat caused by those or other reactions, organic acids generated by the biodegradation process can cause corrosion of metals, volume changes due to the mechanical compression mechanism can cause raveling processes; and

- Liquid and gas pore pressures dissipation: similar to the consolidation mechanism of soils, it represents the deformation obtained by the dissipation of pore pressures of liquids and gases from the interior of the landfill due to loading, which can take considerable time.

Among all those mechanisms, only the first and the last ones (mechanical compression and pore pressure dissipation) are related to imposed loads. All others are related to the environment inside the waste mass (in terms of oxygen, moisture, temperature, among others) and are linked to biochemical transformations inside the landfill.

The evaluation and prediction of settlements should take into consideration the presented mechanisms; however, the quantification of the several factors involved is very complex or even impossible. So, empirical formulations and classical expressions are commonly utilized for settlement estimations associated with laboratory tests and field monitoring.

The factors that determine the processes described above are the MSW composition, size, height and operation of the waste landfill, deposition rate, MSW pre-treatments, initial unit weight, compaction, saturation, efficiency of drainage systems (superficial and internal), leachate level fluctuations, and biological constraints (pH, temperature, and moisture) (Sowers, 1973; Coumoulos and Koryalos, 1997). Another important factor to be taken into consideration is the landfill age (time elapsed since landfill closure). Older landfills tend to have smaller settlements in a given time interval when compared to newer ones.

Biodegradation is considered one of the principal processes and also one of the most complex, and can last for decades. Biodegradation in conventional landfills can be divided into five phases, which occur chronologically: aerobic decomposition phase, transition or hydrolytic phase, acid anaerobic decomposition (acetogenic) phase, methanogenic anaerobic decomposition phase, and maturation phase. A detailed description of these phases is presented in Section 2.2.1 in Chapter 2.

Regarding pore pressure dissipation, one can speculate that probably this is not the mechanism that induces settlements the most. However, for Latin American landfills, there are some indications that this process happens, according to De Abreu (2000). Figure 3.1 presents the evolution of leachate and gas pressures for a piezometer from a Brazilian sanitary landfill located near an area where loading due to stocking of cover soil has occurred. The trend of increasing leachate and gas pressures in both chambers of the piezometer (10 m and 20 m depth) during the loading process can be noticed.

Bono (1991), as reported by Debnath (2000), reported that the activation of a gas extraction system caused localized settlements of 1.0 to 1.5 ft within a 70-ft-thick landfill over a period of six months.

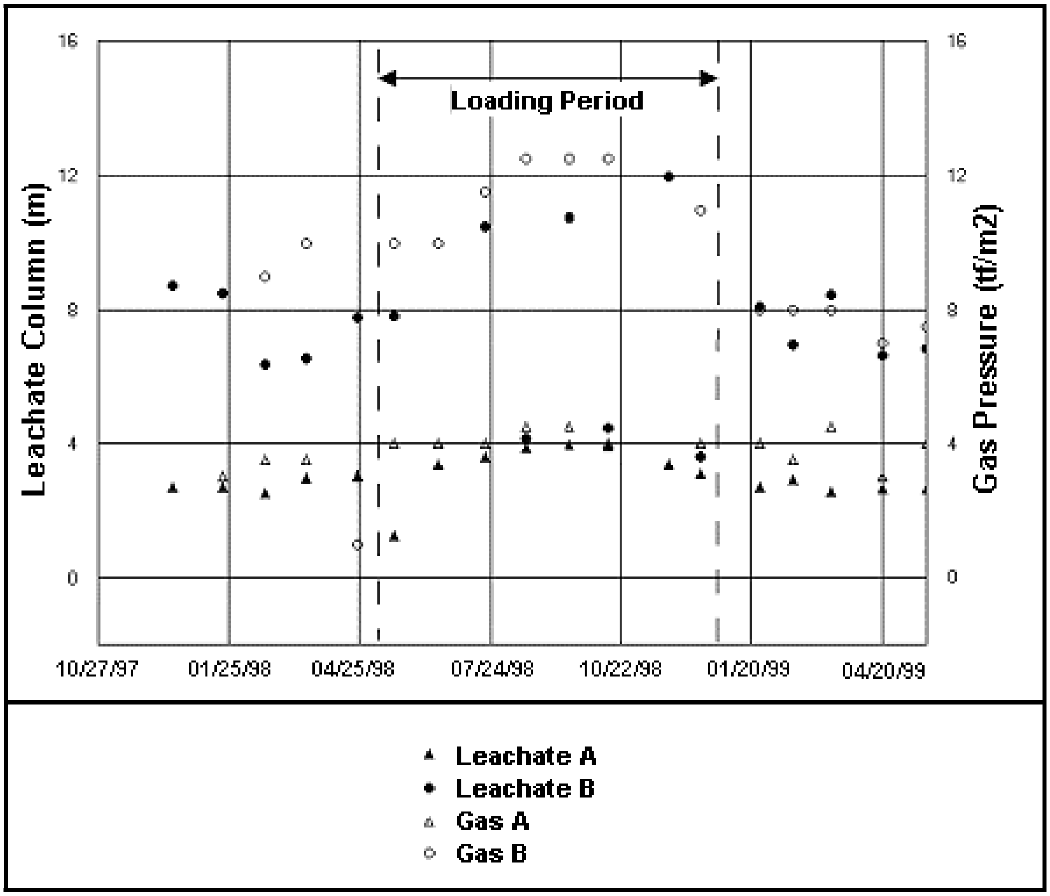

Figure 3.1 – Graph of Leachate and Gas Pressures Variations in Piezometer Located in a Brazilian Sanitary Landfill during Loading Process

Raveling, which is the most complex and difficult process to be quantified, in practice is normally ignored or aggregated to the other processes, mainly the part corresponding to the migration of daily cover soil particles through the MSW voids. However, the model in which the cover soil layers are independent and do not interact with MSW disposed layers is not valid since the migration of soil particles is real. If the soil layers initially represent 20% of the total geometric volume of an ordinary sanitary landfill, after loading they will represent approximately 5% of the total thickness of the landfill, as presented by Morris and Woods (1990), Figure 3.2.

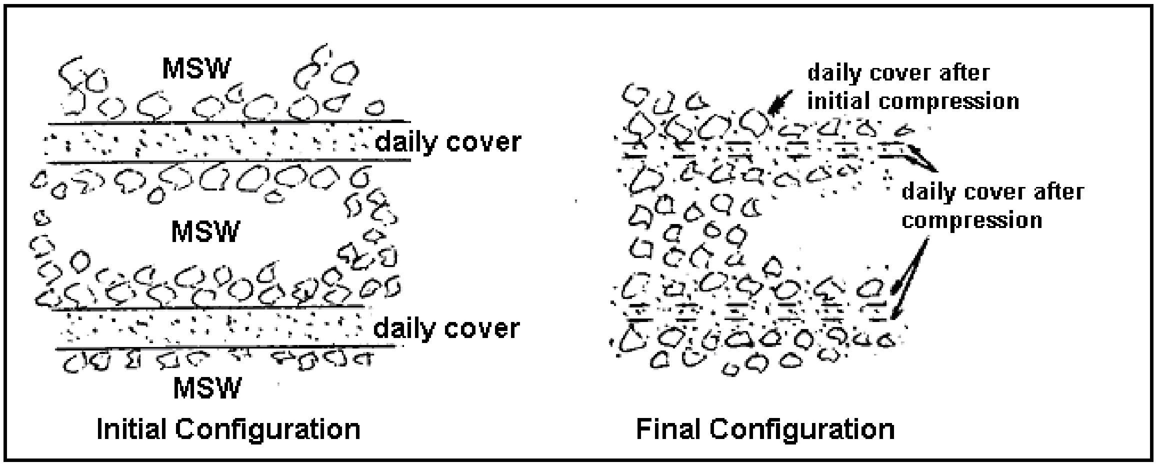

Figure 3.2 – Raveling due to Daily Soil Cover (Morris and Woods, 1990)

However, other factors induce raveling and combustion mechanisms, which take place after episodes that are favorable to MSW degradation, as for example, sudden leachate level fluctuations, or floods caused by heavy rains, or pipe ruptures, among other unpredictable events, and make the estimation of the settlement associated with those factors impossible.

According to most authors, the settlement of sanitary landfills, independently of their mechanisms, can be divided into three phases with time similar to classical Soil Mechanics: initial compression, primary compression, and secondary compression (Wall and Zeiss, 1995).

- <u>Initial compression</u>: corresponds to the settlement that occurs when an external load is applied to the landfill, commonly associated with the void reduction between particles and particle sizes due a load imposed. This type of settlement is analogous to the elastic compression that occurs in soils and it is instantaneous, finishing soon after disposal.

- <u>Primary compression</u>: corresponds to the settlement due to the dissipation of pore pressures and gases in the voids, occurring quickly and ending approximately 30 days after applying the load (Sowers, 1973) or sooner

(Boutwell and Fiore, 1995). Although it is known that both the primary compression and the secondary compression occur simultaneously, the magnitude of the primary compression is higher than the secondary in this initial period, and then becomes equal after 30 days, and modeled separately. Terzaghi defined primary compression (or primary consolidation) as the settlement that occurs by the dissipation of water between particles of a saturated material when an external load is applied. However, that phenomenon does not generally apply totally with landfills due to the absence of the a saturated state in the MSW. Furthermore, some authors describe the MSW with permeability properties similar to gravels and sands without the development of pore pressures in the leachate and with the gases quickly escaping from the waste mass. This is not true for Latin American landfills, as seen in Figure 3.1. However, several authors (Sowers, 1973; Landva and Clark, 1990; Rao et al., 1977; Morris and Woods, 1990) applied Terzaghi's classical consolidation theory for the primary compression of MSW and report some success.

- <u>Secondary compression</u>: consists of settlement corresponding to "creep", slow deformation of MSW components and biological degradation (biodegradation). Secondary compression is responsible for most of the settlement in landfills, which can last decades and reach 25% or more of the initial total thickness of the landfill.

Figure 3.3 shows the phases described above.

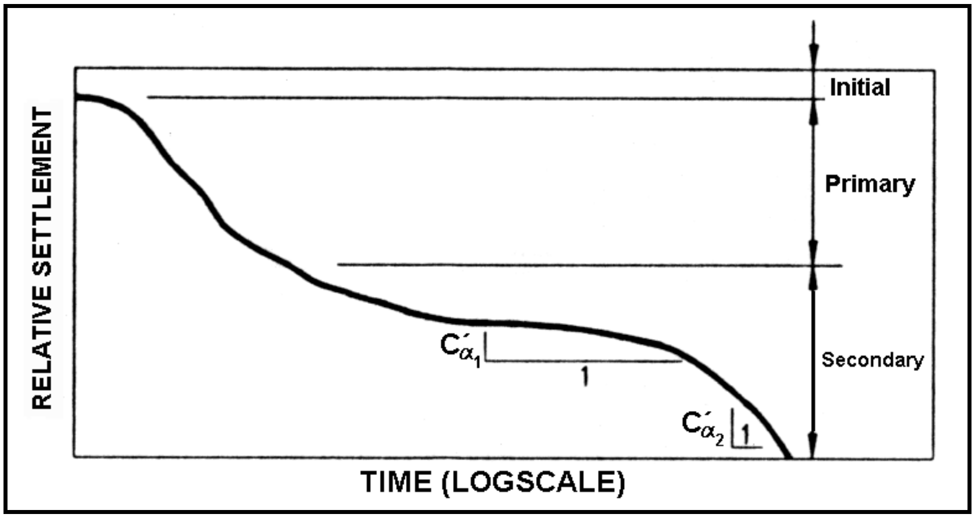

Figure 3.3 – MSW Compressibility Phases (Boutwell and Fiore, 1995; modified)

Grisolia and Napoleoni (1996) divided the MSW settlements into five phases: initial deformation with macroporosity reduction, residual settlement of high deformable materials, slow deformation and organic matter decomposition, decomposition concluded, and residual deformation, as presented in Figure 3.4. Note, however, that the phase division presented by Grisolia and Napoleoni corresponds to sub-phases of the phases commonly used.

Manassero et al. (1996) pointed out that the MSW settlements behavior is very similar to that from peaty soils, which has immediate settlement, followed by large additional settlements with low or no dissipation of pore pressures. However, the MSW secondary compression also has a strong component due to biological decomposition.

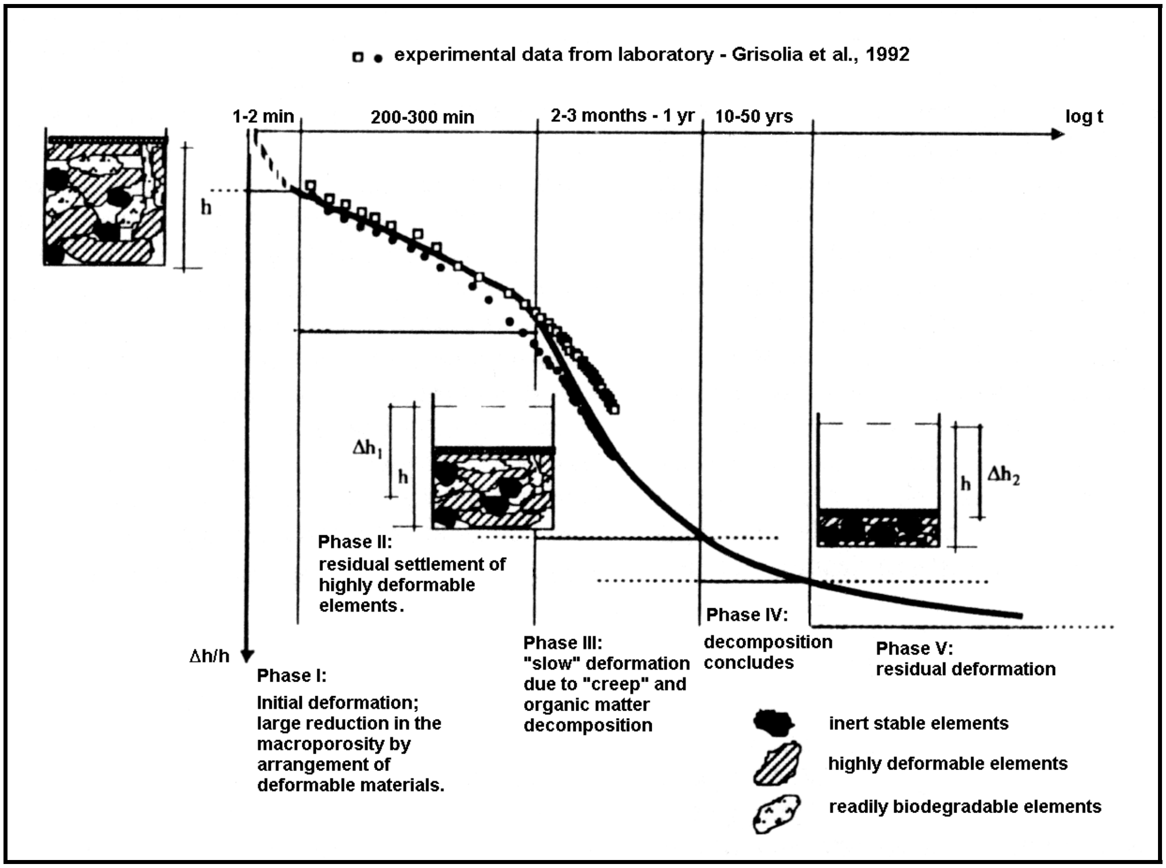

Figure 3.4 – MSW Compressibility Phases According to Grisolia and Napoleoni (1996)

The division of settlements development into phases can be very questionable since the present knowledge does not allow concluding that such phases are independent, and it is certain that they are overlapped. So, graphs such as the ones presented in Figures 3.3 and 3.4, as well as the utilization of the terms “initial compression”, “primary compression”, and “secondary compression” extracted from soil deformation processes suggest that the phases occur dissociated in time, and, in principle, should not be applied to MSW. Furthermore, the “creep” phenomenon is presented exclusively in the secondary compression phase (together with the biodegradation phenomenon). However, for MSW that mechanism is also a consequence of the compression of solid particles, and, therefore, also occurring with loading.

Coduto and Huitric (1990) presented a division of the settlement mechanisms, which, in practice, are related to the phases of settlement development. Those phases correspond to:

- Consolidation: corresponds to the mechanism of dissipation of pore pressures;

- Compaction: corresponds to the mechanism of mechanical compression; and

- Shrinkage: corresponds to the mechanisms associated with “solid losses” – biodegradation and physical-chemical changes.

Note, however, that the raveling and interaction mechanisms can occur in any one of those three phases and are unpredictable, and, therefore, shall be incorporated into the three phases.

The three phases would start together in the case of an instantaneous loading (t=0) after the disposal of the residues. However, there is not enough knowledge to conclude how long each would last. One can understand that, generally speaking, the beginning and ending of each phase is strongly linked to the history of disposal and to the operation of the sanitary landfill.

Generally speaking, the settlements can reach 30% of the total initial thickness of waste landfills (Sowers, 1973). However, values between 25% and 50% are presented by Wall and Zeiss (1995), 10% and 25% are presented by Van Meerten et al. (1995), and values between 20% and 25% are presented by Comoulos and Koryalos (1998).

3.3. Models for Settlement Prediction

3.3.1. Classical Model

3.3.1.1. Sowers Proposition

The Sowers (1973) concept corresponds to the model proposed in the Terzaghi's classical theory for soils and adapted to MSW with the utilization of the parameters obtained from field observations as well as from laboratory tests.

However, one may note that in the development of Terzaghi's classical theory, some simplified hypotheses are adopted that do not always satisfactorily model the MSW behavior. The hypotheses correspond to the validity of Darcy's Law, complete saturation, homogeneous solid particles, insignificant compressibility of the solid and liquid particles, independence of some of the properties with increasing or decreasing of the effective stress, one-dimensional compression, one-dimensional flow, and a linear relationship between stress and void ratio (Taylor, 1948). Despite all those constraints, this is the most utilized model for MSW settlement estimation and prediction.

Dividing the MSW settlement into three phases (initial, primary, and secondary), the initial phase can be analog modeled as for soil elastic settlements:

$$\frac{\Delta h_i}{H_0} = \frac{\Delta\sigma}{E} \qquad \text{...} (3.1)$$

Δh_i ... settlement correspondent to the initial compression,
H_0 ... initial thickness of MSW layer,
$\Delta\sigma$ ... increase in the vertical stress applied to MSW,
E ... MSW elastic modulus.

However, in the case of MSW, the initial compression is sometimes treated together with the primary compression since they are both the result of the same effect (the applied load). Also, it can be very difficult to estimate the elastic modulus due to its immediate nature.

The MSW primary compression can be described in terms of the compression index C_c. The decrease in the void ratio during the primary compression is the result of the increase in the vertical effective stress and can be expressed by the equation:

$$\Delta e = -C_c \cdot \log\left(\frac{\sigma'_0 + \Delta\sigma}{\sigma'_0}\right) \qquad (3.2)$$

Δe ... variation in the void ratio,

C_c ... primary compression index,

σ'_0 ... initial vertical effective stress,

$\Delta\sigma$ ... increase in the vertical stress.

The compression index C_c is related to the initial void ratio (e_0) and can vary between 0.15 e_0 to 0.55 e_0. The upper limit corresponds to MSW containing large quantities of food waste and high decomposable materials. The lower limit corresponds to less resilient materials (Sowers, 1973). Compared to soils, the primary compression indices for peaty soils are approximately 30% larger than the maximum values observed in sanitary landfills. Figure 3.5 presents the variation on the primary compression index C_c as a function of the initial void ratio.

One should remember that the primary compression index C_c has the same meaning utilized in geotechnical engineering, but is not necessarily related to the existence of a “virgin straight line”. It can be better understood as a “secant coefficient” for a given stress interval (Santos and Presa, 1995).

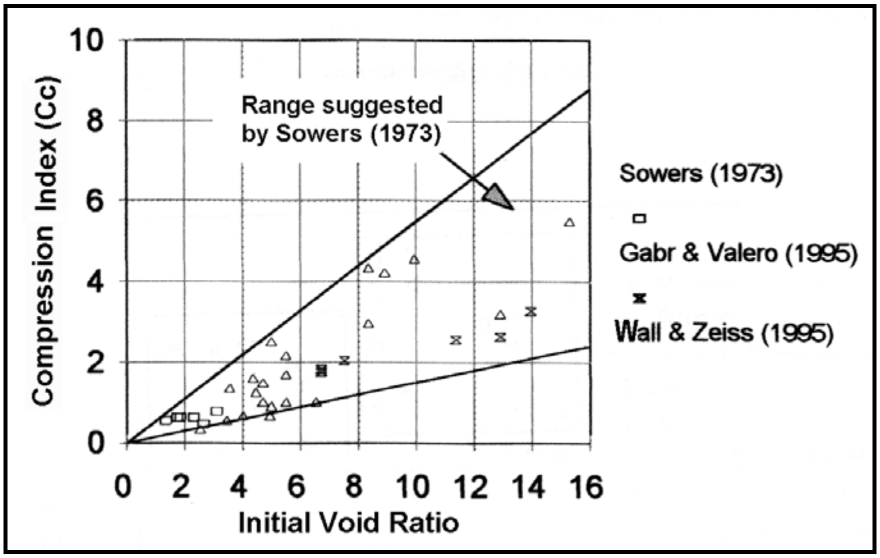

Figure 3.5 - Variation on the Primary Compression Index C_c as a Function of the Initial Void Ratio e_0 (Carvalho, 1999)

However, the MSW void ratio is difficult to estimate and for values greater than 3, typical in sanitary landfills, an extensive range for C_c is presented in Figure 3.5. So, the great majority of authors (for example, Sowers, 1973; Yen and Scanlon, 1975; Grisolia and Napoleoni, 1996) prefer to utilize the coefficient of primary compression C'_c in the analysis of primary compression, as presented in the following procedure:

From Soil Mechanics theory:

$$-\frac{\Delta e}{1+e_0}=\frac{\Delta h}{H_0} \quad \text{...}(3.3)$$

Equation 3.2 can be rewritten as:

$$\frac{\Delta h_p}{H_0}=\frac{C_c}{1+e_0}\cdot \log\left(\frac{\sigma'_0+\Delta\sigma}{\sigma'_0}\right) \quad \text{..}(3.4)$$

Δh_p ... primary compression phase settlement.

Introducing the coefficient of primary compression C'_c as being:

$$C'_c = \frac{C_c}{1+e_0} \quad \text{...}(3.5)$$

and substituting Equation 3.5 into Equation 3.4:

$$\frac{\Delta h_p}{H_0} = C'_c \cdot \log\left(\frac{\sigma'_0 + \Delta\sigma}{\sigma'_0}\right) \quad \text{...}(3.6)$$

The reduction of the primary compression equation to only one parameter obtained from laboratory tests, as well as from field monitoring, can be verified in Equation 3.6.

Table 3.1 presents typical values for the coefficient of primary compression C'_c obtained from laboratory tests and/or field monitoring.

The MSW secondary compression, associated with "creep" and biodegradation phenomena, is expressed in terms of the secondary compression index C_α, in which a decrease in the void ratio during the secondary compression is related to the time elapsed between the initial time (t_i) and the final time (t_f):

$$\Delta e = -C_\alpha \cdot \log\left(\frac{t_f}{t_i}\right) \quad \text{...}(3.7)$$

t_f ... analysis time,

t_i ... time when the primary compression finishes.

Table 3.1 - Typical Values for the Coefficient of Primary Compression C'_c Obtained by Laboratory Tests or Field Monitoring

Researcher	C'_c Range	Remarks
Sowers (1)	0.13 – 0.47	Large Scale Pilot Tests and Full Size Instrumented Fills
Burlingame (1)	0.05 – 0.35	Laboratory Test (C'_c = 0.35); Field Monitoring (C'_c = 0.05 to 0.25)
Gordon (1)	0.05 – 0.10	
Rao et al. (1)	0.16 – 0.24	Laboratory Tests – 0.60 m diameter Consolidometer – Increase of C'_c with increase of initial density (1.4-2.4 kN/m^3)
Landva and Clark (1)	0.17 – 0.35	Laboratory Tests – 0.45 m diameter Consolidometer C'_c = 0.35 for fresh waste
Charles (1)	0.10 – 0.19	
Converse (2)	0.25 – 0.30	
Zoino (2)	0.15 – 0.33	Field Monitoring – Landfill Initial Height=1.8 m
Oweis and Khera (2)	0.08 – 0.22	
Mariano (3)	0.40	
Carvalho and Vilar (4)	0.17 – 0.23	Laboratory Tests – 0.37 m diameter Consolidometer
Debnath (5)	0.29 – 0.35	Laboratory Tests – 0.60 m diameter Consolidometer – Larger values for fresh waste-Initial densities = 1.7-11.3 kN/m^3

(1) PHILLIPS et al. (1993)
(2) WALL and ZEISS (1995)
(3) MARIANO (1999)
(4) CARVALHO and VILAR (1998)
(5) DEBNATH (2000)

C_α is an index related to the biodegradation potential and to the physical-chemical alterations. Its value increases with the amount of organic content, the degradation potential of the residues, and the environmental characteristics of the landfill (moisture, heat, oxygen presence, etc). The values are low for inert residues and unfavorable degradation conditions inside the landfill (Sowers, 1973). The secondary compression index C_α is related to the initial void ratios (e_0), and can vary between 0.03 e_0 and 0.09 e_0. Figure 3.6 shows this relationship.

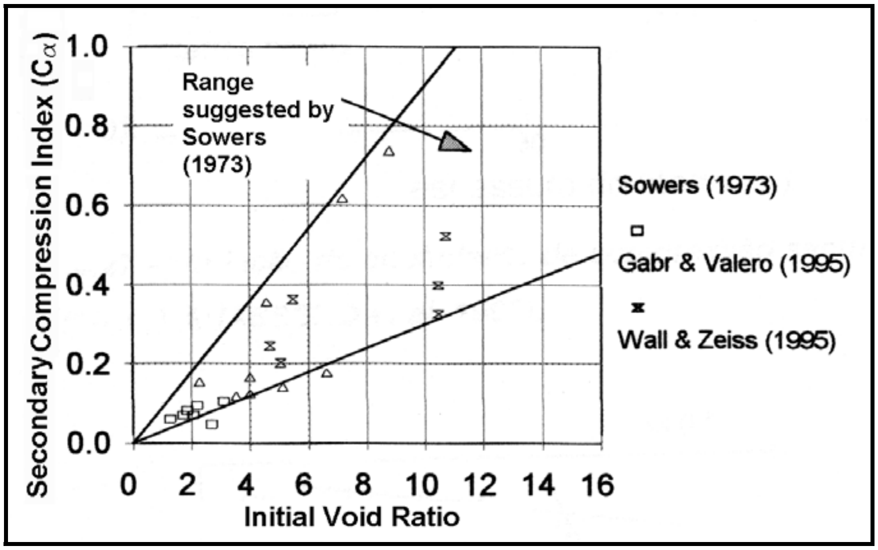

Figure 3.6 - Variation on the Secondary Compression Index C_α as a Function of the Initial Void Ratio e_0 (Carvalho, 1999)

Analogous to the primary compression, the secondary compression settlement can be written as:

$$\frac{\Delta h_s}{H_0} = C'_\alpha \cdot \log\left(\frac{t_f}{t_i}\right) \qquad (3.8)$$

in which C'_α corresponds to the coefficient of secondary compression and Δh_s is the settlement correspondent to the secondary compression phase.

$$C'_\alpha = \frac{C_\alpha}{1 + e_0} \qquad (3.9)$$

Table 3.2 presents typical values for the coefficient of secondary compression obtained by laboratory tests and field monitoring.

Table 3.2 - Typical Values for the Coefficient of Secondary Compression Obtained by Laboratory Tests and Field Monitoring

Researcher	C'_α range	Notes
Sowers (1)	0.025 – 0.075	Large Scale Pilot Tests and Full Size Instrumented Fills
Burlingame (1)	0.008 – 0.022	
Walker et al. (1)	0.04 – 0.08	
Yen et al. (1)	0.06 – 0.14	Field Studies - 3 Landfills in California
Watts et al. (1)	0.02 – 0.23	
Edil et al. (1)	0.012 – 0.075	Field Studies - 4 Sanitary Landfills
Gifford et al. (1)	0.020	
Rao et al. (2)	0.012 – 0.046	Field Studies – Experimental Cells, Large Scale Test Cells, Municipal Landfill
Converse (2)	0.07	
Zoino (2)	0.013 – 0.030	
Landva et al. (2)	0.0005 – 0.029	
Keene (3)	0.014 - 0.034	
Oweis and Khera (3)	0.02 – 0.24	
Cartier and Baldit (3)	0.30 – 0.55	
Palma (3)	0.02 –0.16	Field Studies
El-Fadel and Al-Rashed (3)	0.015 –0.25	
Mariano (3)	0.06 – 0.89	
Carvalho and Vilar (4)	0.002 – 0.016	Laboratory Tests – 0.37 m diameter Consolidometer
Debnath (5)	0.0009 – 0.018	Laboratory Tests – 0.60 m diameter Consolidometer

(1) PHILLIPS et al. (1993)
(2) WALL and ZEISS (1995)
(3) MARIANO (1999)
(4) CARVALHO and VILAR (1998)
(5) DEBNATH (2000)

Some authors that have developed models derived from the classical model are presented herein. Fasset et al. (1994), as reported by Manassero et al. (1996), presented an expression combining the primary and secondary compressions:

$$\frac{\Delta h}{H_0} = C'_c \cdot \log\left(\frac{\sigma'_0 + \Delta\sigma}{\sigma'_0}\right) + C'_\alpha \cdot \log\left(\frac{t_f}{t_i}\right) \quad \text{...(3.10)}$$

Bjarngard and Edgers (1990), as reported by Manassero et al. (1996), presented the secondary compression subdivided into two sub-phases, through the adjustment of two straight lines, and introduced the intermediate coefficient of secondary compression ($C'_{\alpha 1}$) and a final coefficient of secondary compression ($C'_{\alpha 2}$):

$$\frac{\Delta h}{H_0} = C'_c \cdot \log\left(\frac{\sigma'_0 + \Delta\sigma}{\sigma'_0}\right) + C'_{\alpha 1} \cdot \log\left(\frac{t_m}{t_i}\right) + C'_{\alpha 2} \cdot \log\left(\frac{t_f}{t_m}\right) \quad \text{..................................(3.11)}$$

t_m ... time when the intermediate secondary compression begins.

Burlingame (1986), as reported by Manassero et al. (1996), suggested the adoption of a coefficient that englobes the primary and secondary compressions (C'_b):

$$\frac{\Delta h}{H_0} = C'_b \cdot \log\left(\frac{\sigma'_0 + \Delta\sigma}{\sigma'_0}\right) \cdot \log\left(\frac{t_f}{t_i}\right) \quad \text{..(3.12)}$$

All presented models use indexes and coefficients for mono-logarithmic relationships in order to make the settlement estimates easier. However, it must be remembered that such simplifications do not mean that those indexes and coefficients are constant. In the majority of cases, this is not what happens (similar to what happens to soils) for the coefficient of primary compression on high values of applied stress as shown in Figure 3.7, as well as for the coefficient of secondary compression as shown in Figure 3.8. Other aspects of Equations 3.10, 3.11, and 3.12 will be discussed in Section 3.3.1.2.

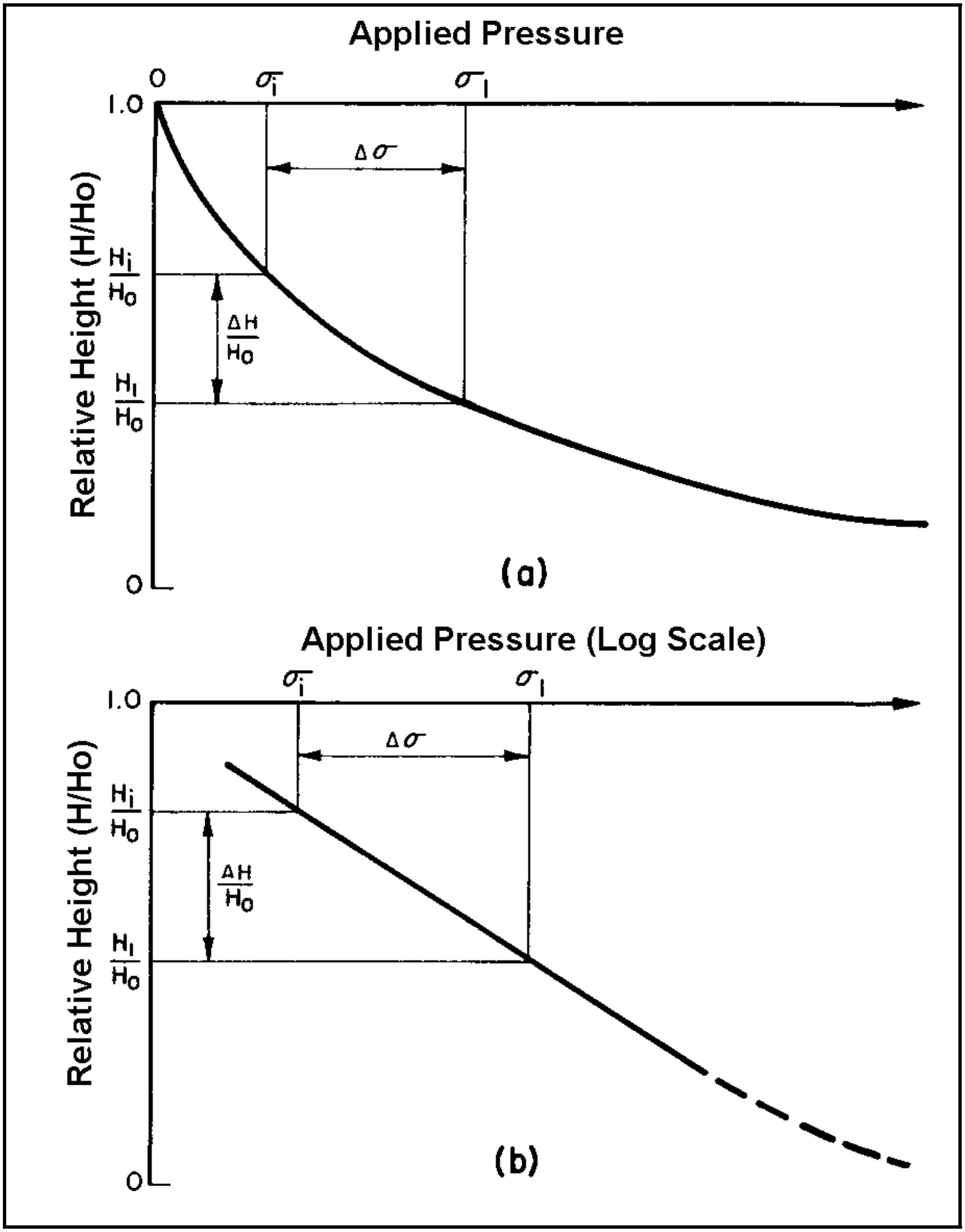

Figure 3.7 – Relative Settlement as a Function of Applied Stresses (a) and as Log of Applied Stress (b), Demonstrating the Non-Linearity of the Coefficient of Primary Compression for High Stress Values (Rao et al., 1977)

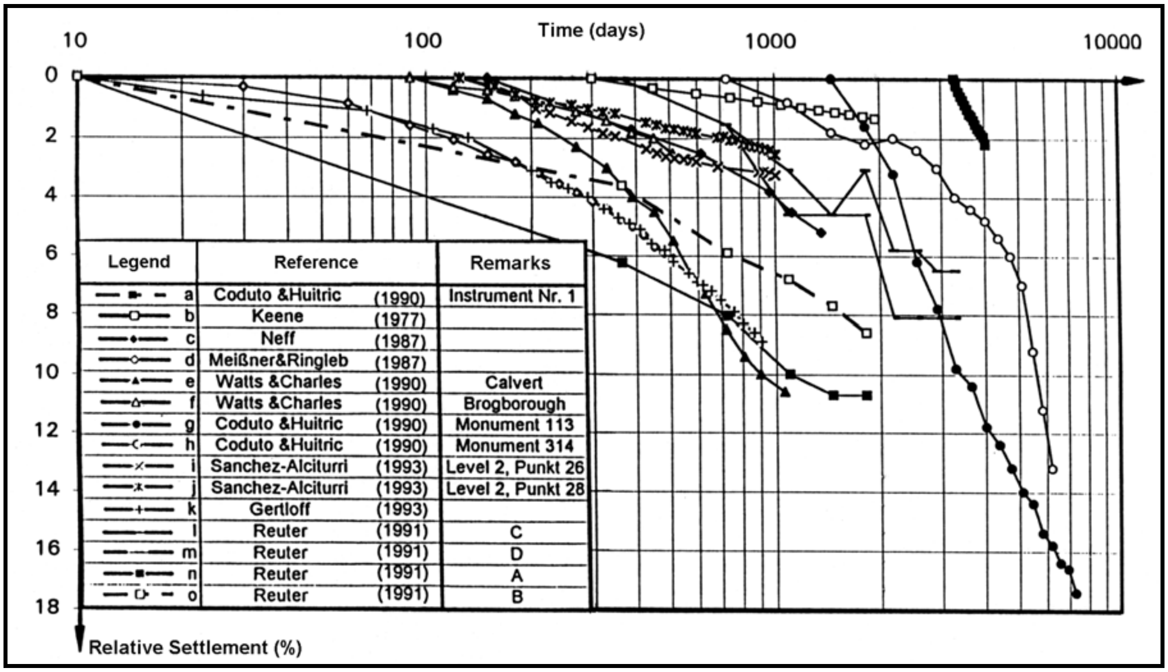

Figure 3.8 – Surface Monitoring of Several Waste Landfills, Demonstrating the Non-Linearity with Log of Time of the Coefficient of Secondary Compression in Several Cases (Kockel et al., 1997)

3.3.1.2. Criticism of the Classical Model as Applied to MSW

Despite the fact that the classical model is the most used model for MSW settlement estimates and prediction, some criticism of it is presented below.

As presented in Section 3.3.1.1, the classical model hypotheses adopted for soils are not adequate for MSW. So, the Sowers Proposition, reviewed by other authors, must be understood as an empirical adjusted model, since it does not represent the phenomenon that occurs in reality. On the secondary compression representation, the C'_α parameter is not constant with log-time. Also, there are difficulties in establishing what the MSW initial thickness is (especially in old landfills), necessary to the formulation.

As pointed out in Section 3.2, the division into the settlement phases of initial, primary, and secondary causes confusion for the comprehension of settlement mechanisms as well as for settlement predictions.

That deficiency can be noted more when the initial and primary compressions are modeled together. The primary compression is determined by an expression in which there is no time variable (Equation 3.6), and must be considered as a final value; however, it is considered to last 30 days (Sowers, 1973). In practice, it is difficult to isolate the primary compression from secondary compression because, differently from most soils, for MSW there is not a clear distinction in the settlement curve as a function of time for the two processes.

Furthermore, the adoption of the time is somewhat vague when the primary compression is finished (t_i, Equation 3.7). Some authors arbitrarily adopt t_i equal to 30 days, like Sowers (1973); others do not explain what t_i to use (Simoes et al., 1996).

Similarly, some authors utilize the initial thickness (H_0, Equation 3.8) as being the thickness at the beginning of instrumentation (Carvalho, 1999). Others use the thickness after the conclusion of primary compression (Morris and Woods, 1990), and the great majority of them do not explain which thickness was used. Using Equation 3.8, from the curve of relative settlements as a function of time, an angular coefficient C'_α is obtained, which is dependent on the H_0 adopted. Thus, the same data can furnish different values of C'_α, depending on the initial thickness adopted. C'_α values can only be compared only if the same criteria are applied, and the values presented in Table 3.2 and in Figure 3.8 must be reviewed with these restrictions.

Fasset et al. (Equation 3.10), as presented by Manassero et al. (1996) and Stulgis et al. (1995), has two components. One is load dependent and the other is time dependent. It must be noted that the equation is conceptually incorrect because it is valid only for times t_f larger than t_i, and for a constant C'_c, which is not reported. Also, if C'_α is constant with log-time, then the relative settlement curve would be a straight line function with time. This is not true, as shown in Figure 3.8.

Figure 3.9 presented by Pinto (1999) shows the settlement as a function of time for a superficial mark on a Brazilian landfill, in which there is available data only between 196 to 685 days after closure. It can be noticed that when the curve is extrapolated for any date near to the landfill closure, a negative relative settlement is obtained, which is nonsense. Probably the real curve follows the behavior of the dashed curve.

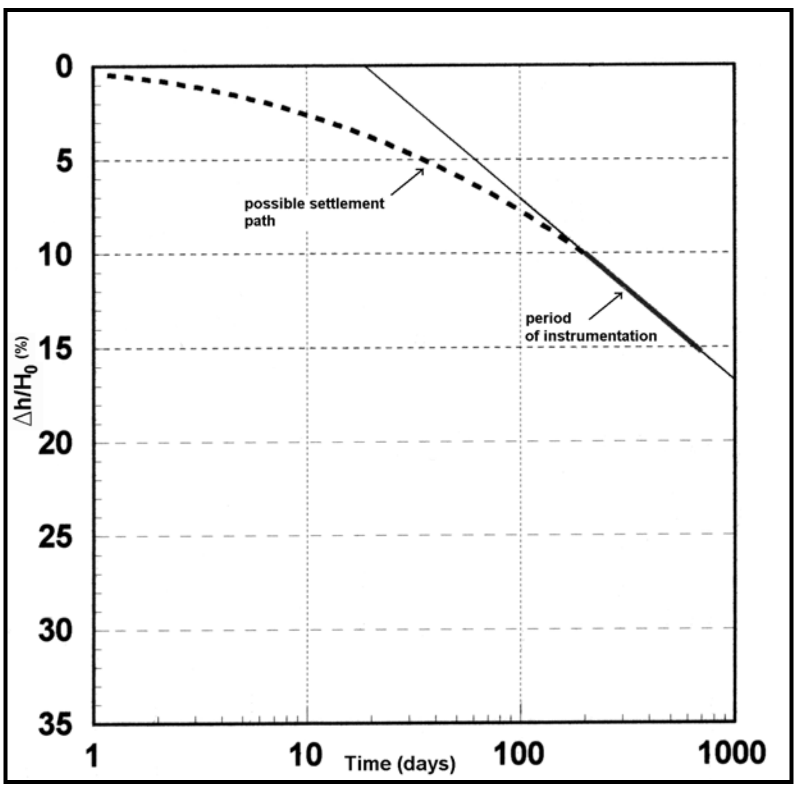

Figure 3.9 – Non-linearity of the Relationship Relative Settlement as a Function of Log-Time (Pinto, 1999)

Similarly, the Bjarngard and Edgers expression (Equation 3.11), presented by Manassero et al. (1996) and Stulgis et al. (1995), is only valid for C'_c constant with the log of the applied stress and for times greater than t_m (t_m>t_i). However, the authors present the curve from the graph of the relative settlement as a function of time approximated by two straight lines with coefficients equal to $C'_{\alpha 1}$ and $C'_{\alpha 2}$. De Abreu (2000), Carvalho (1999), Stulgis et al. (1995), Boutwell Jr. and Fiore (1995), and Manassero et al. (1996) verified the existence of two components exhibiting a quasi-linear behavior with log-time, with different inclinations.

Carvalho (1999), citing Konig and Jessberger (1997) and Manassero et al. (1996), presented the change in the behavior of the curve related to the first coefficient ($C'_{\alpha 1}$) as being associated with the "creep" phenomenon, and the second coefficient ($C'_{\alpha 2}$) as being associated with the "creep" phenomenon plus the effect of MSW degradation, and justifying why $C'_{\alpha 2}$ would be much greater than $C'_{\alpha 1}$. However, this justification is not valid as explained in the following comments:

- Innumerable functions that do not follow a logarithmic law (for example, linear, exponential, polynomial functions), present a distortion in their respective curves whenever presented in semi-logarithmic graphs, with two or more components very distinctive, which can be approximated by segments of straight lines. It does not have a physical meaning. <u>It is a characteristic of the non-logarithmic functions</u>.

- Which is the real meaning of C'_{α}? According to Sowers (1973), C_{α} is an index related to the biodegradation potential and to physical-chemical alterations, and its value can be as high as the organic contents present and how the degradation potential of the residues, given the environmental characteristics of the landfill (moisture, heat, oxygen presence, etc). However, and according to the literature (e.g., Palma, 1995), C'_{α} is not constant and changes with time. As can be seen on Figure 3.10, it is clear that C'_{α} increases with time until it becomes practically constant, while the settlement rate decreases (note that certainly after some time C'_α obligatorily will decrease, otherwise the settlement would be infinite, which is nonsense). That means that C'_α does not have a physical meaning. A landfill under unfavorable conditions to degradation can have a larger C'_α than another one under favorable conditions, and for that it is only necessary that the C'_α of both landfills be compared at different times.

In other words, in order to compare two C'_α obtained in two different landfills, it is necessary to know what is the time of the analysis. Furthermore, C'_α must be considered as a parameter with an origin in a mere adjustment of the curve relative settlement as a function of log-time.

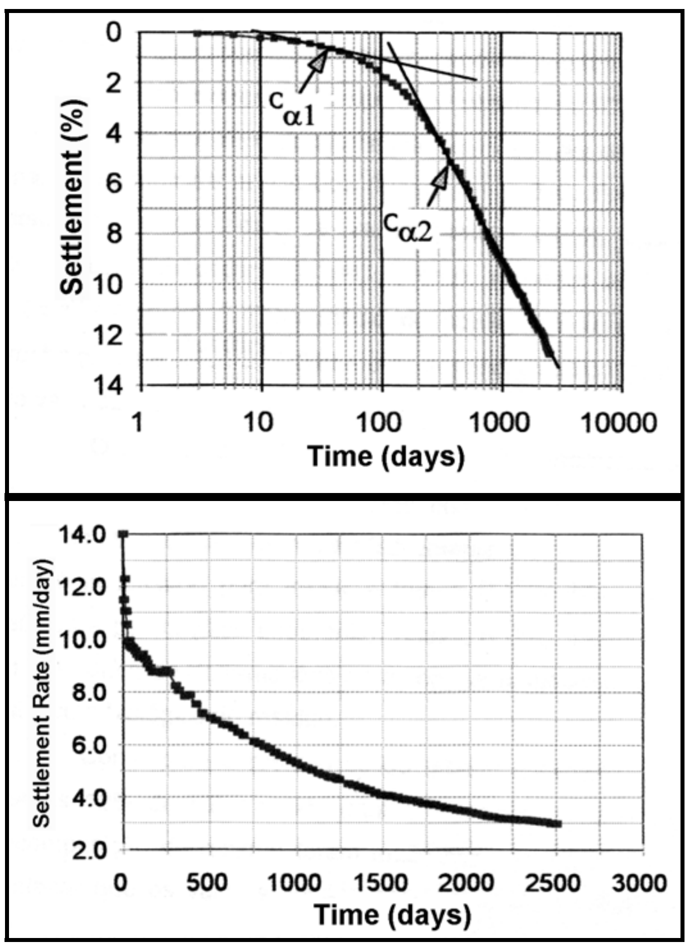

Figure 3.10 – Graphs of MSW Relative Settlement and Settlement Rate (Carvalho, 1999)

Wall and Zeiss (1995) did not notice the influence of biodegradation on the development of secondary settlements during 225 days based on the observation of the behavior of MSW settlement through the use of cells under two different conditions (one favorable to biodegradation and the other unfavorable). Even so, one cannot consider anything regarding the physical meaning of the variation of C'_{α} with time.

MSW settlements do not have an obligation to follow a logarithmic law and the truth is that they do not and cannot follow it. Since the settlement mechanisms between soils and MSW are different, it must be remembered that in several cases even soils do not follow a logarithmic law. It would be more reasonable to use other types of functions (for example, hyperboles or exponentials) than starting from a non-linear relationship, proceed to adjust it into a semi-log graph that also does not present linearity, and try to approximate it by segments of straight lines.

3.3.1.3. Yen and Scanlon's Logarithmic Model

The logarithmic model for settlements prediction based on settlement rates was introduced by Yen and Scanlon (1975) from field studies carried out by the authors through a nine-year observation period of the settlements of three California waste landfills after closure. The objectives of the research were to analyze the data, looking for common trends relating to settlement as a function of time for the years following construction of the three landfills. Based on field data, the factors that influence the settlement mechanisms were verified to furnish a real database in which future theoretical models could be tested.

The settlement analysis can be executed from the "instantaneous settlement rate" (v_{rp}). The instantaneous settlement rate expresses the settlement variation in a short period of time, for instance, one month.

$$v_{rp} = -\frac{\Delta H}{\Delta t} \quad \text{..........(3.13)}$$

ΔH ... change in elevation of survey monument

Δt ... elapsed time between topographic surveys

The variable time herein utilized is the "median age of a fill column", measured as the time elapsed between the date of the topographic survey and the date when the thickness of the "fill column" is 50% of the total.

The average age of the fill column (t_1) can be estimated as:

$$t_1 = t - \frac{t_c}{2} \quad \text{..........(3.14)}$$

t_c ... time elapsed since the beginning of landfill construction.

The term "fill column" is defined as the thickness of waste and cover soil disposed below a survey monument and above the natural soil. Other utilized variables are: H_f is the final thickness of "fill column" after construction, and t_c is the total construction time.

It must be noticed that Equation 3.14 is only valid if the curve landfill construction height is a linear function of time, which is not true for waste landfills with cells with different disposal areas.

Figure 3.11 shows what is described above.

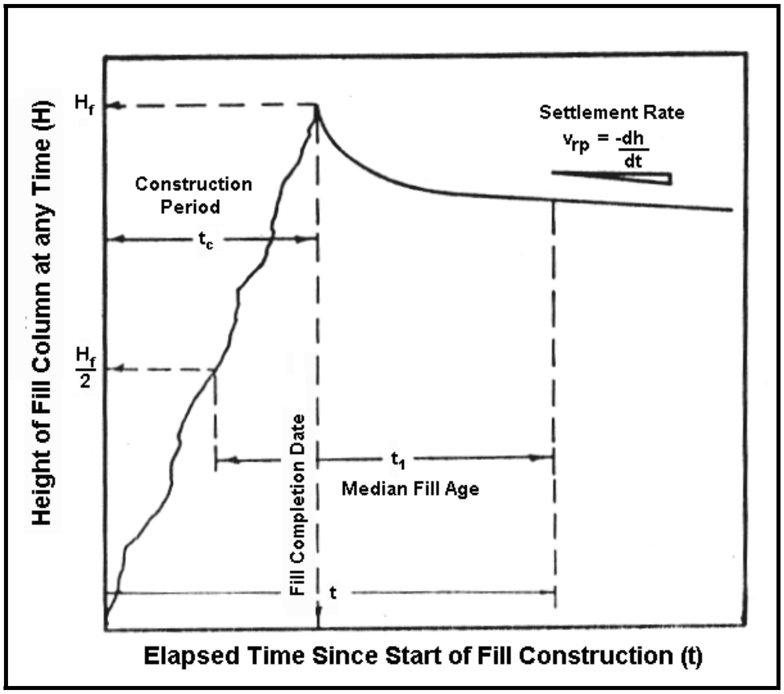

Figure 3.11 – Schematic Diagram: Landfill Thickness as a Function of Time (Yen and Scanlon, 1975)

Settlement data were plotted into graphs of instantaneous settlement rate (v_{rp}) as a function of time, as presented in Figure 3.12, and from regression analyses for different landfill column thicknesses and construction times. The authors presented the following relationship:

$$v_{rp} = m - n \cdot \log t_1 \qquad (3.15)$$

where m and n are empirical constants.

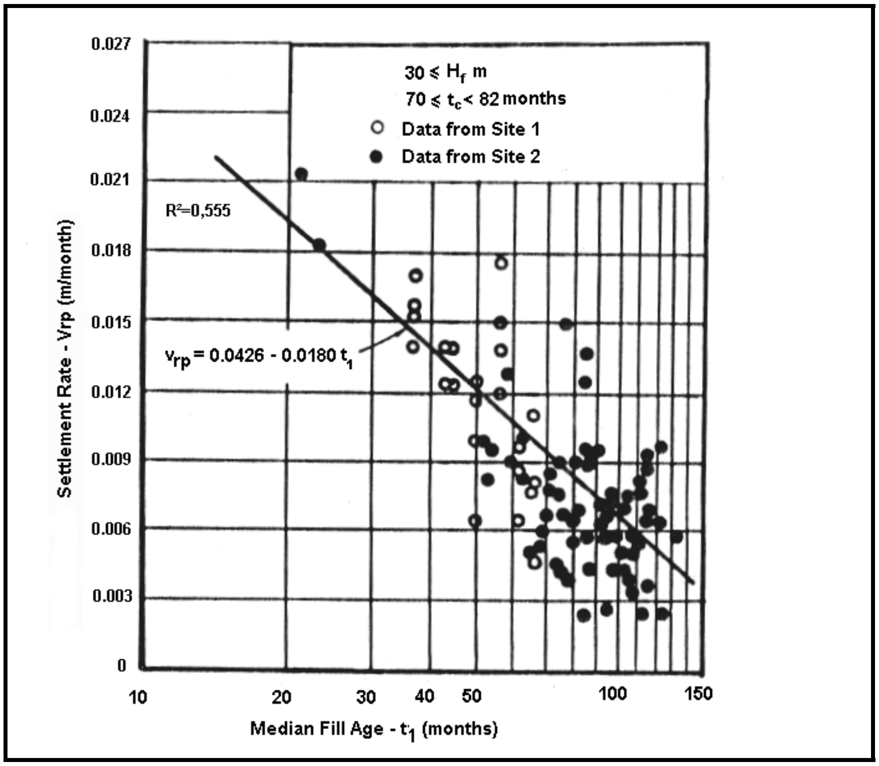

Figure 3.12 – Graph of Instantaneous Settlement Rate as a Function of the Average Age of the Fill Column for Fill Column Thickness Greater than 30 m and Construction Time Between 70 and 82 Months (Yen and Scanlon, 1975)

Table 3.3 presents the average values for settlement rates from the results of the regression analyses for different landfill column thicknesses and construction times.

According to Yen and Scanlon (1975), for values above 27 m, there is no difference for the settlement rates. This can be explained due to the fact that in large depths there is a predominance of an anaerobic environment, in which biodegradation occurs slower than for points closer to the surface with aerobic conditions favorable to biodegradation, and, therefore, generating larger settlement rates. Based on this reasoning, however, smaller thicknesses would have larger settlement rates, which do not occur since the anaerobic phase is dominant even for small depths. Palma (1995) also disagrees with this explanation.

Table 3.3 - Average Values for Settlement Rates from the Results of the Regression Analyses for Different Landfill Column Thicknesses and Construction Times (Yen and Scanlon, 1975; Modified) – Values in mm/day

	t_1 < 40 months			40 ≤ t_1 < 60 months			60 ≤ t_1 < 80 months			80 ≤ t_1 < 100 months			100 ≤ t_1 < 120 months		
H_f (m)	t_c ≤12 months	24≤t_c≤50 months	70≤t_c≤82 months	t_c ≤12 months	24≤t_c≤50 months	70≤t_c≤82 months	t_c ≤12 months	24≤t_c≤50 months	70≤t_c≤82 months	t_c ≤12 months	24≤t_c≤50 months	70≤t_c≤82 months	t_c ≤12 months	24≤t_c≤50 months	70≤t_c≤82 months
<12	-	-	-	0.162	0.162	0.152	0.162	0.101	0.091	0.081	0.121	-	-	-	-
12-24	0.303	-	0.303	0.101	0.263	0.293	0.091	0.121	0.162	-	0.121	0.081	-	-	0.152
25-31	0.505	-	-	0.303	-	0.404	0.364	-	0.253	-	-	0.222	-	-	0.202
>31	-	-	0.576	-	-	0.414	-	-	0.253	-	-	0.253	-	-	0.202

(-) indicates that less than 3 superficial marks data are available and, therefore, m value was not calculated.

Sohn and Lee (1994) utilized the study of Yen and Scanlon (1975), and also data from Rao et al. (1977), to present a formulation which allows the estimation of parameters *m* and *n* of Equation 3.15, based only on the landfill final thickness (H_f). Figure 3.13 presents the graph of parameters *m* and *n*, when plotted as a function of the landfill final thickness (H_f).

The following relationships are obtained by linear regression:

$$m = 0{,}00095 \cdot H_f + 0{,}00969 \qquad (3.16)$$

$$n = 0{,}00035 \cdot H_f + 0{,}00501 \qquad (3.17)$$

where H_f is in meters, and the settlement rate is in meters/month.

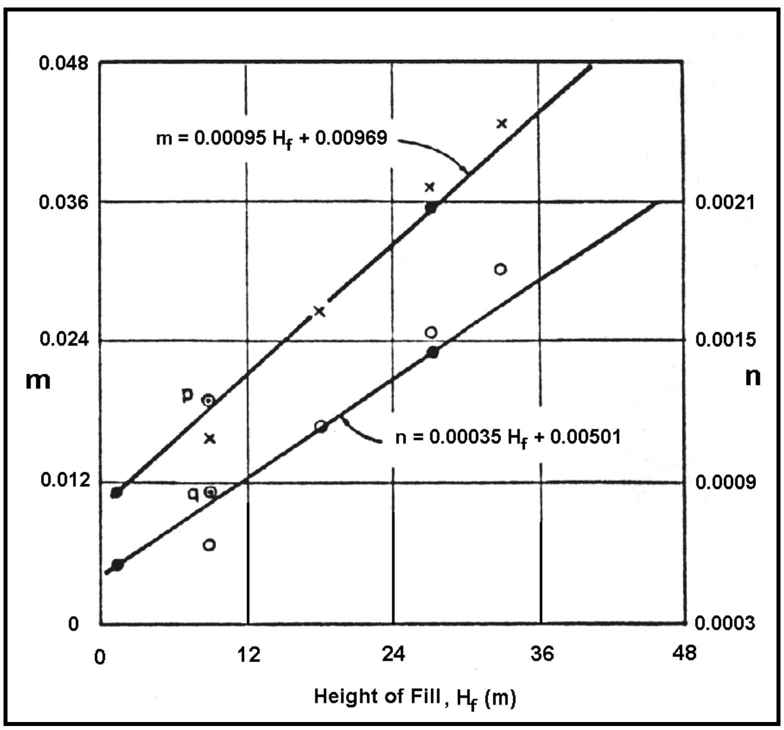

Figure 3.13 – Determination of Parameters of Settlement Rate (Sohn and Lee, 1994; modified)

However, the main weakness of this model is that even for the study of Yen and Scanlon (1975), there is a very weak correlation among the data. Other authors that utilized this model, such Ling et al. (1998), Carvalho (1999), De

Abreu (2000), and Marques (2001) have also obtained very weak correlations for their data, which did not validate this approach.

3.3.1.4. Attenuation Equation Application

Coumoulos and Koryalos (1997) discussed the studies of Yen and Scanlon (1975) and Sohn and Lee (1994) and noticed that the formulation produces settlement rates equal to zero after a certain period of time. This is not observed in the field. The authors' proposition is an equation that landfill settlements can be approximated by a straight line as a function of the logarithm of time, as in Equation 3.8.

By the derivation of Equation 3.8, the attenuation equation is produced:

$$y = \frac{d\left(\Delta h / H\right)}{dt} = \frac{0{,}434 \cdot C'_{\alpha}}{t} \quad \text{...}(3.18)$$

where *y* is the vertical strain rate, being expressed in %/month or %/year.

Figure 3.14 schematically presents the time-settlement curves of two similar "columns" with different construction periods. Figure 3.15 shows the attenuation curves from columns A and B presented on Figure 3.14.

The main advantage of this model is that data from different points on the landfill with different characteristics can be grouped and compared. It must be noted, however, that the accuracy of *y* depends on the accuracy of C'_{α}, which is not constant, as seen before.

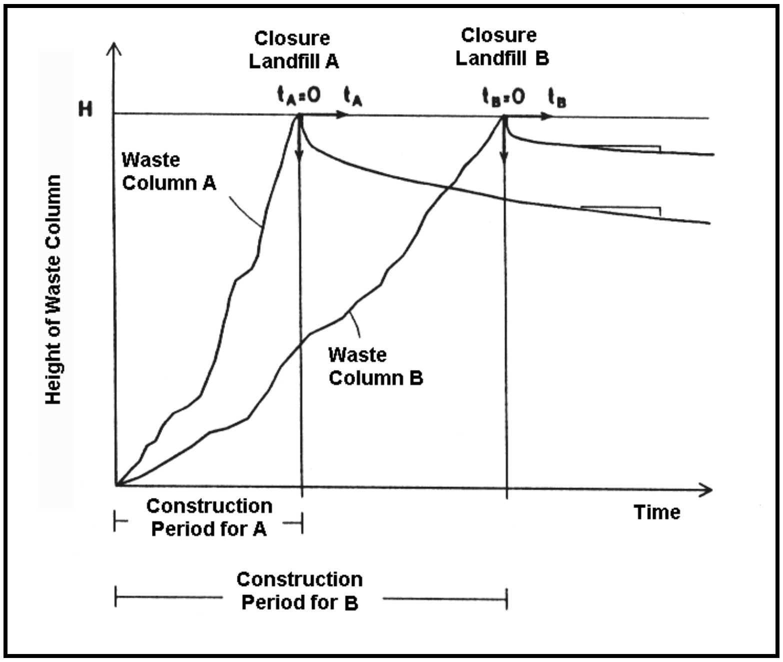

Figure 3.14 – Schematic Curves Time-Settlement of Two Similar Waste Columns with Different Construction Times (Coumoulos and Koryalos, 1997)

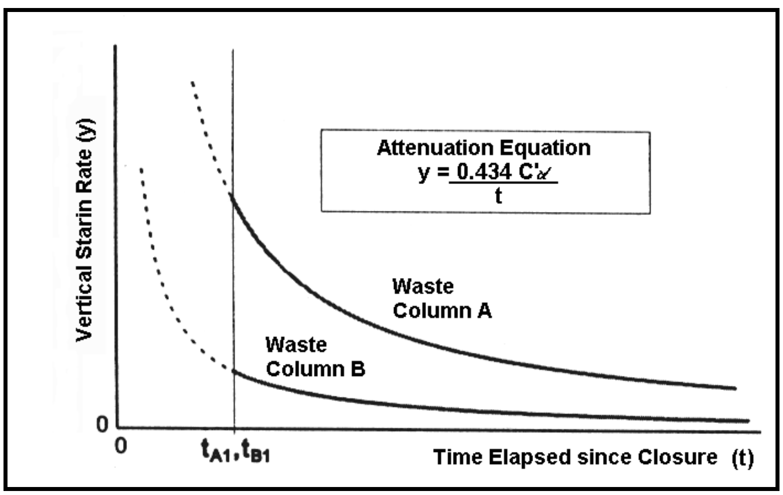

Figure 3.15 – Attenuation Curves of Strain Rates (Coumoulos and Koryalos, 1997)

Whenever the closure date is not known, which can happen with older sanitary landfills, a procedure can be done by placing the time elapsed since landfill closure (t_{c^*}), which is unknown, into the attenuation equation, as shown in Equation 3.19, and with the execution of three readings in any two different intervals of time. So, C'_{α} and t_{c*} can be determined as presented in Figure 3.16 and Equations 3.19, 3.20, and 3.21.

$$y = \frac{0{,}434 \cdot C'_{\alpha}}{t_{c*} + t} \qquad \text{(3.19)}$$

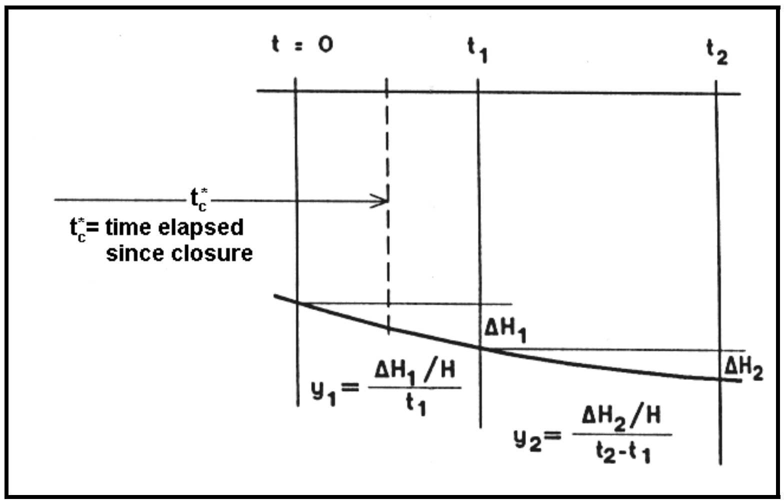

Figure 3.16 – Methodology for C'_{α} and t_{c*} Determination (Coumoulos and Koryalos, 1997)

$$y_1 = \frac{0{,}434 \cdot C'_{\alpha}}{t_{c*}} \qquad \text{(3.20)}$$

$$y_2 = \frac{0{,}434 \cdot C'_{\alpha}}{t_{c*} + t_2/2} \qquad \text{(3.21)}$$

By using the presented methodology, and applying it to several published data from North American and European landfills, the authors present attenuation curves with adjustment in the C'_{α} interval between 0.02 and 0.25, as shown in Figure 3.17.

The Coumoulos and Koryalos proposition presents the same problems mentioned in the Sowers proposition review (C'_{α} not constant, non-validity of logarithmic law), since both have the same conceptual origin.

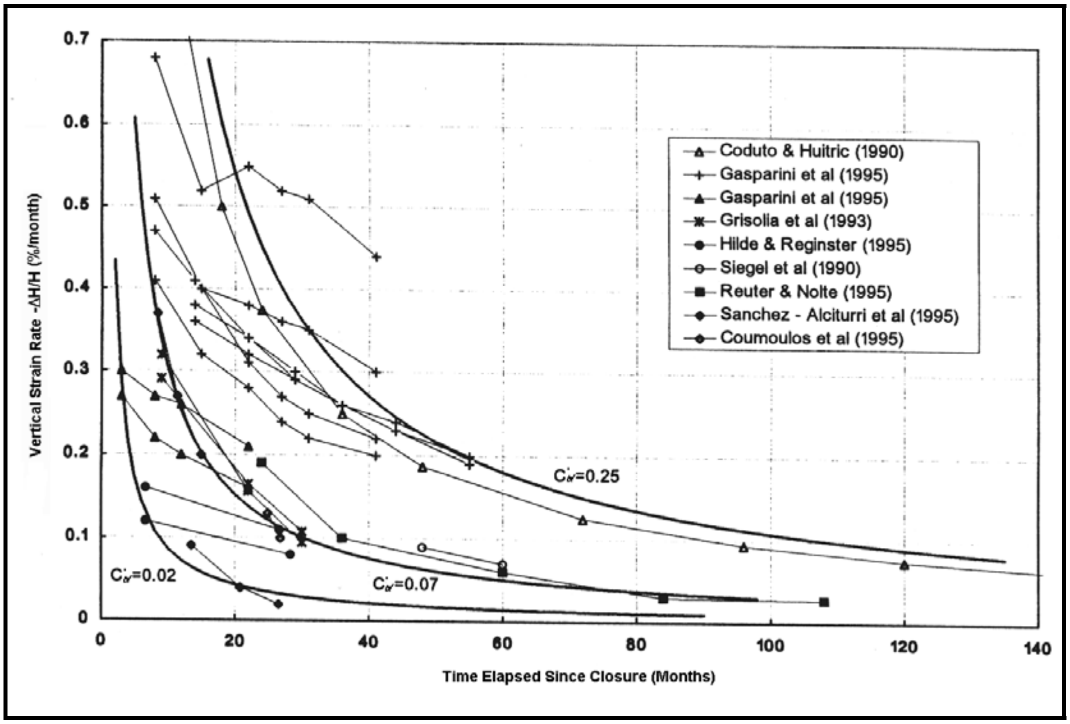

Figure 3.17 – Adjustment from Several Published Data to the Attenuation Curve Proposed by Coumoulos and Koryalos (1997)

3.3.2. Zimmerman et al. Mathematical Model

The behavior time-settlement of MSW was modeled by Zimmerman et al. (1977) by simultaneously utilizing two equations. The first represents the dissipation of pore-pressures with time and is based on the theory of mixture, which takes into consideration the effects of finite strain, biological and chemical transformations, and the time variation of saturation. The second equation represents the "creep" behavior and is modeled using a rheological model with parameters that change as the settlements develop, taking into account large strains.

The simplified hypotheses adopted are: homogeneous material, completely saturated, with incompressible fluid and incompressible solid particles, but degradable, as well as the consideration of a linear relationship between void ratio and the logarithm of permeability, and between void ratio and the logarithm of effective stress.

Whenever compression occurs in total saturation conditions, constant compressibility and permeability, low deformations, and without chemical and biological alterations, the model is reduced to Terzaghi's Model.

In the modeling of "creep" phenomenon, a rheological model is proposed, composed of a double dashpot with a non-linear spring to simulate the physical mechanism of MSW primary and secondary compression phases. In this model, the MSW is considered as a random agglomerate of structural entities with micro-pores inside larger structures containing macro-pores.

Figure 3.18 presents the rheological model introduced, in which the top non-linear spring represents the macro-compressibility of the MSW structure and the bottom part simulates the micro-compressibility of the porous structure. The pistons, which are also porous, represent the average permeability of the macro-pores (top part) and micro-pores (bottom part).

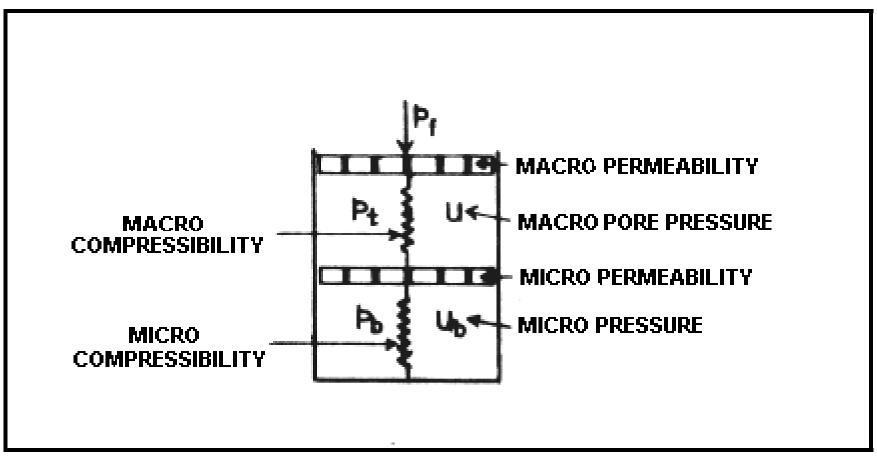

Figure 3.18 – Rheological Model Based on the Mechanism of Macro and Micro-Pores (Zimmerman, 1977)

- The equation that represents the dissipation of pore pressure is:

$$\frac{e_0 - e_f}{e_{t0} - e_{tf}} \cdot \frac{k_0 \cdot (1+e_0)^2}{\gamma_w \cdot a_{v0}} \cdot \frac{\partial}{\partial x}\left[\frac{\left(\frac{k_f}{k_0}\right)^{\frac{e_0-e}{e_0-e_f}} \cdot \left(\frac{\sigma_f}{\sigma_0}\right)^{\frac{e_{t0}-e_t}{e_{t0}-e_{tf}}}}{1+e}\right] \cdot \frac{\partial e_t}{\partial x} + \frac{e \cdot \phi_0 \cdot C}{\gamma_w \cdot \delta_s} \cdot e^{-C \cdot t} = -\frac{\partial e}{\partial t}$$

..........................(3.22)

k_0 ... initial permeability

k_f ... final permeability

a_{v0} ... initial value of compressibility

γ_w ... unit weight of waster

δ_s ... specific gravity of solids

σ_0 ... initial effective stress

σ_f ... final effective stress

ϕ_0 ... initial weight of the decomposable part of the waste

C ... rate constant for decomposition

e ... void ratio

e_0 ... initial void ratio

e_f ... final void ratio

e_t ... time-independent portion of void ratio at any time

e_{t0} ... initial value of the time-independent portion of void ratio

e_{tf} ... final value of the time-independent portion of void ratio

- Equation that represents the “creep” phenomenon:

$$\frac{\partial e_b}{\partial t} = -A_0 \sigma_0 (1+e_0)\left(\frac{\sigma_f}{\sigma_0} - 1\right)\left(\frac{A_f}{A_0}\right)^{\frac{e_{b0}-e_b}{e_{b0}-e_{bf}}} \sinh\left\{\frac{B_0 \cdot \left(\frac{B_f}{B_0}\right)^{\frac{e_{b_0}-e_b}{e_{b_0}-e_{bf}}}}{\left(\frac{\sigma_f}{\sigma_0} - 1\right)} \cdot \left[\left(\frac{\sigma_f}{\sigma_0}\right)^{\frac{e_{t0}-e_t}{e_{t0}-e_{tf}}} - \left(\frac{\sigma_f}{\sigma_0}\right)^{\frac{e_{b0}-e_b}{e_{b0}-e_{bf}}}\right]\right\}$$

......................(3.23)

A_0 e B_0 ... initial rheological parameters

A_f e B_f ... final rheological parameters

e_b ... time-dependent portion of void ratio at any time

e_{b0} ... initial value of the time-dependent portion of void ratio

e_{bf} ... final value of the time-dependent portion of void ratio

Equations 3.22 and 3.23 define the MSW compressibility for the case of 100% saturation, but do not take into consideration the generation of gases and liquids.

The presented model is non-linear, but can be solved by numerical methods. The determination of the many parameters (total of 19) that, according to the authors, can be done through laboratory and field tests is very complicated and it is uncertain that it will correctly reproduce the phenomenon.

Parametric studies were conducted with some of the variables in order to establish limit values that can be considered as constants, without the introduction of large errors in the equations' solving process. The idea was to reduce the number of variables without loss of the precision of the model. Although the idea was good, only a few incipient studies were done (Chen et al., 1977).

It is obvious that the model can also be applied through the fundamental equations for the cases of non-constant saturation and considerations of gas and liquid generation, resulting in equations much more complex, though.

Despite the fact that this model takes into consideration most of the settlement mechanisms presented in Section 3.2, the excessive number of parameters required and the difficulties of obtaining them make this model impractical for practice.

3.3.3. Isotaches Model (or Model abc)

The Isotaches model, developed in the Netherlands by Den Haan (1994) for utilization in the study of secondary (or secular, according to Den Haan) compression of soft clays and peats, was re-written and adapted by Van Meerten et al. (1995) to be applied in sanitary landfills. The model has, in logarithm scale, the specific volume (V), the vertical effective stress (σ_v), and the "intrinsic time" as having a linear relationship.

The model proposed by Van Meerten (1995), introducing some concepts of Den Haan's theory, is presented as the following:

The equation that describes the total natural strain can be written as:

$$\frac{d\varepsilon^H}{dt} = \frac{d\varepsilon_d{}^H}{dt} + \frac{d\varepsilon_s{}^H}{dt} \qquad \text{.......(3.24)}$$

ε^H ... total natural specific strain,

$\varepsilon_d{}^H$ ... natural specific strain due to direct compression (loadings),

$\varepsilon_s{}^H$ ... natural specific strain due to secular compression.

The natural specific strain (ε^H) is expressed in logarithmic terms, therefore being presented in a different way to the linear specific strain of Cauchy (ε^C), and following the expression:

$$\varepsilon^H = -\int \frac{dV}{V} = -\ln\frac{V}{V_0} \qquad \text{.......(3.25)}$$

V ... specific volume (ratio between the total volume and the solids volume),

V_0 ... initial specific volume.

$$V = 1 + e \qquad \text{.......(3.26)}$$

e ... void ratio

The Isotaches model is also denominated “Model abc”, due to the compressibility parameters denominated as a, b, and c - which have some relationship with the coefficient of primary and secondary compressions (C'_c and C'_α, respectively) from the classical model.

The terms “direct compression” and “secular compression” are adopted by Van Meerten to express, respectively, the primary compression and the secondary compression, since the effects of both do not occur separately. So, independently when they occur, the “direct compression” is related only with the part of settlement due to loading action, and the “secular compression”, with the slow deformation that happens with time.

For the primary (or direct) compression, Den Haan (1996) arbitrarily assumes that the strain rate due to the primary compression ($d\varepsilon_d^H/dt$) is linearly related to the vertical effective stress (σ'_v) and the rate of increasing vertical effective stress ($d\sigma'_v/dt$).

The strain due to the direct compression can be written as:

$$\frac{d\varepsilon_d^{\ H}}{dt} = a \cdot \frac{d\ln(\sigma'_v)}{dt} = \frac{a}{\sigma'_v} \cdot \frac{d\sigma'_v}{dt} \quad \text{...(3.27)}$$

a ... direct compression index,

σ'_v ... vertical effective stress.

For the effect of secular compression, the Isotaches model is introduced. Den Haan shows that for a large variety of normally consolidated soils, the following equation can be applied:

$$V = V_1 \cdot \left(\sigma'_v - \sigma'_{vs}\right)^{-b} \quad \text{...(3.28)}$$

V ... specific volume (ratio between the total volume and the solids volume),

V_1 ... specific volume related to the reference value of effective stress σ'_{vs}+1 kPa ,

σ'_{vs} ... vertical effective stress in which the asymptote of the curve specific volume as a function of the vertical effective strain is verified,

b ... secular compression index or natural compression index.

Figure 3.19 shows this relationship.

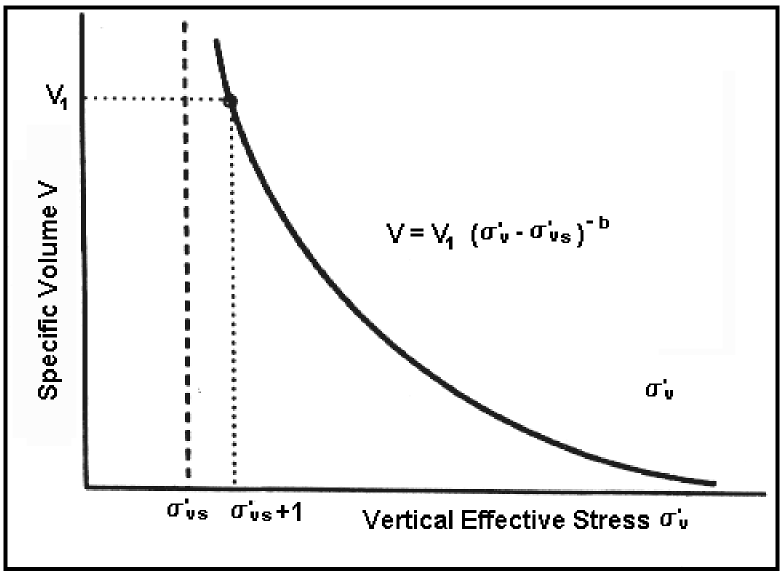

Figure 3.19 – Curve Specific Volume as a Function of Vertical Effective Stress (Den Haan, 1994)

The natural compression index *b* can also be written as:

$$b = \frac{-\Delta \ln(V)}{\Delta \ln(\sigma'_v)} \quad \text{.......(3.29)}$$

Similarly, Den Haan (1994) observing the curves of strain as a function of time for secular compression of Dutch peats could conclude that:

- The curves of linear strain as a function of log-time are not linear (Figure 3.20a) and demonstrates again the non-linearity of C_α and C'_α.

- Although, most of the time, the curves of natural strain as a function of log-time are non-linear (Figure 3.20b), the author noticed that they have some parallelism, which could indicate the existence of a secular compression coefficient constant with time.

- There is a linear relationship between the natural deformation and the logarithmic of the linear strain rate (Figure 3.20c). So, the following equation can be written:

$$\varepsilon_s{}^H - \varepsilon_{s0}{}^H = c \cdot \left(\ln\left(\frac{d\varepsilon_{s0}{}^H}{dt} \right) - \ln\left(\frac{d\varepsilon_s{}^H}{dt} \right) \right) \quad \text{...(3.30)}$$

c ... coefficient of secular compression rate or natural secondary compression index,

$\varepsilon_s{}^H$ ... secular natural specific strain,

$\varepsilon_{s0}{}^H$ ... initial secular natural specific strain.

Then the concept of intrinsic time (τ) is introduced, as being:

$$\tau = \frac{c}{d\varepsilon_s{}^H / dt} \quad \text{..(3.31)}$$

So, combining the Equations 3.25, 3.28, and 3.30, the model proposes that vertical effective stress, specific volume, and intrinsic time follow a linear relationship when in logarithmic scale, and can be expressed as:

$$\frac{d\varepsilon_s{}^H}{dt} = \frac{d\varepsilon_{s0}{}^H}{dt} \cdot \left(\frac{V \cdot \sigma'_v{}^b}{V^*_1} \right)^{1/c} \quad \text{...(3.32)}$$

V^*_1 ... correspondent specific volume to σ'_v = 1 kPa

Or, presenting Equation 3.32 in terms of initial specific volume (V_0) and initial vertical effective stress (σ'_{v0}):

$$\frac{d\varepsilon_s{}^H}{dt} = \frac{d\varepsilon_{s0}{}^H}{dt} \cdot \left(\frac{V \cdot \sigma'_v{}^b}{V_0 \cdot \sigma'_{v0}{}^b} \right)^{1/c} \quad (3.33)$$

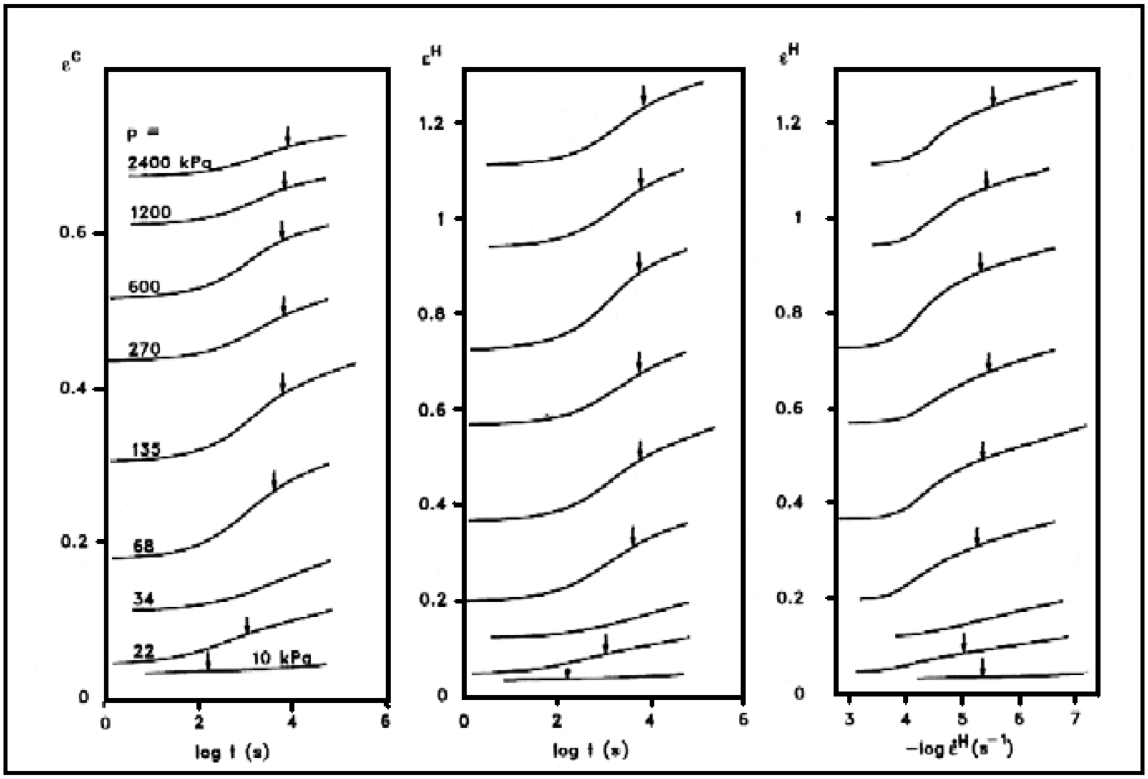

Figure 3.20 – Curves Settlement as a Function of Time for Dutch Peats. (a) Linear Strain as a Function of Log-Time. (b) Natural Strain as a Function of Log-time. (c) Natural Strain as a Function of the Logarithm of Natural Strain Rate. The arrows approximately indicate the ending of the primary consolidation by Taylor's method (Den Haan, 1994)

Figure 3.21 shows the generic surface obtained for linear and logarithm scales. Figure 3.22 presents the same relationship on the plane specific volume – vertical effective stress, with isotaches representing the curves of the same rate of secular natural specific strain (or same intrinsic time).

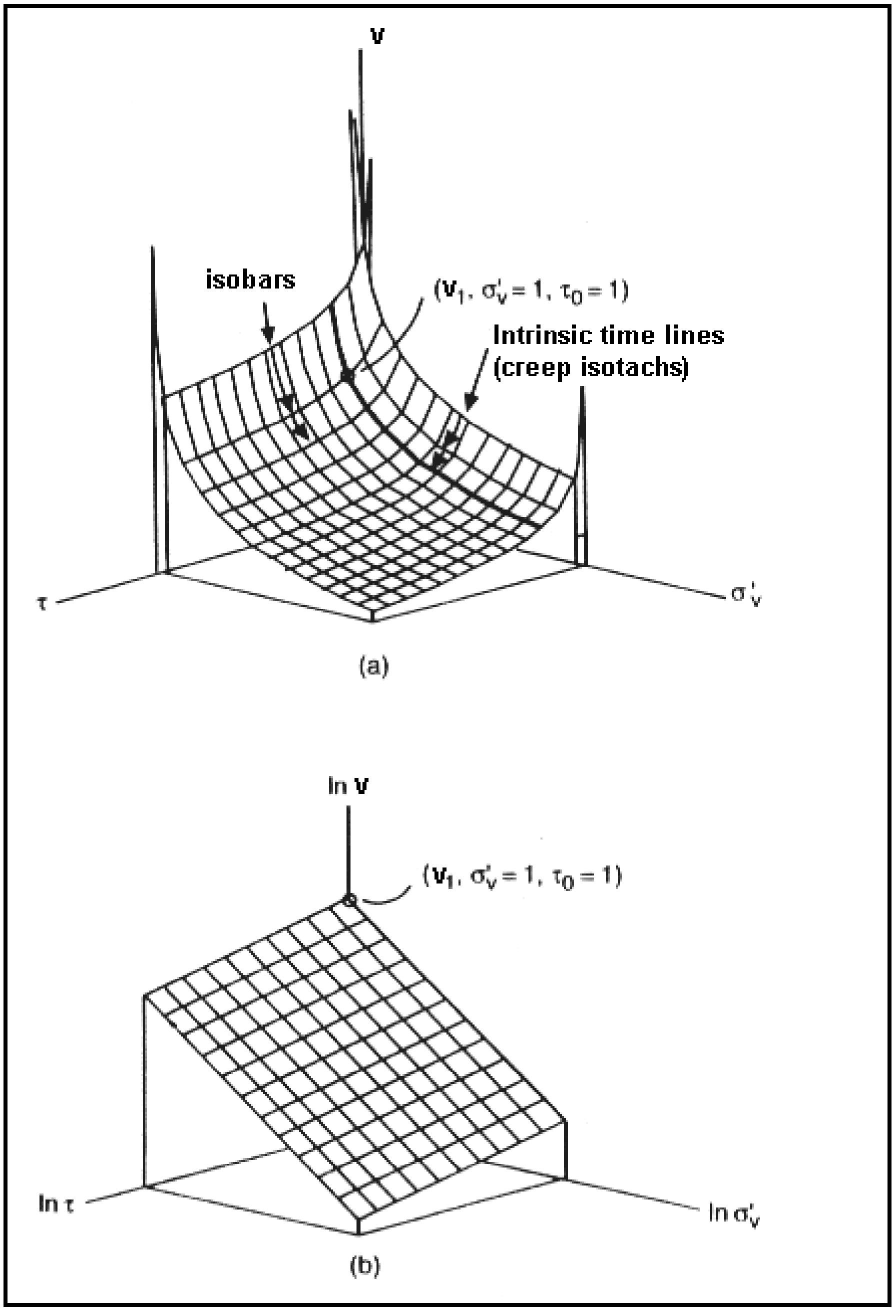

Figure 3.21 – Surfaces Obtained in Natural Scale (a), and Logarithm Scale (b) from the Relationship Vertical Effective Stress – Specific Volume – Intrinsic Time (Den Haan, 1996)

Note that the axis referent to intrinsic time can be substituted for the rate of secular natural specific strain.

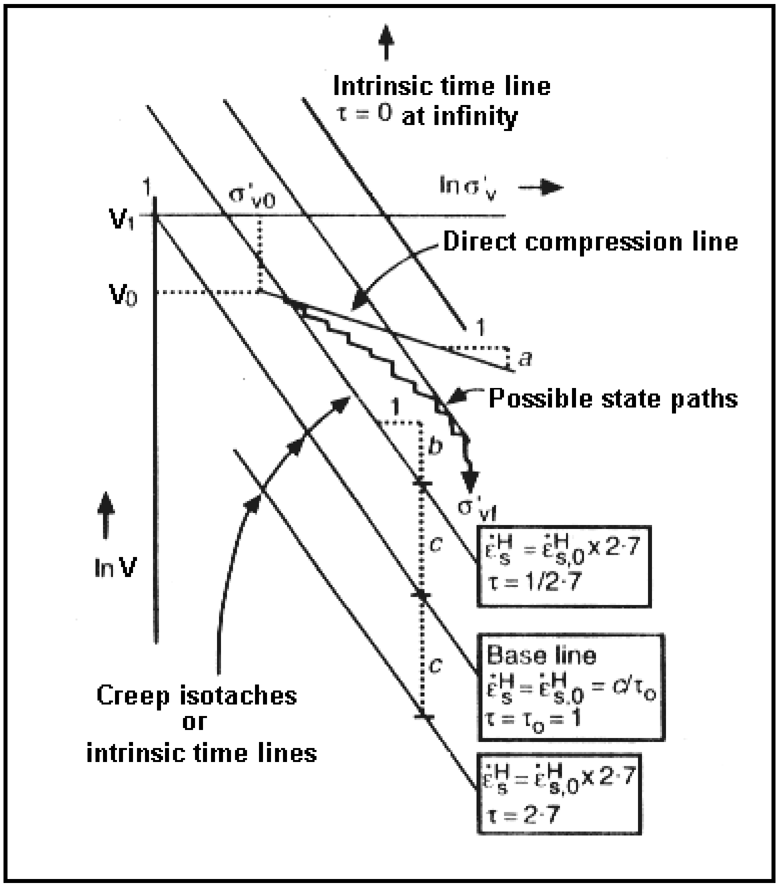

Figure 3.22 – Projections of the Isotaches of Secular Natural Specific Strain to the Specific Volume – Vertical Effective Stress Plane (Den Haan, 1996)

So, substituting Equations 3.27 and 3.33 into Equation 3.24, Van Meerten et al. (1995) presents the base equation for prediction of settlements in waste landfills:

$$\frac{V}{V_0} = \left[\left(\frac{\sigma'_v}{\sigma'_{v0}} \right)^{\frac{a}{c}} + \left(\frac{\sigma'_v}{\sigma'_{v0}} \right)^{\frac{a}{c}} \cdot \frac{d\varepsilon_{s0}/dt}{c} \cdot \int_0^t \left(\frac{\sigma'_v}{\sigma'_{v0}} \right)^{\frac{b-a}{c}} \cdot dt \right]^{-c} \quad \text{......................................(3.34)}$$

V_0 ... initial specific volume,

σ'_{v0} ... initial vertical effective stress.

Equation 3.34 can be simplified for the case of one-dimensional compression (considering the volume per area unit), relating the increasing vertical effective stress to the thickness of disposed MSW, and adopting the landfill construction as Figure 3.23.

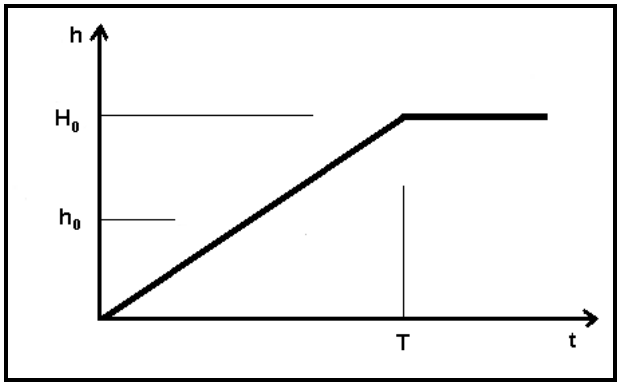

Figure 3.23 – Simplification of Landfill Construction Rates without Settlement Consideration (Van Meerten et al., 1995)

Considering H_0 as the landfill final height, without settlement consideration, and T as the time for landfill construction, the formulation for determination of a waste landfill height (h) at a given time t is:

$$h = \int_{\sigma'_{v_0}}^{\sigma'_{v_0}+\gamma\cdot h_0} \left[\left(\frac{\sigma'_v}{\sigma'_{v_0}}\right)^{\frac{b}{c}} + \left(\frac{\sigma'_v}{\sigma'_{v_0}}\right)^{\frac{b}{c}} \cdot \frac{d\varepsilon_{s0}/dt \cdot T}{c} \cdot \left\{ \frac{\sigma'_v}{\gamma\cdot H_0} \cdot \frac{1-\left(\sigma'_{v_0}/\sigma'_v\right)^{\frac{c+b-a}{c}}}{\frac{c+b-a}{c}} + \frac{t}{T} - \frac{h_0}{H_0} \right\} \right]^{-c} \cdot \frac{d\sigma'_v}{\gamma}$$

......................(3.35)

γ ... initial MSW unit weight,

h_0 ... landfill height without settlement consideration at a time t.

However, Equation 3.35 can be very difficult to apply for a practical case due to difficulties associated with the parameters utilized. So the model is transformed into an instrument of mathematical adjustment for being utilized with field surveys.

Grouping the common factors on Equation 3.35, the following relationships can be written:

$$N = \frac{\gamma \cdot H_0}{\sigma'_{v_0}} \quad \text{..(3.36)}$$

N ... number of constructive layers in landfill

$$P = \left(\frac{\sigma'_v}{\sigma'_{v_0}} \right) \quad \text{..(3.37)}$$

$$E_i = \frac{d\varepsilon_{s0}/dt \cdot T}{c} \quad \text{..(3.38)}$$

E_i ... initial strain factor

$$\alpha = \frac{a}{c} \quad \text{..(3.39)}$$

α ... direct compression factor

$$\beta = \frac{b}{c} \quad \text{..(3.40)}$$

β ... secular compression factor

From those new parameters, the final equation for prediction of settlements in sanitary landfills can be written:

$$\frac{h}{H_0} = \frac{1}{N} \cdot \int_1^{1+N \cdot \frac{h_0}{H_0}} \left[1 + P^b \cdot E_i \cdot \left\{ \frac{t}{T} - \frac{h_0}{H_0} - \frac{P}{N} \cdot \frac{1 - P^{\alpha-\beta-1}}{\alpha - \beta - 1} \right\} \right]^{-c} \cdot dP \quad \text{........................(3.41)}$$

Therefore, the settlement is the difference between the height H_0 (without settlements) and the height h determined on Equation 3.41.

According to the authors, the direct compression index α has a low influence and can be assumed as zero. This can only be adopted due to the direct compression component being incorporated most of the time to the slow deformation of the landfill. The construction of a landfill is generally slow (and, therefore, the construction of each cell cannot be considered as an instantaneous load). Furthermore, the initial strain factor E_i can be adopted as any fixed value to make the model dependent only on two parameters (β and c), because E_i is very difficult to be distinguished from the other scale parameters.

The last consideration reduces the model to an adjustment of settlement-time curves, becoming impossible for use in sanitary landfills design.

Van Meerten et al. (1995) presented an example of the application of Equation 3.41 from the settlement monitoring of a sanitary landfill in The Netherlands. The monitoring of approximately 3.5 years recorded settlement data that was the input to the modeling and adjustment of a curve of settlement prediction as a function of time. Figure 3.24 shows the results obtained as a graph of height versus time.

However, it must be noticed that the model herein presented does not directly incorporate the effects of biodegradation, which, in a first approximation, are included in the formulation of secular compression.

This model could be very interesting for application to MSW. It introduces the concept in which every point of the landfill has its own relationship: vertical effective stress – specific strain – time. However, the excessive simplification adopted by Van Meerten et al. (1995) reduced the model into an adjustment of the curve settlement-time.

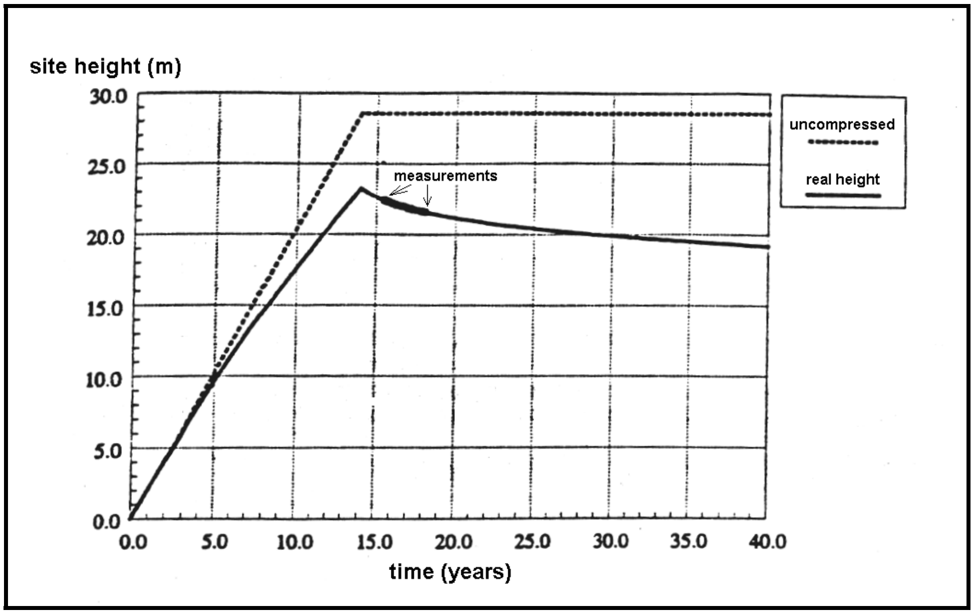

Figure 3.24 – Prediction of the Height of Kragge Landfill, Holland, Using Historical Data Series from Settlement Monitoring (Van Meerten et al., 1995)

3.3.4. Gibson and Lo Rheological Model

Due to the difficulties in separating the effects of primary and secondary compressions, Edil et al. (1990) adopted models that combine the two settlement stages for application in waste landfills. One of those models is the rheological model proposed by Gibson and Lo in 1961 for application to peaty soils. Peats as well as MSW present similar settlement behavior: fast initial and primary compressions, but with much lower values when compared to secondary compression.

The rheological model consists in the representation, by analogy, of the primary compression by a spring, which expresses the immediate deformation and the secondary compression by a piston and a spring, which expresses the slow deformation, linked in parallel and in series, as shown on Figure 3.25.

Whenever there is a load increment (for example, the construction of new cells), the spring, which linearly deforms and has a* as a constant, immediately is compressed. The spring arrangement compression, which has b* as a constant, is retarded due to the dashpot (which has λ/b* as viscosity and is arranged in parallel), and the loading applied to the spring with constant b* is progressively transferred to the dashpot. After a certain period of time (at the end of the secondary compression), all loading will be absorbed only by the springs, and the dashpot will not sustain any part of the total applied loading.

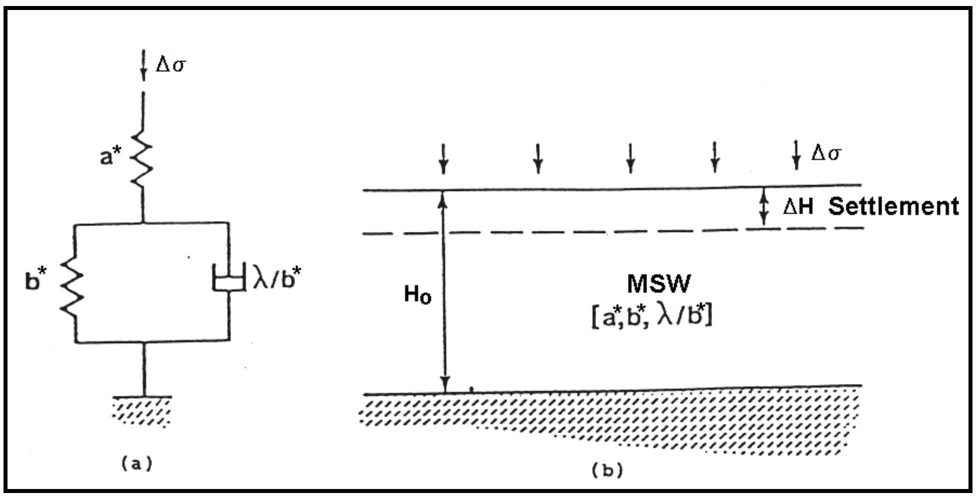

Figure 3.25 – Schematic Representation of the Rheological Model (Edil et al., 1990)

Equation 3.42 presents the relationship time-settlement of the presented model.

$$\Delta h = H_0 \cdot \Delta\sigma \cdot \left[a^* + b^* \cdot \left(1 - e^{-\frac{\lambda}{b^*} \cdot t} \right) \right] \qquad \text{...(3.42)}$$

Δσ ... increasing vertical effective stress

t ... time elapsed since loading beginning

Edil et al. (1990) in their study obtained reasonable results when compared the model to the observed settlements of four sanitary landfills with very different characteristics in Wisconsin, Michigan, and Connecticut.

Figures 3.26, 3.27, and 3.28 show graphs of the variation of parameters a*, b*, and λ/b*. Table 3.4 presents the variation range for parameters a*, b*, and λ/b* obtained from the study.

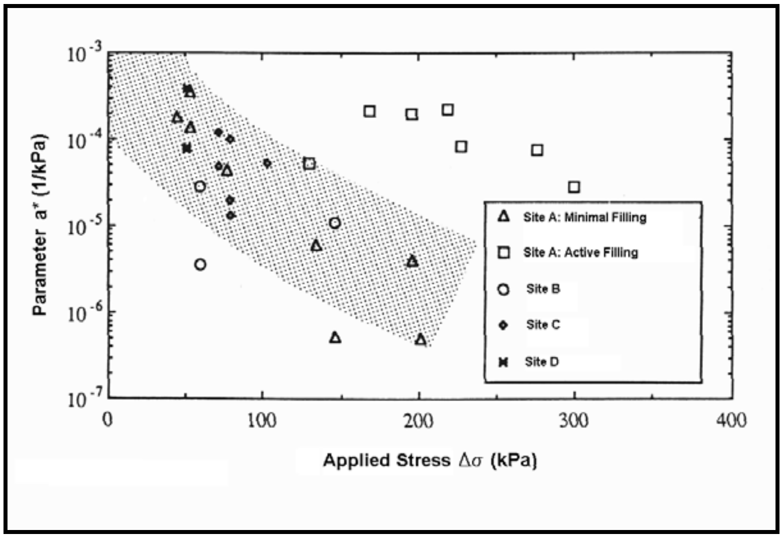

Figure 3.26 – Graph of Parameter a* as a Function of the Applied Stress ($\Delta\sigma$) (Edil et al., 1990)

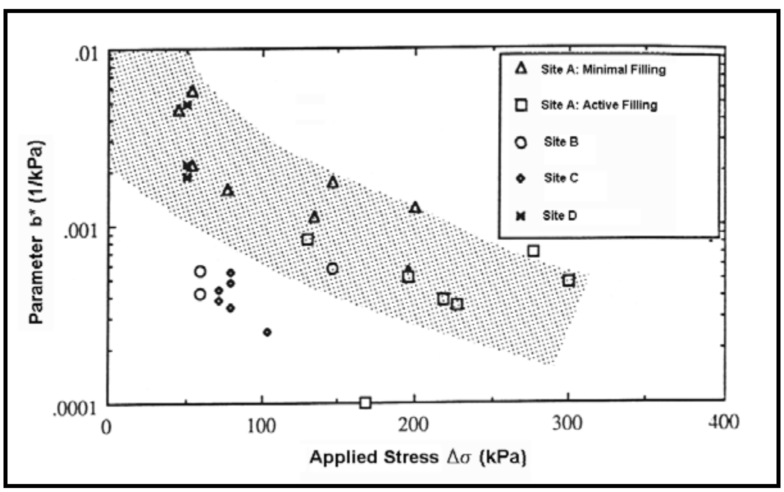

Figure 3.27 – Graph of Parameter b* as a Function of the Applied Stress ($\Delta\sigma$) (Edil et al., 1990)

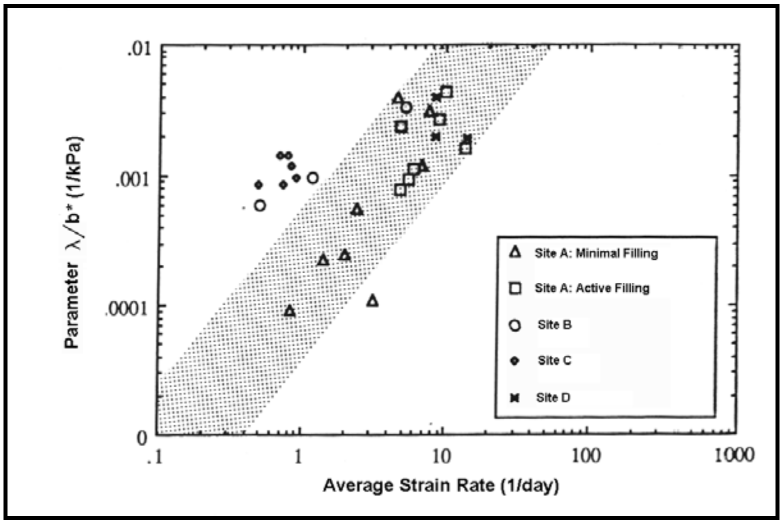

Figure 3.28 – Graph of Parameter λ/b* as a Function of the Average Strain Rate (Edil et al., 1990)

Table 3.4 – Variation Range of the Gibson and Lo Rheological Model Parameters in the Study of Edil et al. (1990)

Landfill	Characteristics	Variation range for a* (1/kPa)	Variation range for b* (1/kPa)	Variation range for λ/b* (1/dias)
A	Wisconsin; MSW with ages of 0 to 4 years; concluded and under operation cells; Δσ between 45 kPa and 200 kPa	$5{,}11\times10^{-7}$ to $3{,}52\times10^{-4}$	$1{,}00\times10^{-4}$ to $5{,}87\times10^{-3}$	$9{,}20\times10^{-5}$ to $4{,}30\times10^{-3}$
B	Michigan; MSW with ages of 1 to 16 years; concluded and under operation cells; Δσ between 59 kPa and 146 kPa	$3{,}60\times10^{-6}$ to $2{,}80\times10^{-5}$	$4{,}10\times10^{-4}$ to $5{,}70\times10^{-4}$	$6{,}00\times10^{-4}$ to $3{,}30\times10^{-3}$
C	Connecticut; MSW with ages of 40 to 50 years; excavated and recompacted MSW; Δσ between 72 kPa and 103 kPa	$1{,}30\times10^{-5}$ to $1{,}20\times10^{-4}$	$2{,}50\times10^{-4}$ to $5{,}40\times10^{-4}$	$8{,}40\times10^{-4}$ to $1{,}40\times10^{-3}$
D	Pilot landfill and additional area composed by old MSW; Δσ= 51 kPa	$1{,}30\times10^{-5}$ to $1{,}20\times10^{-4}$	$1{,}90\times10^{-3}$ to $4{,}90\times10^{-3}$	$1{,}90\times10^{-3}$ to $4{,}00\times10^{-3}$

According to the authors, the adoption of the Gibson and Lo Rheological Model for settlement prediction has resulted in errors of 0 to 21% for the settlement plates monitored in landfill A at the end of two years after the prediction was made.

One of the advantages of this model is the introduction of an exponential model that follows a similar curve for waste biodegradation. However, there are some difficulties in evaluating $\Delta\sigma$ after a certain period of time for landfills which there is no history of disposal. A good application of this model can be in cases where there is an application of loads almost "immediately" as, for example, the stocking of soil cover on finished cells.

3.3.5. Creep Law Model

One of the simplest ways of determining the relationship time-strain under constant loading, and one of the more utilized ones to determine the strain behavior of several engineering materials is the denominated Creep Law. According to that law, the relationship time-settlement can be expressed as:

$$\Delta h = H_0 \cdot \Delta\sigma \cdot m^* \cdot \left(\frac{t}{t_{r^*}}\right)^{n^*} \qquad \text{...(3.43)}$$

m* ... reference compressibility,

n* ... rate of compression,

t_{r^*} ... reference time introduced into the equation to make the variable time dimensionless (t_{r^*} = 1 day, the authors suggest).

Edil et al. (1990) also applied the model to the study of the four sanitary landfills mentioned in Section 3.3.4. According to the authors, the use of this model produced better adjustments to the data series in comparison to the ones obtained by the Rheological Model of Gibson and Lo with errors between 0 and 14% for landfill A.

Table 3.5 presents the variation range for parameters m* and n* obtained from the study.

As the Gibson and Lo Rheological model, the Creep Law model also has some difficulties for application to older landfills where the disposition historically is unknown.

It can be noticed from Equations 4.46 and 4.47 that when one imposes $\Delta\sigma=0$, the settlement also is zero, meaning that the MSW final settlements would be exclusively loading-dependent, which is not correct.

Table 3.5 – Variation Range of the Creep Law Model Parameters in the Study of Edil et al. (1990)

Landfill	**Characteristics**	**Variation Range for m* (1/kPa)**	**Variation Range for n* (t_r= 1 day)**
A	Wisconsin; MSW with ages of 0 to 4 years; concluded and under operation cells; $\Delta\sigma$ between 45 kPa and 200 kPa	$7{,}52\times10^{-8}$ to $1{,}38\times10^{-4}$	0,297 to 1,170
B	Michigan; MSW with ages of 1 to 16 years; concluded and under operation cells; $\Delta\sigma$ between 59 kPa and 146 kPa	$7{,}85\times10^{-7}$ to $8{,}83\times10^{-6}$	0,648 to 0,779
C	Connecticut; MSW with ages of 40 to 50 years; excavated and recompacted MSW; $\Delta\sigma$ between 72 kPa and 103 kPa	$1{,}10\times10^{-5}$ to $6{,}48\times10^{-5}$	0,264 to 0,465
D	Pilot landfill and additional area composed by old MSW; $\Delta\sigma$= 51 kPa	$4{,}69\times10^{-5}$ to $8{,}57\times10^{-5}$	0,486 to 0,666

3.3.6. Empirical Exponential Models

The exponential model presented by Gandolla et al. (1994) represents the adjustment of a decreasing exponential curve with time to the data. The equation type is:

$$\Delta h = a_k \cdot H_0 \cdot \left(1 - e^{-k\cdot t}\right) \qquad \text{.......(3.44)}$$

where a_k e k are parameters to be estimated by non-linear interpolation.

This model has as advantage in the use of a function with the same type of the one presented for waste biodegradation, although the authors do not mention this parallelism, as pointed out by Espinace et al. (1999).

A similar model is presented by Park and Lee (1997), as reported by Marques (2001), that reduces the component ($a_k \cdot H_0$) into only one parameter, denominated as the total compression that can be developed due to waste biodegradation.

Manassero and Pasqualini (1993), as reported by Manassero et al. (1996), presented a very simple graphical procedure, illustrated in Figure 3.29, for plotting the data and for determining the parameters of Equation 3.44, similar to Asaoka's method for prediction of soil settlements. Note that Asaoka's method is valid only for the phase of primary consolidation of soils.

However, Asaoka's method does not utilize an exponential equation as Equation 3.44, but the equation of the classical model, which expresses the primary consolidation in soils. In Manassero and Pasqualini's procedure, the theoretical formulation is different from the classical model and is based on Equation 3.44. That methodology is only valid if landfill elevation readings are regular and executed in constant time intervals – same Δt – (each month, for example); otherwise, the method is not practical.

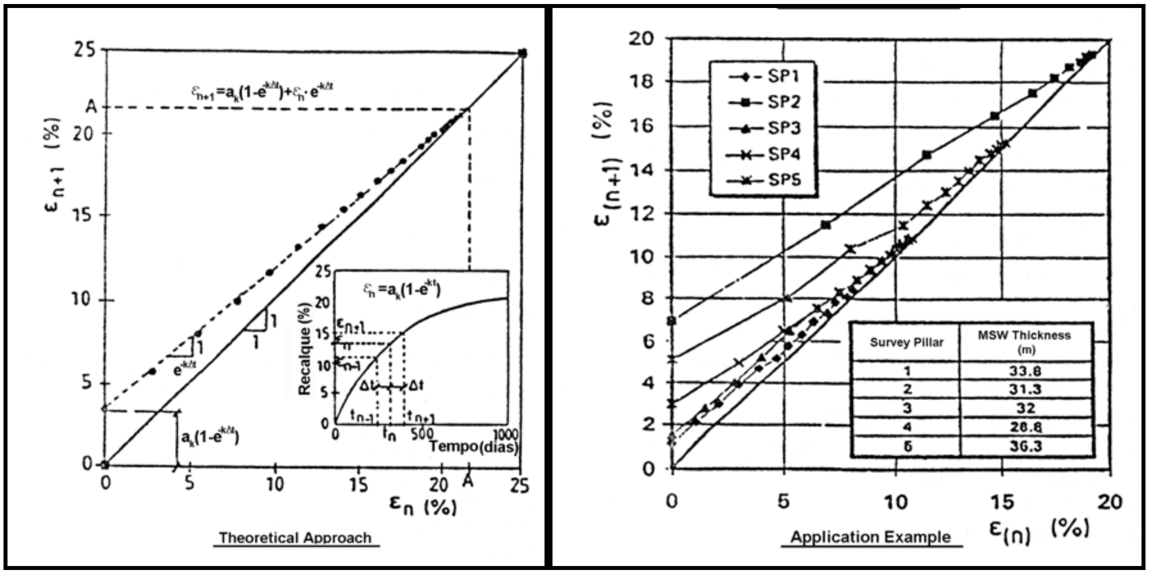

Figure 3.29 – Manassero and Pasqualini's Procedure (Manassero et al., 1996)

Sagaseta (1993), as reported by Manassero et al. (1996), re-wrote Equation 3.44 in another way, but with the same meaning:

$$\Delta h = \Delta h_{\infty} \cdot \left(1 - e^{-t/t_r}\right) \qquad \text{.........(3.45)}$$

Δh_{∞} ... final settlement,

t_r ... reference time (assumed by the authors as the time necessary to reach 63% of the final settlement).

Similarly, Sanchez-Alciturri et al. (1993) re-wrote Equation 3.44 for the same model:

$$\Delta h = \frac{r_0}{k} \cdot H_0 \cdot \left(1 - e^{-k \cdot t}\right) \qquad \text{.........(3.46)}$$

where r_0 is a parameter representing the initial settlement rate.

Edgers et al. (1992), as reported by Carvalho (1999), presented a similar model to the one presented by Gandolla et al. (1994); however they mention that the exponential decay of settlements has fundament in waste biodegradation, relating them to the change in the number of bacteria along time.

3.3.7. Meruelo's Model

The exponential model of Meruelo, developed in 1994 and 1995 by the Geotechnical Group of the University of Cantabria, Spain, and of the Catholic University of Valparaiso, Chile, represents long-term settlements (depending on time), considering the degradation processes occurring in solid wastes (Espinace et al., 1999).

The main hypotheses of the model, presented by Palma (1995) are the following:

- Of all settlement mechanisms, the model treats only the one due to the degradation of waste.

- From the biodegradation phases, the anaerobic phase is responsible by settlement development in the long term.

- The process of anaerobic degradation is conditioned by the hydrolytic phase, and the rate of this phase is the one that governs the general rate of the process.

- The hydrolysis rate of a degradable mass element varies inversely with the remaining mass (not degradable), and the relationship between them is defined by the hydrolysis coefficient (k_h), under constant environmental conditions and only dependent on the moisture content.

- The degradable solid mass transforms into liquids and gases.

- The rate of waste received by the landfill is constant in time.

- The decrease in solid mass is only partially observed as a decrease in volume.

- The decrease in solid mass can obey two processes: one, under constant volume, with decreasing density; and the other, under constant density, with decreasing volume, and, therefore, developing settlements.

- For a given waste type and compaction way, the relationship between a decrease in solid mass and the decrease in volume is constant.

The final equation presented by Palma (1995) for Meruelo's model is:

$$\Delta h = \alpha \cdot H \cdot OMC \cdot \left[1 - \frac{1}{k_h \cdot T_c} \cdot \left(e^{-k_h \cdot (t - T_c)} - e^{-k_h \cdot t} \right) \right] \quad (3.47)$$

α ...	coefficient of mass losses that are transformed into settlements,
OMC ...	organic material content,
T_C ...	waste landfill construction time,
k_h ...	coefficient of hydrolysis,
H ...	waste landfill thickness.

Note that for $t=T_c$ the equation does not go to zero, since the model predicts the waste degradation starting from the beginning of the landfill operation and not at the end of disposition, which makes sense. Palma (1995) designates H, generically, as the thickness of the landfill, but theoretically H represents the final "virtual" landfill thickness, considering that no settlements have occurred.

Since the thicknesses of older landfills sometimes are unknown, De Abreu (2000) re-writes Equation 3.47 in his study, presenting an expression that can be utilized based on the thickness of the landfill when the geotechnical monitoring starts:

$$\Delta h_{t_i-t} = \frac{\alpha \cdot (H_i + \Delta h_{0-i}) \cdot OMC}{k_h \cdot T_c} \cdot (1 - e^{k_h \cdot T_c}) \cdot \left(e^{-k_h \cdot t} - e^{-k_h \cdot t_i}\right) \qquad \text{.......................(3.48)}$$

$\Delta h_{ti\text{-}t}$ ... settlement between the time when the superficial mark is installed and any posterior time,

t_i … time when the superficial mark is installed,

H_i ... landfill thickness when the superficial mark is installed.

$\Delta h_{0\text{-}i}$ is defined as the settlement developed between the initial time and the time when the superficial mark is installed and can be calculated by:

$$\Delta h_{0-i} = \frac{\alpha \cdot H_i \cdot \aleph \cdot OMC}{1 - \alpha \cdot \aleph \cdot OMC} \qquad \text{...(3.49)}$$

where

$$\aleph = 1 - \frac{1}{k_h \cdot T_c} \cdot \left(e^{-k_h \cdot (t_i - T_c)} - e^{-k_h \cdot t_i}\right) \qquad \text{..(3.50)}$$

Meruelo's model has as an advantage the representation of the degradation process, which can be a determinant factor of the long-term settlements, with parameters that have a physical meaning. Probably, this model can be better applied to old landfills, where the mechanisms of mechanical compression and pore-pressure dissipation can have little or no influence.

The disadvantage of the model is in its present lack of representative published parameter values (α, k_h). Palma (1995) lists the value ranges obtained for each parameter:

- OMC - 25%
- α - 0.12 to 0.50
- k_h – 0.0003 to 0.003 days^{-1}

De Abreu (2000) presented the results for an old landfill in Brazil when Meruelo's model is utilized. The author reported values between 0.0007 and 0.0013 days^{-1} for k_h, The values of α obtained were distorted due to particularities of the landfill construction in conflict with some of the hypotheses of the model (such as the constant rate of waste received).

3.3.8. Hyperbolic Model

The Hyperbolic model presented by Ling et al. (1998) represents the review of the model utilized in studies of embankments over soft soils presented by Tan et al. (1991). The expression utilized for MSW settlements is given by:

$$\Delta h_{tm-t} = \frac{t}{\frac{1}{v_{rp0}} + \frac{t}{\Delta h_{\infty}}} \qquad \text{(3.51)}$$

t ... difference between a given time and the time at the beginning of readings,

Δh_{tm-t} ...difference between the settlement at a given time and the settlement at the beginning of readings,

v_{rp0} ... initial settlement rate (at the beginning of readings),

Δh_{∞} ... final settlement.

Parameters v_{rp0} and Δh_{∞} can be easily obtained through linear regression, transforming Equation 3.51 into a linear expression, as shown on Equation 3.52 and Figure 3.30.

$$\frac{t}{\Delta h_{tm-t}} = \frac{1}{v_{rp0}} + \frac{t}{\Delta h_{\infty}} \qquad \text{(3.52)}$$

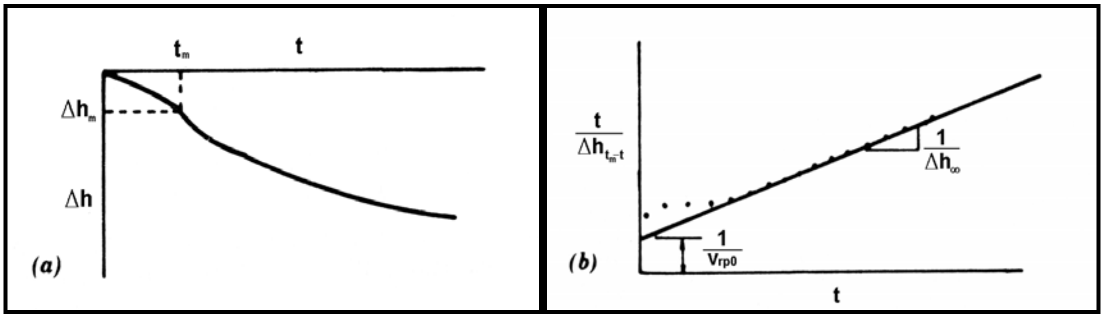

Figure 3.30 – Schematic Representation of the Hyperbolic Model Plottings

It can be noted that parameter $t/\Delta h_{ti\text{-}t}$ represents the inverse of the average velocity between the settlements on times t_i and t.

The authors comment that in practice a final settlement of approximately 80% to 95% of Δh_∞ occurs. The time to reach 95% of the settlements can be calculated through the equation:

$$t_{95\%} = 19 \cdot \frac{\Delta h_\infty}{v_{rp0}} \quad \text{.....(3.53)}$$

The advantages of this method are:

- Flexibility – can be applied from any given initial point (for example, the beginning of the readings), even when there are changes in the loading (however, the analysis must be reinitiated).

- Practicability – very easy to be applied and there is the possibility of analyzing the history of eventual loadings by using the curve.

The disadvantages are:

- Impossibility of settlement predictions in the design phase, since it is necessary to have settlement data to plot the final curve.

- According to Tan et al. (1991), the model has good accuracy when the settlements reach 30% to 40% of the total settlements for soils. For MSW, some time is necessary for the data fit the final straight line.

Ling et al. (1998) applied the Hyperbolic model into a case study using the settlement data of landfills in Southeastern Wisconsin, Meruelo, and Los Angeles. The authors related a better adjustment using this model when compared to the Yen and Scanlon and “Creep” Law also utilized on their case study.

Boscov and De Abreu (2000) described the excellent results of the Hyperbolic Model when applied for settlement prediction in a Brazilian landfill. The difference between the settlement predicted by De Abreu (2000) and the real settlements measured 1 year after were approximately 2%. Marques (2001) in his study of a pilot landfill also obtained very good results with the model.

3.3.9. Composite Rheological Model (Marques, 2001)

Based on the several models presented on the literature, Marques (2001) presents a model considering the primary and secondary compression mechanisms, governed by rheological parameters with physical meaning, explaining in a clear and consistent way the time parameter. The model is named composite since it has an individual formulation for each compression mechanism, adapted from existent solutions.

The proposed formulation is:

$$\frac{\Delta h}{H} = C'_c \cdot \log\left(\frac{\sigma_0 + \Delta\sigma}{\sigma_0}\right) + \Delta\sigma \cdot b \cdot \left(1 - e^{-ct'}\right) + E_{dg} \cdot \left(1 - e^{-dt''}\right) \quad \text{......................(3.54)}$$

b … coefficient of secondary mechanical compression, c … secondary mechanical compression rate,

E_{dg} … total compression due waste degradation,

d … secondary biological compression rate,

t' … time elapsed since loading application,

t" … time elapsed since waste disposal.

One of the best features of the model is that the primary compression formulation is introduced as an "immediate compression". This "immediate compression" is independent of time, based on the observation that the respective process is linear for curves of void ratio as a function of the logarithm of the applied stress, as related by Carvalho (1999). It is interesting to note that Debnath (2000) also noticed the linearity of those curves in his study for lower values of stress (<400 kN/m^2). However, for higher values of stress they are certainly non-linear, as presented on Figure 3.7 (Rao et al., 1977).

The possibility of using a computer program (called MSWSET by Marques, 2001), for describing and computing the compression conditions layer by layer is also a very good feature of this method.

In his study, Marques (2001) obtained average values of 0.1061 for C'_c; 5.72×10^{-4} for parameter b; 1.79×10^{-3} for parameter c; 0.1585 for E_{dg}; and 1.14×10^{-3} for parameter d.

3.3.10. Other Models for MSW Settlement Prediction

Gomes et al. (1998), as reported by Carvalho (1999), present a regression analysis for settlement prediction on three reactive cells with a diameter of 1 m, monitored during 448 days, and without additional loadings. Equation 3.55 and Figure 3.31 present the adjustment executed.

$$\Delta h = 10^{-6} \cdot t^3 - 0{,}0009 \cdot t^2 + 0{,}2738 \qquad \text{...(3.55)}$$

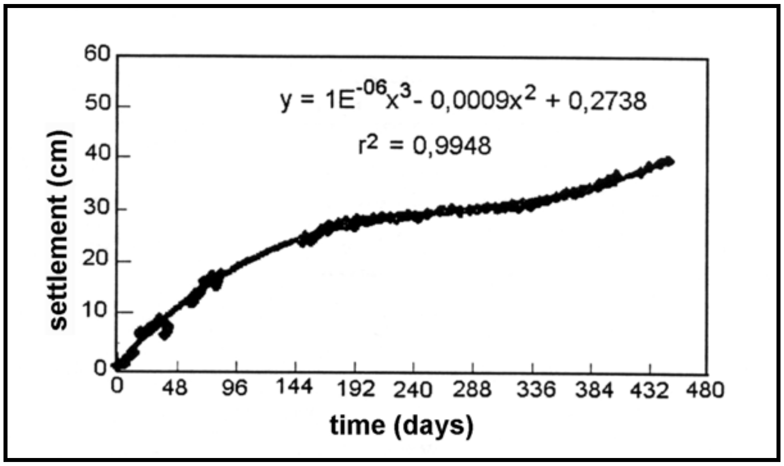

Figure 3.31 – Adjustment of Curve Settlement as a Function of Time Done by Gomes et al., 1998 (Carvalho, 1999)

Obviously the adjustment is valid only for the case studied by the authors.

Kockel et al. (1997) present a study of the variability of settlement parameters for MSW. They use a stress-dependent constrained modulus ($E=a+b\cdot\sigma$) for determination of the settlements due to loading, and parameters $C'_{\alpha}1$ and $C'_{\alpha}2$ from Bjarngard and Edgers' equation (Equation 3.11) for determination of settlements due to time. Variation ranges for settlements were established using the maximum and minimum limits presented in literature. This application can be used for settlement prediction in the design phase and also in the monitoring phase. The authors comment that the variation range is very extensive due to the several factors that influence the settlements on landfills. Table 3.6 and Figure 3.32 synthesize the studied presented.

Table 3.6 – Variation Range for MSW Compression Parameters

	Loading Parameters		Time Parameters		
	a (kN/m)	b	$C'_{\alpha 1}$	$C'_{\alpha 2}$	t_m (days)
Number of Observations	21	21	16	20	20
Average	-200	11.7	0.030	0.102	425
Standard Deviation	206	1.72	0.017	0.077	472
Superior Limit (95% confidence level)	-106	12.5	0.039	0.138	645
Inferior Limit (95% confidence level)	-294	10.9	0.021	0.066	204

It must be noted that this study is supported, even partially, by the Classical model, and, therefore, must be faced with restrictions.

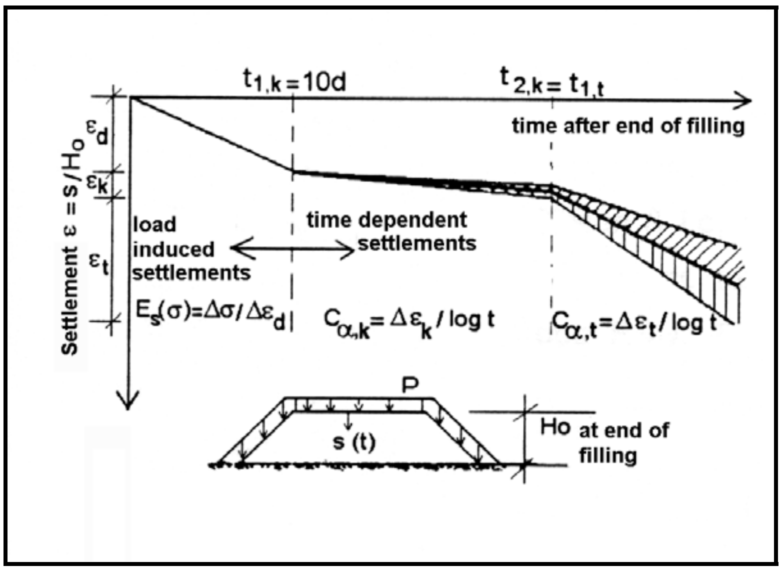

Figure 3.32 – Idealized Time Settlement Behavior of a MSW Landfill According to Kockel et al., 1997

3.3.11. Development of New Models

Studies of new models that try to analyze the compressibility of sanitary landfills aim to review existent models and implement new models that can more realistically interpret the involved mechanisms, treating the MSW as a new geotechnical unit (Simoes et al., 1996). However, the introduction of a model that satisfies all the conditions involved in the phenomena and the development of realistic parameters that can be obtained by laboratory tests or field monitoring is not an easy task.

Soler et al. (1995) discuss the procedures to be adopted in the field and in the laboratory for obtaining a conceptual model in which the sanitary landfills can be represented by a porous medium containing inert particles and biomass, reviewing the settlement mechanisms.

Simoes and Campos (1998) present the models more commonly utilized in geotechnical engineering and the possibility of application to MSW. The models presented are the classical model, consolidation with finite strain, consolidation in unsaturated media, consolidation with finite strain in unsaturated media, and empirical models.

The classical model is presented in Section 3.3.1. The model using consolidation with finite strain, the equilibrium and continuity equations must be verified, adding that for MSW there are mass losses. Since sanitary landfills are generally unsaturated and, most of the time, have as design constraints infiltration minimization and the use of superficial and internal drainage systems, the use of theories that utilize the consideration of consolidation in unsaturated media needs to be utilized, according to the authors. Swarbrick (1995) presents a formulation for consolidation with finite strain in unsaturated media, however admitting incompressible particles with uniform dimensions and densities under isothermal conditions. Therefore this needs review if utilized for MSW.

Zimmerman’s Mathematical model, developed in 1972, is maybe the model that more realistically expresses all involved mechanisms, however, with parameters not so simple to be obtained.

3.4. Settlement Studies in Bioreactor Landfills

There are very few studies about settlement in bioreactor landfills in the general literature. One of the first studies was presented by Wall and Zeiss (1995), where six landfill test cells of 0.57 m in diameter and at a height of 1.7 m were constructed to model both settlement and decomposition over a period of 225 days. Three cells were designed to simulate bioreactor landfills, while the other three simulated conventional landfills. Comparisons among the six cells during the period of study indicated that there was no significant increase in the settlement rate due to biodegradation.

Townsend et al. (1996) observed the effects of leachate recycling on stabilization and settlements of a North-Central Florida landfill, using the technique of infiltration ponds. Settlement was measured in wet and dry (treated and untreated, respectively) areas of the landfill over a period of 21 months. The authors reported that for the dry area a volume loss of 3.82% occurred and for the wet area the volume loss corresponded to 5.65% of the original volume, therefore, a slight increase when compared to the dry area.

Reinhart and Townsend (1998) reported settlements on cells with recirculation in pilot-scale landfills at Sonoma County, CA, which settled as much as 20% of the initial thickness, while dry cells settled less than 8%. The authors also mentioned settlements in wet cells at the Mountain View Landfill, CA, reaching 13 to 15%, while control dry cells settled 8 to 12% over a 4-year period.

Espinace et al. (1999) presented in their study the utilization of two 0.80 m-diameter lysimeters for evaluating the MSW settlements. The first lysimeter was operated as a conventional landfill, and the second as a recirculating landfill, but with leachate pre-treatment using an anaerobic digester. As part of the same study, the authors monitored the settlements of a 6 m-height pilot landfill located in Valparaiso, Chile. The landfill was also operated as a bioreactor landfill with leachate pre-treatment using an anaerobic digestor. Espinace et al. (1999) presented that the settlement on the recirculating lysimeter was almost two and on-half times greater than on the conventional landfill after approximately 100 days from the beginning of operation. It must be pointed out that in Latin American countries the content of organic matter in the MSW is much greater than in the United States.

Gabr et al. (2000) proposed an approach for development of a model that could be utilized with bioreactor landfills. The authors proposed to divide the MSW settlements into two parts: Early Stages of Decomposition, when the compressibility of waste is governed by changes in void ratio due to solids loss and material physical size and stiffness with no consideration to hydrodynamic lag effect; and the second, Later Stage of Decomposition, as decomposition takes place, the material leads to an increase in the surface area. With the leachate recycle approach, Terzaghi's model with primary and secondary settlement may then be applied at that point.

Yuen and Styles (2000) presented the findings of a settlement investigation conducted at a MSW landfill in Melbourne, Australia. One of the objectives of the project studied by the authors was to evaluate the full-scale landfill settlements on two different sections of the landfill, one operated as a bioreactor landfill with raw leachate recirculation, and the other operated as a conventional landfill. Both sections were approximately 18 m high and settlement plates were installed at heights of 4 m, 6m, 8 m, 10 m, and 18 m inside the landfill. The settlement monitoring showed that at the 4 m height, there was no difference in the settlements between the sections; however, for all other sections an effective difference in settlements could be noticed. After approximately 3.5 years of monitoring, the settlement in the section with leachate recirculation reached about 1.1 m (6% of the total thickness), while for the other section, the settlement reached about 0.60 m (3% of the total thickness), or almost half of the first section.

CHAPTER 4

EXPERIMENTAL PHASE

4.1. Introduction

As part of the research project being carried out at the University of New Orleans (Facultative Landfill Bioreactor Project), a pilot-scale plant was installed in a fenced area adjacent to the Engineering Building at the University of New Orleans in New Orleans, Louisiana.

The pilot-scale plant consists of three lysimeters filled with MSW prepared with identical composition, a leachate treatment plant, and a biofilter to control odor emissions. The three lysimeters are operated under different conditions: one is being operated as a facultative bioreactor landfill with external leachate pre-treatment prior to recirculation, the second is being operated as an anaerobic landfill bioreactor with straight raw leachate recirculation, and the third one is the control unit operated as a conventional (dry) landfill. The leachate treatment plant composed of an electrocoagulation/settling unit and two fluidized bed reactors in series is used in conjunction with the facultative bioreactor lysimeter.

The lysimeters were filled in May 2002. Settlement was measured in each of the lysimeters between May 2002 and May 2003. Leachate samples from the lysimeters were taken between August 2002 and May 2003 and analyzed for several parameters at the UNO Environmental Engineering Laboratory and at the Schlieder Urban Environmental Systems Center Analytical Laboratory. Gas composition in the lysimeters was analyzed using a portable gas analyzer specifically designed for waste landfills.

In addition to settlement measurements in the three lysimeters to compare and evaluate the compressibility mechanism of landfills, compressibility tests were run at the UNO Geotechnical Laboratory by using a large-dimension chamber specifically designed for solid wastes. Gradually controlled loads were applied to waste samples using PVC/steel plate discs and hydraulic jacks to compress them. Vertical displacements were measured to evaluate and study the compressibility mechanism of MSW. Compressibility tests were performed on MSW with the same composition of the waste placed in the lysimeters at dry (initial) and wet (final) conditions.

4.2. The UNO Pilot Plant

The lysimeters as well as all units on the pilot treatment plant were purchased from CGvL Engineers and assembled in an area adjacent to the Engineering Building at the University of New Orleans Lakefront Campus in December 2001. Pictures of the pilot plant construction and set-up are presented in Appendix A.

The design of the lysimeters as well as the leachate treatment plant were initiated in July 2001 and lasted through November 2001. Project staff worked through December 2001 in the construction and installation of the three lysimeters, each unit of the treatment plant and pumping/piping system.

4.2.1. Lysimeters Construction and Set-up

The three lysimeters are made of flanged PVC schedule-40 pipes, 600 mm (24 in) in diameter and 3.05 m (10 feet) tall, furnished with schedule-80 top and bottom flanges, and PVC flange plates 38 mm (1.5 in) thick. Rubber gaskets seal the joints between the flanges and plates at the top and bottom of the lysimeters. The lysimeters were positioned on a platform made of steel and wood and elevated above the ground approximately 46 cm (18 in). The platform was necessary to allow collection of the leachate generated by the lysimeters in three covered plastic leachate receiving tanks (80 liters), located underneath them. Picture 1 in Appendix A shows the lysimeters at the pilot plant.

At the bottom of each lysimeter, a 38 mm (1.5 in) valve was installed. An orifice was placed on the cover of the receiving tank, and a 38 mm-diameter hose installed to make the connection between the receiving tank and the valve. Each receiving tank also contained an overflow orifice to send the excess of leachate to the sewage system. Leachate is collected in the receiving tank, and depending on the lysimeter operation mode, is pumped by centrifugal pumps with 27 m^3/day capacity.

In the Facultative Bioreactor Lysimeter (Lysimeter 1), the leachate produced is stored and pumped to the leachate treatment plant, with flow controlled by a flowmeter (Georg Fischer, Type SK10, 2.2 gpm capacity) and a valve. The pump is operated with an on-off manual switch. Immediately after the pump, a sampling port is positioned.

In the Lysimeter with Leachate Recirculation (Lysimeter 2), the leachate produced is stored and pumped back to the top of the lysimeter, with flow controlled by a flowmeter with the same characteristics as the one used on Lysimeter 1. The pump is also operated with an on-off manual switch. A sampling port is also available immediately after the pump.

In the Conventional Lysimeter (Lysimeter 3), the leachate produced is controlled by a manual valve and goes directly to the sewer line. A sampling port is available at the receiving tank.

There are also three ports, each 1.9 cm in diameter, at the top of each lysimeter to measure the settlements. A gas-sampling valve is present at the top of each lysimeter, as well.

A biofilter was installed for odor control since the plant is located on-campus. A blower with an on-off manual switch was installed to send the gases to the biofilter.

4.2.2. Pilot Treatment Plant Construction and Set-Up

The treatment plant was designed to treat 2.7 m^3/day of leachate produced in the Facultative Bioreactor Lysimeter, and consists of four main components: an electrocoagulation unit, a clarifier, an anaerobic fluidized bed reactor (AFBR), and an aerobic nitrification fluidized bed reactor (Cadenas, 2002). Figure 4.1 presents a schematic of the leachate treatment plant. Picture 2 in Appendix A shows the leachate treatment plant.

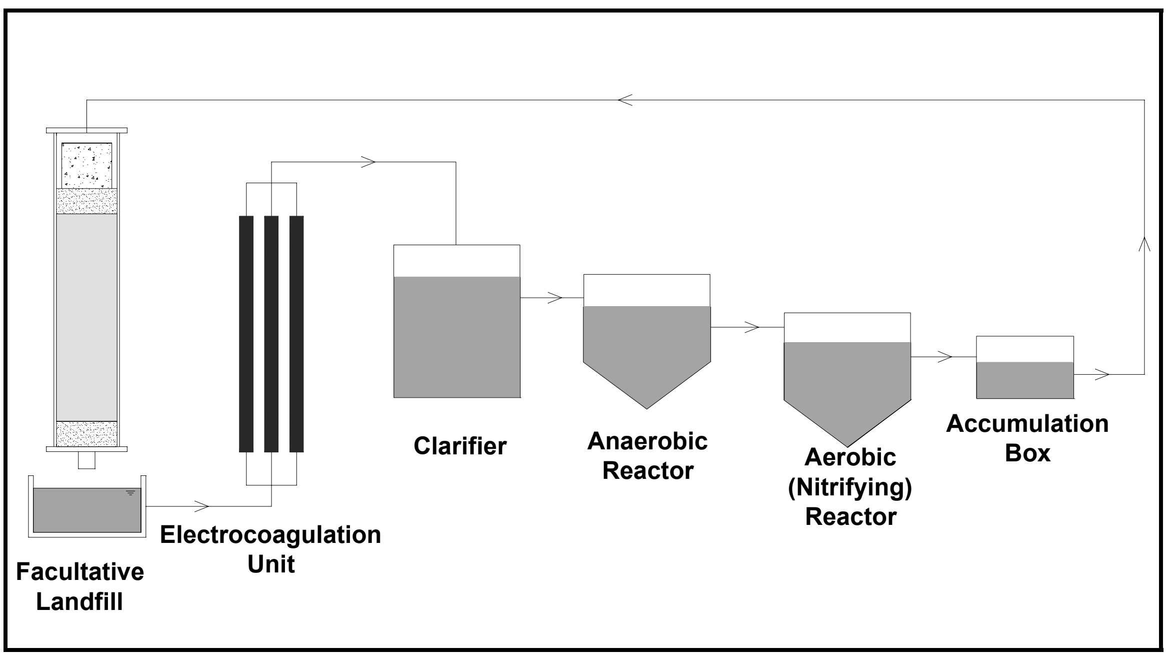

Figure 4.1 – Schematic of the Leachate Treatment Plant

The electrocoagulation unit is the first component of the treatment plant. It consists of three electrocoagulation reaction chambers with concentric vertical aluminum tubes. This unit removes heavy metals and some colloidal and suspended organic matter in the leachate that comes from the facultative landfill bioreactor and can handle up to 3.27 m^3/d (1.1 m^3/d each reaction chamber). The aluminum tubes are connected to a 40-volt, 20-amp direct current rectifier. Inner and outer tubes act as anodes and cathodes and are connected to the negative and positive outputs of the rectifier, respectively. Metal compounds are precipitated by the electron flow between the tubes while the colloidal particles are coagulated by aluminum cations (coagulant) produced in the anode. If some sediment is produced, it is removed as sludge.

The second component of the treatment plant is a gravity conical-bottom clarifier, with 0.8 m^3 capacity, constructed of low-density polyethylene and furnished with a 0.2 m center well, a 2 RPM motorized bottom scraper, and three baffled overflow ports. The electrocoagulation unit effluent enters the clarifier at the top. The material that settles in this reactor goes to the sludge tank through the motorized ball valve operating with a timer. There is a sampling port in the line between the clarifier and the AFBR.

The anaerobic fluidized bed reactor (AFBR), with 0.8-m^3 capacity, is constructed of low-density polyethylene, with a conical bottom configuration and contains 0.4 m^3 of granular activated carbon media. The granular activated carbon particles offer a large surface per unit bed volume for anaerobic biofilm to attach. This reactor has internal recirculation and flow distribution components. The bed is fluidized by means of the upflow created with effluent recirculation achieved with a recirculation pump (an electric motor-driven pump) and a motor with starter and overload protection. The effluent is directed to the pump through a valve and then returns to the AFBR through the effluent recirculation line. In this line, there is a pressure indicator with isolating valve, an air-bleeding valve, and a valve that allows the effluent to enter into the AFBR. There are a sampling port in the line that goes from the AFBR to the nitrification fluidized biological reactor (NIT-FBR) and two others for bed level sampling.

The fourth component of the treatment plant is an aerobic nitrification fluidized bed reactor (NIT-FBR), with 0.8 m^3 capacity, constructed of low-density polyethylene with a conical bottom configuration and contains 0.4 m^3 of granular activated carbon media as support for the nitrifying bacteria. The function of this reactor is the nitrification of the ammonia present in the leachate. This reactor has internal recirculation achieved with an electric motor-driven pump on the same way of the AFBR unit. The effluent feeds by gravity the suction side of a pump and goes to an oxygen saturator unit. The oxygen-saturated effluent emerges from the O_2 – SAT and is mixed with the raw effluent line into the overall recycle line. There is a sampling port for the final effluent and two others for bed level sampling. The aerobic nitrification fluidized bed reactor effluent goes to the treated effluent tank and is pumped back to the facultative landfill bioreactor with the treated leachate return pump (a centrifugal pump).

A detailed description of the leachate treatment plant was presented by Cadenas (2002).

4.2.3. Start-Up Process

The first step of the process was to determine the type and source of waste that would be used on the project. To obtain representative samples, since the nature of MSW is very heterogeneous, with composition highly variable depending on the city, on the season, etc., it was opted to use a fabricated waste matrix noted as the “synthetic MSW” matrix. The components of the “synthetic MSW” utilized were collected between October 2002 and February 2003 from different sources and mixed according to the United States average residential and commercial discarded MSW composition (EPA, 2000). The composition of the MSW, as well as the sources of the materials are presented in Table 4.1.

Table 4.1 – Composition and Description of MSW Utilized

Material	% by weight	Description
Paper and cardboard	31.1	newspapers, magazines, office papers, boxes
Glass	5.9	jars, wine bottles
Metals - ferrous	5.1	steel and iron debris cutted into small pieces
Metals - non ferrous	1.7	aluminum cans
Plastics	13.4	soda bottles, bags, water bottles
Rubber and leather	3.8	tires in small pieces, old shoes
Textiles	4.7	old clothes, rags
Wood	7.1	lumber pieces
Food wastes	13.6	food waste collected in cafeterias
Yard trimmings	9.6	grass and vegetation collected on campus
Others	4.0	soil, ashes

After the total quantity of materials necessary to create the MSW matrix was collected, the component particles were manually reduced to a maximum size of 20 cm to be compatible with the size of the lysimeters utilized. Then the components were weighted (according to the composition presented in Table 4.1), manually mixed, compacted and placed in large plastic contractor bags (170 liters capacity, each one – Picture 9 in Appendix A). Each batch utilized 14.7 kg of material components. A total of 18 bags (batches) were utilized. Table 4.2 presents the quantities of materials utilized in each batch.

Table 4.2 – Quantities of Materials Utilized in Each Batch

Material	%, weight	Batch (kg)
Paper and Cardboard	31.1	4.6
Glass	5.9	0.9
Metals - Ferrous	5.1	0.7
Metals - Non Ferrous	1.7	0.2
Plastics	13.4	2.0
Rubber and Leather	3.8	0.6
Textiles	4.7	0.7
Wood	7.1	1.0
Food Wastes	13.6	2.0
Yard Trimmings	9.6	1.4
Others	4.0	0.6
TOTAL	**100.0**	**14.7**

In filling the lysimeters, a scaffold was used (Picture 8 in Appendix A). The lysimeters were first filled with a 25-cm layer of pea-gravel, dropped from the top, and leveled using a wood post with dimensions 10 cm by 10 cm by 300 cm. A circular wire mesh (stainless steel, 0.8 mm opening) was placed above the pea gravel. The synthetic MSW was loaded into the lysimeters and manually compacted using the same wood post used to regularize the layer of pea gravel (Pictures 10 and 11 in Appendix A).

The contents of six bags, consisting of 88.2 kg of MSW, were loaded inside each lysimeter, achieving approximately a 2 m thickness. A second mesh was placed immediately above the waste and approximately 12.5 cm of pea-gravel was placed above the mesh. A PVC manifold for leachate recirculation and rainfall simulation was placed above the waste and another layer of 12.5 cm of pea-gravel was placed on top.

Both lysimeters 1 and 2 have flexible hose connections to the leachate/water distribution manifold, such that the manifold can move along with the waste as it consolidates and settles as result of waste stabilization and loading processes (Pictures 12, 13, and 14 in Appendix A).

A concrete weight with a diameter of 51 cm and a height of 46 cm was placed over the final layer of pea-gravel to simulate the load conditions at a landfill. Each concrete block was approximately 255 kg. Prior to placing the concrete blocks inside the lysimeters with a crane-truck, four PVC guides were attached to each block to avoid an uneven vertical movement of the concrete block inside the lysimeter (Pictures 15 and 16 in Appendix A).

Figure 4.2 presents a schematic of the lysimeters as simulated landfills. Table 4.3 shows the thickness, mass, and unit weight of the MSW layer before and after placing the pea-gravel layer and concrete blocks.

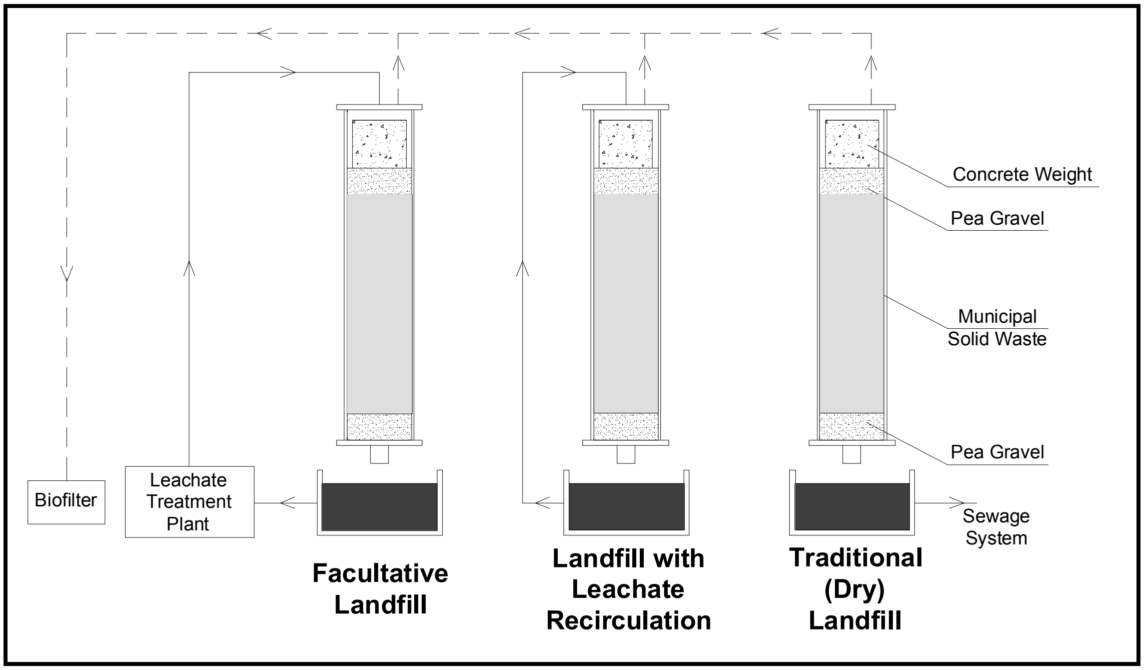

Figure 4.2 – Schematic of Lysimeters as Simulated Landfills

Table 4.3 – MSW Characteristics Before and After Placing Weights in the Lysimeters

	Lysimeter 1	Lysimeter 2	Lysimeter 3
Initial MSW Layer Thickness (cm)	234.5	221.6	221.6
Initial MSW Mass (kg)	88.2	88.2	88.2
Initial MSW Unit Weight (kN/m^3)	1.29	1.36	1.36
MSW Layer Thickness immediately after placing pea-gravel and weight (cm)	198.9	198.8	203.8
MSW Unit Weight immediately after placing pea-gravel and weight (kN/m^3)	1.52	1.52	1.48

The lysimeters were closed and in June 2002 all of them were filled with tap water with the bottom valve closed until complete saturation of the waste was achieved. Then, the valve was opened and the volume of initial leachate generated was measured and pumped to the sewer line, bringing the waste to field capacity.

Regarding the leachate treatment plant, several problems occurred during the start-up process between January and August 2002. Some of the problems encountered were leakages in some units, successive clogging of pipes and the pump in the anaerobic fluidized bed reactor, failures in the oxygen generator unit, and electrical problems. The leachate treatment plant was started with artificial substrate in January and February 2002, and reinitiated after a complete revision in May 2002.

4.2.4. Operational Procedures

In July 2002, water started to be added at a regular basis (typically twice a week) to simulate rainfall precipitation. The water added corresponds to 50% of the annual average rainfall precipitation of New Orleans, which is equal to the addition of 3.85 liters of water per week (1.93 liters twice a week) to each lysimeter. The value of 50% of the annual average rainfall precipitation was used to simulate the water that actually infiltrates in the waste mass of a landfill (therefore subtracting the effects of runoff, evapo-transpiration, etc.). Although this value can be considered high for landfills with composite cap, it allows enough leachate to be generated in the dry lysimeter for analysis.

Leachate was generated July 11th 2002 from Lysimeter 1 and in July 22nd 2002 from Lysimeters 2 and 3 with a few drops, initially. On August 5th the leachate sampling program started. On August 22nd leachate from Lysimeter 1 was sent to the treatment plant for the first time, and untreated leachate was recycled in Lysimeter 2.

Leachate generated from Lysimeters 1 and 2 was sent to the treatment plant or recirculated, depending on the lysimeter, typically on a daily basis (5 days a week, from Monday to Friday) during the period of August 22nd 2002 – May 30th 2003. Treated leachate was recycled to Lysimeter 1 also on a daily basis and at the same quantity and rate as the untreated recycled to Lysimeter 2. The leachate flow rates were controlled with flowmeters (Georg Fischer, Type SK10, 2.2 gpm capacity).

The quantity of raw leachate or treated leachate applied to the lysimeters varied with time, initially depending on the quantities of leachate produced. At the end of the experiment, the rate of leachate recycled was changed for both lysimeters to study the effects of the rate on the parameters studied.

During the period of December 15th 2002 through January 16th 2003, recirculation was not practiced to study the effect of this procedure on the parameters studied. During the September 25th-27th 2002 period, recirculation was also not practiced due to presence of Tropical Storm Isidore in the New Orleans metropolitan area. Also, during those periods water simulating rainfall was not added to the lysimeters.

Table 4.4 presents the rates of raw or treated leachate recirculation during the period studied.

Table 4.4 – Rates of Leachate Recirculation (Raw or Treated) Practiced on Lysimeters 1 and 2

Period	**Leachate Recirculation Rate Practiced**	
	Lysimeter 1	**Lysimeter 2**
August 22nd – September 13th 2002	15 liters/day	15 liters/day
September 16th – September 24th 2002	23 liters/day	23 liters/day
September 25th – September 27th 2002	No recirculation practiced	No recirculation practiced
September 30th – October 4th 2002	30 liters/day	30 liters/day
October 7th – November 1st 2002	38 liters/day	38 liters/day
November 4th – December 13th 2002	45 liters/day	45 liters/day
December 15th 2002 – January 16th 2003	No recirculation practiced	No recirculation practiced
January 17th – February 7th 2003	45 liters/day	45 liters/day
February 7th – April 24th 2003	45 liters/day	180 liters/day
April 25th – May 30th 2003	15 liters/day	180 liters/day

The only major problem encountered with the leachate treatment plant, during the operational period was in the oxygen generator unit, which had to be repaired during January 2003.

The biofilter, installed during the construction of the pilot plant, was not used during the period of study since no problems associated with odors was perceived. The gas valves located at the top of the lysimeters remained closed during all periods of the experiment.

4.2.5. Leachate Sampling and Analysis

4.2.5.1. Sampling Procedures

The leachate sampling phase started in August 2002 and lasted through May 2003. Sample collection for the leachates generated by the three lysimeters was carried out on a weekly/bi-weekly basis during the period of August – December 2002, and on a monthly basis during the period of January – May 2003. Sampling was typically done in the morning.

Although there were sampling ports to collect the leachate in each one of the receiving tanks, it was opted to collect the leachate generated directly into glass sampling containers placed in the previous day of sampling immediately after the hoses that connect the lysimeter bottom valves to the receiving tanks. This procedure avoided the mixing of the old leachate already in the tanks with the recent leachate generated. Thus, the leachate collected represents the leachate that is effectively generated between the previous day and the day of sampling. Sampling of the final effluent sent to the facultative lysimeter was also performed on a regular basis.

After collection, all samples were taken for analysis to the Environmental Engineering Laboratory (between August 2002 and November 2002) or to the Schlieder Urban Environmental Systems Center Analytical Laboratory (between November 2002 and May 2003), both located at the University of New Orleans. The samples were stored in a refrigerator prior to being analyzed on the same day in most of the cases. In some few cases, some of the analyses were conducted the following day after sampling, but always within a 24-hour period.

A Cadenas (2002) study included samples that were also collected at several different points int the leachate treatment plant during the August – November 2002 period: from the electrocoagulation unit, from the clarifier unit, from the anaerobic fluidized bed reactor (AFBR) unit, from the aerobic nitrification fluidized bed reactor (NIT-FBR) unit, and from the final effluent sent to Lysimeter 1.

4.2.5.2. Laboratory Analyses

As reported before, all leachate analyses were conducted either at the Environmental Engineering Laboratory (between August 2002 and November 2002) or at the Schlieder Urban Environmental Systems Center Laboratory (between November 2002 and May 2003), both located at the University of New Orleans. The equipment used in both laboratories was the same for all of the analysis periods.

Several parameters were measured: Chemical Oxygen Demand (COD - total and filtrated), 5-day Biochemical Oxygen Demand (BOD_5), pH, Ammonia-Nitrogen, Nitrate, Total Kjeldahl Nitrogen (TKN), Total Volatile Acids (TVA), Total Phosphorus, Chloride, Total Suspended Solids (TSS), Volatile Suspended Solids (VSS), and Metals (Iron and Aluminum). The parameters were chosen for analysis because they can be good indicators of the state of MSW stabilization, and can be compared with values presented in the literature.

Chemical Oxygen Demand (COD)

The COD is defined as a measure of the organic matter contained in a sample in terms of the oxygen required for the chemical oxidation of these materials. Method 5220C of the Standard Methods for the Examination of Water and Wastewater (APHA, 1998) – Closed Reflux, Titrimetric Method – was used to determine the COD. This test was performed each time samples were collected (at a weekly/bi-weekly basis during the period of August – December 2002 and at a monthly basis during the period of January – May 2003). All COD tests were done with triplicate specimens.

Since this is one of the most important tests performed, 10 samples of a COD standard solution (with concentration 1000 ± 50 mg/L) were used to verify the precision and accuracy of the method. For the method utilized, with the concentration tested and at a dilution of 1:5 (also used for the actual samples), the average value obtained was 985 mg/L with a standard deviation of 33 mg/L.

In some samples, filtered COD tests were also performed to determine the amount of "dissolved" organic matter in the filtered samples. Although filtrated COD readings are assumed to correspond to true dissolved organic matter, it must be noted that a large fraction of colloidal particles pass through the filter. The samples were filtered through a Hach No. 30 qualitative filter paper, with a pore size of 0.45 micron, using vacuum filtration micropore glassware.

5-Day Biochemical Oxygen Demand (BOD_5)

BOD determination is an empirical test in which standardized laboratory procedures are used to determine the relative requirements of effluents. The test measures the oxygen utilized during a specified incubation period for the biochemical degradation of the organic material (carbonaceous demand) and the oxygen utilized to oxidize inorganic material such as sulfides and ferrous iron. Method 5210B of the Standard Methods for the Examination of Water and Wastewater (APHA, 1998) – 5-Day BOD Test – was used. This test was performed three times: at the beginning, in the middle, and at the end of the period of study. BOD parameters can be used together with COD parameters to evaluate the stabilization of waste.

pH

The pH measures the intensity of the acidic or basic character of a solution, corresponding to the hydrogen ion activity. This parameter was measured using a pH meter (Orion, Model 420A plus) with resolution of 0.001 and accuracy of ±0.005. The pH meter was calibrated at each time of use and operated following the recommendations given by the manufacturer. This test was performed each time samples were collected (at a weekly/bi-weekly basis during the period of August – December 2002 and at a monthly basis during the period of January – May 2003).

Ammonia-Nitrogen (NH_3)

Method 4500-NH_3 C of the Standard Methods for the Examination of Water and Wastewater (APHA, 1998) – Nesslerization Method – was used. A Direct Reading Spectrophotometer (Hach, Model DR/2000) was used, and distillation was applied to the samples prior to analysis, since high concentrations were expected. This test was typically performed every month during the period of study.

Five samples of an ammonia-nitrogen standard solution (with concentration 150 mg/L) were used to verify the precision and accuracy of the method. For the method utilized, with the concentration tested and distilled, the average value obtained was 125 mg/L with a standard deviation of 32 mg/L.

Nitrate-Nitrogen (NO_3^-)

Method 4500-NO_3 E of the Standard Methods for the Examination of Water and Wastewater (APHA, 1998) – Cadmium Reduction Method – with a Direct Reading Spectrophotometer (Hach, Model DR/2000) was used. This test was performed at least twice a month, during the period of August – September 2002, and every month during the period of October 2002 – May 2003.

Five samples of a nitrate-nitrogen standard solution (with concentration 10 mg/L) were used to verify the precision and accuracy of the method. For the method utilized and the concentration tested, the average value obtained was 10.8 mg/L with a standard deviation of 1.2 mg/L.

Total Kjeldahl Nitrogen (TKN)

The TKN measures the total of organic nitrogen concentration plus ammonia nitrogen concentration in a sample. Method 8075 of the Hach DR/2000 Spectrophotometer Procedures Manual (Hach, 1994) – Nessler Method – with a Direct Reading Spectrophotometer (Hach, Model DR/2000) was used. Digestion, using hydrogen peroxide and Digesdahl equipment, was performed on the samples prior to analysis. This test was performed twice a month for the period of August – September 2002 and every two months for the period of January 2003 – May 2003.

Total Volatile Acids (TVA)

Method 8120 of the Hach DR/2000 Spectrophotometer Procedures Manual (Hach, 1994) – Esterification Method – with a Direct Reading Spectrophotometer (Hach, Model DR/2000) was used. Filtration was applied using Hach No. 30 qualitative filter paper, with a pore size of 0.45 micron, and vacuum filtration micropore glassware. This test was performed typically once a month during the period of August – December 2002, and at the end of the study (May 2003).

Total Phosphorus (TP)

Phosphorus is essential to the growth of organisms and can be the nutrient that limits the primary productivity of a body of water.

Method 8190 of the Hach DR/2000 Spectrophotometer Procedures Manual (Hach, 1994) – Acid Persulfate Digestion Method – with a Direct Reading Spectrophotometer (Hach, Model DR/2000) was used. Digestion, using sulfuric acid, was performed on the samples prior to analysis. This test was performed typically once a month during the period of study.

Chloride

Chloride is one of the major inorganic anions present in wastewater and in the leachate. Chloride is considered as a "conservative" and inert parameter, and would only be removed from conventional landfills via washout. Therefore, it can be used as a "tracer" parameter. Method 8113 of the Hach DR/2000 Spectrophotometer Procedures Manual (Hach, 1994) – Mercuric Thiocyanate Method – with a Direct Reading Spectrophotometer (Hach, Model DR/2000) was used. This test was performed twice a month during the period of August – September 2002, and every month during the period of October 2002 – May 2003, typically.

Total Suspended Solids (TSS)

The TSS test was performed to quantify the amount of solid matter suspended in the samples. Method 2540D of the Standard Methods for the Examination of Water and Wastewater (APHA, 1998) – Total Suspended Solids

Dried at 103-105^{o}C – was applied. The samples were filtered through a Hach No. 30 qualitative filter paper, with a pore size of 0.45 micron, using vacuum filtration micropore glassware. After filtration, the solids remaining in the 0.45-micron pore size filter paper were dried at 105^{o}C for one hour. This test was performed twice a month, during the period of August – September 2002, and every month during the period of October 2002 – May 2003, typically.

Volatile Suspended Solids (VSS)

The VSS test was performed to quantify the volatile fraction of solids in the samples. Method 2540E of the Standard Methods for the Examination of Water and Wastewater (APHA, 1998) – Fixed and Volatile Solids Ignited at 550^{o}C – was applied. Solids remaining in the filter paper after the suspended solids test were ignited at 550^{o}C in a muffle furnace. The remaining solids represent the fixed fraction, while the weight lost represents the volatile fraction. This test was performed twice a month, during the period of August – September 2002, and every month during the period of October 2002 – May 2003, typically.

Metals – Iron and Aluminum

The total concentrations of iron and aluminum were the parameters analyzed for metals utilized in the study due to the characteristics of the synthetic MSW prepared (rich in iron and aluminum components). Iron was analyzed following Method 8008 of the Hach DR/2000 Spectrophotometer Procedures Manual (Hach, 1994) – FerroVer Method Using Powder Pillows. Aluminum was analyzed following Method 8012 of the same manual – Aluminum Method. A Direct Reading Spectrophotometer (Hach, Model DR/2000) was used for both analyses. Previous digestion of the samples using Digesdahl equipment was carried out. These tests were performed twice a month, during the period of August – September 2002, and every month during the period of October 2002 – May 2003, typically.

4.2.5.3. Summary of Sampling Procedures and Laboratory Analyses

Table 4.5 presents a summary of sampling procedures and laboratory analysis, indicating the sampling date and parameters analyzed for the leachate generated in the three lysimeters.

Table 4.5 - Summary of Sampling Program for Leachate Generated by Lysimeters

Sampling Date	COD	BOD	pH	Ammonia	Nitrate	TKN	TVA	Total Phosphorus	Chloride	TSS and VSS	Metals
08/05/2002	X		X					X	X		
08/08/2002	X		X	X	X	X		X	X		
08/14/2002			X				X			X	X
08/19/2002	X		X	X	X	X	X				
08/26/2002	X	X	X					X	X	X	X
09/03/2002	X		X	X	X	X	X				
09/09/2002	X		X					X	X	X	X
09/16/2002	X		X		X	X					
09/23/2002	X		X					X	X	X	X
10/07/2002	X		X	X	X						
11/04/2002	X		X				X	X	X	X	X
11/11/2002	X		X	X	X						
11/18/2002	X		X					X	X	X	X
12/02/2002	X	X	X	X	X		X	X	X	X	X
01/20/2003	X		X	X	X	X		X	X	X	X
02/13/2003	X		X	X	X			X	X	X	X
03/10/2003	X		X	X	X	X		X	X	X	X
04/15/2003	X		X	X	X			X	X	X	X
05/15/2003	X	X	X	X	X	X	X	X	X	X	X

4.2.6. Gas Analysis

Gas composition in the lysimeters was analyzed using a portable gas analyzer specifically designed for waste landfills (Gas Data LMSx). The gas analyzer works with infrared detector and can measure fractions of methane, carbon dioxide, and oxygen as a percentage of the total gas.

The equipment was calibrated by the manufacturer when it was purchased. The analyzer is equipped with an internal gas pump that sends the gas/air from the probe to the equipment. It was operated using an external membrane filter to remove aerosol droplets of water, totally blocking liquids that accidentally could be sucked up by the probe.

The readings were made by connecting the probe furnished with the equipment at the port designed for gas sampling and opening the respective valve, located at the top of each lysimeter. The analysis of gas was typically performed every one or two weeks since the equipment was purchased. The first reading was done on November 7th, 2002.

4.2.7. Settlement Measurements

Settlement of the waste inside the three lysimeters was measured weekly during the period of May – October 2002, every two weeks during October – December 2002, and monthly after that to compare and evaluate the long-term compressibility mechanism of landfills. Settlements were measured introducing a wood ruler with precision of 1 mm to the three ports located at the top of each lysimeter, designed to measure the displacement of the concrete weights. The average of the three readings is considered as the average distance between the top of the lysimeter and the top of the concrete weight. The difference between this value at a given time and the initial value is the settlement of waste at that particular time.

4.2.8. Leachate Production Measurements

Leachate production was measured daily or every 2 weeks, depending on the lysimeter, during the period of study. For Lysimeters 1 and 2, the leachate production was measured daily due to the procedures of leachate recirculation. For Lysimeter 3, the production was measured every two weeks, typically.

However, after October 2002, the production of leachate generated by Lysimeters 1 and 2 was not measured since overflow started to occur in the respective receiving tanks. The receiving tank beneath Lysimeter 3 was emptied at a regular basis (typically every month) before overflow occurred, making possible the measurement of leachate production.

Leachate evaporation inside the tanks is believed to be very small since the receiving tanks are closed and wrapped with plastic bags to minimize infiltration of rainwater.

4.3. Compressibility Tests

The main purpose of the laboratory compressibility tests was to evaluate the MSW compressibility in order to predict short-term settlements due to loading. Several MSW compressibility tests were performed at the University of New Orleans in the past using a large-dimension chamber specifically designed for solid wastes. Reviewing the results of those tests, it was noticed that there is a pattern for the stress-strain relation for immediate settlements. In addition, new compressibility tests with the same composition of the waste placed in the lysimeters were performed for the MSW at the initial dry condition and the final wet condition.

4.3.1. Equipment Description

The equipment utilized for the tests was the same used by Debnath (2002). The equipment fabrication and experimental set-up was done at the Geotechnical Engineering Laboratory of the Department of Civil and Environmental Engineering, University of New Orleans. The chamber is 61 cm (2 ft) in diameter and approximately 120 cm tall with a blind flange at the bottom and open at the top.

The main body of the chamber is open at the top and is made of PVC pipe with 1.9 cm (3/4 in) thick walls. The bottom 36 cm (1/4 in) portion of this part is 1.9-cm (3/4 in) thicker (3.8 cm – 1-1/2 in) than the lower part for stability since the maximum lateral pressures (and, thus, cell wall stresses) occur at high loads when the sample is compressed to about this height. At the bottom end, there is a steel flange attached to the bottom plate on the outer surface of the chamber.

The base of the chamber supports the overall test chamber. It is made of a PVC ring 61 cm (2 ft) in diameter (external), 15 cm (6 in) high, and 1.9 cm (3/4 in) thick. The PVC ring was attached to the chamber by aluminum angles at two points.

The bottom plate of the chamber is a circular PVC plate 81 cm (31 ¾ in) in diameter and 2.5 cm (1 in) thick. There are twenty 3.2 cm (1-3/4 in) diameter holes around the periphery. One circular steel flange (or ring) with the same number, size, and position of holes as the bottom plate is placed on the top of the bottom plate surrounding the chamber. The bottom plate is attached to the flange by a set of nuts, bolts, and washers. The connection between these two parts is sealed with a rubber gasket in between. There is a 2.5 cm (1 in) diameter opening on the bottom plate to pass out the leachate from the chamber.

A steel loading frame was used to hold the test chamber and as a reaction for the hydraulic jack during the application of load. The size of the frame utilized is 330 cm (10.8 ft) by 106 cm (3.5 ft) by 173 cm (5.67 ft) (length x width x height).

Two types of hydraulic jacks were used for applying load to the samples, depending on the load capacity needed. This included one jack system with a 20-ton capacity (10 tons in each ram) and a second with a 50-ton (25 tons in each ram) capacity.

Figure 4.3 presents a schematic of the compressibility test set-up. Figures 4.4 and 4.5 schematically present a detail of the bottom part of the chamber and the loading frame used, respectively. Pictures 1, 2, and 3 in Appendix B show the compressibility test set-up.

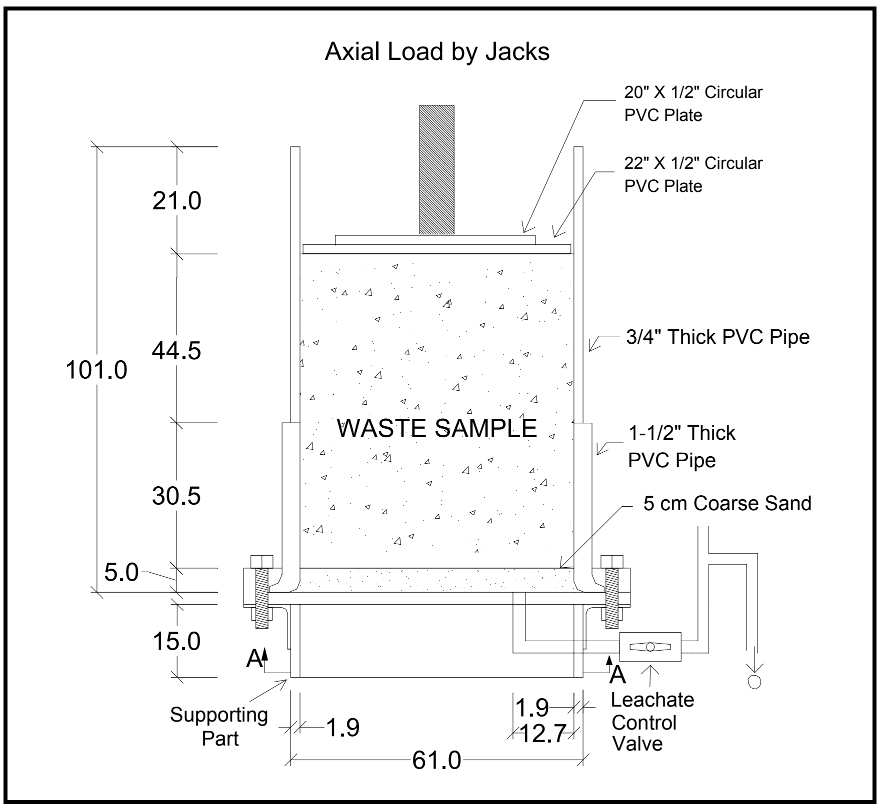

Figure 4.3 - MSW Compressibility Test Set-up (measures in centimeters, except otherwise indicated; not to scale)

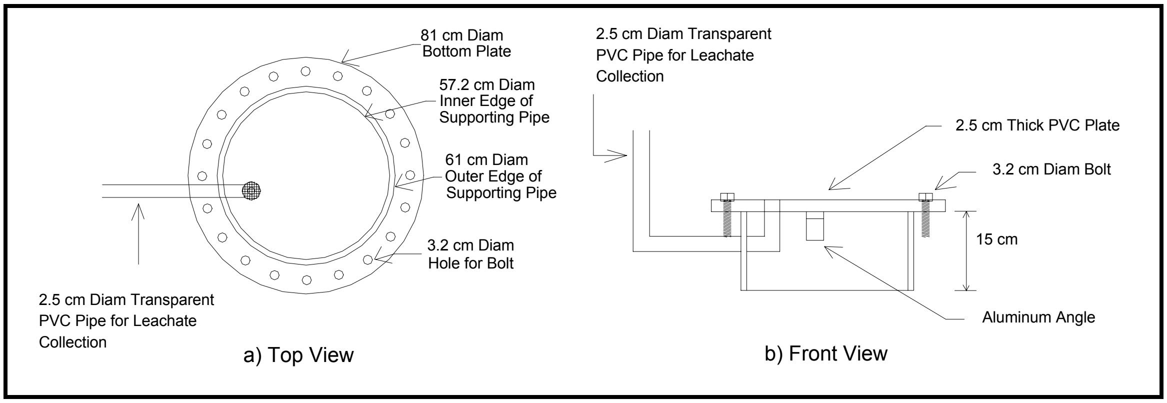

Figure 4.4 – Detail of Bottom Part of the Chamber (not to scale)

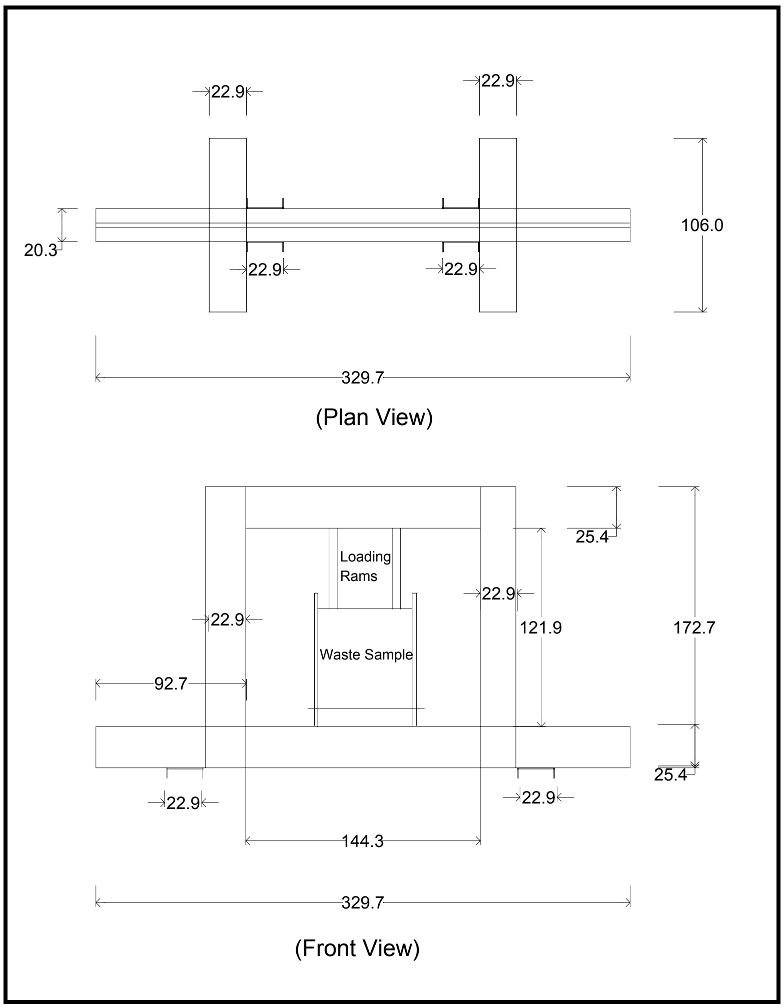

Figure 4.5 – Schematic of Loading Frame Used (measures in centimeters; not to scale) (Debnath, 2000; modified)

4.3.2. Chamber Preparation and Tests Procedure

The chamber was prepared for each test by placing a 5-cm thick layer of coarse sand at the bottom of the chamber (manually leveled) and putting a geotextile fabric immediately over it to isolate the sand from the waste.

A layer of approximately 75 cm (divided in four sub-layers of 20 cm or less) of MSW prepared with the same composition utilized in the lysimeters was placed in the chamber. Compaction was applied by tamping with a wooden block on each sub-layer of the sample, which made the surface of each sub-layer horizontal. Another piece of geotextile fabric was placed on the final layer of waste.

After loading the chamber with the prepared MSW, a pair of PVC and steel circular plates was placed on the top of the geotextile. The bottom plate was a perforated PVC plate 56 cm (22 in) in diameter and 1.3 cm (1/2 in) thick. A perforated steel plate 51 cm (20 in) in diameter and 1.3 cm (1/2 in) thick was placed over the PVC plate.

The loading device (pair of hydraulic jacks) was positioned between the plate and the frame and the load was applied slowly. The first test was performed under dry conditions initially using a 20-ton jack system. However, an oil leak was noticed in the system at the beginning of the test and the system was replaced with a 50-ton jack system.

The vertical compression due to each load was measured at three different points. Successive increments of load were applied to the waste with measurements of the displacement of the plates. The displacements of the plates were measured in three different points from the top of the chamber with a ruler with a precision of 1 mm. The load pattern (total stress applied to the MSW) for the compressibility tests was typically 5, 10, 20, 25, 50, 100, 150, and 350 kPa.

The second test was performed under wet conditions immediately after the plates were placed on the top of the waste. The chamber was saturated with tap water and then drained.

CHAPTER 5

RESULTS PRESENTATION

5.1. Environmental Study

5.1.1. Leachate Composition

As mentioned in Section 4.2.5, several parameters were measured on the leachate generated by the three lysimeters: Chemical Oxygen Demand (COD - total and filtered), 5-day Biochemical Oxygen Demand (BOD_5), pH, Ammonia-Nitrogen, Nitrate, Total Kjeldahl Nitrogen (TKN), Total Volatile Acids (TVA), Total Phosphorus, Chloride, Total Suspended Solids (TSS), Volatile Suspended Solids (VSS), and Metals (Iron and Aluminum). In addition, Nitrate was also measured for the effluent that was injected into the Facultative Lysimeter (Lysimeter 1).

For the following tables and graphs, “day 0 (zero)” was considered as the first day that leachate was observed as being generated by the lysimeters, i.e., July 11th 2002.

5.1.1.1. Chemical Oxygen Demand (COD)

Tables 5.1 and 5.2 respectively present the concentration of total COD and filtered COD measured on the leachate generated by the lysimeters. Figures 5.1 and 5.2 present the graphs of concentration as a function of time for total and filtered COD, respectively.

Table 5.1 – Concentrations of Total COD Measured in Lysimeters

Date	Days (*)	COD (mg/L)		
		Lysimeter 1	Lysimeter 2	Lysimeter 3
8/5/2002	25	2400	1920	112
8/8/2002	28	992	1568	1920
8/19/2002	39	1824	1632	1312
8/26/2002	46	2080	2400	1440
9/3/2002	54	960	1248	1376
9/9/2002	60	544	768	256
9/16/2002	67	448		
9/23/2002	74	96	768	608
10/7/2002	88	32	544	512
11/4/2002	116	32	416	352
11/11/2002	123	112	496	400
11/18/2002	130	112	480	
12/2/2002	144	96	560	560
1/20/2003	193	104	420	570
2/13/2003	217	72	412	538
3/10/2003	242	80	342	512
4/15/2003	278	72	366	472
5/15/2003	308	51	396	436

(*) Days elapsed since leachate production started

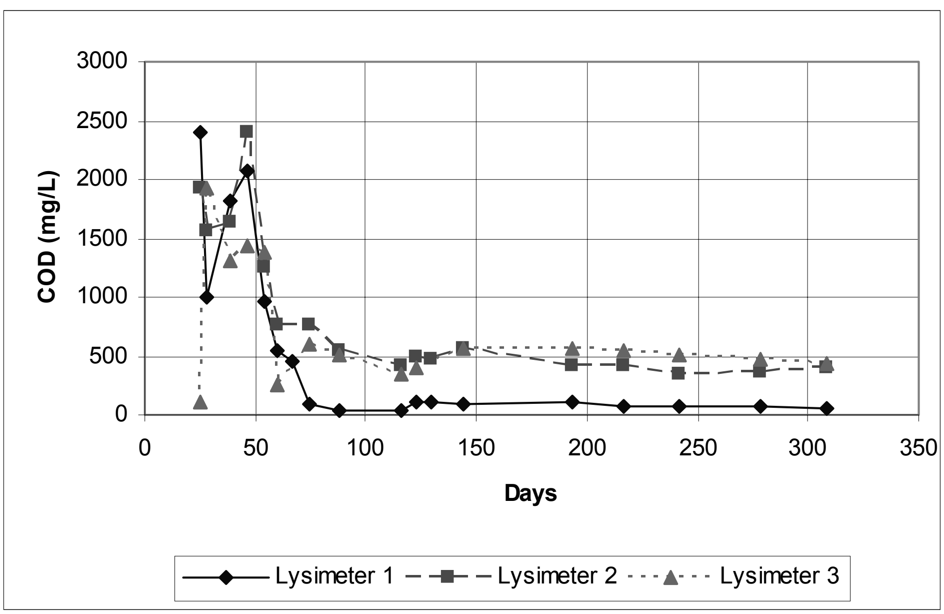

Figure 5.1 – Total COD Concentrations as a Function of Time

From Table 5.1 and Figure 5.1, it can be noted that total COD concentrations fluctuate typically between 1,000 and 2,400 mg/L during the first 50 days, and then dropped quickly to a concentration of about 500 mg/L for Lysimeters 2 (straight recirculation) and 3 (dry lysimeter), and to 100 mg/L for Lysimeter 1 around day 75. After that day, COD concentrations decrease very slowly for the three lysimeters, finishing at the end of the experiment at a 50 mg/L level for Lysimeter 1, at a 400 mg/L level for Lysimeter 2, and at a 440 mg/L level for Lysimeter 3.

Table 5.2 – Concentrations of Filtered COD Measured in the Lysimeters

Date	Days (*)	Filtered COD (mg/L)		
		Lysimeter 1	Lysimeter 2	Lysimeter 3
12/2/2002	144	16	288	448
1/20/2003	193	20	282	462
2/13/2003	217	12	243	440
3/10/2003	242	15	150	432
4/15/2003	278	11	162	388
5/15/2003	308	12	120	371

(*) Days elapsed since leachate production started

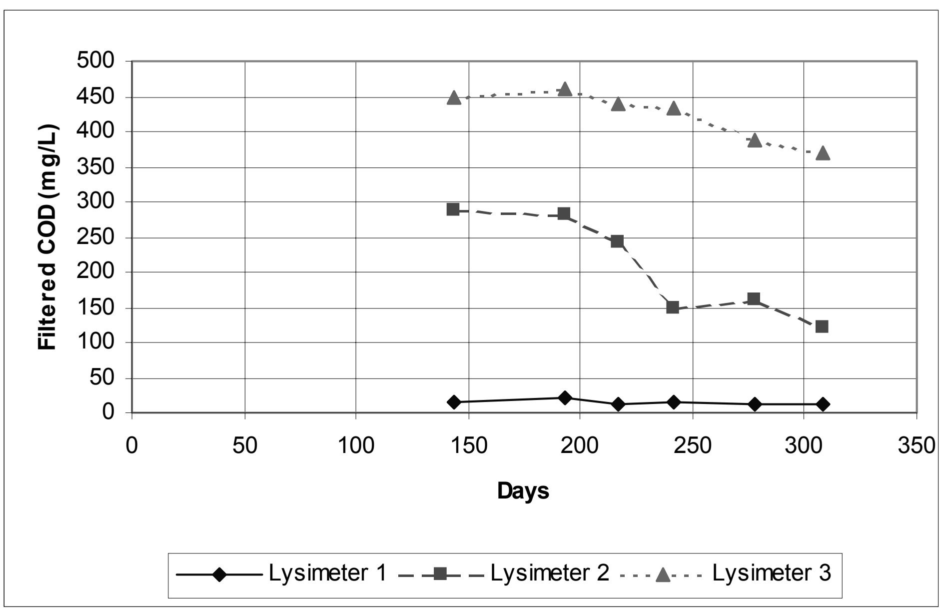

Figure 5.2 – Filtered COD Concentrations as a Function of Time

From Table 5.2 and Figure 5.2, it can be noted that filtered COD concentrations stayed at a 10 to 20 mg/L level for Lysimeter 1, but dropped from 288 mg/L to 120 mg/L for Lysimeter 2, and from 448 mg/L to 371 mg/L for Lysimeter 3.

5.1.1.2. 5-Day Biochemical Oxygen Demand (BOD_5)

Table 5.3 and Figure 5.3 present the concentration of BOD_5 measured on three different dates for the leachate generated by the lysimeters.

Table 5.3 – Concentrations of BOD_5 Measured in the Lysimeters

Date	Days (*)	BOD (mg/L)		
		Lysimeter 1	Lysimeter 2	Lysimeter 3
8/26/2002	46	1974	2023	1420
12/2/2002	144	62	422	457
5/15/2003	308	11	211	266

(*) Days elapsed since leachate production started

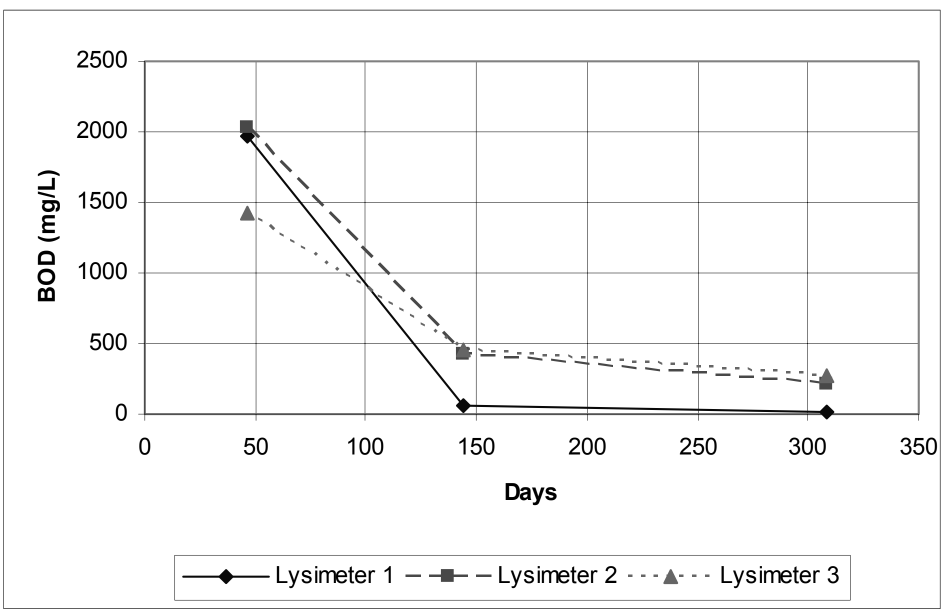

Figure 5.3 – BOD_5 Concentrations as a Function of Time

From Table 5.3 and Figure 5.3 it can be noticed that BOD_5 concentrations dropped from a 1,500 to 2,000 mg/L level to a 400 to 450 mg/L level around day

150, and then to a 210 to 270 mg/L level at the end of the study for Lysimeters 2 and 3. For Lysimeter 1, the concentrations dropped from a 2,000 mg/L level to a 60 mg/L level on day 144, and then to 11 mg/L at the end of the study.

5.1.1.3. pH

Table 5.4 and Figure 5.4 present the pH values measured throughout the period of study.

Table 5.4 – pH Values Measured in the Three Lysimeters

Date	Days (*)	pH		
		Lysimeter 1	Lysimeter 2	Lysimeter 3
8/5/2002	25	8.03	8.27	8.60
8/8/2002	28	7.82	7.98	7.68
8/14/2002	34	7.72	8.12	
8/19/2002	39	7.53	8.09	7.67
8/26/2002	46	7.28	7.97	7.54
9/3/2002	54	7.08	7.98	7.65
9/9/2002	60	7.10	8.13	7.92
9/16/2002	67	7.58	7.86	8.15
9/23/2002	74	7.07	7.86	8.02
10/7/2002	88	7.65	7.36	7.45
11/4/2002	116	6.83	7.78	8.43
11/11/2002	123	7.22	7.08	8.61
11/18/2002	130	7.16	7.46	
12/2/2002	144	7.44	7.46	7.41
1/20/2003	193	7.35	7.62	7.64
2/13/2003	217	7.20	7.61	7.43
3/10/2003	242	7.42	7.46	7.52
4/15/2003	278	7.48	7.47	7.55
5/15/2003	308	7.51	7.60	7.62

(*) Days elapsed since leachate production started

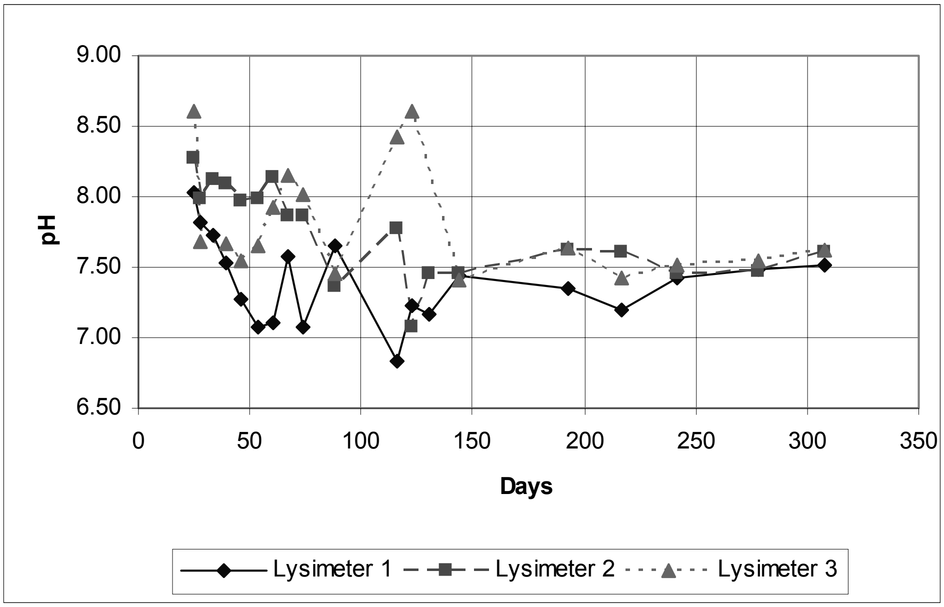

Figure 5.4 – pH Values as a Function of Time

From Table 5.4 and Figure 5.4, it can be noticed for the three lysimeters that pH values started around 8 to 8.6, fluctuated at a 7 to 8 range until day 144, and then stabilized around 7.5 to 7.6 at the end of the study.

5.1.1.4. Ammonia-Nitrogen (NH_3)

Table 5.5 and Figure 5.5 present the concentrations of ammonia-nitrogen measured throughout the period of study.

Ammonia-nitrogen concentrations started around 300 mg/L for Lysimeter 1, quickly dropping to 34 mg/L at day 88, and then remained at a range of 24 to 42 mg/L until the end of the study. For Lysimeters 2 and 3, concentrations started around 90 to 100 mg/L and very slowly decreased to a 80 mg/L level for Lysimeter 2 and to 70 mg/L level for Lysimeter 3.

Table 5.5 – Concentrations of Ammonia-Nitrogen Measured in the Lysimeters

Date	Days (*)	Ammonia-Nitrogen (mg/L)		
		Lysimeter 1	Lysimeter 2	Lysimeter 3
8/8/2002	28	304	98	88
8/19/2002	39	194	156	82
9/16/2002	67	72	62	94
10/7/2002	88	34	92	110
11/11/2002	123	42	90	100
12/2/2002	144	24	95	103
1/20/2003	193	26	74	86
2/13/2003	217	32	88	92
3/10/2003	242	35	82	81
4/15/2003	278	27	82	76
5/15/2003	308	26	82	68

(*) Days elapsed since leachate production started

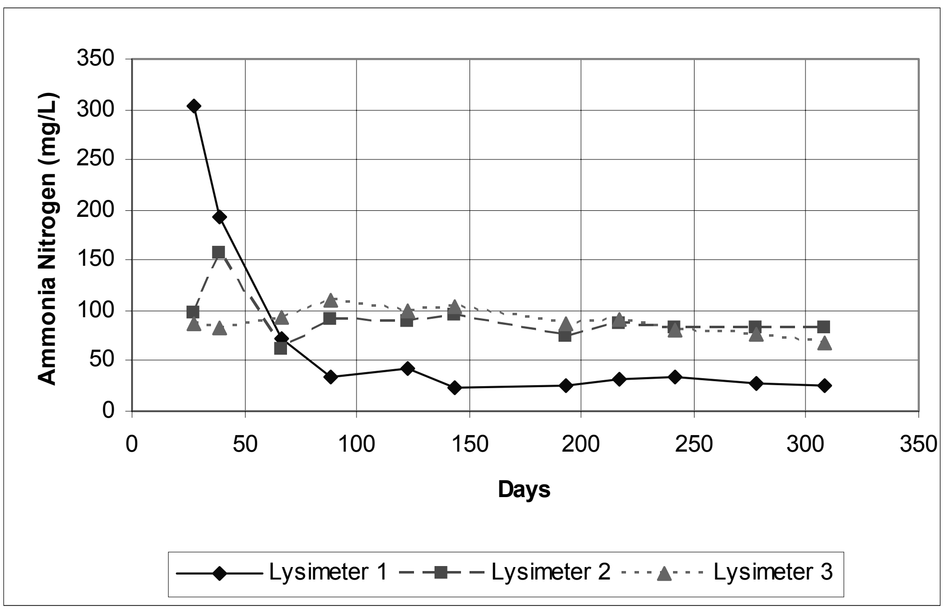

Figure 5.5 – Ammonia-Nitrogen Concentrations as a Function of Time

5.1.1.5. Nitrate-Nitrogen (NO_3)

Table 5.6 presents the concentrations of nitrate-nitrogen measured on the lysimeters throughout the period of study.

Table 5.6 – Concentrations of Nitrate-Nitrogen Measured in the Lysimeters

Date	Days (*)	Nitrate (mg/L)		
		Lysimeter 1	Lysimeter 2	Lysimeter 3
8/8/2002	28	<0.05	<0.05	<0.05
8/19/2002	39	<0.05	<0.05	<0.05
9/3/2002	54	<0.05	<0.05	<0.05
9/16/2002	67	1.00	0.80	1.10
10/7/2002	88	0.10	<0.05	<0.05
11/11/2002	123	<0.01	<0.01	<0.01
12/2/2002	144	<0.01	<0.01	<0.01
1/20/2003	193	<0.01	<0.01	<0.01
2/13/2003	217	<0.01	<0.01	<0.01
3/10/2003	242	0.50	<0.01	<0.01
4/15/2003	278	1.20	0.02	<0.01
5/15/2003	308	<0.01	<0.01	<0.01

(*) Days elapsed since leachate production started

Nitrate-nitrogen concentrations on the lysimeters remained very low throughout the study for the three lysimeters, below the equipment detection limits in the vast majority of times analyzed. However, values around 1 mg/L can be pointed out for the three lysimeters on day 67 and for Lysimeter 1 between days 242 and 248.

Table 5.7 and Figure 5.6 present the concentrations of nitrate-nitrogen measured on the treated effluent injected to Lysimeter 1 during the study.

Table 5.7 - Concentrations of Nitrate-Nitrogen Measured in Treated Effluent Sent to Lysimeter 1

Date	Days (*)	Nitrate (mg/L)
		Lysimeter 1
12/2/2002	144	4.0
1/20/2003	193	2.0
2/13/2003	217	10.0
3/10/2003	242	12.0
4/15/2003	278	24.0
5/15/2003	308	22.0

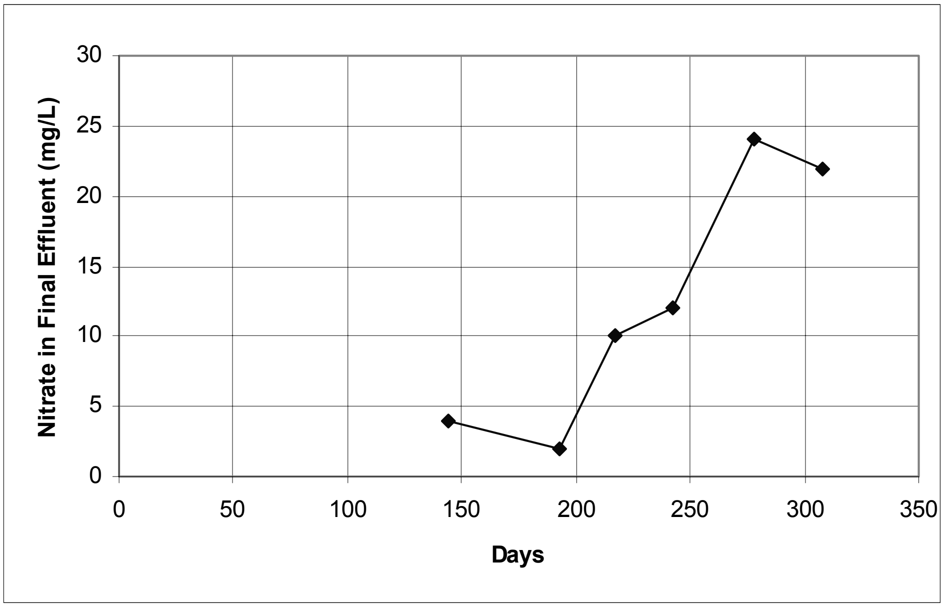

Figure 5.6 – Nitrate-Nitrogen Concentrations as a Function of Time Measured in Treated Effluent Sent to Lysimeter 1

From Table 5.7 and Figure 5.6, it can be noticed that nitrate-nitrogen concentrations in the effluent sent to Lysimeter 1 increased from 2 to 4 mg/L level on day 150 to a 25 mg/L level at the end of the study.

5.1.1.6. Total Kjeldahl Nitrogen (TKN)

Table 5.8 and Figure 5.7 present the concentrations of TKN measured throughout the period of study.

Table 5.8 – Concentrations of TKN Measured in the Lysimeters

Date	**Days (*)**	**Total Kjeldahl Nitrogen (mg/L)**		
		Lysimeter 1	**Lysimeter 2**	**Lysimeter 3**
8/8/2002	28	308	135	135
8/19/2002	39	240	315	255
9/3/2002	54	150	129	71
9/16/2002	67	90	78	48
1/20/2003	193	63	218	255
3/10/2003	242	225	322	287
5/15/2003	308	44	272	317

(*) Days elapsed since leachate production started

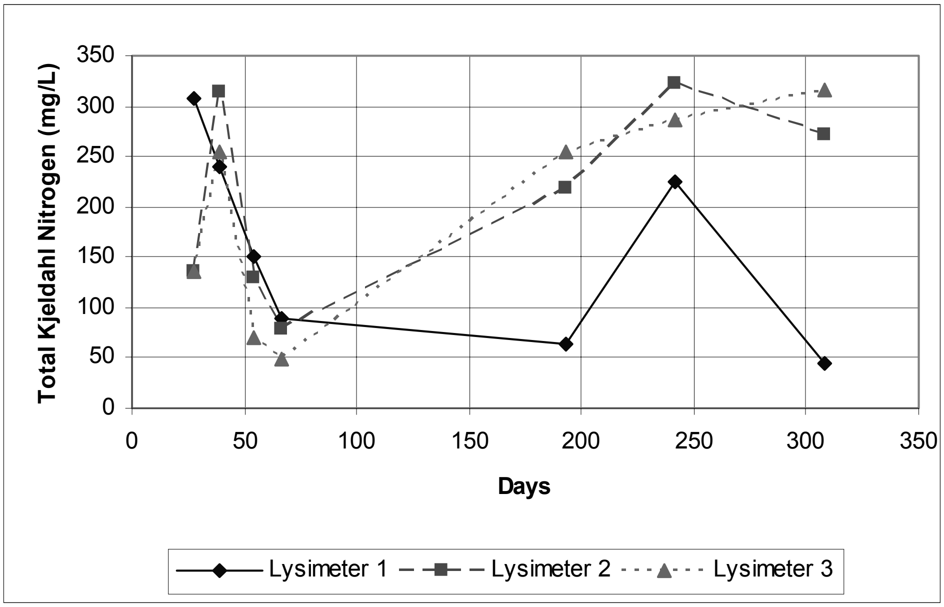

Figure 5.7 – TKN Concentrations as a Function of Time

TKN concentrations started around 300 mg/L for Lysimeter 1, decreasing to a 63 mg/L value on day 193, and then to a 44 mg/L value at the end of the study. A spiked value of 225 mg/L is noticed on day 242. For Lysimeters 2 and 3, TKN concentrations started increasing from 135 mg/L to a 250-315 mg/L level, then decreasing to a 50-80 mg/L level on day 67 and increasing again to a 320 mg/L level at the end of the study.

5.1.1.7. Total Volatile Acids (TVA)

Table 5.9 and Figure 5.8 present the concentrations of TVA measured throughout the period of study.

Table 5.9 – Concentrations of TVA Measured in the Lysimeters

Date	Days (*)	Total Volatile Acids (mg/L)		
		Lysimeter 1	Lysimeter 2	Lysimeter 3
8/14/2002	34	325	495	
8/20/2002	40	470	672	780
9/3/2002	54	17	36	45
9/16/2002	67	63	63	6
11/4/2002	116	1	4	1
12/2/2002	144	1	2	1
5/15/2003	308	2	3	1

(*) Days elapsed since leachate production started

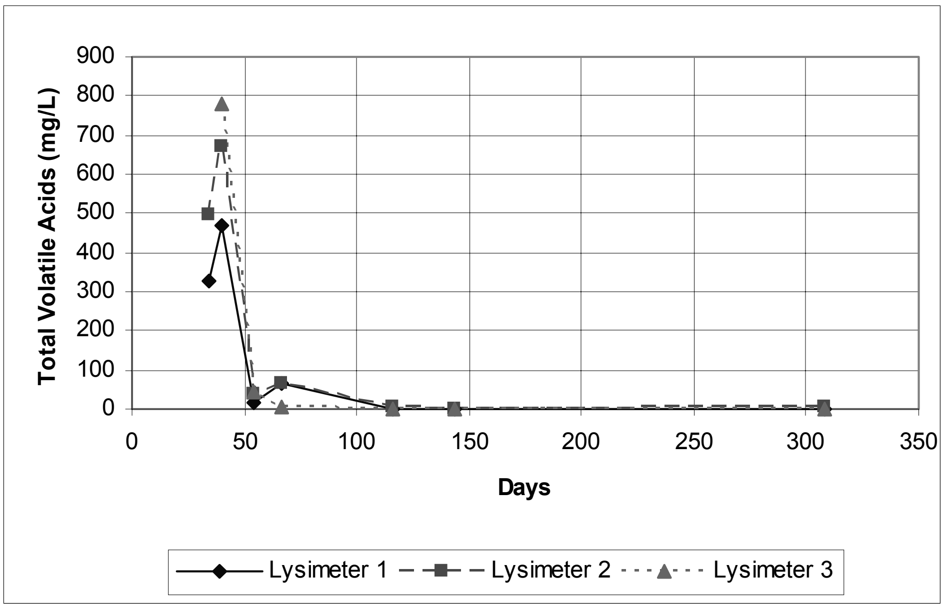

Figure 5.8 – TVA Concentrations as a Function of Time

TVA concentrations increased from the start and peaked on day 40 with values of 470, 672, and 780 mg/L for lysimeters 1, 2, and 3, respectively. The concentrations quickly decreased to a 40-60 mg/L level around day 60 and then decreased even more to a 1-4 mg/L level on day 116, staying at this level until the end of the study.

5.1.1.8. Total Phosphorus (TP)

Table 5.10 and Figure 5.9 present the concentrations of TP measured throughout the period of study.

Table 5.10 – Concentrations of Total Phosphorus Measured in the Lysimeters

Date	Days (*)	Total Phosphorus (mg/L)		
		Lysimeter 1	Lysimeter 2	Lysimeter 3
8/5/2002	25	1.2	0.5	0.5
8/8/2002	28	4.1	2.9	4.5
8/26/2002	46	10.0	18.8	6.3
9/9/2002	60	11.5	12.0	4.5
9/23/2002	74	6.0	1.5	0.5
11/4/2002	116	2.5	9.0	6.4
11/18/2002	130	2.5	8.4	3.2
12/2/2002	144	4.8	3.5	2.5
1/20/2003	193	2.3	4.6	1.7
2/13/2003	217	3.7	4.4	0.2
3/10/2003	242	2.6	4.4	0.8
4/15/2003	278	2.6	5.2	0.1
5/15/2003	308	2.5	3.6	0.7

(*) Days elapsed since leachate production started

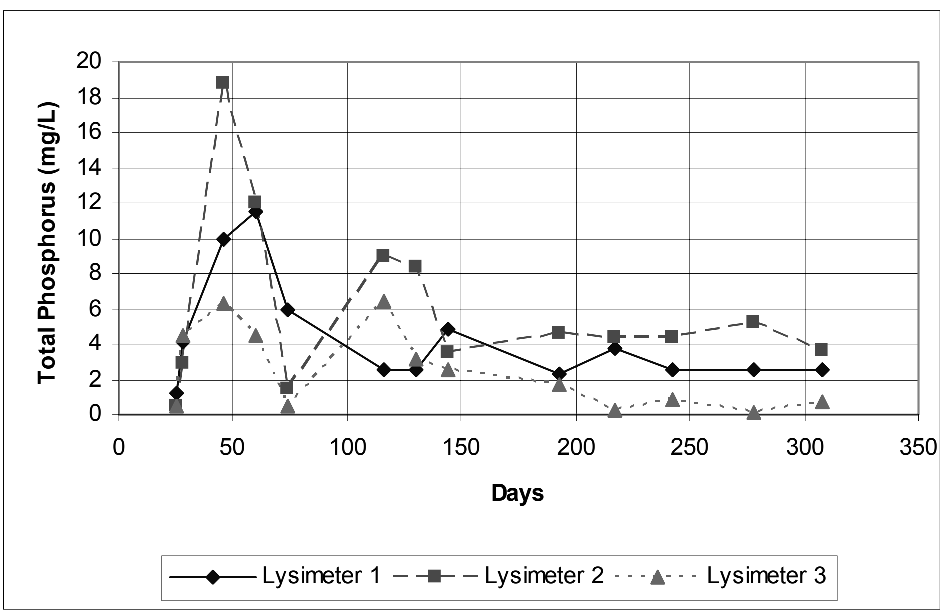

Figure 5.9 – Total Phosphorus Concentrations as a Function of Time

Total phosphorus concentrations started at a 1 mg/L level for Lysimeter 1, and quickly increased, peaking at 11.5 mg/L on day 60. The concentration then decreased to a 2.5 mg/L value on day 116, and stayed around this value until the end of the study. For Lysimeter 2, the TP concentration started at 0.5 mg/L, and then increased to 18.8 mg/L on day 46. It then fluctuated at a 1.5 to 9.0 mg/L range between days 74 and 144, and stayed at a 4 to 5 mg/L level until the end of the period of study. For Lysimeter 3, TP concentrations started at 0.5 mg/L, and increased to 6.3 mg/L on day 46. The concentrations then fluctuated at a 0.5 to 6.4 mg/L range between days 74 and 193, and staying at less than 1 mg/L until the end of the period of study.

5.1.1.9. Chloride

Table 5.11 and Figure 5.10 present the concentrations of chloride measured throughout the period of study.

Table 5.11 – Concentrations of Chloride Measured in the Lysimeters

Date	Days (*)	Chloride (mg/L)		
		Lysimeter 1	Lysimeter 2	Lysimeter 3
8/5/2002	25	355	245	130
8/8/2002	28	363	320	238
8/26/2002	46	278	563	375
9/9/2002	60	185	435	355
9/23/2002	74	125	465	260
11/4/2002	116	98	353	188
11/18/2002	130	153	398	162
12/2/2002	144	172	375	144
1/20/2003	193	186	382	123
2/13/2003	217	212	258	108
3/10/2003	242	145	264	84
4/15/2003	278	234	252	103
5/15/2003	308	244	288	66

(*) Days elapsed since leachate production started

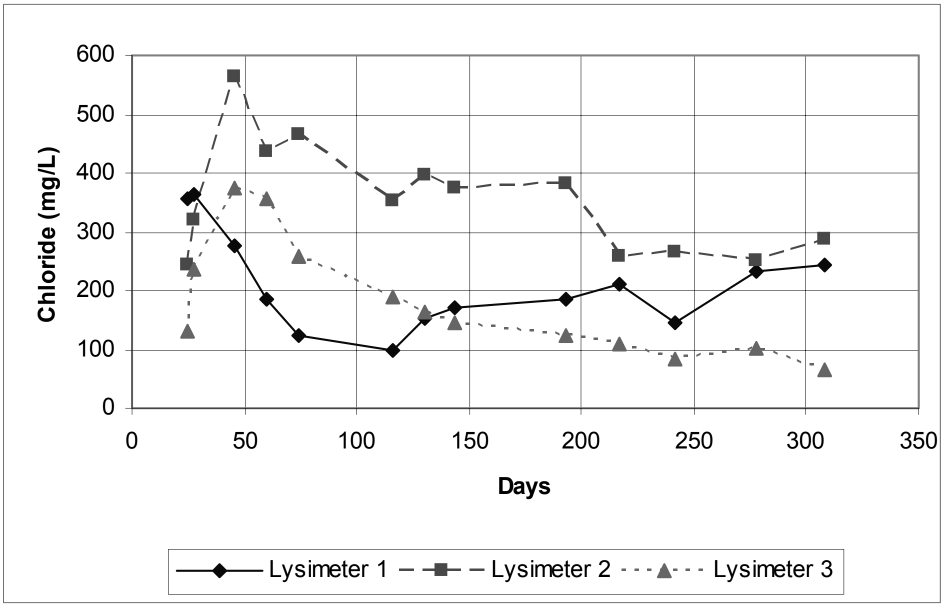

Figure 5.10 – Chloride Concentrations as a Function of Time

For Lysimeter 1, chloride concentrations started at 355 mg/L, decreased to 98 mg/L level on day 116, and then increased again, finishing at 244 mg/L at the end of the period of study. Lysimeters 2 and 3, respectively, started with concentrations of 245 mg/L and 130 mg/L, increasing to 563 mg/L and 375 mg/L on day 46, and then gradually decreased to approximate values of 250 mg/L and 70 mg/L at the end of the study.

5.1.1.10. Total Suspended Solids (TSS)

Table 5.12 and Figure 5.11 present the concentrations of TSS measured throughout the study.

Table 5.12 – Concentrations of TSS Measured in the Lysimeters

Date	Days (*)	Total Suspended Solids (mg/L)		
		Lysimeter 1	Lysimeter 2	Lysimeter 3
8/15/2002	35	75	166	139
8/26/2002	46	120	2037	127
9/9/2002	60	48	248	162
9/23/2002	74	72	92	200
11/4/2002	116	58	147	379
11/18/2002	130	100	118	
12/2/2002	144	12	72	60
1/20/2003	193	83	76	48
2/13/2003	217	112	142	63
3/10/2003	242	70	65	68
4/15/2003	278	211	63	95
5/15/2003	308	129	181	83

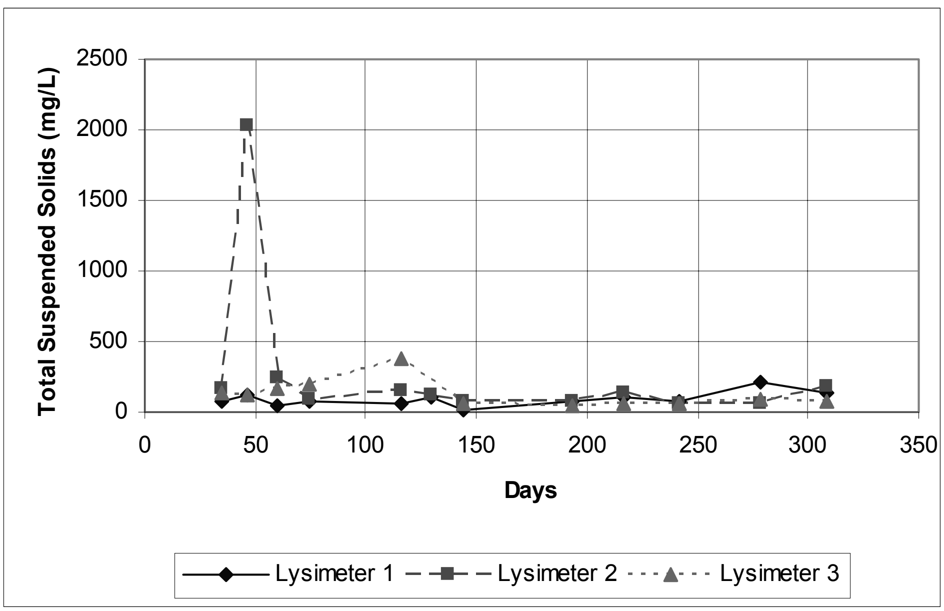

Figure 5.11 – TSS Concentrations as a Function of Time

Figure 5.12 presents the TSS concentrations as a function of time for the 0 to 500 mg/L range.

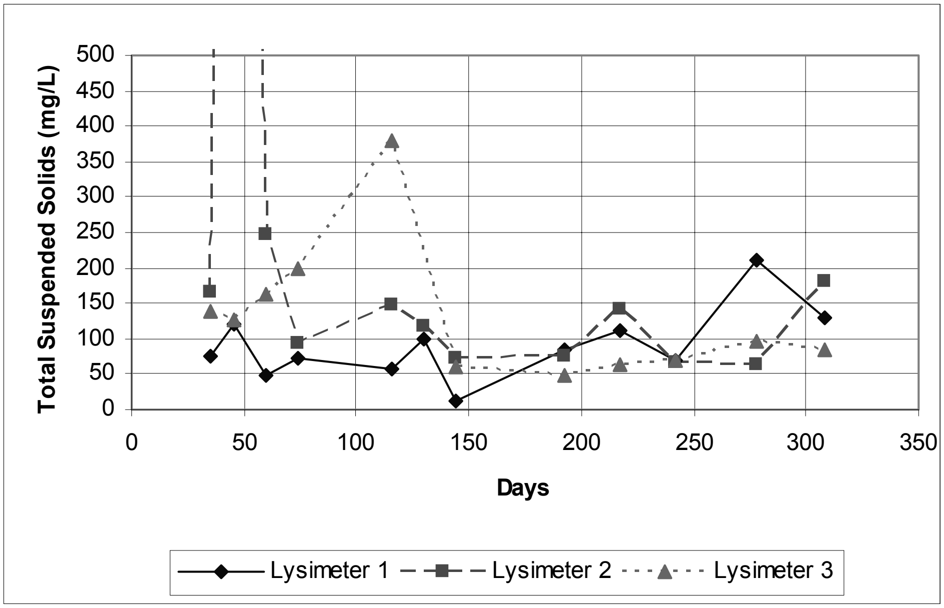

Figure 5.12 – TSS Concentrations as a Function of Time for 0 to 500 mg/L Range

In general, the TSS concentrations stayed between 50 mg/L and 200 mg/L throughout the study for all lysimeters. Clear exceptions were made to Lysimeter 2 that presented a value around 2,000 mg/L on day 46 and to Lysimeter 3, with a concentration value of 379 mg/L on day 116.

5.1.1.11. Volatile Suspended Solids (VSS)

Table 5.13 and Figure 5.13 present the concentrations of VSS measured throughout the study.

Figure 5.14 presents the VSS concentrations as a function of time for the 0 to 200 mg/L range.

Table 5.13 – Concentrations of VSS Measured in the Lysimeters

Date	Days (*)	Volatile Suspended Solids (mg/L)		
		Lysimeter 1	Lysimeter 2	Lysimeter 3
8/15/2002	35	61	102	82
8/26/2002	46	59	1090	67
9/9/2002	60	38	162	54
9/23/2002	74	2	98	22
11/4/2002	116	39	83	148
11/18/2002	130	57	66	
12/2/2002	144	8	42	44
1/20/2003	193	64	33	17
2/13/2003	217	61	58	32
3/10/2003	242	30	17	52
4/15/2003	278	37	16	54
5/15/2003	308	25	55	53

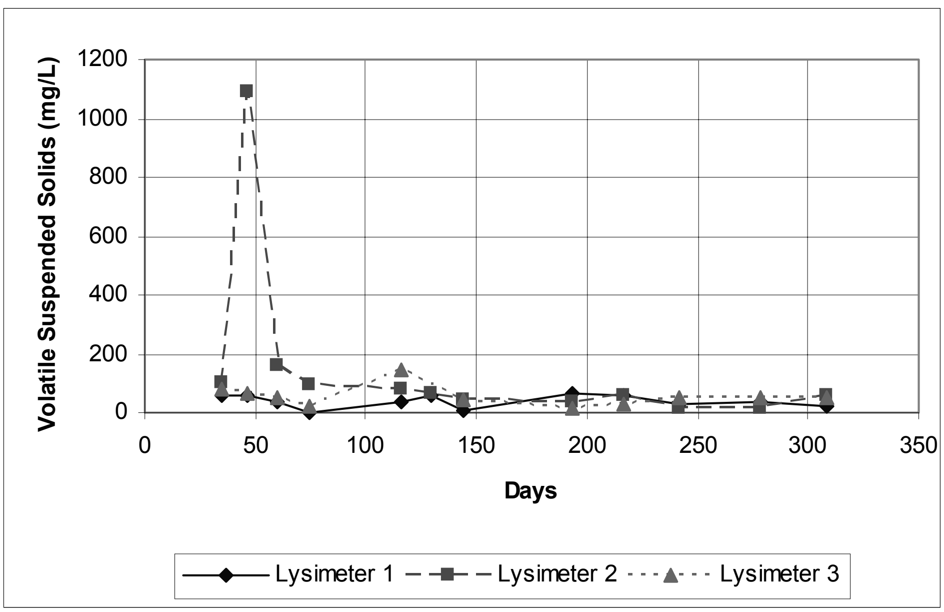

Figure 5.13 – VSS Concentrations as a Function of Time

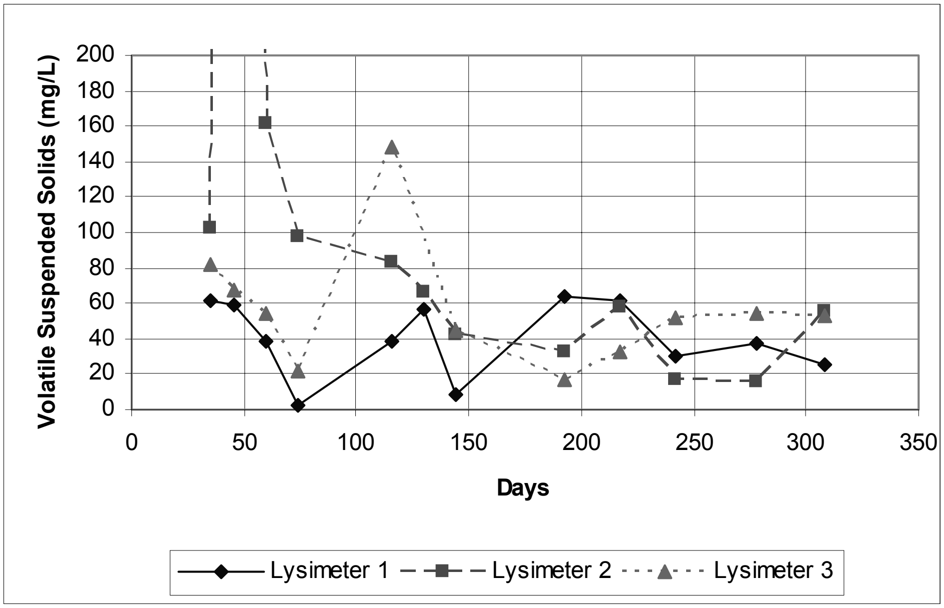

Figure 5.14 – VSS Concentrations as a Function of Time for 0 to 200 mg/L Range

In general, the VSS concentrations stayed between 20 mg/L and 80 mg/L throughout the study for all lysimeters. Clear exceptions were made to Lysimeter 2 that presented a value around 1,100 mg/L on day 46 and to Lysimeter 3, with a concentration value of 148 mg/L on day 116.

5.1.1.12. Iron

Table 5.14 and Figure 5.15 present the concentrations of iron measured throughout the study.

Table 5.14 – Concentrations of Iron Measured in the Lysimeters

Date	Days (*)	Iron (mg/L)		
		Lysimeter 1	Lysimeter 2	Lysimeter 3
8/14/2002	34	4.0	5.9	
8/26/2002	46	11.5	18.0	22.8
9/9/2002	60	22.5	15.0	4.0
9/23/2002	74	5.8	12.5	5.5
11/4/2002	116	6.0	2.5	4.0
11/18/2002	130	4.2	6.4	2.7
12/2/2002	144	2.5	8.2	4.2
1/20/2003	193	2.8	7.3	4.6
2/13/2003	217	3.2	7.1	4.2
3/10/2003	242	2.6	7.2	3.6
4/15/2003	278	2.5	8.8	2.9
5/15/2003	308	3.4	6.7	2.9

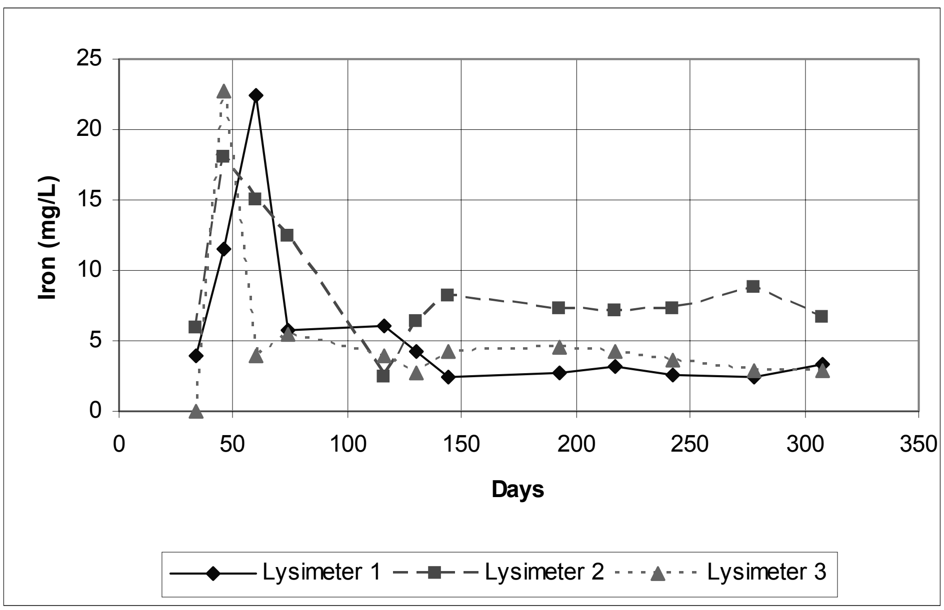

Figure 5.15 – Iron Concentrations as a Function of Time

Iron concentrations started at a 4 mg/L value for Lysimeter 1, and quickly increased, peaking at 22.5 mg/L on day 60, and then decreased to a 2.5 mg/L value on day 144, staying around this value until the end of the study. For Lysimeter 2, the iron concentration started at 5.9 mg/L, increased to 18.0 mg/L on day 46, decreased to 2.5 mg/L on day 116, and finally increased and stayed at a 6 to 8 mg/L level until the end of the study. For Lysimeter 3, iron concentrations peaked at 22.8 mg/L on day 46, immediately decreasing to a 3 to 5 mg/L range, and staying at this value until the end of the study.

5.1.1.13. Aluminum

Table 5.15 and Figure 5.16 present the concentrations of aluminum measured throughout the study.

Table 5.15 – Concentrations of Aluminum Measured in the Lysimeters

Date	Days (*)	Aluminum (mg/L)		
		Lysimeter 1	Lysimeter 2	Lysimeter 3
8/14/2002	34	0.10	0.20	
8/26/2002	46	0.10	0.08	0.14
9/9/2002	60	0.50	1.00	0.20
9/23/2002	74	0.10	0.10	0.10
11/4/2002	116	0.12	0.06	0.02
11/18/2002	130	0.36	0.20	0.04
12/2/2002	144	0.28	0.13	0.11
1/20/2003	193	0.11	0.12	0.21
2/13/2003	217	<0.01	0.17	0.05
3/10/2003	242	0.06	0.05	0.06
4/15/2003	278	0.19	0.05	0.13
5/15/2003	308	0.06	0.05	0.05

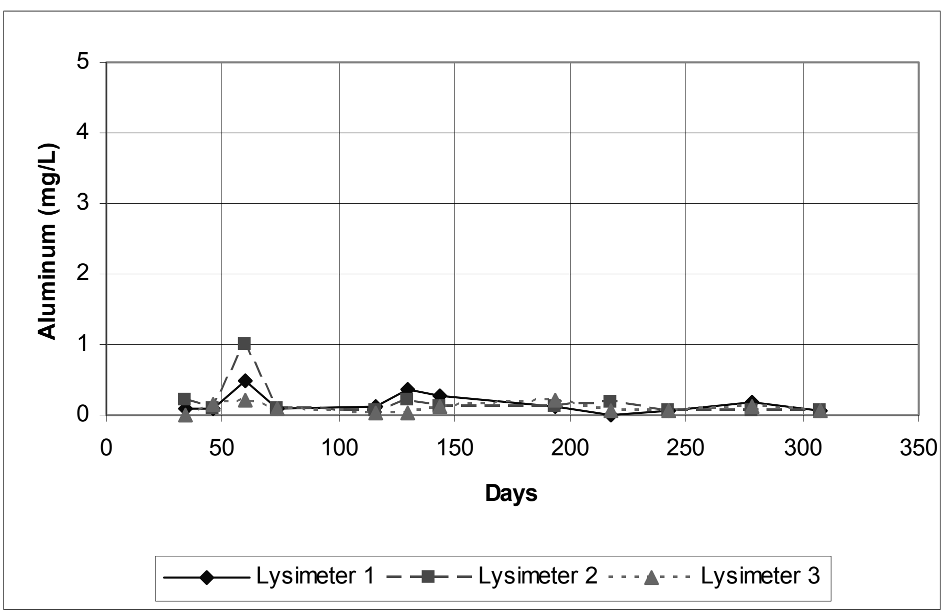

Figure 5.16 – Aluminum Concentrations as a Function of Time

In general, aluminum concentrations stayed at very low values (< 0.20 mg/L) for all lysimeters throughout the study. The maximum value noticed was equal to 1 mg/L for Lysimeter 2 on day 60.

5.1.2. Leachate Generation

As mentioned in Section 4.2.3, all lysimeters were brought to field capacity by saturating them with tap water and then draining. The initial volumes of water placed inside and drained were measured and the moisture content at field capacity could be calculated for all lysimeters. Table 5.16 presents the calculation of the moisture content at field capacity on dry and wet basis for each lysimeter.

Table 5.16 – Calculation of Moisture Content at Field Capacity

Lysimeter	Volume of Water In (L)	Volume of Liquids Out (L)	Volume of Water Retained (L)	Dry Waste Weight (kg)	Waste Volume (m^3)	Moisture Content (%) - Field Capacity	
						Dry Basis	Wet Basis
1	409.3	359.3	50.0	88.2	0.581	56.7	36.2
2	421.8	403.5	18.3	88.2	0.580	20.7	17.2
3	417.5	385.8	31.7	88.2	0.595	35.9	26.4

As mentioned in Section 4.2.8, leachate production was measured at a regular basis for the three lysimeters. For Lysimeters 1 and 2 the leachate production was measured daily due to the procedures of leachate recirculation. For Lysimeter 3 the production was measured typically twice a month. After October 2002, the production of leachate generated by Lysimeters 1 and 2 was not measured, since overflow started to occur in the respective receiving tanks.

Tables 5.17, 5.18, and 5.19 present the balance of liquids introduced into and generated by Lysimeters 1, 2, and 3, respectively.

Table 5.17 – Balance of Liquids in Lysimeter 1

Week	Date - Beginning of Week	Water Added (L)	Recirculation Rate (L/day)	Leachate Introduced (L)	Accumulated Liquids Added (L)	Accumulated Leachate Generated (L)	Ratio Leachate/ Liquids
1	7/1/2002	3.85	0	0	3.85	0.0	0.00
2	7/8/2002	3.85	0	0	7.70	0.0	0.00
3	7/15/2002	3.85	0	0	11.55		
4	7/22/2002	3.85	0	0	15.40		
5	7/29/2002	3.85	0	0	19.25		
6	8/5/2002	3.85	0	0	23.10	19.3	0.84
7	8/12/2002	3.85	0	0	26.95	22.6	0.84
8	8/19/2002	3.85	0	0	30.80	25.3	0.82
9	8/26/2002	3.85	15	75	109.65	103.1	0.94
10	9/2/2002	3.85	15	75	188.50	181.3	0.96
11	9/9/2002	3.85	15	75	267.35	258.9	0.97
12	9/16/2002	3.85	23	115	386.20	376.9	0.98
13	9/23/2002	1.93	0/23(*)	46	434.13	423.4	0.98
14	9/30/2002	1.93	30	150	586.06	574.3	0.98
15	10/7/2002	3.85	38	190	779.91	767.4	0.98
16	10/14/2002	3.85	38	190	973.76	960.2	0.99
17	10/21/2002	3.85	38	190	1167.61	1153.3	0.99

(*) Recirculation not done in 3 days during T.S. Isidore

Table 5.18 – Balance of Liquids in Lysimeter 2

Week	Date - Beginning of Week	Water Added (L)	Recirculation Rate (L/day)	Leachate Introduced (L)	Accumulated Liquids Added (L)	Accumulated Leachate Generated (L)	Ratio Leachate/ Liquids
1	7/1/2002	3.85	0	0	3.85	0.0	0.00
2	7/8/2002	3.85	0	0	7.70	0.0	0.00
3	7/15/2002	3.85	0	0	11.55	0.0	0.00
4	7/22/2002	3.85	0	0	15.40	0.0	0.00
5	7/29/2002	3.85	0	0	19.25		
6	8/5/2002	3.85	0	0	23.10	11.0	0.48
7	8/12/2002	3.85	0	0	26.95	14.1	0.52
8	8/19/2002	3.85	0	0	30.80	18.6	0.60
9	8/26/2002	3.85	15	75	109.65	97.2	0.89
10	9/2/2002	3.85	15	75	188.50	175.3	0.93
11	9/9/2002	3.85	15	75	267.35	252.7	0.95
12	9/16/2002	3.85	23	115	386.20	371.0	0.96
13	9/23/2002	1.93	0/23(*)	46	434.13	418.4	0.96
14	9/30/2002	1.93	30	150	586.06	571.2	0.97
15	10/7/2002	3.85	38	190	779.91	764.1	0.98
16	10/14/2002	3.85	38	190	973.76	957.8	0.98
17	10/21/2002	3.85	38	190	1167.61	1151.5	0.99

(*) Recirculation not done in 3 days during T.S. Isidore

Table 5.19 – Balance of Liquids in Lysimeter 3

Week	Date - Beginning of Week	Water Added (L)	Recirculation Rate (L/day)	Leachate Introduced (L)	Accumulated Liquids Added (L)	Accumulated Leachate Generated (L)	Ratio Leachate/ Liquids
1	7/1/2002	3.85	0	0	3.85	0.0	0.00
2	7/8/2002	3.85	0	0	7.70	0.0	0.00
3	7/15/2002	3.85	0	0	11.55	0.0	0.00
4	7/22/2002	3.85	0	0	15.40	0.0	0.00
5	7/29/2002	3.85	0	0	19.25		
6	8/5/2002	3.85	0	0	23.10	13.8	0.60
7	8/12/2002	3.85	0	0	26.95	17.5	0.65
8	8/19/2002	3.85	0	0	30.80		
9	8/26/2002	3.85	0	0	34.65	25.2	0.73
10	9/2/2002	3.85	0	0	38.50		
11	9/9/2002	3.85	0	0	42.35	32.2	0.76
12	9/16/2002	3.85	0	0	46.20		
13	9/23/2002	1.93	0	0	48.13		
14	9/30/2002	1.93	0	0	50.06		
15	10/7/2002	3.85	0	0	53.91	43.2	0.80
16	10/14/2002	3.85	0	0	57.76		
17	10/21/2002	3.85	0	0	61.61	50.8	0.82
18	10/28/2002	1.93	0	0	63.54		
19	11/4/2002	3.85	0	0	67.39	55.2	0.82
20	11/11/2002	3.85	0	0	71.24		
21	11/18/2002	3.85	0	0	75.09	62.9	0.84
22	11/25/2002	3.85	0	0	78.94		
23	12/2/2002	3.85	0	0	82.79	69.9	0.84
24	12/9/2002	3.85	0	0	86.64	73.8	0.85
25	12/16/2002	0	0	0	86.64		
26	12/23/2002	0	0	0	86.64		

Table 5.19 – Balance of Liquids in Lysimeter 3 (cont.)

Week	Date - Beginning of Week	Water Added (L)	Recirculation Rate (L/day)	Leachate Introduced (L)	Accumulated Liquids Added (L)	Accumulated Leachate Generated (L)	Ratio Leachate/ Liquids
27	12/30/2002	0	0	0	86.64		
28	1/6/2003	0	0	0	86.64		
29	1/13/2003	3.85	0	0	90.49	76.8	0.85
30	1/20/2003	3.85	0	0	94.34		
31	1/27/2003	3.85	0	0	98.19	84.6	0.86
32	2/3/2003	3.85	0	0	102.04		
33	2/10/2003	3.85	0	0	105.89	91.7	0.87
34	2/17/2003	3.85	0	0	109.74		
35	2/24/2003	3.85	0	0	113.59	99.6	0.88
36	3/3/2003	3.85	0	0	117.44		
37	3/10/2003	3.85	0	0	121.29	107.1	0.88
38	3/17/2003	3.85	0	0	125.14		
39	3/24/2003	3.85	0	0	128.99	114.7	0.89
40	3/31/2003	3.85	0	0	132.84		
41	4/7/2003	3.85	0	0	136.69	122.1	0.89
42	4/14/2003	3.85	0	0	140.54		
43	4/21/2003	3.85	0	0	144.39	129.7	0.90
44	4/28/2003	3.85	0	0	148.24		
45	5/5/2003	3.85	0	0	152.09	137.5	0.90
46	5/12/2003	3.85	0	0	155.94		
47	5/19/2003	3.85	0	0	159.79	145.1	0.91

Figures 5.17, 5.18, and 5.19 present the accumulated leachate production as a function of time for Lysimeters 1, 2, and 3, respectively.

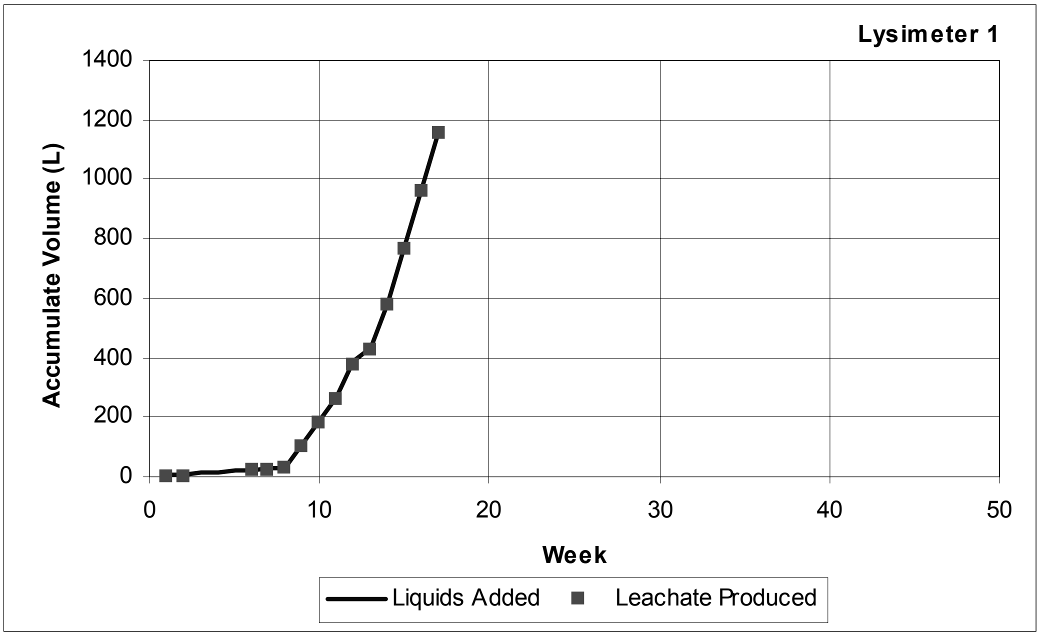

Figure 5.17 – Accumulated Leachate Generation as a Function of Time for Lysimeter 1

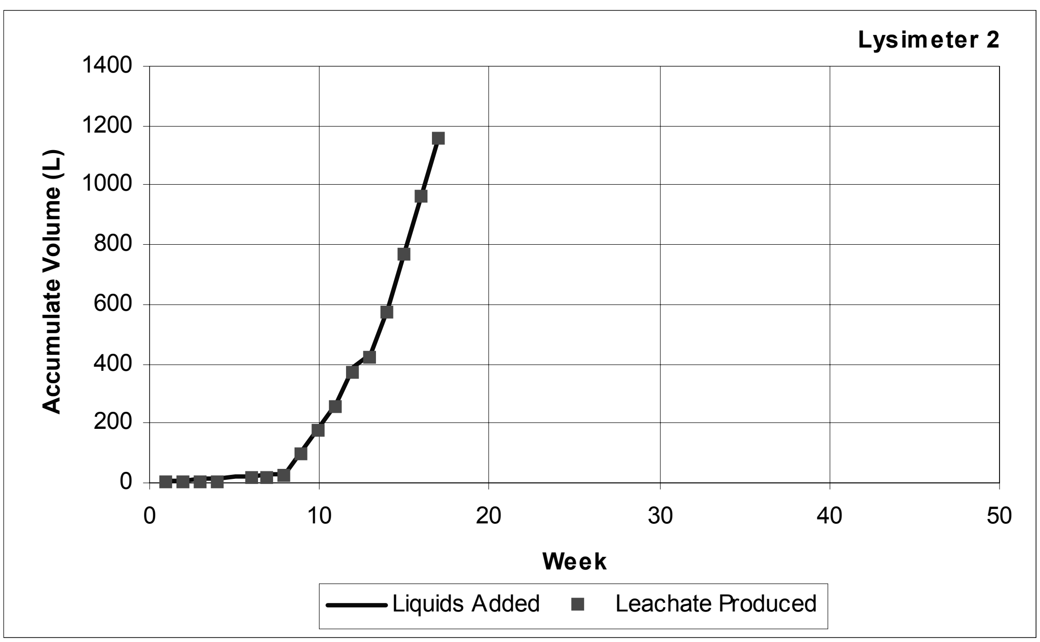

Figure 5.18 – Accumulated Leachate Generation as a Function of Time for Lysimeter 2

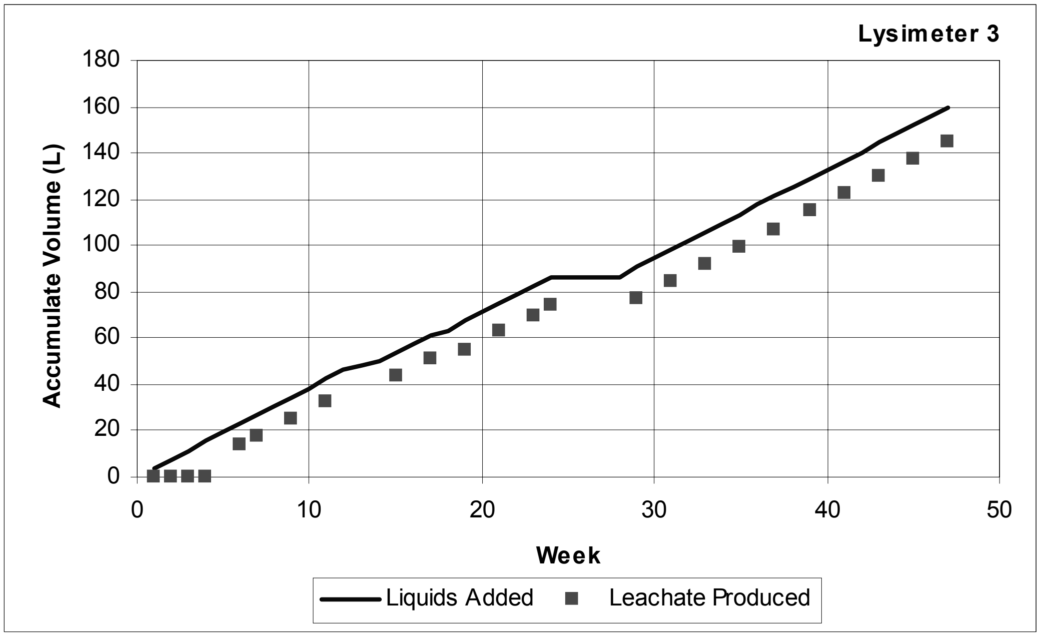

Figure 5.19 – Accumulated Leachate Generation as a Function of Time for Lysimeter 3

As seen from Tables 5.17, 5.18, and 5.19 and from Figures 5.17, 5.18, and 5.19, Lysimeters 1 and 2 produced about 1,200 liters of leachate in 17 weeks, while Lysimeter 3 produced a little more than 60 liters during the same period.

5.1.3. Gas Composition

As mentioned in Section 4.2.6, gas composition as a percentage of volume was measured in the three lysimeters for methane, carbon dioxide, and oxygen during the period of study. Table 5.20 presents the gas composition measured in the three lysimeters, while Figures 5.20, 5.21, and 5.22 present the variation of gas composition as a function of time for Lysimeters 1, 2, and 3, respectively. For those table and graphs, day 0 (zero) was considered as the first day that leachate was observed being generated by the lysimeters, i.e., July 11th 2002. The "other gases" entry corresponds to the difference of 100% and the sum of methane, carbon dioxide, and oxygen percentages.

Table 5.20 - Gas Composition Measured in the Lysimeters

		Lysimeter 1				Lysimeter 2				Lysimeter 3			
Date	**Days**	**CH_4 (%)**	**CO_2 (%)**	**O_2 (%)**	**Other Gases (%)**	**CH_4 (%)**	**CO_2 (%)**	**O_2 (%)**	**Other Gases (%)**	**CH_4 (%)**	**CO_2 (%)**	**O_2 (%)**	**Other Gases (%)**
11/7/2002	119	0	8	6	86	25	47	0	28	23	54	0	23
11/11/2002	123	1	9	8	82	24	41	0	35	24	52	0	24
11/18/2002	130	2	7	5	86	28	43	2	27	23	61	2	14
11/26/2002	138	1	7	3	89	23	44	0	33	21	63	4	12
12/2/2002	144	2	7	5	86	27	48	0	25	29	67	2	2
12/9/2002	151	1	12	3	84	32	63	3	2	27	72	0	1
1/17/2003	190	16	23	0	61	19	67	0	14	20	76	2	2
2/4/2003	208	23	21	3	53	23	58	1	18	25	63	0	12
2/17/2003	221	5	18	2	75	24	63	2	11	24	66	0	10
3/6/2003	238	3	15	3	79	24	48	0	28	26	68	2	4
3/17/2003	249	1	11	1	87	27	61	2	10	25	53	0	22
3/31/2003	263	4	13	6	77	35	64	0	1	23	72	1	4
4/14/2003	277	3	5	11	81	37	55	0	8	27	71	0	2
4/28/2003	291	1	7	5	87	38	54	3	5	28	60	0	12
5/6/2003	299	2	7	8	83	36	53	0	11	26	71	1	2
5/22/2003	315	7	8	2	83	39	53	0	8	26	72	0	2

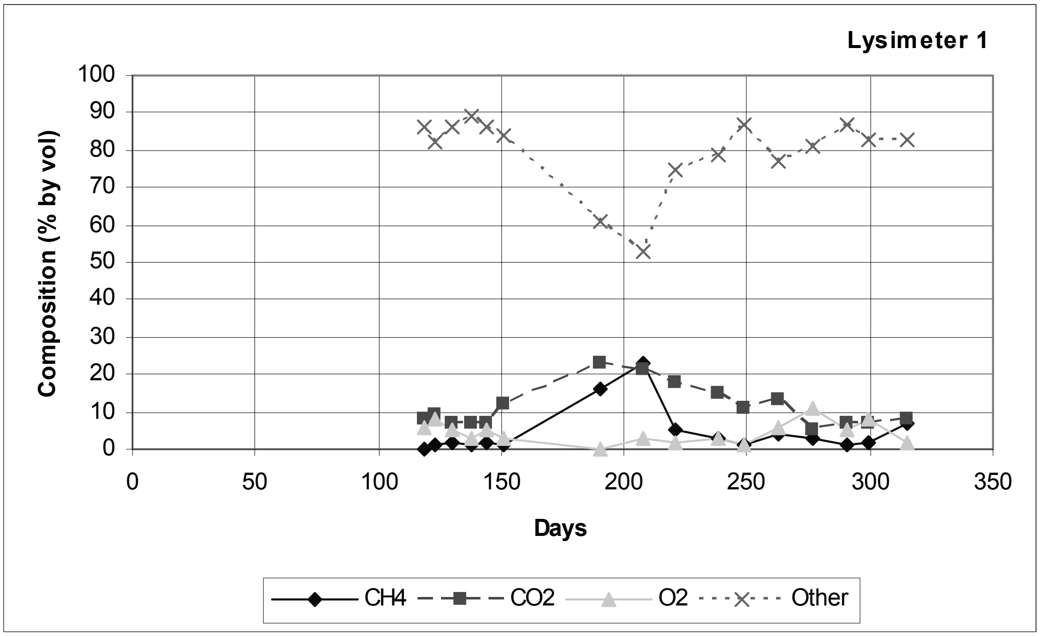

Figure 5.20 – Gas Composition as a Function of Time in Lysimeter 1

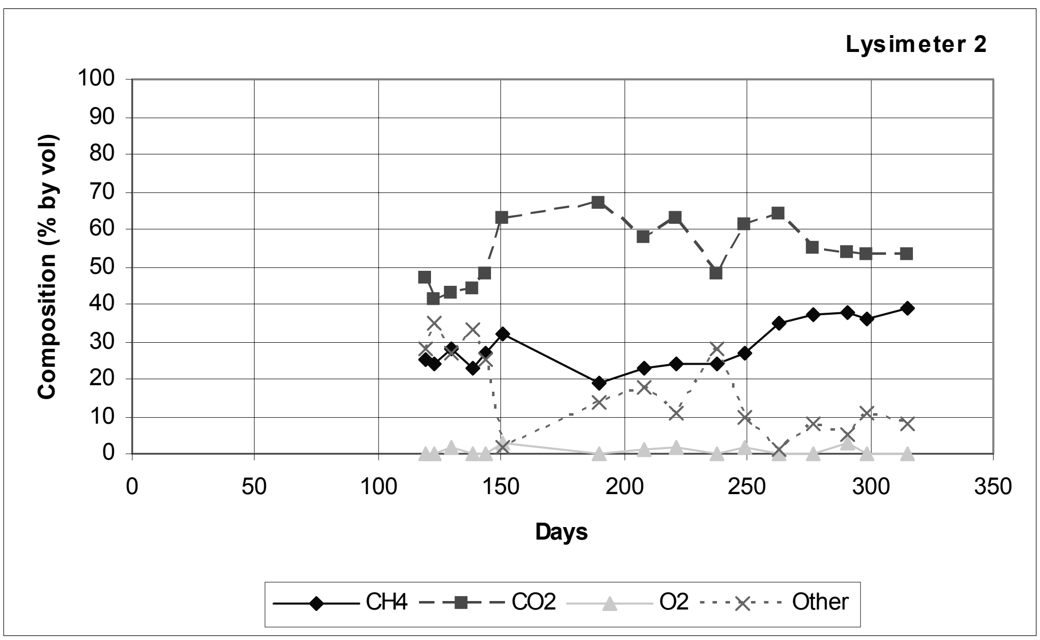

Figure 5.21 – Gas Composition as a Function of Time in Lysimeter 2

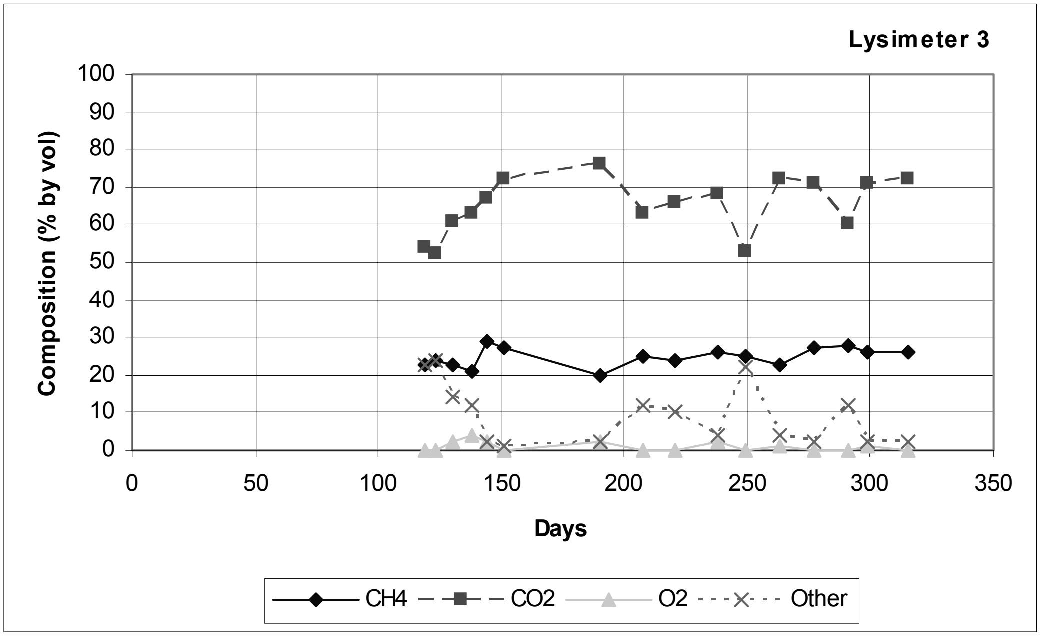

Figure 5.22 – Gas Composition as a Function of Time in Lysimeter 3

The analysis of gas composition results will be presented in Chapter 6.

5.2. Geotechnical Study

5.2.1. Settlement Measurements

As mentioned in Section 4.2.7, settlement of the waste inside the three lysimeters was measured at regular intervals to compare and evaluate the long-term compressibility mechanism of landfills.

The settlements in the three lysimeters are herein divided into two categories: immediate settlement and time-dependent settlement. The immediate settlement was measured immediately after the layer of pea gravel and the concrete weight were placed inside each lysimeter. The time-dependent settlement corresponds to the settlement observed after the immediate settlement was measured (as a function of time). The total settlement corresponds, therefore, to the sum of the immediate and time-dependent settlements. Tables 5.21 and 5.22 present, respectively, the total settlement and the total relative settlement (as a percentage value - dividing the total settlement by the initial thickness of the waste layer). Figures 5.23 and 5.24 show the total settlement and total relative settlement as a function of time, respectively.

Table 5.21 – Total Settlement Measured in the Lysimeters

Date	**Days**	**Total Settlement (cm)**		
		Lysimeter 1	**Lysimeter 2**	**Lysimeter 3**
5/2/2002	0	35.6	22.8	17.8
5/7/2002	5	37.5	27.9	22.9
5/11/2002	9	38.8	29.8	24.2
5/28/2002	26	42.0	32.6	28.6
6/15/2002	44	44.5	36.1	32.4
6/20/2002	49	45.4	36.8	
6/21/2002	50	45.8	37.1	33.0
6/24/2002	53	46.1	37.7	33.4
6/28/2002	57	46.7	38.7	34.0
7/3/2002	62	47.7	39.6	34.8
7/8/2002	67	48.9	40.3	35.6
7/15/2002	74	50.2	41.5	36.2
7/22/2002	81	51.8	42.8	37.5
7/29/2002	88	53.7	43.4	38.3
8/5/2002	95	54.3	43.8	38.6
8/12/2002	102	55.1	44.2	38.8
8/19/2002	109	55.9	45.0	39.1
8/26/2002	116	56.6	45.3	39.7
9/2/2002	123	56.9	46.0	39.9
9/9/2002	130	57.2	46.1	40.0
9/16/2002	137	58.1	46.9	40.2
9/23/2002	144	58.8	47.6	40.3
9/30/2002	151	58.9	48.2	40.5
10/7/2002	158	59.4	48.2	40.8
10/14/2002	165	59.4	48.2	40.8
10/30/2002	181	59.6	48.7	41.3
11/11/2002	193	60.0	49.2	41.6
11/25/2002	207	60.0	50.1	41.6
12/9/2002	221	60.4	50.4	41.9
1/17/2003	260	60.7	51.4	42.4
2/17/2003	291	61.3	52.6	43.5
3/17/2003	319	62.6	53.3	45.1
4/14/2003	347	63.2	54.6	46.4
5/27/2003	390	64.8	55.5	47.3

Table 5.22 –Total Relative Settlement Measured in the Lysimeters

Date	Days	Total Relative Settlement (%)		
		Lysimeter 1	Lysimeter 2	Lysimeter 3
5/2/2002	0	15.2	10.3	8.0
5/7/2002	5	16.0	12.6	10.3
5/11/2002	9	16.5	13.4	10.9
5/28/2002	26	17.9	14.7	12.9
6/15/2002	44	19.0	16.3	14.6
6/20/2002	49	19.4	16.6	
6/21/2002	50	19.5	16.7	14.9
6/24/2002	53	19.6	17.0	15.1
6/28/2002	57	19.9	17.5	15.3
7/3/2002	62	20.3	17.9	15.7
7/8/2002	67	20.9	18.2	16.1
7/15/2002	74	21.4	18.7	16.3
7/22/2002	81	22.1	19.3	16.9
7/29/2002	88	22.9	19.6	17.3
8/5/2002	95	23.2	19.7	17.4
8/12/2002	102	23.5	20.0	17.5
8/19/2002	109	23.8	20.3	17.6
8/26/2002	116	24.1	20.5	17.9
9/2/2002	123	24.3	20.7	18.0
9/9/2002	130	24.4	20.8	18.1
9/16/2002	137	24.8	21.2	18.1
9/23/2002	144	25.1	21.5	18.2
9/30/2002	151	25.1	21.8	18.3
10/7/2002	158	25.3	21.8	18.4
10/14/2002	165	25.3	21.8	18.4
10/30/2002	181	25.4	22.0	18.6
11/11/2002	193	25.6	22.2	18.8
11/25/2002	207	25.6	22.6	18.8
12/9/2002	221	25.7	22.8	18.9
1/17/2003	260	25.9	23.2	19.1
2/17/2003	291	26.1	23.8	19.6
3/17/2003	319	26.7	24.0	20.4
4/14/2003	347	27.0	24.6	20.9
5/27/2003	390	27.6	25.0	21.4

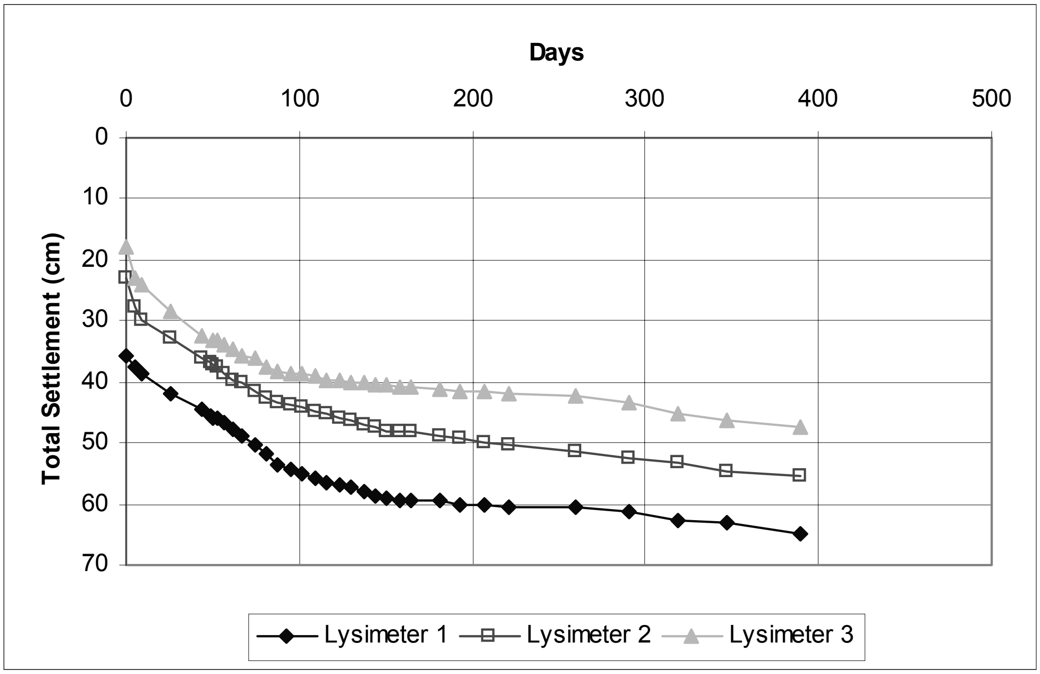

Figure 5.23 – Total Settlement as a Function of Time

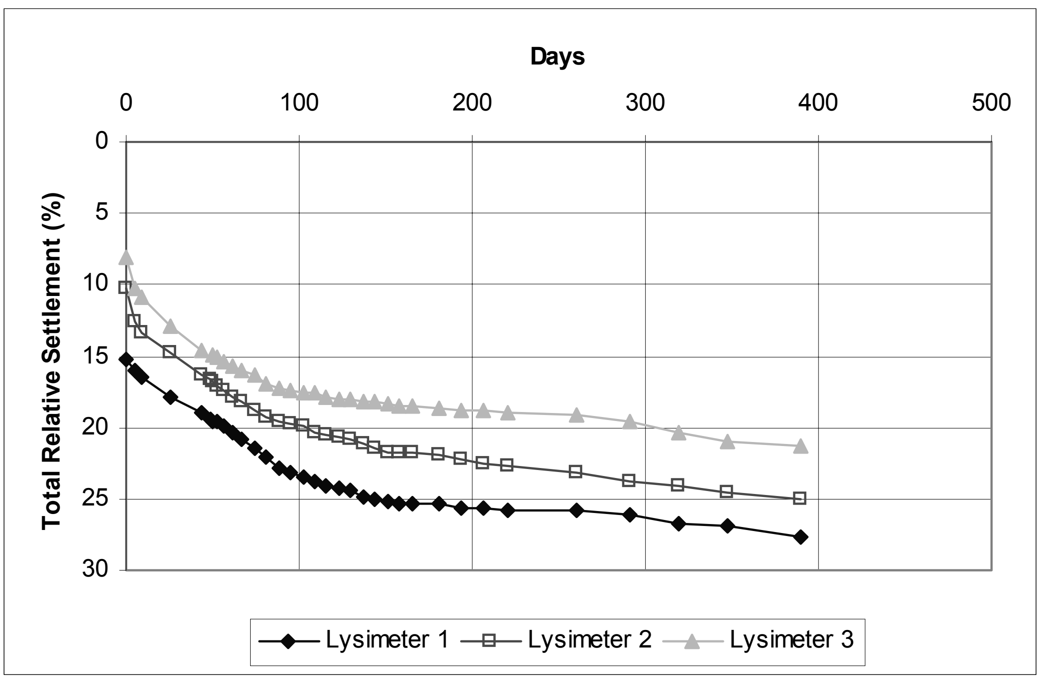

Figure 5.24 – Total Relative Settlement as a Function of Time

Tables 5.23 and 5.24 present, respectively, the time-dependent settlement and the time-dependent relative settlement (as a percentage value - dividing the time-dependent settlement by the initial thickness of the waste layer). Figures 5.25 and 5.26 show the time-dependent settlement and time-dependent relative settlement as a function of time, respectively.

Table 5.23 – Time-Dependent Settlement Measured in the Lysimeters

Date	Days	Time-Dependent Settlement (cm)		
		Lysimeter 1	Lysimeter 2	Lysimeter 3
5/2/2002	0	0	0	0
5/7/2002	5	1.9	5.1	5.1
5/11/2002	9	3.2	7.0	6.4
5/28/2002	26	6.4	9.8	10.8
6/15/2002	44	8.9	13.3	14.6
6/20/2002	49	9.8	14.0	
6/21/2002	50	10.2	14.3	15.2
6/24/2002	53	10.5	14.9	15.6
6/28/2002	57	11.1	15.9	16.2
7/3/2002	62	12.1	16.8	17.0
7/8/2002	67	13.3	17.5	17.8
7/15/2002	74	14.6	18.7	18.4
7/22/2002	81	16.2	20.0	19.7
7/29/2002	88	18.1	20.6	20.5
8/5/2002	95	18.7	21.0	20.8
8/12/2002	102	19.5	21.4	21.0
8/19/2002	109	20.3	22.2	21.3
8/26/2002	116	21.0	22.5	21.9
9/2/2002	123	21.3	23.2	22.1
9/9/2002	130	21.6	23.3	22.2
9/16/2002	137	22.5	24.1	22.4
9/23/2002	144	23.2	24.8	22.5
9/30/2002	151	23.3	25.4	22.7
10/7/2002	158	23.8	25.4	23.0
10/14/2002	165	23.8	25.4	23.0
10/30/2002	181	24.0	25.9	23.5
11/11/2002	193	24.4	26.4	23.8
11/25/2002	207	24.4	27.3	23.8
12/9/2002	221	24.8	27.6	24.1
1/17/2003	260	25.1	28.6	24.6
2/17/2003	291	25.7	29.8	25.7
3/17/2003	319	27.0	30.5	27.3
4/14/2003	347	27.6	31.8	28.6
5/27/2003	390	29.2	32.7	29.5

Table 5.24 – Time-Dependent Relative Settlement Measured in the Lysimeters

Date	Days	Time-Dependent Relative Settlement (%)		
		Lysimeter 1	Lysimeter 2	Lysimeter 3
5/2/2002	0	0.0	0.0	0.0
5/7/2002	5	0.8	2.3	2.3
5/11/2002	9	1.4	3.2	2.9
5/28/2002	26	2.7	4.4	4.9
6/15/2002	44	3.8	6.0	6.6
6/20/2002	49	4.2	6.3	
6/21/2002	50	4.3	6.4	6.9
6/24/2002	53	4.5	6.7	7.0
6/28/2002	57	4.7	7.2	7.3
7/3/2002	62	5.1	7.6	7.7
7/8/2002	67	5.7	7.9	8.0
7/15/2002	74	6.2	8.5	8.3
7/22/2002	81	6.9	9.0	8.9
7/29/2002	88	7.7	9.3	9.2
8/5/2002	95	8.0	9.5	9.4
8/12/2002	102	8.3	9.7	9.5
8/19/2002	109	8.7	10.0	9.6
8/26/2002	116	8.9	10.2	9.9
9/2/2002	123	9.1	10.5	10.0
9/9/2002	130	9.2	10.5	10.0
9/16/2002	137	9.6	10.9	10.1
9/23/2002	144	9.9	11.2	10.2
9/30/2002	151	10.0	11.5	10.2
10/7/2002	158	10.2	11.5	10.4
10/14/2002	165	10.2	11.5	10.4
10/30/2002	181	10.2	11.7	10.6
11/11/2002	193	10.4	11.9	10.7
11/25/2002	207	10.4	12.3	10.7
12/9/2002	221	10.6	12.5	10.9
1/17/2003	260	10.7	12.9	11.1
2/17/2003	291	11.0	13.5	11.6
3/17/2003	319	11.5	13.8	12.3
4/14/2003	347	11.8	14.3	12.9
5/27/2003	390	12.5	14.8	13.3

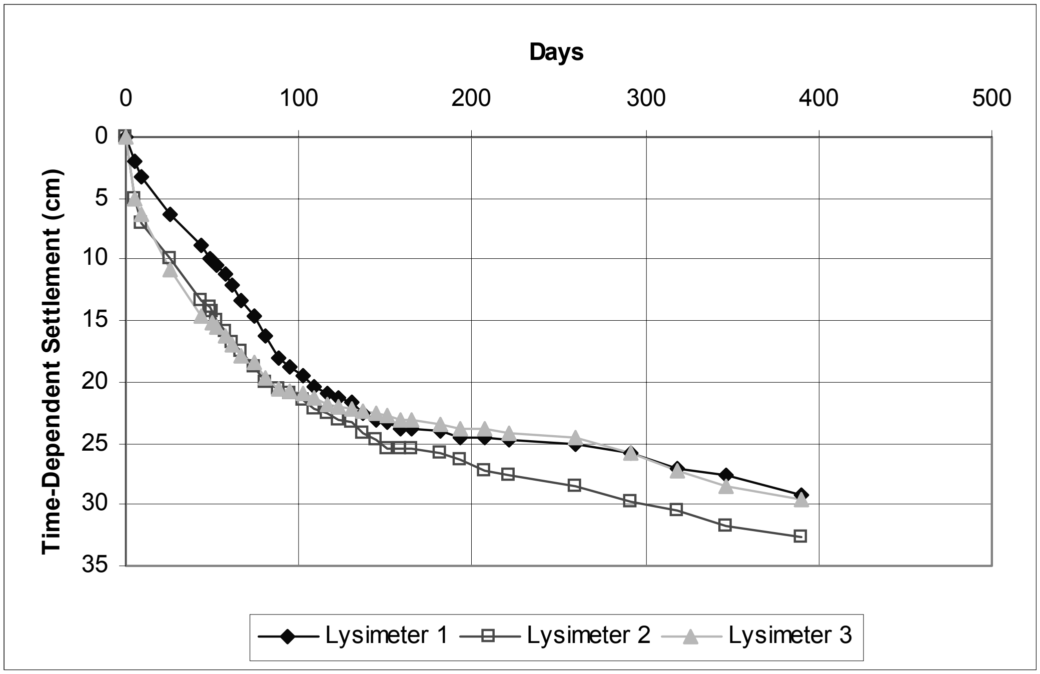

Figure 5.25 – Time-Dependent Settlement as a Function of Time

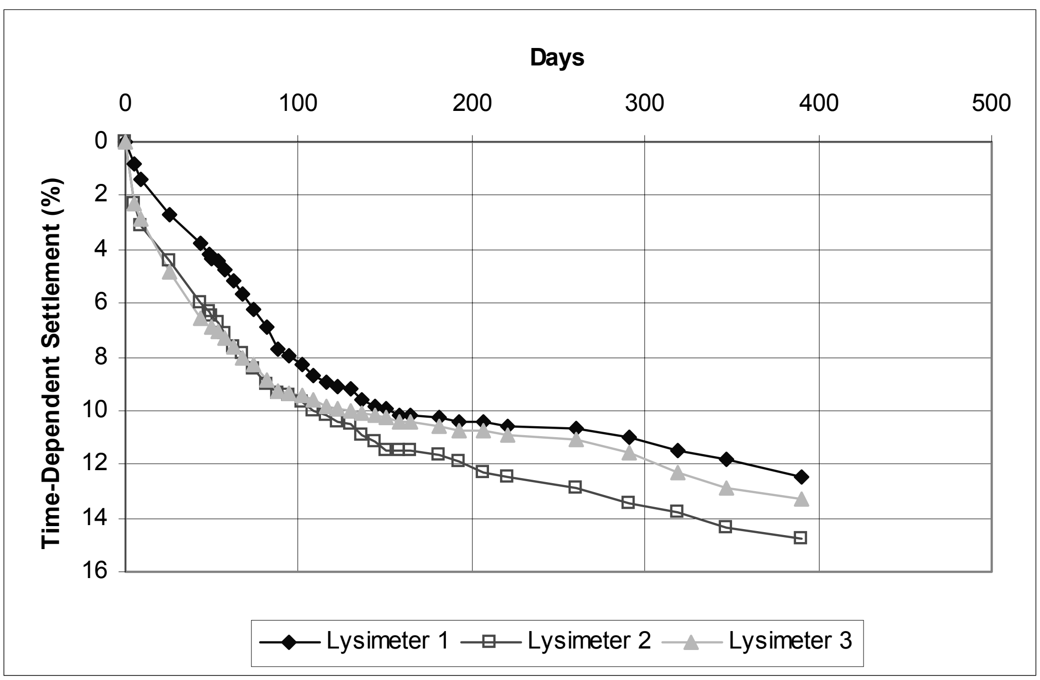

Figure 5.26 – Time-Dependent Relative Settlement as a Function of Time

Figures 5.27 and 5.28 show the same graphs presented in Figures 5.25 and 5.26, respectively, but in a log-time scale, as usual in classical Soil Mechanics.

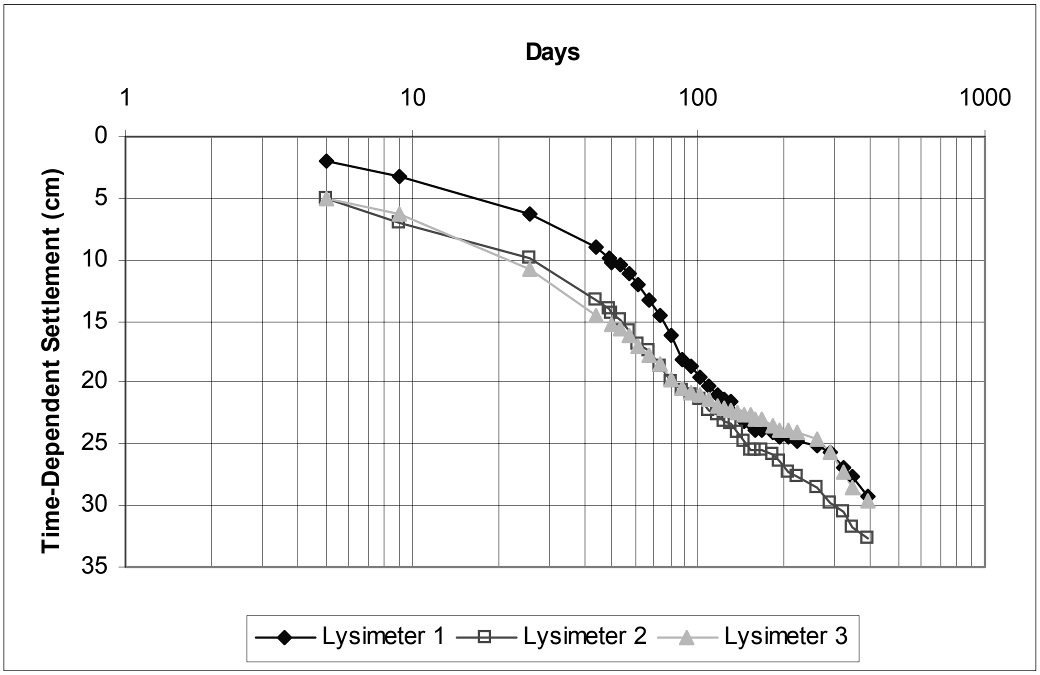

Figure 5.27 – Time-Dependent Settlement as a Function of Time (Log-Scale)

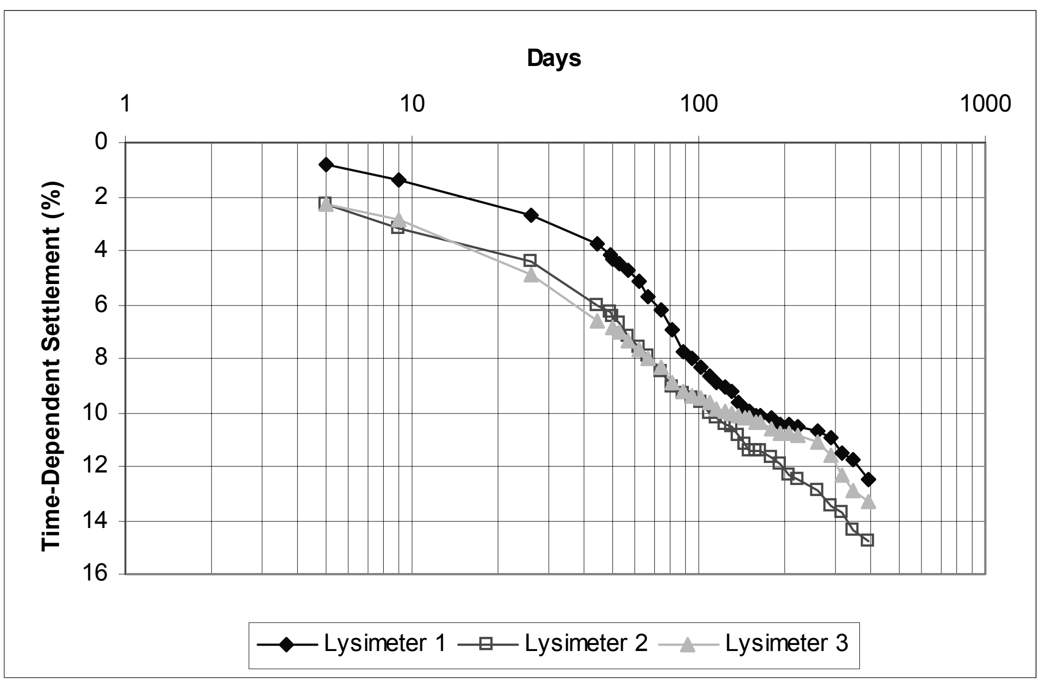

Figure 5.28 – Time-Dependent Relative Settlement as a Function of Time (Log-Scale)

Table 5.25 presents the settlement rates in the three lysimeters (as the difference of settlements between two measurements divided by the number of days between settlement measurements). Figure 5.29 presents the evolution of settlement rates in time. Figure 5.30 shows the same graph presented in Figure 5.29, but restricting the settlement rates in the 0 to 4 mm/day range.

Table 5.25 – Settlement Rates in the Lysimeters

Date	Days	Settlement Rate (mm/day)		
		Lysimeter 1	Lysimeter 2	Lysimeter 3
5/2/2002	0			
5/7/2002	5	3.8	10.2	10.2
5/11/2002	9	3.2	4.8	3.2
5/28/2002	26	1.9	1.7	2.6
6/15/2002	44	1.4	1.9	2.1
6/20/2002	49	1.9	1.3	
6/21/2002	50	3.2	3.2	1.1
6/24/2002	53	1.1	2.1	1.1
6/28/2002	57	1.6	2.4	1.6
7/3/2002	62	1.9	1.9	1.6
7/8/2002	67	2.5	1.3	1.6
7/15/2002	74	1.8	1.8	0.9
7/22/2002	81	2.3	1.8	1.8
7/29/2002	88	2.7	0.9	1.1
8/5/2002	95	0.9	0.5	0.5
8/12/2002	102	1.1	0.7	0.2
8/19/2002	109	1.1	1.1	0.5
8/26/2002	116	0.9	0.5	0.9
9/2/2002	123	0.5	0.9	0.2
9/9/2002	130	0.5	0.2	0.2
9/16/2002	137	1.4	1.1	0.2
9/23/2002	144	0.9	0.9	0.2
9/30/2002	151	0.2	0.9	0.2
10/7/2002	158	0.7	0.0	0.5
10/14/2002	165	0.0	0.0	0.0
10/30/2002	181	0.1	0.3	0.3
11/11/2002	193	0.4	0.4	0.3
11/25/2002	207	0.0	0.7	0.0
12/9/2002	221	0.2	0.2	0.2
1/17/2003	260	0.1	0.2	0.1
2/17/2003	291	0.2	0.4	0.4
3/17/2003	319	0.5	0.2	0.6
4/14/2003	347	0.2	0.5	0.5
5/27/2003	390	0.4	0.2	0.2

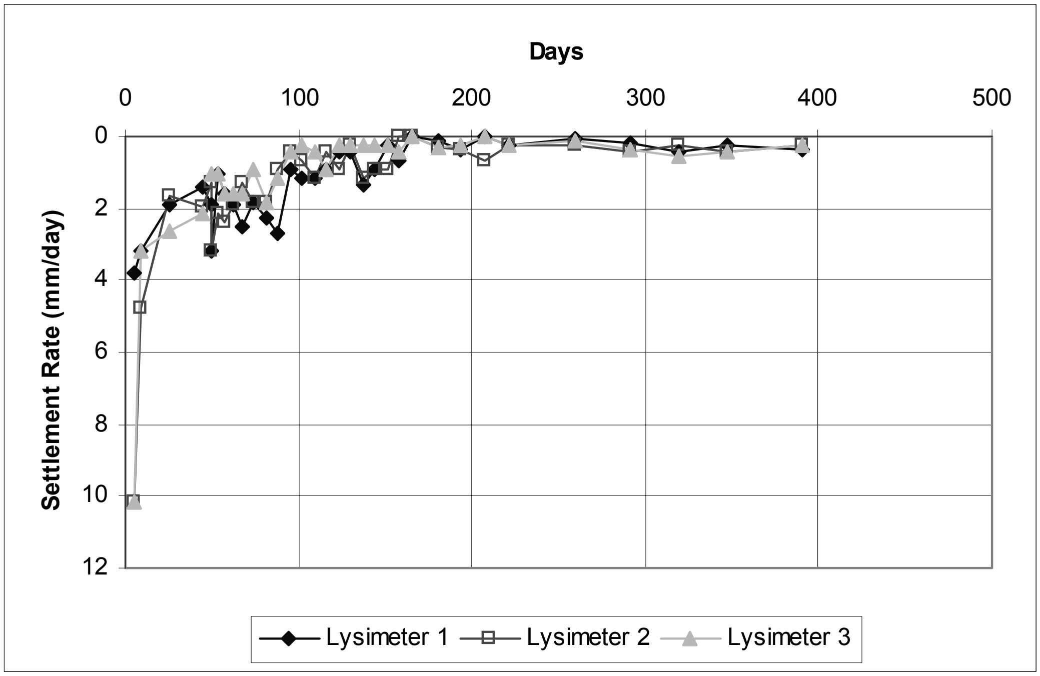

Figure 5.29 – Settlement Rates as a Function of Time

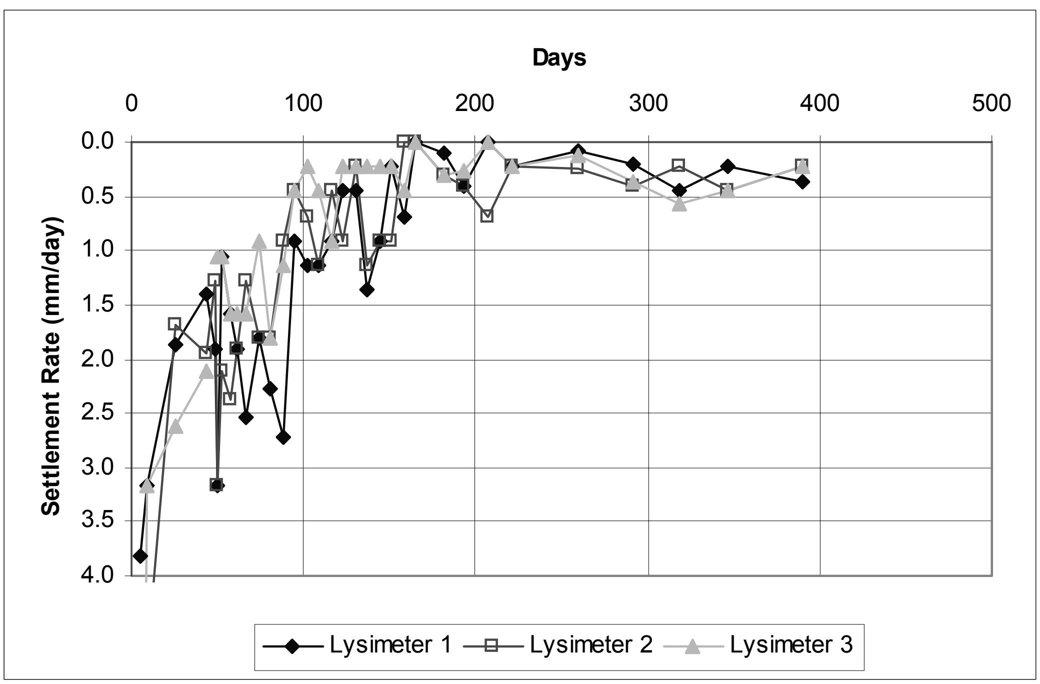

Figure 5.30 – Settlement Rates as a Function of Time (0 to 4 mm/day range)

5.2.2. Compressibility Tests

As mentioned in Section 4.3, two compressibility tests were done to evaluate the MSW immediate settlements under dry and wet conditions, in addition to several similar tests performed at the University of New Orleans in the past.

Table 5.26 presents the results of the test performed under dry conditions, while Table 5.27 presents the results for the wet test. The tables show the jack type used, the stress in the point located at the middle of the waste layer, the average settlement (average of the measurement in 3 points), the strain (or relative settlement, the settlement divided by the initial thickness of the waste layer), and the waste layer thickness, volume, and unit weight as the MSW is compressed.

Table 5.26 – Compressibility Test Results for Dry Waste

Jack Type	Stress (KPa)	Average Settlement (cm)	Strain (m/m)	Thickness (cm)	Volume (m^3)	Unit Weight (kN/m^3)
	1.3	0.0	0.000	73.9	0.189	0.78
2 x 10 ton	5.2	11.9	0.161	62.0	0.159	0.93
1 x 10 ton	12.5	25.3	0.342	48.6	0.125	1.18
1 x 10 ton	19.4	30.2	0.408	43.8	0.112	1.31
2 x 25 ton	25.7	33.2	0.450	40.7	0.104	1.41
2 x 25 ton	49.6	43.0	0.582	30.9	0.079	1.86
2 x 25 ton	97.5	50.4	0.681	23.6	0.060	2.44
2 x 25 ton	146.9	56.0	0.757	18.0	0.046	3.20
2 x 25 ton	349.9	59.6	0.806	14.4	0.037	4.00

Table 5.27 – Compressibility Test Results for Wet Waste

Jack Type	Stress (*) (KPa)	Average Settlement (cm)	Strain (m/m)	Thickness (cm)	Volume (m^3)
	1.3	0.0	0.000	75.1	0.192
2 x 25 ton	5.7	12.8	0.170	62.3	0.159
2 x 25 ton	10.3	21.5	0.286	53.6	0.137
2 x 25 ton	21.3	32.3	0.430	42.8	0.110
2 x 25 ton	25.3	34.6	0.461	40.5	0.104
2 x 25 ton	50.4	44.1	0.587	31.0	0.079
2 x 25 ton	95.5	51.3	0.683	23.8	0.061
2 x 25 ton	147.2	57.2	0.762	17.9	0.046
2 x 25 ton	367.7	60.3	0.803	14.8	0.038

The waste unit weight for the wet test could not be obtained since water was added to the chamber and then drained, therefore changing the initial unit weight. The quantities of water added and drained were not measured, not making possible a mass balance.

Figures 5.31 and 5.32 present the stress-strain graphs for the dry test in normal and log-scale, respectively.

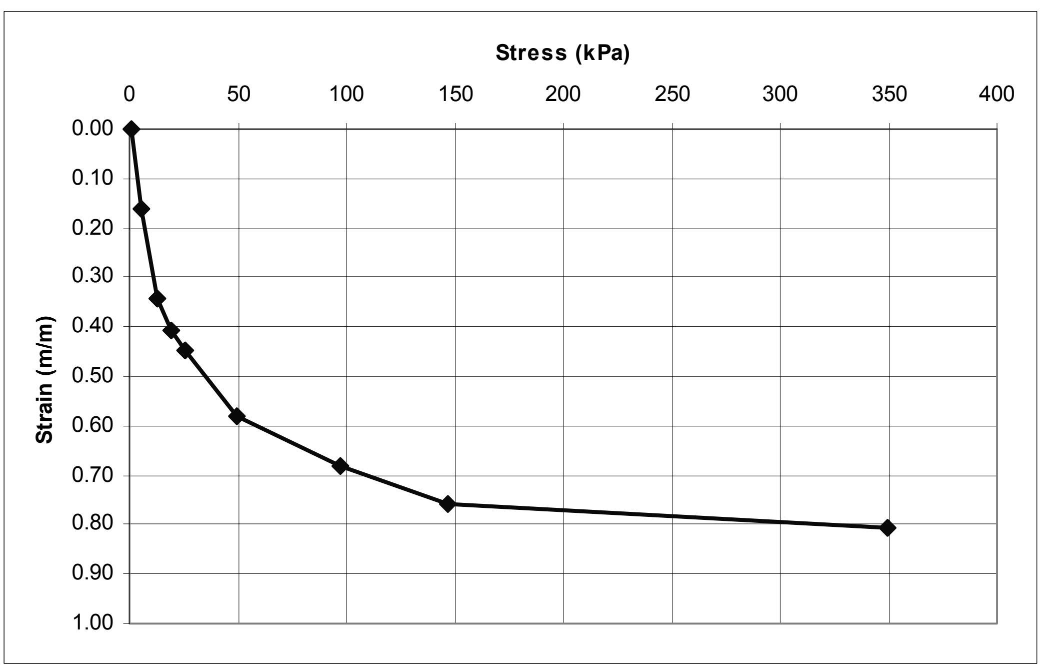

Figure 5.31 – Stress-Strain Relationship for Dry Test in Normal Scale

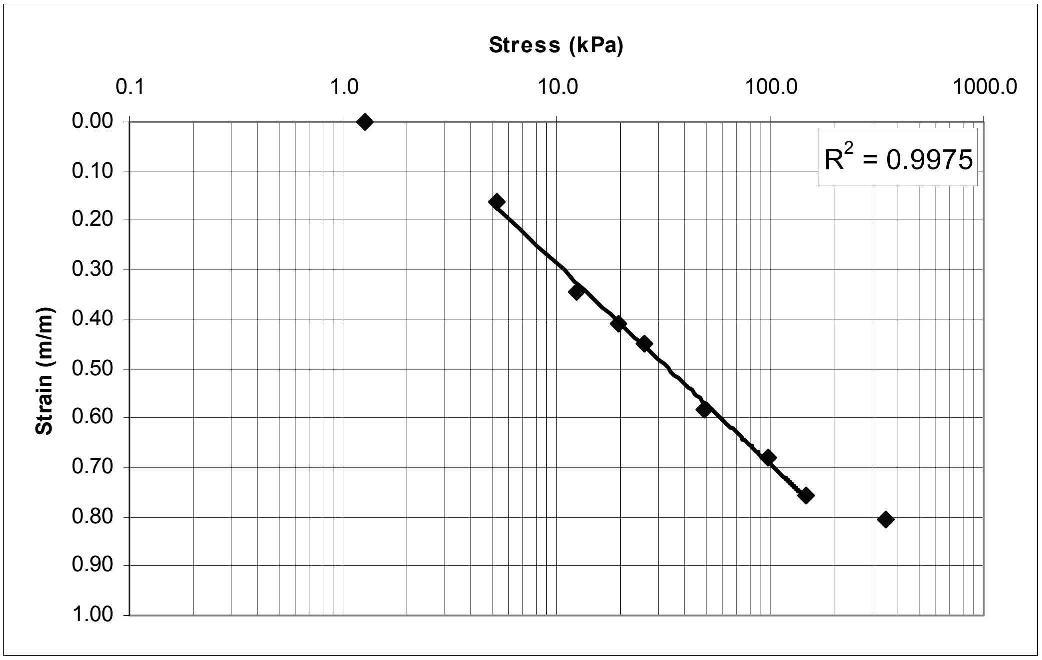

Figure 5.32 – Stress-Strain Relationship for Dry Test in Logarithmic Scale

The coefficient of primary compression C'_c was calculated for the dry test, based on the linear portion of the curve presented in Figure 5.32, which is equal to 0.41.

Figures 5.33 and 5.34 present the stress-strain graphs for the wet test in normal and log-scale, respectively.

The coefficient of primary compression C'_c was calculated for the wet test, based on the linear portion of the curve presented in Figure 5.34, which is equal to 0.41.

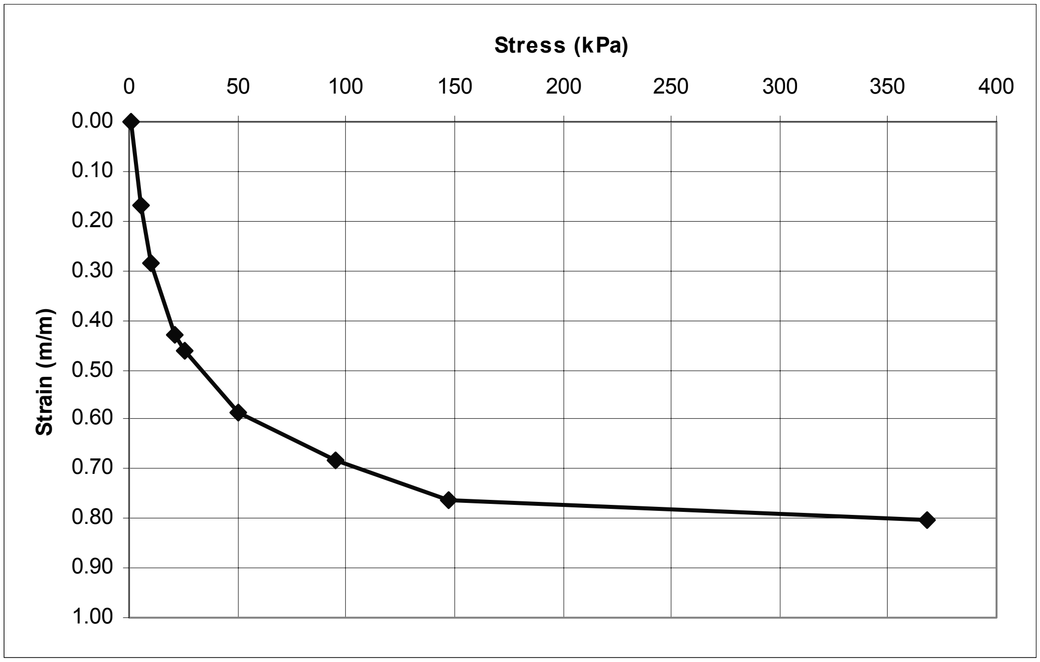

Figure 5.33 – Stress-Strain Relationship for Wet Test in Normal Scale

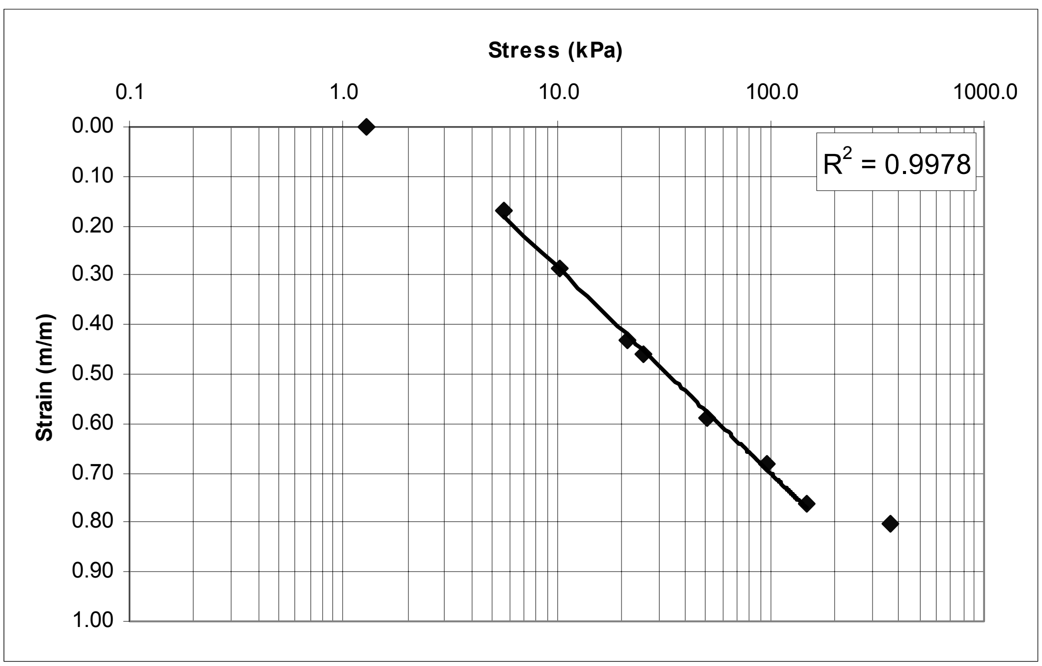

Figure 5.34 – Stress-Strain Relationship for Wet Test in Logarithmic Scale

As seen from Figures 5.31 to 5.34, the results from the dry and wet tests are very similar, with both tests resulting in the same coefficient of primary compression C'_c (equal to 0.41).

CHAPTER 6

DISCUSSION OF THE RESULTS

6.1. Environmental Study

6.1.1. Leachate Composition Analysis

This section analyzes the parameters measured on the leachate generated by the three lysimeters, individually, in comparison with other parameters, and in comparison with other studies. This section is divided into three parts: Organic Matter Stabilization, Nitrogen Management, and Other Parameters.

6.1.1.1. Organic Matter Stabilization

As stated before, the COD is defined as a measure of organic matter in terms of the oxygen required for the chemical oxidation of these materials. From Table 5.1 and Figure 5.1, it was noticed that total COD concentrations fluctuate typically between 1,000 and 2,400 mg/L during the first 50 days, and then dropped quickly to a concentration of about 500 mg/L for Lysimeters 2 (straight recirculation) and 3 (dry lysimeter), and to 100 mg/L for Lysimeter 1 (facultative lysimeter) around day 75. After that day, COD concentrations decreased very slowly for the three lysimeters, finishing at the end of the experiment at a 50 mg/L level for Lysimeter 1, at a 400 mg/L level for Lysimeter 2, and at a 440 mg/L level for Lysimeter 3.

Figure 6.1 presents the ratio of total COD to maximum COD for the three lysimeters as a function of time.

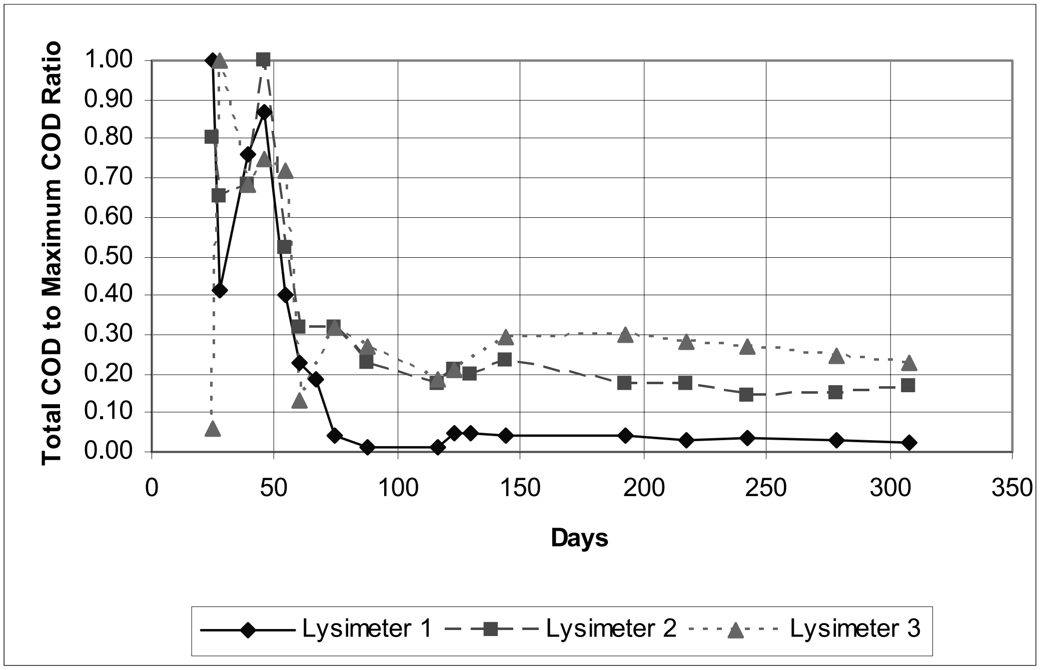

Figure 6.1 - Ratio of Total COD to Maximum COD as a Function of Time

Figure 6.1 shows that the facultative lysimeter (Lysimeter 1) was the most effective in reducing the total COD in time, with a reduction of 98%. This was followed by the lysimeter with straight recirculation (Lysimeter 2), with a reduction of approximately 85%, and the conventional lysimeter (Lysimeter 3), with a reduction of approximately 75%.

Filtered COD was measured and is compared to the total COD for all lysimeters in Figure 6.2, using the ratio of filtered COD to total COD. From this figure, it can be noticed that the amount of "dissolved" organic matter in total organic matter is much lower for Lysimeter 1 when compared to the other lysimeters. Lysimeter 2 presented the second best performance, with the ratio decreasing quickly after day 211, the day when the leachate recirculation rate was increased. Lysimeter 3 presented a stable ratio of 0.80 to 0.85 with a very slight increase during the period measured. Lysimeter 1 also presented stabilization of the ratio, but around a much lower value (0.20, approximately).

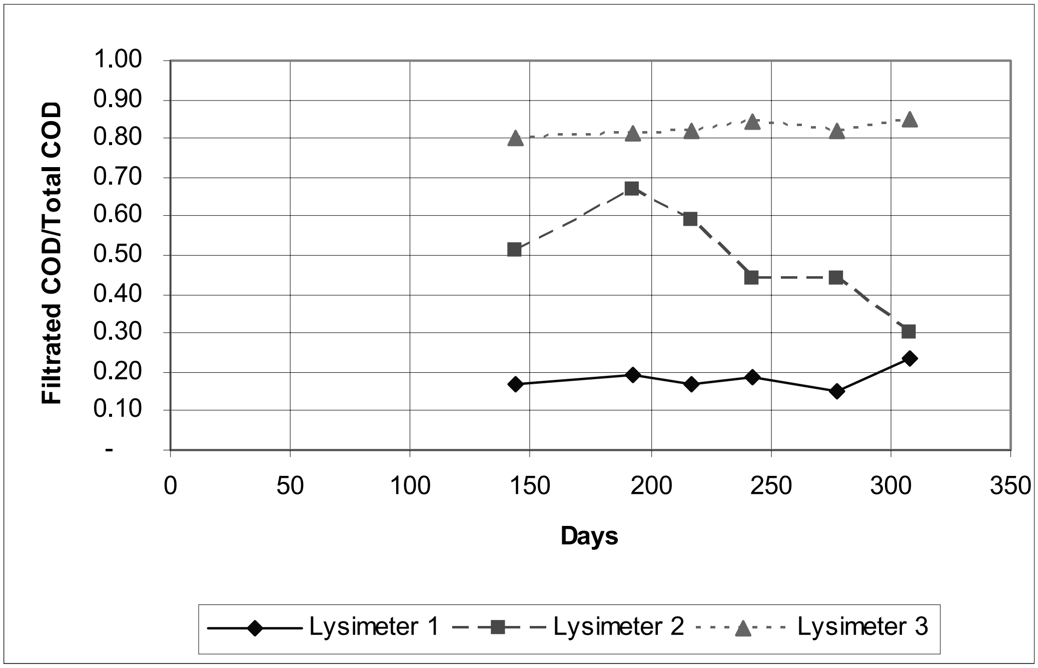

Figure 6.2 - Ratio of Filtered COD to Total COD as a Function of Time

Another way to measure the stabilization of the organic matter is to analyze the BOD-to-COD ratio. The BOD-to-COD ratio can be used as an indicator of waste stabilization, and the lower the value, the more stable the waste (Reinhart and Townsend, 1998), since BOD represents mainly the organic matter that can be decomposed biologically. Figure 6.3 presents the BOD-to-COD ratio as a function of time.

As seen in Figure 6.3, the BOD-to-COD ratio decreased at constant rates for the three lysimeters. However, the values presented on the Facultative Lysimeter is much lower than the other two (approximately 0.2 versus 0.6). This shows another indication that the facultative lysimeter processed and stabilized the organic matter in the waste much faster.

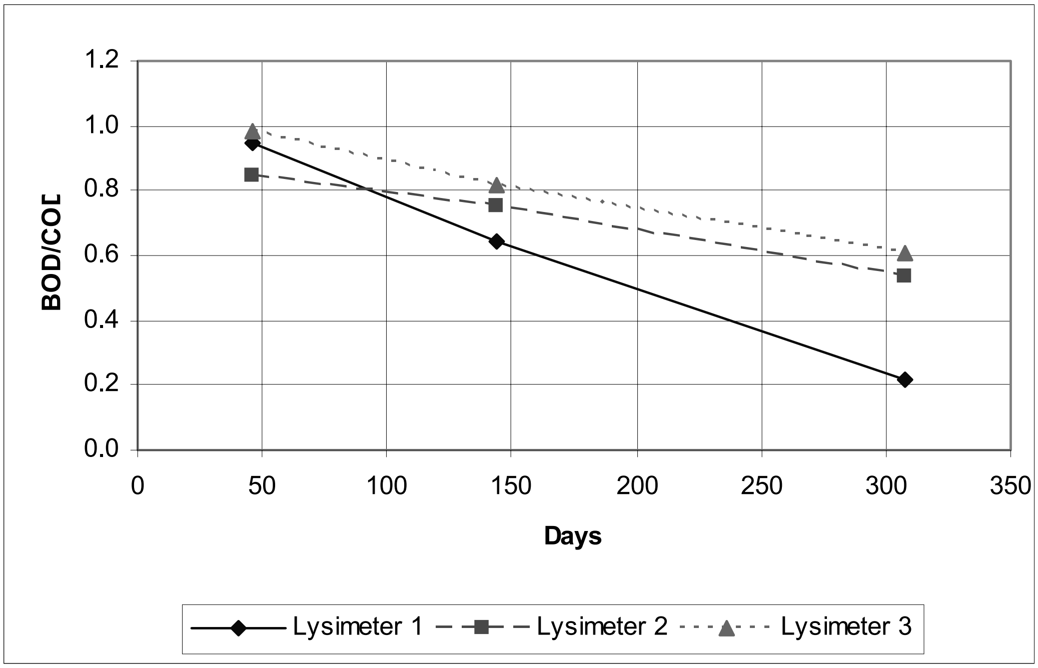

Figure 6.3 – BOD-to-COD Ratio as a Function of Time

However, another issue must be addressed: COD is removed not only by biological conversion, but also via washout. It is clear that in recirculating landfills there is minimum washout since the leachate is reintroduced to the landfill. But in single pass landfills, washout can play an important role in reducing COD, even more if the rate of liquids infiltrating the landfill is high (the case of the facultative lysimeter). One way to confirm if the COD is being removed by biological conversion or washout is to analyze the COD-to-chloride ratio. Chloride is a very stable compound, and mainly removed by washout, so a decrease of COD-to-chloride ratio in time means that COD is being removed not only by washout, but also by biological activity.

Figure 6.4 presents the COD-to-chloride ratio as a function of time.

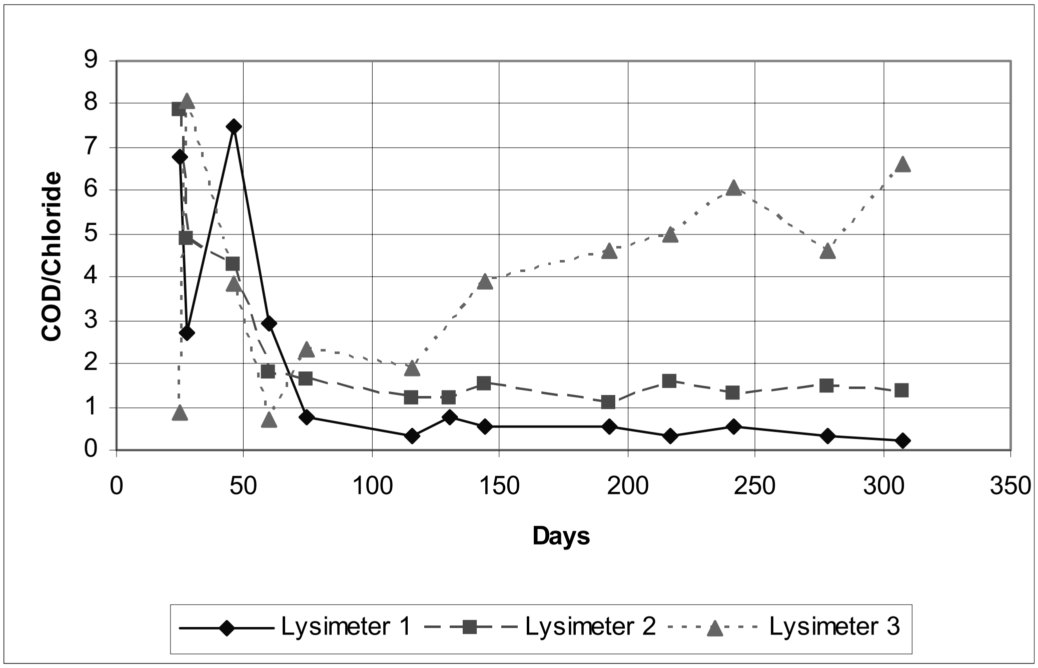

Figure 6.4 – COD-to-Chloride Ratio as a Function of Time

From Figure 6.4, it can be noticed that the COD-to-chloride ratio decreased very quickly and to lower values for Lysimeters 1 and 2, indicating that biological activity plays an important role in removing COD in those lysimeters (note that no washout is expected in Lysimeter 2). In Lysimeter 3, the ratio first decreased and then increased again, showing that biological activity removed the COD in the first days, but after that, slow decomposition took place and the biological activity in this lysimeter is minimum.

In general, TSS and VSS concentrations stayed at low values during the period of study. Although VSS could be considered, initially, as a parameter to verify waste stabilization regarding organic matter content, Reinhart and Townsend (1998) point out that low volatile solids content is a misleading parameter with respect to waste stability and, therefore, cannot be used to analyze organic matter stabilization in MSW. However, it is interesting to note the high values of TSS and VSS (2,037 mg/L and 1,090 mg/L) for Lysimeter 2 on day 46. There is no explanation for those high values other than the heterogeneity of the leachate during the first initial days. It must be noted that the highest value of COD was also obtained on this same day for the same sample. Laboratory analyses for TSS and VSS were remade at the same day, using the same sample, and the values were confirmed.

Comparison to Other Studies

The values of COD and BOD obtained for the leachate of the three lysimeters are typically lower than the values presented in the general literature (see Tables 2.2 to 2.7, Chapter 2). However, COD and BOD values are within the range presented by Reinhart and Townsend (1998) (Table 2.6 for conventional landfills and landfills with leachate recirculation). It must be noted, that during the initial procedures for bringing the waste to field capacity, the lysimeters were filled with water to the top and then drained, so certainly part of the COD and BOD could have been removed via washout. Of course, this type of operation does not happen in full-scale landfills where most of the published data were obtained.

Another cause for the low COD and BOD values could be the wire mesh (stainless steel, 0.8 mm opening) located at the bottom parts of the lysimeters, that “filter” the leachate, lowering COD and BOD values, as well as TSS and VSS values.

TSS values obtained for the lysimeters are typically within the minimum values in the range presented in the general literature or lower, according to Tables 2.2 to 2.4.

Chloride values obtained for all lysimeters in this study are located within the range presented in the general literature, according to Tchobanoglous et al. (1993) and Reinhart and Townsend (1998), and below the range presented by

Qasim and Chiang (1994). Again, this could be due to washout during the initial procedures for bringing the waste to field capacity.

6.1.1.2. Nitrogen Management

As seen in Figure 5.5, ammonia-nitrogen concentrations were significantly lower for Lysimeter 1 leachate, but were constant throughout the study for the other two lysimeters. Figure 6.5 presents the variation of the ratio ammonia-nitrogen concentration to initial concentration for the three lysimeters as a function of time.

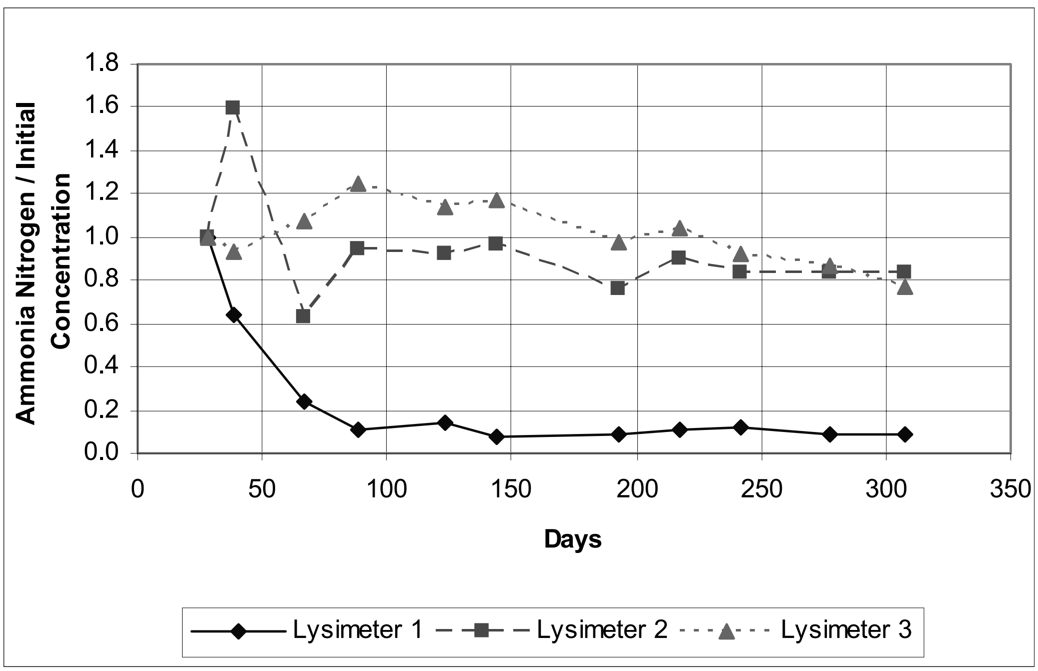

Figure 6.5 – Variation of the Ratio of Ammonia-Nitrogen Concentration to Initial Concentration as a Function of Time

From Figure 6.5 it can be seen that the reduction in ammonia-nitrogen concentrations was approximately 90% for Lysimeter 1 at the end of the period of study, while the reduction for the other two lysimeters was approximately only 20%.

Nitrate-nitrogen concentrations stayed low throughout the period of study (Table 5.6). Maximum values reached 1.2 mg/L in Lysimeter 1, while the concentrations of the treated effluent sent to the same lysimeter reached 24 mg/L. Figure 6.6 presents the ratio between the concentration of nitrate-nitrogen in the leachate and the concentration of the same component in the effluent sent to Lysimeter 1, as a function of time.

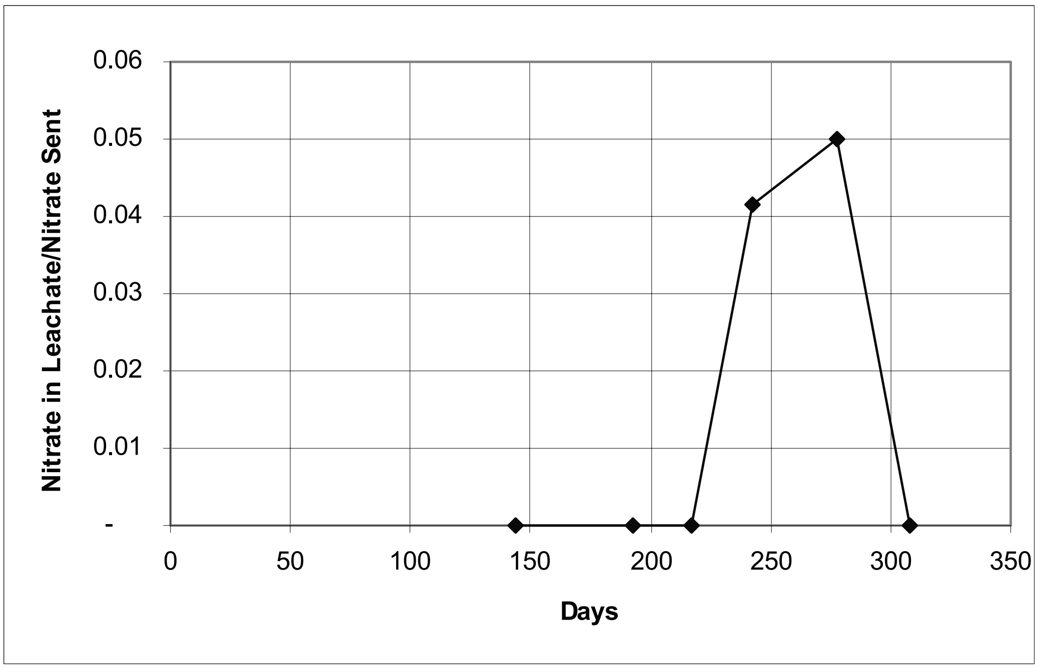

Figure 6.6 – Variation of the Ratio between the Concentrations of Nitrate-Nitrogen in the Leachate and in the Effluent Sent to Lysimeter 1 as a Function of Time

From Figure 6.6, a reduction of 95% in the concentration of nitrates injected in the lysimeter can be noticed. The reduction in theory could be associated with one of the following factors: 1) conversion of the nitrates into nitrogen gas by denitrifying bacteria (denitrification), 2) conversion of the nitrates into ammonia (depending of the pE-pH conditions), 3) accumulation of nitrates inside the lysimeter, and 4) dilution due to rainfall simulation. Conversion of the nitrates back into ammonia can occur only at very low pHs, which is not the case. Also, ammonia concentrations had decreased in time. Accumulation inside the lysimeter can also be discarded since there is no mechanism to make the nitrates accumulate inside it. Although some dilution can occur due to the rainfall simulation, the quantities of effluent sent to the lysimeter are much larger than the quantity of water added to it in a proportion of approximately a minimum of 1:20 (at the beginning of the study), thus, not justifying this possibility. The only possible explanation for the reduction of nitrates is that they are being converted into nitrogen gas, due to denitrification, using the organic matter inside the lysimeter. The denitrification process will be further discussed in the gas analysis section (Section 6.1.3).

Regarding TKN (Table 5.8 and Figure 5.7) there was a significant reduction in the concentrations for Lysimeter 1. The TKN of the other two lysimeters decreased and then experienced a substantial increase over time (after day 67). Figure 6.7 presents the ratio of ammonia-nitrogen to TKN for the lysimeters as a function of time.

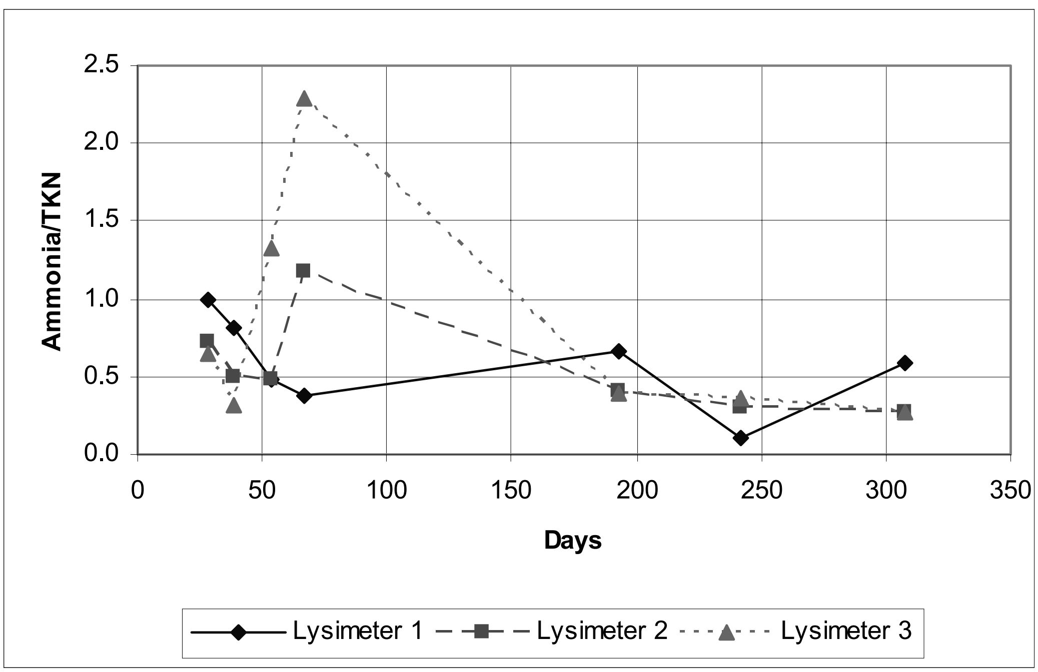

Figure 6.7 - Ammonia-Nitrogen to TKN Ratio as a Function of Time

Since the TKN concentrations are in theory the sum of ammonia nitrogen concentrations and organic nitrogen concentrations, the ratio should never be greater than 1. Values greater than 1 were noticed in two occasions for Lysimeter 3, and in one occasion for Lysimeter 2 around day 60. There is no explanation for these values, other than the heterogeneity of leachate during the first days and despite all the rigorous procedures for sampling. Laboratory analyses for TKN were remade for these three cases using the same sample, and the values confirmed.

Disregarding those values, the ammonia-nitrogen to TKN ratio stayed typically between 0.3 and 0.6 throughout the study. Therefore, organic nitrogen represented about 40% to 70% of the TKN values.

Comparison to Other Studies

The values of ammonia-nitrogen and TKN obtained for the leachate of the three lysimeters are typically within the range of values presented in the general literature, with Lysimeter 1 producing the lowest values in the range. Nitrate concentrations for the three lysimeters were below or in the low range presented by the literature.

6.1.1.3. Other Parameters

Values of pH obtained for the three lysimeters were in the higher range presented in the general literature. With the exception of one occasion, all values of pH are greater than 7, which indicates that the first three phases (aerobic, transition, and acid anaerobic) finished very early in the study for all lysimeters. This can also be noticed by analyzing the TVA concentrations, which had a significant decrease after approximately 50 days for the three lysimeters, indicating the end of the acid anaerobic phase. The TVA concentrations obtained for the leachates of the three lysimeters were situated at the low range of the general literature, typical of methanogenic anaerobic or post-methanogenic phases.

The metals, iron and aluminum, were also produced in low concentration values. Iron concentrations peaked very early in the study (around days 46 to 60) and then decreased to low values, staying with low values during the remaining time of the study. Aluminum concentrations were typically lower than 0.2 mg/L throughout the study for all lysimeters. The probable reason for those low values is that metals are typically mobile at low pH values, which was not observed during the period of study.

Total phosphorus stayed below or at the low range presented in the literature typical of mature landfills, with the lowest values presented in Lysimeter 3. This also indicates that the three lysimeters reached an advanced phase very early in the study.

6.1.2. Leachate Production Analysis

The leachate production and initial moisture content for the wastes in the lysimeters are examined in this section.

The moisture contents at field capacity for the wastes were measured when the lysimeters were filled with water, according to the procedure explained in Section 4.2.3. The results are presented in Table 5.16 of Section 5.1.2.

The moisture content at field capacity varied between 17% (Lysimeter 2) and 36% (Lysimeter 1) on a wet basis. Although it was expected in a first analysis that the values should be the same, since waste types and composition and start-up processes were the same for the three lysimeters, the moisture content in Lysimeter 1 was more than double that of Lysimeter 2. It must be noted, however, that the ammounts of water added to saturate the three lysimeters were almost the same (409 L to 422 L), a difference of approximately 3%. The same applies to the volume of liquids drained (359 L to 404 L), a difference of about 12%.

The difference in the moisture content at field capacity between the three lysimeters can be explained due to the relatively low initial unit weight of the waste inside the lysimeters, which can create “pockets” inside the waste mass where water can accumulate in different ways, depending on the arrangement of waste components inside the lysimeters. The largest difference of water retained in the three lysimeters was equal to 32 L, or less than 6% of the volume of waste.

If the moisture contents were measured on a volumetric basis (saturation), instead of gravimetric, the ratio of volume of liquids retained to total volume (volume of waste) is presented in Table 6.1.

Table 6.1 – Moisture Contents at Field Capacity on a Volumetric Basis

Lysimeter	Volume of Water Retained (L)	Waste Volume (L)	Saturation (%)
1	50.0	581.0	8.6
2	18.3	580.0	3.2
3	31.7	595.0	5.3

Of course, the difference in saturation in relative values is still high, but the conclusion is that there is not much difference among the three lysimeters when analyzing the saturation in terms of absolute values.

Regarding leachate production, Lysimeter 1 produced leachate a few drops a little earlier (11 days before) than the other two lysimeters, which can be attributed to the volume of liquids initially retained in this lysimeter in comparison to the other two. Leachate was generated 2 weeks after water was added on a regular basis to this lysimeter.

After recirculation started, Lysimeters 1 and 2 produced much more leachate than Lysimeter 3, which is logical since leachate production is directly related to the quantity of liquids that enter the landfill. Since the quantities of liquids added to Lysimeters 1 and 2 are much greater than Lysimeter 3, operated as a dry landfill, the first two lysimeters should generate much more leachate, as observed. Figure 6.8 presents the comparison of leachate production among the three lysimeters in time.

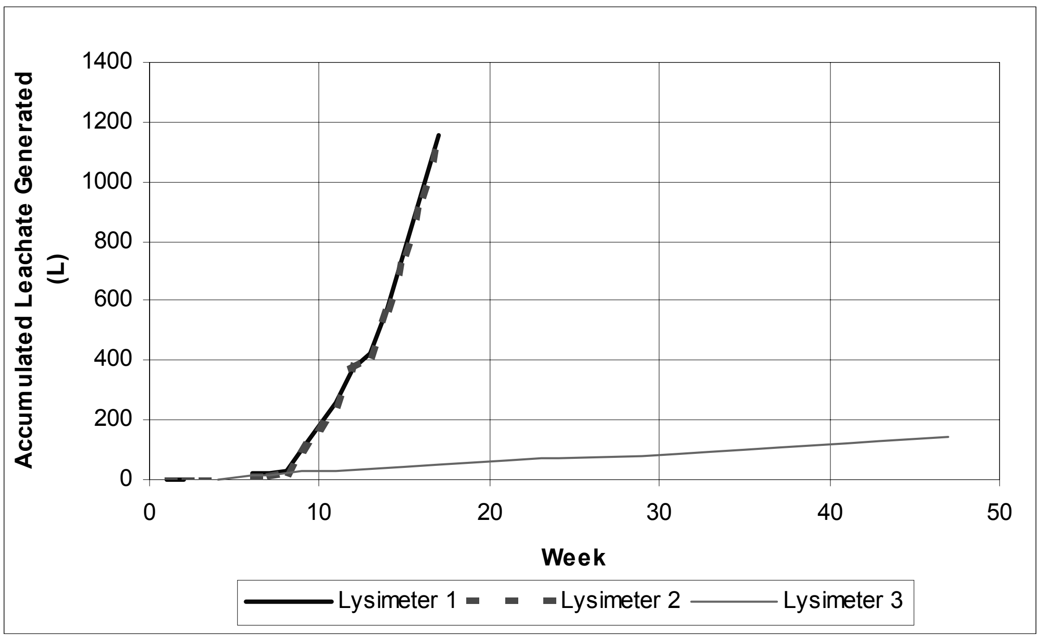

Figure 6.8 – Comparison of Accumulated Leachate Generation as a Function of Time among the Three Lysimeters

Figures 6.9, 6.10, and 6.11 present the leachate generation as a function of liquids added for Lysimeters 1, 2, and 3, respectively. From these figures it can be noticed that liquids added and leachate generation had almost the same values.

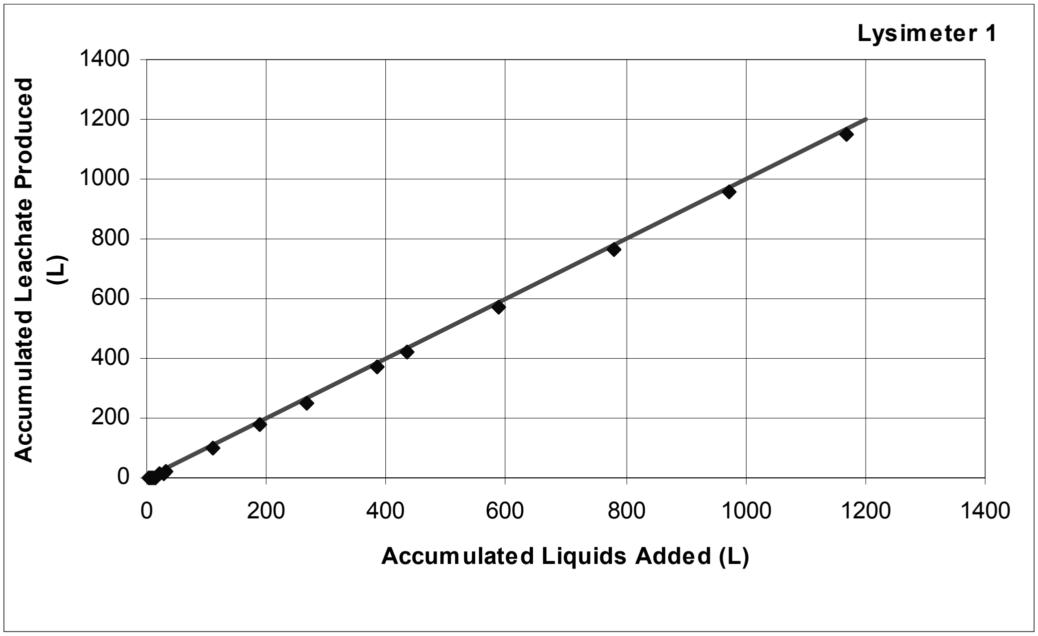

Figure 6.9 – Accumulated Leachate Production as a Function of Accumulated Liquids Added to Lysimeter 1

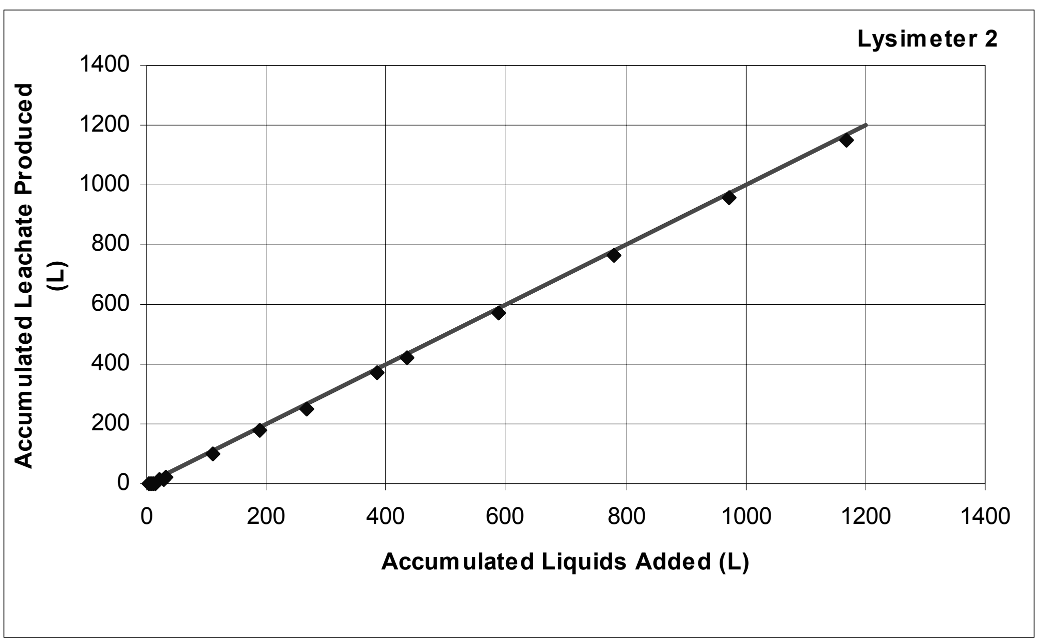

Figure 6.10 – Accumulated Leachate Production as a Function of Accumulated Liquids Added to Lysimeter 2

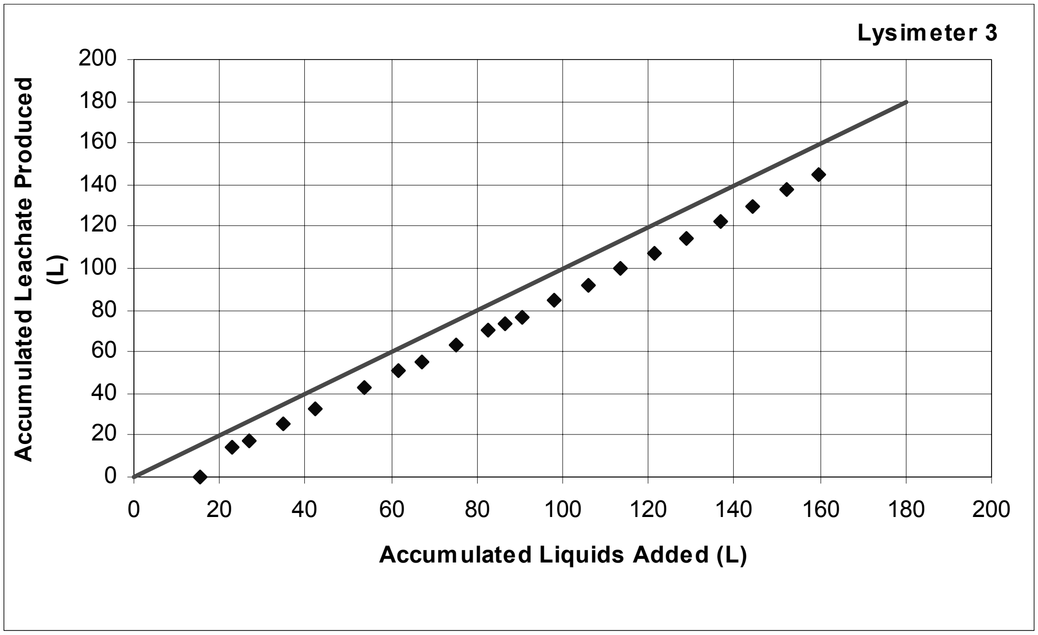

Figure 6.11 – Accumulated Leachate Production as a Function of Accumulated Liquids Added to Lysimeter 3

A comparison of leachate generation versus liquids added can also be made by analyzing the ratio of leachate production to liquids added with time. Figures 6.12, 6.13, and 6.14 present these ratios for Lysimeters 1, 2, and 3, respectively. From these figures and from Tables 5.17 to 5.19, it can be inferred that the ratio of leachate produced to liquids added increases as more liquids are added, with the ratio tending to be 1 in the long term. However, this conclusion is specific to the lysimeters studied since in real landfills there are other mechanisms involved.

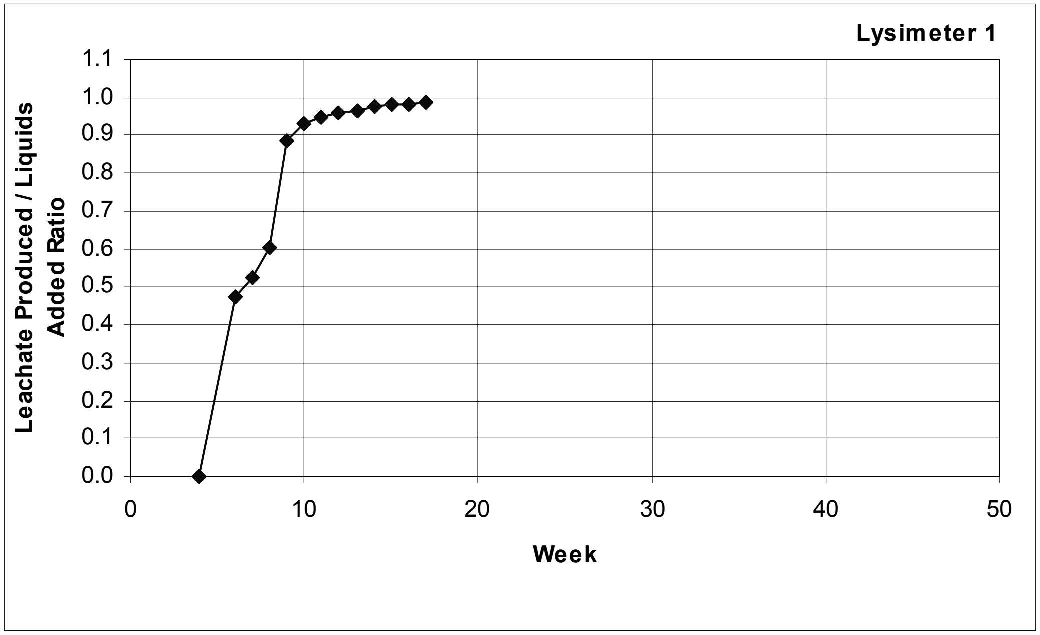

Figure 6.12 - Ratio of Leachate Production to Liquids Added as a Function of Time for Lysimeter 1

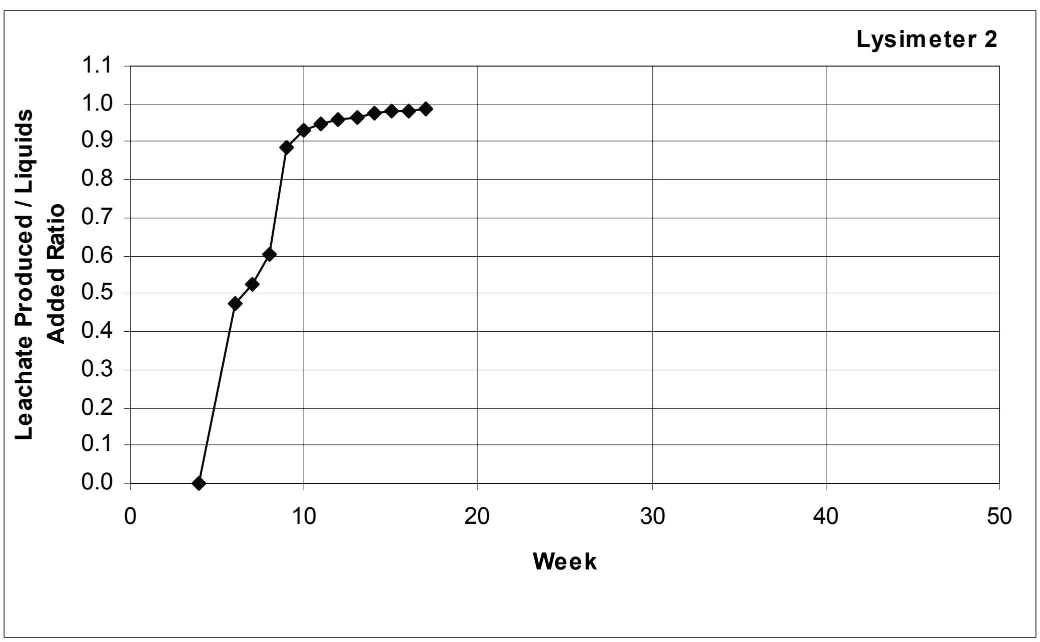

Figure 6.13 - Ratio of Leachate Production to Liquids Added as a Function of Time for Lysimeter 2

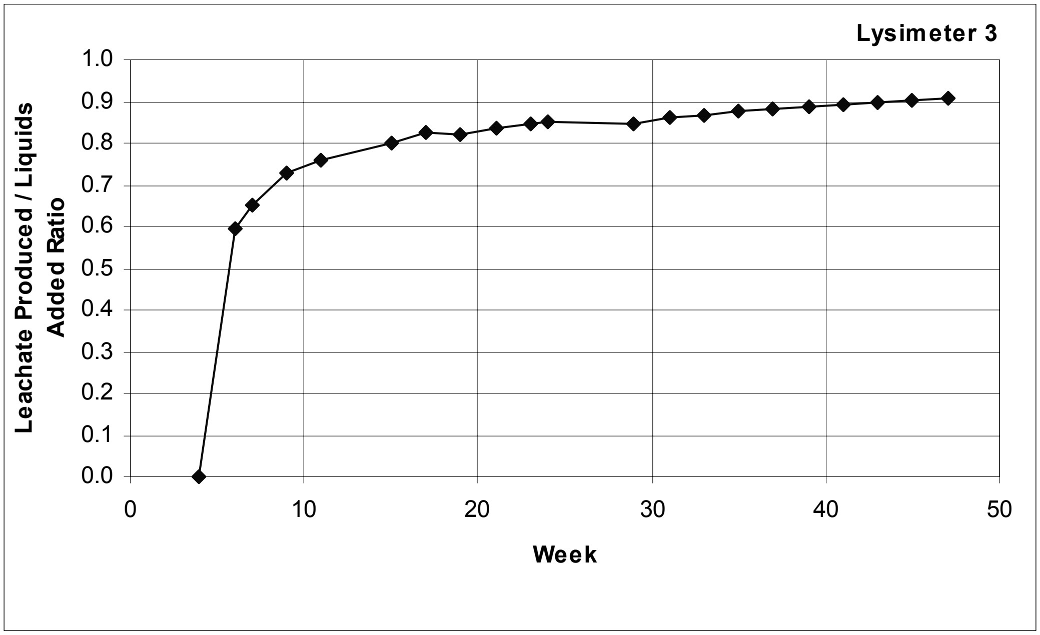

Figure 6.14 - Ratio of Leachate Production to Liquids Added as a Function of Time for Lysimeter 3

6.1.3. Gas Composition Analysis

Table 5.20 and Figures 5.20, 5.21, and 5.22 in Section 5.1.3 presented the gas composition inside the lysimeters in terms of methane, carbon dioxide, oxygen, and other gases. In this table and these figures, a substantial difference can be noticed in Lysimeter 1 from the other two lysimeters.

The fractions of methane and carbon dioxide generated when compared to other gases are much greater in Lysimeters 2 and 3 than in Lysimeter 1, which proves that methanogenesis was established in these two lysimeters. The fraction of methane generated was initially almost the same and around 20% to 30% in Lysimeters 2 and 3. It increased to 40% at the end of the period of study for Lysimeter 2 and remained at the same levels for Lysimeter 3, which indicates that raw leachate recirculation is beneficial to generate better quality gas to energy recovery, in accordance with other studies presented in Chapter 2. The lowest values of methane fraction for the two lysimeters was obtained in January

2003, after a period of almost one month when liquids were not added to the lysimeters, showing the correspondence between the methane fraction and moisture inside the landfill.

The fraction of carbon dioxide generated ranged between 40% and 70% for Lysimeter 2 and between 50% and 80% for Lysimeter 3, with the lowest values at the beginning of the measurement period. The fraction of oxygen ranged between 0% and 4% for Lysimeters 2 and 3, typical of anaerobic environments. Other gases typically ranged between 0 and 35% for Lysimeter 2 and between 0 and 25% for Lysimeter 3, with maximum values at the beginning of the measurement period.

Lysimeter 1 had its own pattern for gas composition, with behavior very different from the other two lysimeters. The measured methane fraction was very low during the measurement period. It was typically 0% to 3%, except during the period in December 2002 and January 2003, when liquids were not added to the lysimeters. These low values, in addition to the low values of carbon dioxide observed, indicate that this lysimeter was operating in non-methanogenic conditions. However, immediately after the end of the period that liquids were not added to the lysimeters, methane fraction peaked to 23% and carbon dioxide to 23%, which is an indication that methanogenesis had occurred inside that lysimeter. Once treated leachate injection was resumed, methane and carbon dioxide fractions decreased very quickly.

Measured oxygen fractions on Lysimeter 1 were greater than the other two lysimeters, and ranged between 0% and 11%. The null value was measured in the same period of time that methane and carbon dioxide peaked, typical of a methanogenic, anaerobic environment.

Other gases fraction had the greatest values in Lysimeter 1, ranging from 53% to 89%, with the lowest values obtained during the period that liquids were not added to the lysimeters. Values at other periods were typically above 80%. The only explanation for those high values, given the conditions that the lysimeter was subjected to, is that denitrification is the main process occurring in the lysimeter. The conversion of nitrate into nitrogen gas is also supported by the leachate analysis presented in Section 6.1.1. The achievement of methanogenic conditions in the lysimeter during a certain period of time demonstrates that the lysimeter is capable of having microbiological activity under anaerobic or aerobic conditions, not operating under “sterile” conditions. This process of denitrification-methanogenesis-denitrification was also verified by Barlaz (2002), as presented in Chapter 2.

A tentative procedure for achieving denitrification and methanogenesis at the same time was attempted at the end of the study by reducing the rate of treated effluent applied to Lysimeter 1. An increase in the methane fraction was noticed to a level not previously observed in the periods when treated effluent was added to the lysimeter, therefore verifying that both conditions can co-exist in the lysimeter.

Another interesting conclusion is that changing the conditions inside the lysimeters could be very quickly achieved, just in a matter of days, which was another important verification in this study.

6.2. Geotechnical Study

6.2.1. Immediate Settlement

Immediate settlement of MSW can be evaluated based on the compressibility tests described in Section 4.3. The coefficient of primary compression C'_c calculated for the dry and wet conditions is equal to 0.41 for both tests performed, according to the results presented in Section 5.2.2. This value is located at the top of the range reported in the general literature for MSW, as presented in Table 3.1. It is believed that in landfill systems, which are basically composed of waste plus soil cover plus drainage materials, the coefficient of primary compression is somewhat lower due to the presence of the more resilient materials such as soils and drainage materials (Debnath, 2000). Another reason for the larger C'_c is the relative low initial unit weight of the MSW tested. Intuitively, the lower the unit weight of a given material, the "softer" it is and, therefore, more compressible. However, this contradicts the study presented by Rao et al. (1977), which presents that the coefficient of primary compression increases with the increase of the initial unit weight.

Another possible factor that led to higher values for the coefficient of primary compression is that the test was performed by applying successive and immediate increments of load, without the possibility of development of "secondary compression" or compression under constant load.

The performed compressibility tests suggest that, probably, for low initial unit weights, moisture content does not affect the waste compressibility since the dry and wet tests led to the same coefficient of primary compression with very similar strain-stress path.

From the application of the load to the waste inside the lysimeters, relative settlements (or strains) of 15.2%, 10.3%, and 8.0% for Lysimeters 1, 2, and 3, respectively, occurred (Table 5.22). For the load applied (255 kg), the respective coefficients of primary compression are 0.18, 0.12, and 0.10. These values are located in the low to mid range of values presented in the general literature.

Although these values, at first analysis, may appear very different from the ones obtained with the compressibility tests, it must be noted that the level of stress at a point located in the middle of the waste layer is low (about 10 kPa). If only the first two points of the dry compressibility test performed are considered (see Figure 5.32), where the levels of stress are low, the local coefficient of primary compression at that level is 0.27, instead of 0.41. It is also necessary to consider that the initial unit weights of the two wastes are somewhat different. Also, some friction between the waste and the lysimeter wall could exist since the ratio of the length-diameter for the waste inside the lysimeter is about 4 to 1, further reducing the coefficient of primary compression of the waste inside the lysimeters. This friction can also be responsible for the difference in the coefficient of primary compression among the lysimeters. The same value was expected since the waste types and compositions, stress levels, and initial unit weight are approximately the same.

As presented in Chapter 2, according to Rao et al. (1977), at high values of stress, the strain versus log-stress curve is non-linear. This was also noticed by Debnath (2000), and is quite logical since at some point the curve must become inflected, otherwise the strain could be greater than 100%, which is nonsense. The same observation applies at very low values of stress, leading to a conclusion that the coefficient of primary compression is non-constant and dependent of the stress applied. This was also observed in both tests performed in this study. At low and high values of stress, the curve strain versus log-stress is non-linear.

The use of a coefficient that is not constant for modeling the compressibility of MSW at values greater than 200 kPa, a stress level reached in landfills with a depth greater than approximately 20 m (if Kavazanjian's Relationship is applied (Boutwell, 2002)), appears not to be reasonable, or, at least, not practical.

In a first approach, the data from the dry and wet tests were modeled using a hyperbolic function. When the data were plotted as a curve of stress divided by strain versus the stress, a strong linear relationship for the data was noted. This was not a surprise since the two variables plotted (x and y axes) are not independent, but help in the study of the significance of the parameters of the curve, as will be shown further. Figures 6.15 and 6.16 present the graphs of stress-to-strain ratio as a function of stress for the dry and wet tests, respectively.

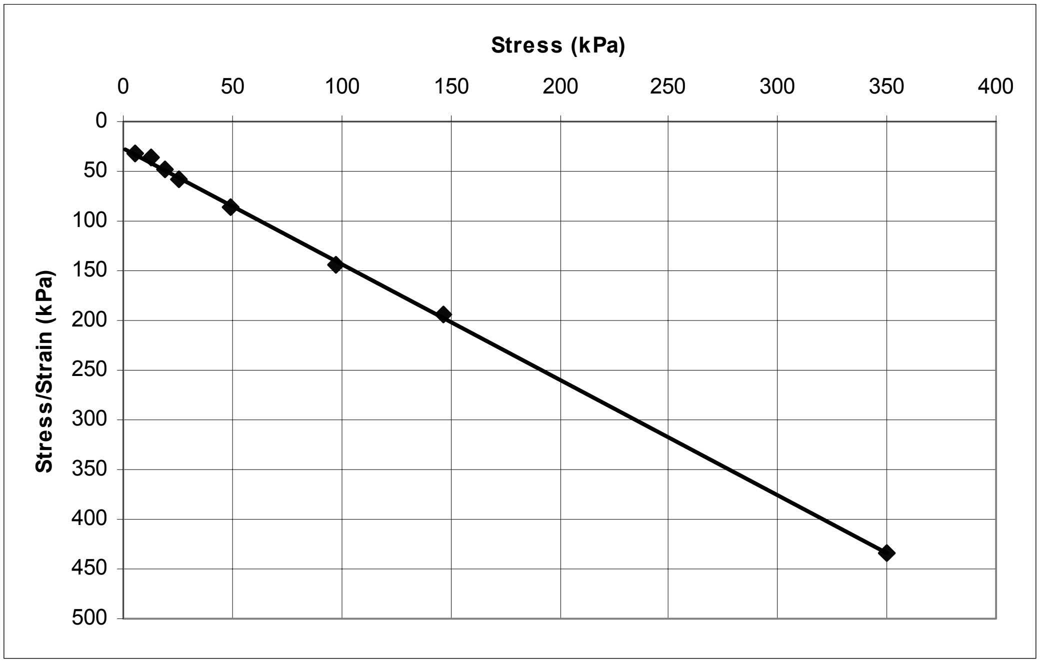

Figure 6.15 - Graph of Stress-to-Strain Ratio as a Function of Stress for the Dry Test

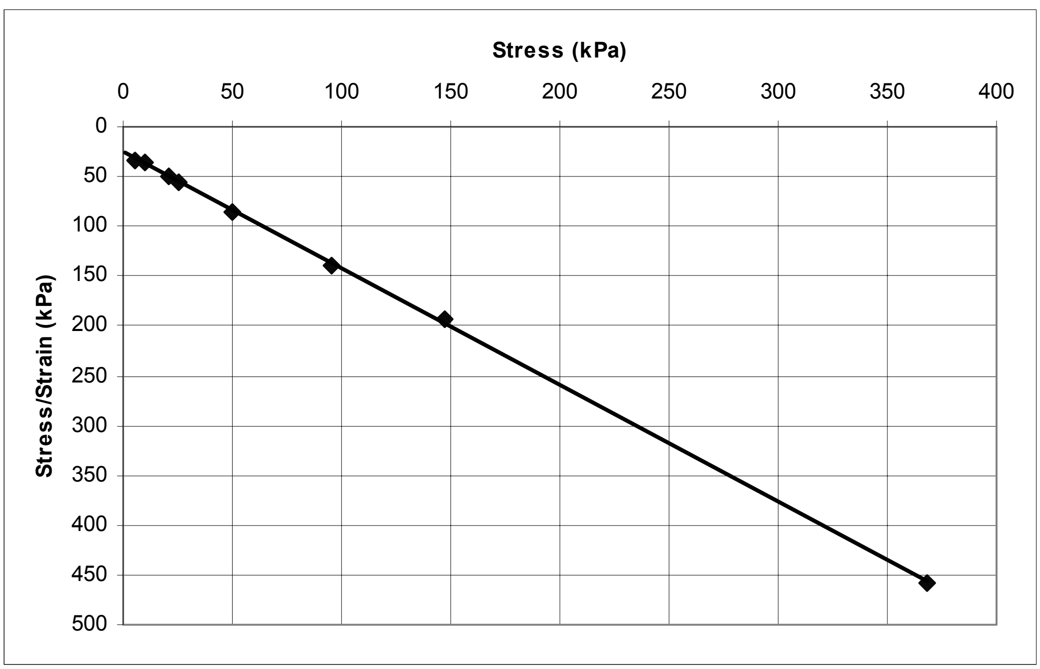

Figure 6.16 - Graph of Stress-to-Strain Ratio as a Function of Stress for the Wet Test

The same procedure was applied to other data published in the general literature. The studies of Debnath (2000) and Carvalho (1999) were added to the present study to understand the immediate settlement mechanism of MSW. Both studies and this investigation utilized compression chambers to study the MSW compressibility.

From Debnath (2000), tests 1 to 4 were utilized since those tests were performed in a similar way to the ones in this study. From Carvalho (1999), tests T2A10 and T2A14 were utilized. Table 6.2 presents some of the characteristics of the tests performed by Debnath (2000), Carvalho (1999), and the present study. Figures 6.17 to 6.22 present the graphs of stress-to-strain ratio as a function of stress for those tests.

Table 6.2 – Characteristics of the Compressibility Tests Utilized in This Study

Author	**Waste Utilized**	**Test**	**Initial Unit Weight (kN/m^3)**	**Maximum Load Applied (kPa) (*)**	**Stress (KPa) at Strain = 0**
This Study	Artificial MSW USA Average Composition	Dry	0.78	350	1.3
		Wet	N/A	368	1.3
Debnath (2000)	Real MSW Old (Tests 1,2,3) and Fresh (Test 4) MSW Louisiana, USA	1	6.3	58	3.3
		2	5	670	3.0
		3	4.5	1628	2.9
		4	1.7	1053	2.3
Carvalho (1999)	Real MSW Old MSW Sao Paulo, Brazil	T2A10	10	640	10.0
		T2A14	14	640	10.0

(*) Maximum load applied utilized in the study

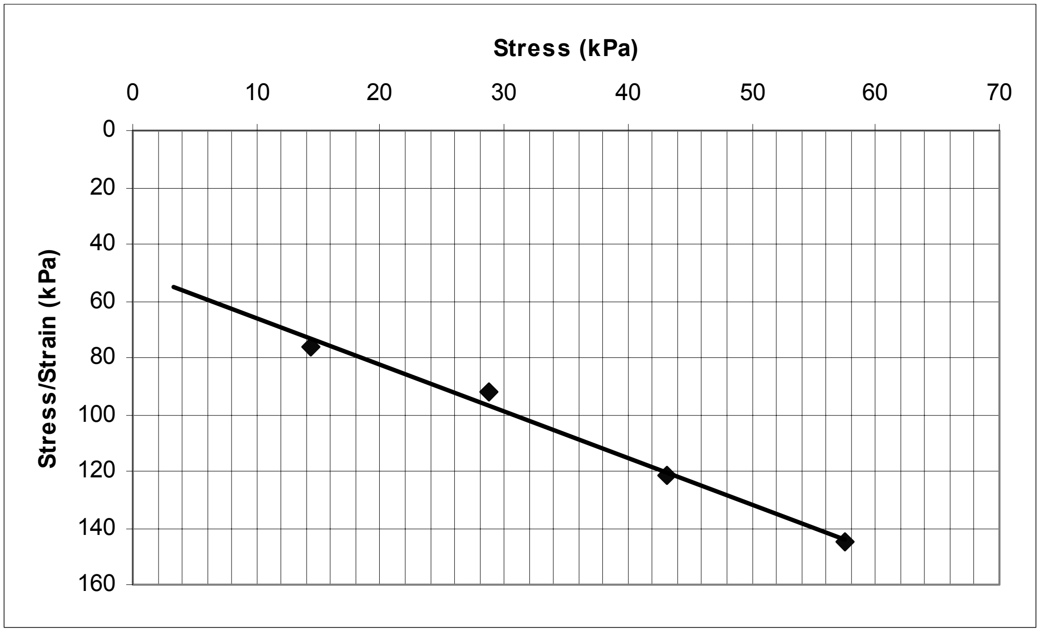

Figure 6.17 - Graph of Stress-to-Strain Ratio as a Function of Stress for Test 1 of Debnath (2000)

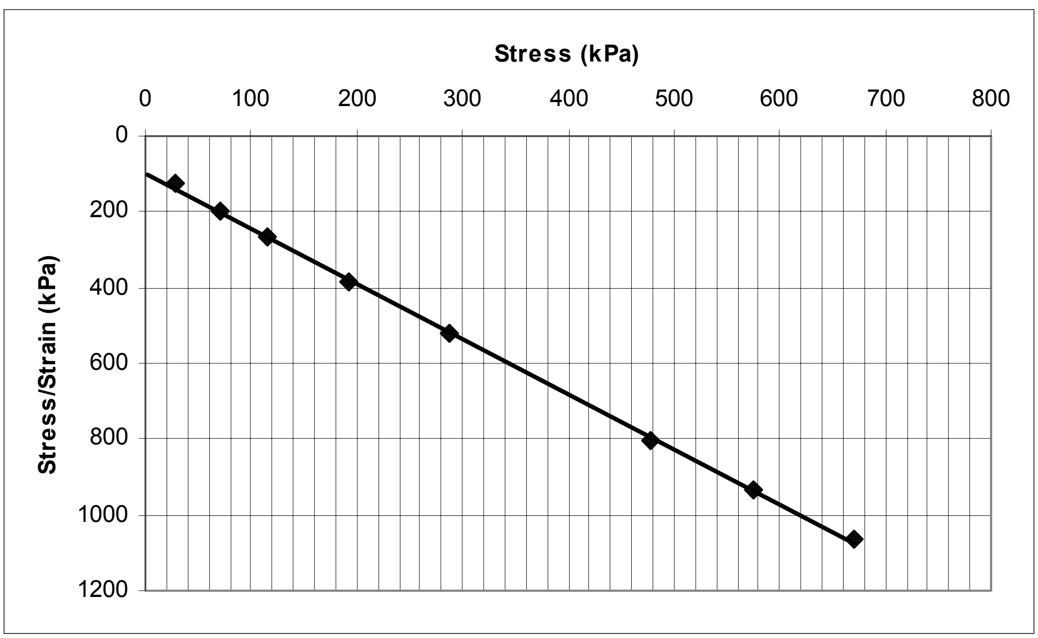

Figure 6.18 - Graph of Stress-to-Strain Ratio as a Function of Stress for Test 2 of Debnath (2000)

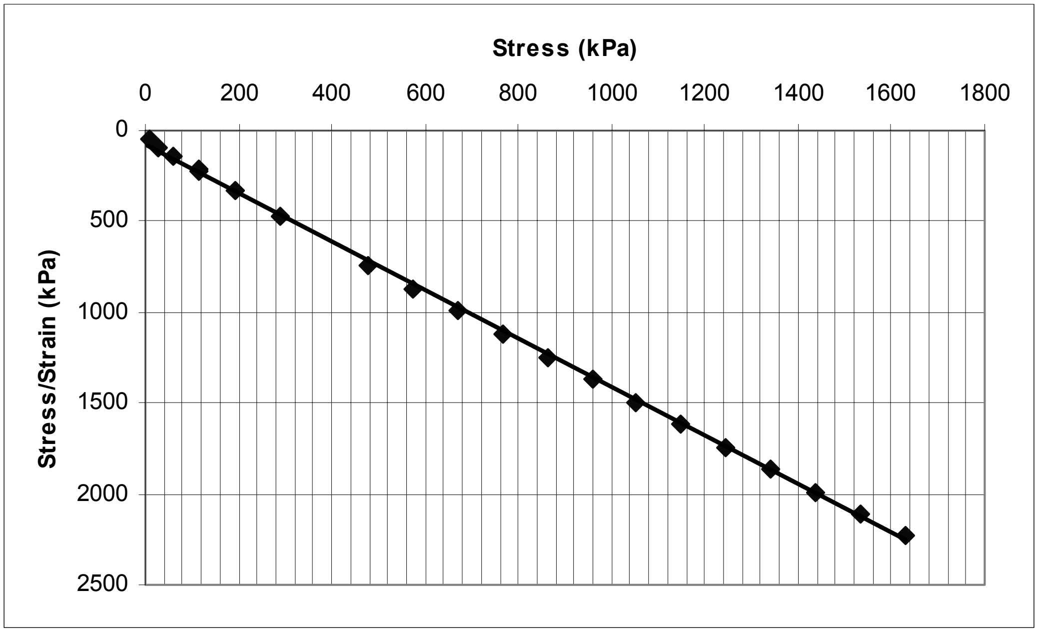

Figure 6.19 - Graph of Stress-to-Strain Ratio as a Function of Stress for Test 3 of Debnath (2000)

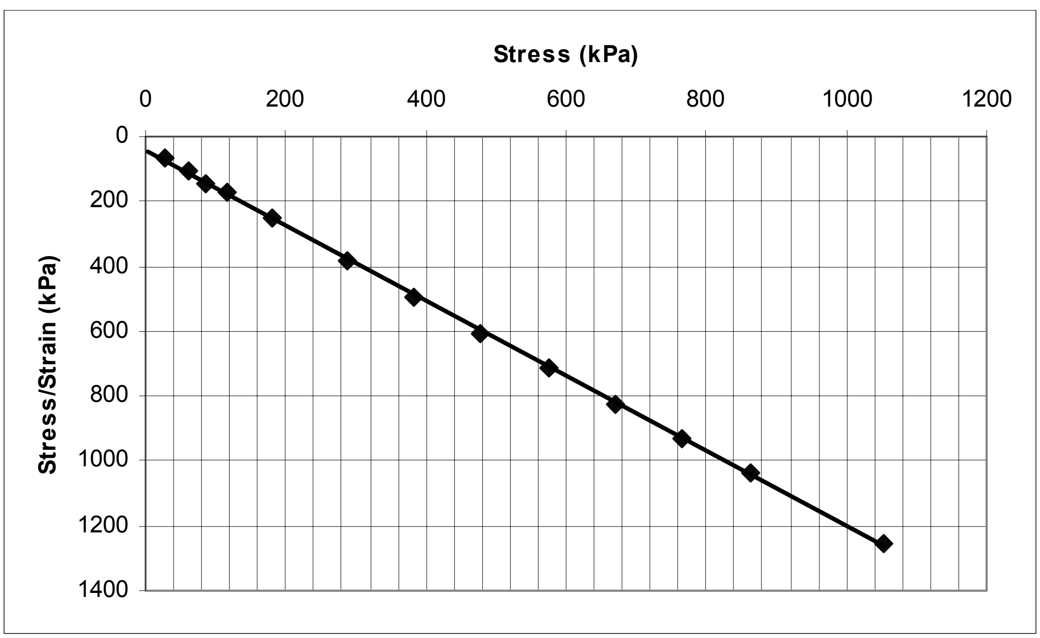

Figure 6.20 - Graph of Stress-to-Strain Ratio as a Function of Stress for Test 4 of Debnath (2000)

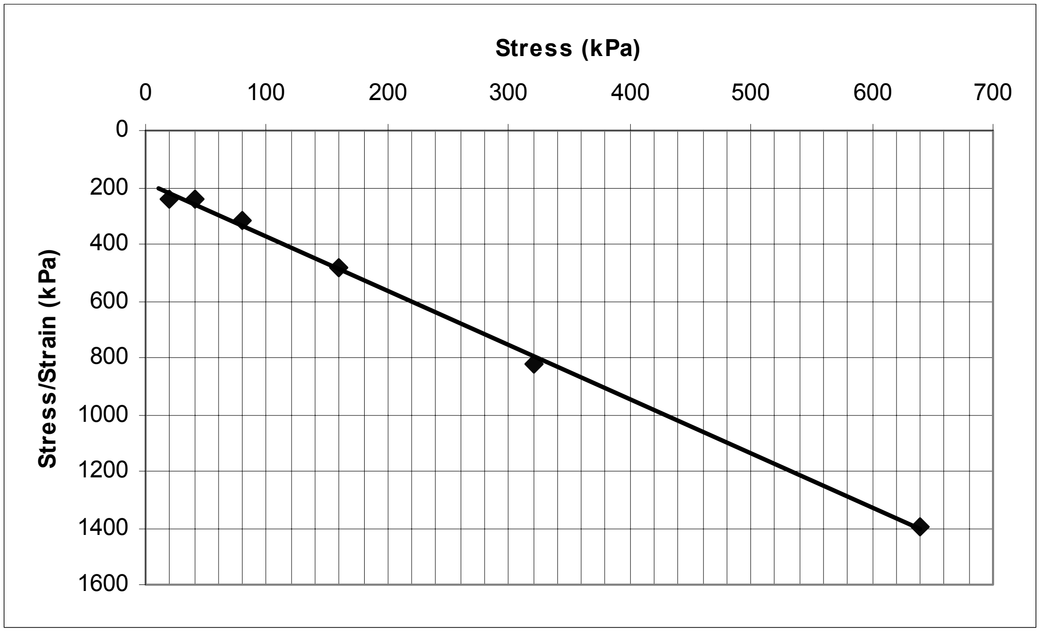

Figure 6.21 - Graph of Stress-to-Strain Ratio as a Function of Stress for Test T2A10 of Carvalho (1999)

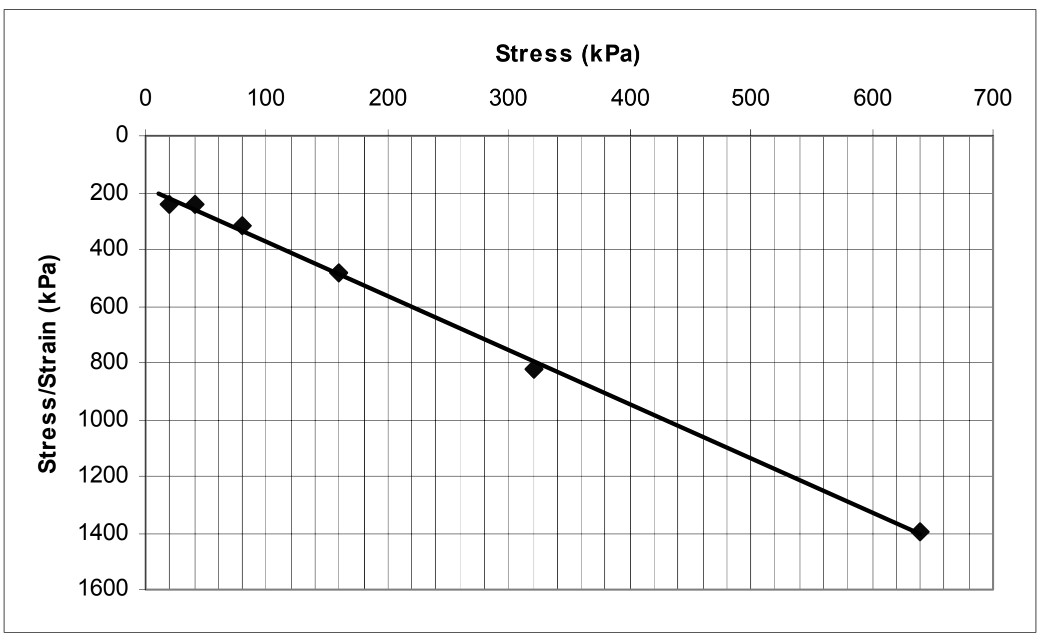

Figure 6.22 - Graph of Stress-to-Strain Ratio as a Function of Stress for Test T2A14 of Carvalho (1999)

From Figures 6.15 to 6.22, it can be observed that the curve stress to strain ratio as a function of stress is a straight line for all cases presented, which as mentioned before, is quite logical. However, the significance of the parameters of the linear function must be determined. In the new model presented herein, the expression of a linear curve for the cases shown can be written as:

$$\frac{\sigma_v}{\varepsilon} = a \cdot \sigma_v + b \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots (6.1)$$

where:

σ_V … total vertical stress at the middle of the layer,

ε … strain,

a, b …parameters of the linear curve.

Since the strain can be written as the ratio between the settlement and the initial thickness of the layer, Equation 6.1 can be rewritten as:

$$\frac{\sigma_v \cdot H_0}{\Delta H_i} = a \cdot \sigma_v + b \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots (6.2)$$

where:

H_0 … thickness of the waste layer,

ΔH_i …immediate settlement.

From Table 6.2, it is important to note that the initial stress (at strain = 0) is not the same for all of the tests (depending on the use of jacks, use of top plates, waste initial unit weight, etc.), which means that the initial thickness of the layer presented is not at $\sigma_V = 0$. The first step to compare the tests is to use the same reference for all the tests. In this case, an artifice is introduced in order to have strain = 0 at initial stress = 0, as follows:

Equation 6.2 can be rewritten as:

$$\Delta H_i = \frac{\sigma_v \cdot H_0}{a \cdot \sigma_v + b} \quad \text{...(6.3)}$$

If H_0 can be associated with the thickness of waste layer at $\sigma_v = 0$, then it may be divided as the sum of two components:

$$H_0 = H_e + \Delta H_{0-E} \quad \text{...(6.4)}$$

where:

H_e … thickness of waste at the beginning of the test,

$\Delta H_{0\text{-}E}$ … settlement between $\sigma_v = 0$ and the σ_v at the beginning of the test.

Equation 6.3 can be written as:

$$\Delta H_i = \frac{\sigma_v \cdot (H_e + \Delta H_{0-E})}{a \cdot \sigma_v + b} - \Delta H_{0-E} \quad \text{..(6.5)}$$

and the settlement between $\sigma_v = 0$ and the σ_v at the beginning of the test can be written as:

$$\Delta H_{0-E} = \frac{\sigma_{ve} \cdot (H_e + \Delta H_{0-E})}{a \cdot \sigma_{ve} + b} \quad \text{..(6.6)}$$

where σ_{ve} is the stress at the beginning of the test. Equation 6.6 can also be written as:

$$\Delta H_{0-E} = \frac{\sigma_{ve} \cdot H_e}{(a-1) \cdot \sigma_{ve} + b} \quad \text{..(6.7)}$$

And substituting Equation 6.7 into Equation 6.5:

$$\Delta H_i = \frac{H_e \cdot \sigma_v \cdot (a \cdot \sigma_{ve} + b)}{[(a-1) \cdot \sigma_{ve} + b)] \cdot (a \cdot \sigma_v + b)} - \frac{\sigma_{ve} \cdot H_e}{(a-1) \cdot \sigma_{ve} + b} \quad \text{..........................(6.8)}$$

which is the final expression to determine the parameters “a” and “b” for the tests studied. These parameters were determined by using regression analysis. Table 6.3 presents the determined parameters “a” and “b” for the tests studied.

Table 6.3 – Parameters “a” and “b” for the Tests Studied

Author	**Waste Utilized**	**Test**	**Initial Unit Weight (kN/m^3)**	**Parameter “a”**	**Parameter “b” (kPa)**
This Study	Artificial MSW USA Average Composition	Dry	0.78	1.17	21
		Wet	N/A		
Debnath (2000)	Real MSW Old (Tests 1,2,3) and Fresh (Test 4) MSW Louisiana, USA	1	6.3	1.68	30
		2	5	1.47	79
		3	4.5	1.34	51
		4	1.7	1.16	36
Carvalho (1999)	Real MSW Old MSW Sao Paulo, Brazil	T2A10	10	2.01	183
		T2A14	14	2.59	506

From Table 6.3, it appears that the parameters “a” and “b” increase with increasing initial unit weight for the tests studied. Figures 6.23 and 6.24 present the graphs of parameters “a” and “b” versus the initial unit weight. These figures show that indeed the parameters “a” and “b” present a strong relationship with the initial unit weight of the waste. This relationship is stronger for parameter “a” than for parameter “b”.

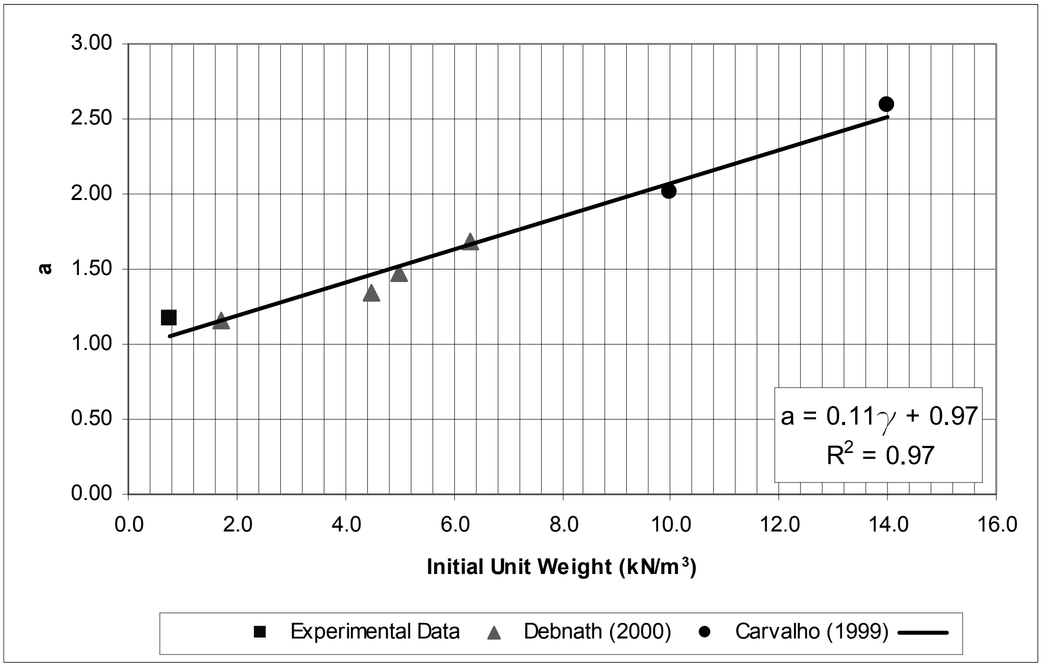

Figure 6.23 - Graph of Parameter “a” versus the Initial Unit Weight

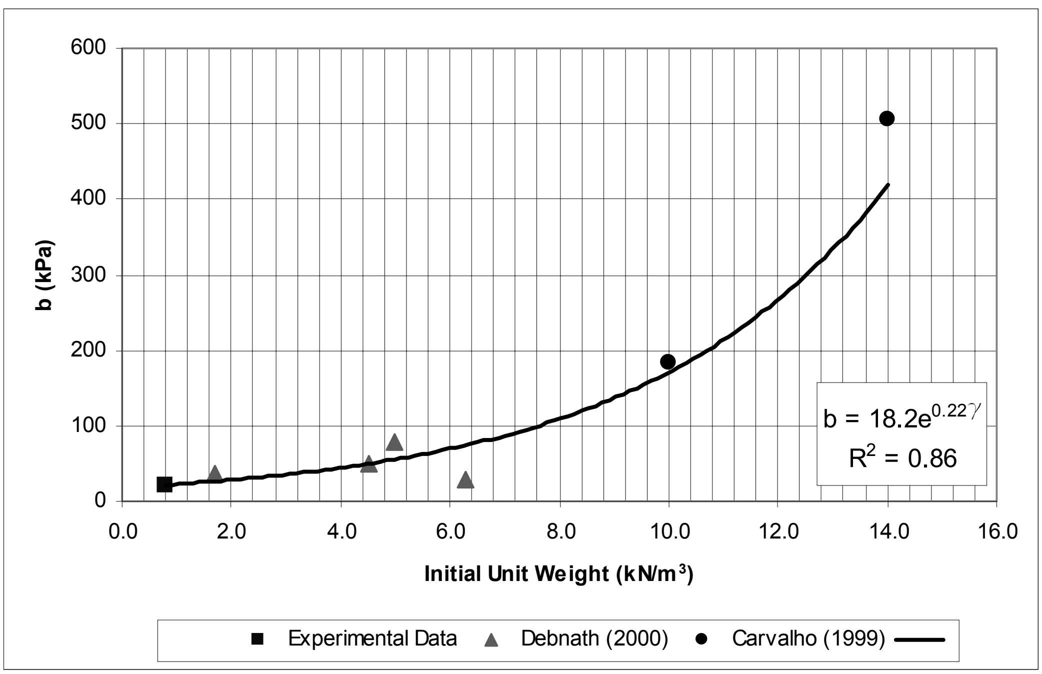

Figure 6.24 - Graph of Parameter “b” versus the Initial Unit Weight

Although it is not possible to analyze the effect of other parameters in the variation of parameters “a” and “b”, such as waste composition or aging due to the limited data available in the literature, it appears that the initial unit weight plays an important role in waste compressibility. Of course, waste composition should also have a significant importance in compressibility behavior. But for regular municipal solid wastes, the initial unit weight can be the main factor.

Additional analyses of parameters “a” and “b”, including their physical meaning, are presented in Section 6.2.3.

6.2.2. Time-Dependent Settlement

From Tables 5.23 and 5.24, as well as Figures 5.25 to 5.28, no substantial difference for the time-dependent settlements among the three lysimeters could be observed. Time-dependent settlements at the end of the period of study were equal to 29.2 cm, 32.7 cm, and 29.5 cm (corresponding to relative settlements of 12.5%, 14.8%, and 13.3%) for Lysimeters 1, 2, and 3, respectively.

Although it was expected that settlements should be greater for the two bioreactor lysimeters, since large quantities of liquids were injected and greater waste stabilization rates were achieved, no substantial differences were noticed. However, some distinction can be noticed between Lysimeters 2 and 3. After starting recirculation in Lysimeter 2 (around day 110), the path of settlements of the waste in the two lysimeters, practically the same until that point, started to change. The same was not observed in Lysimeter 1, however.

An explanation for the observation that the three lysimeters have almost the same time-dependent settlement is that a structure (or skeleton) formed by plastics, glass, and more resilient materials supports the load while the less resilient and more readily decomposable materials are part of the larger voids of more resilient materials. Whenever the less resilient and more readily decomposable materials are being degraded, no effects are mechanically felt in the structure that supports the load.

Settlement rates in the three lysimeters followed the pattern noticed in the general literature, starting with elevated rates and decreasing with time. Initial rates were as large as 10.2 mm/day for Lysimeters 2 and 3, and 3.8 mm/day for Lysimeter 1. It can be noticed, however, that Lysimeter 1, which exhibited the larger immediate settlement, had the lowest initial settlement rate and the lowest time-dependent settlement at the end of the study. At the end of the study, typical values of 0.2 mm/day were observed for the three lysimeters. However, a small acceleration of the settlements could be noticed in the last readings of the three lysimeters and was most significant in Lysimeter 3. There is no apparent reason or cause that can explain that acceleration.

If total settlements are analyzed, the MSW in Lysimeter 1 had the largest settlement (64.8 cm/27.6%), followed by Lysimeter 2 (55.5 cm/25.0%), and then by Lysimeter 3 (47.3 cm/21.4%).

Using the Hyperbolic model (Ling et al., 1998), the term-dependent settlements in the three lysimeters can be modeled. Figures 6.25, 6.26, and 6.27 present the application of the Hyperbolic model to Lysimeters 1, 2, and 3, respectively. Table 6.4 presents the parameters determined for the model.

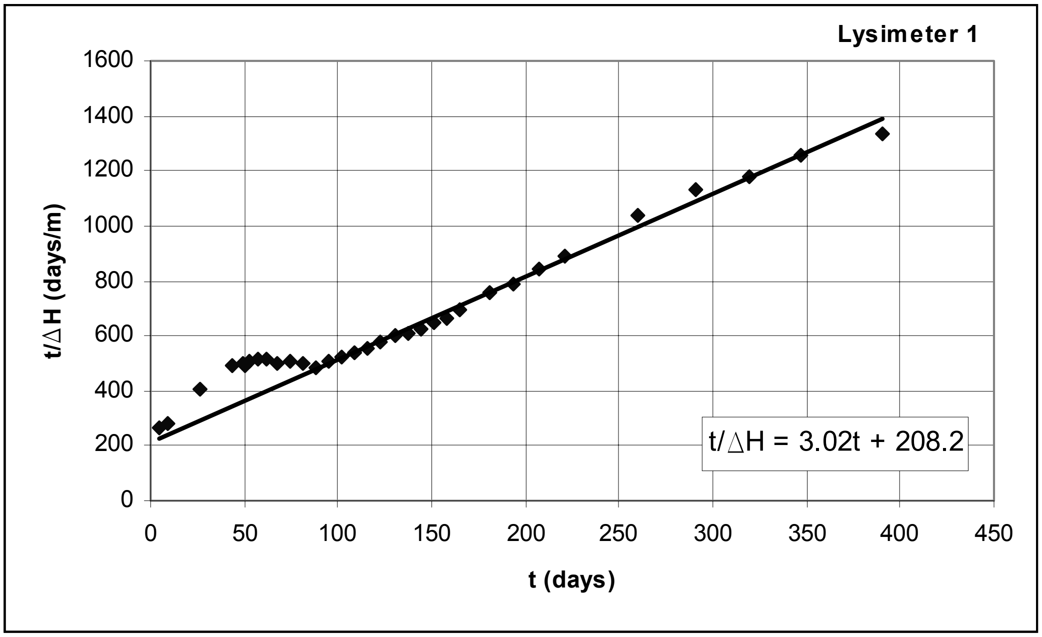

Figure 6.25 – Application of the Hyperbolic Model to Lysimeter 1

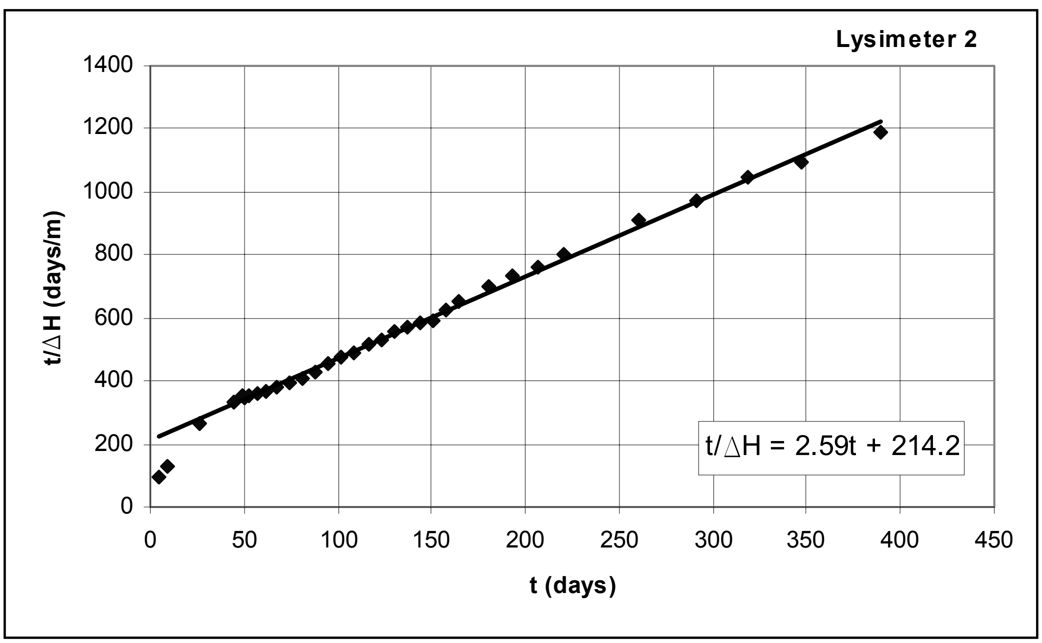

Figure 6.26 – Application of the Hyperbolic Model to Lysimeter 2

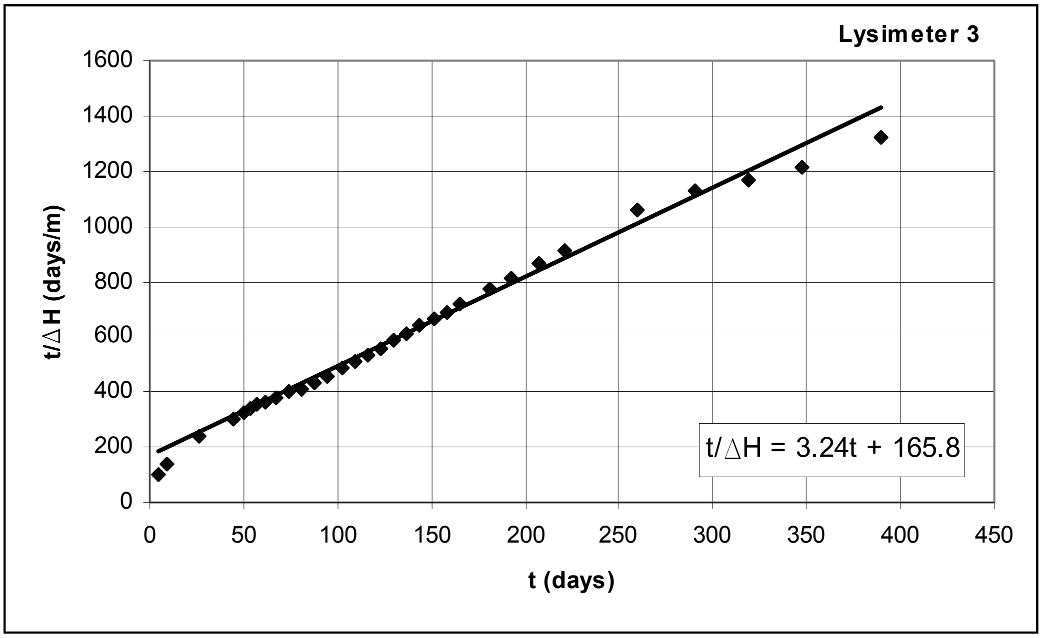

Figure 6.27 – Application of the Hyperbolic Model to Lysimeter 3

Table 6.4 – Hyperbolic Model Parameters for Lysimeters 1, 2, and 3

Lysimeter	Initial Unit Weight (kN/m^3)	Unit Weight after Immediate Settlement (kN/m^3)	ΔH_∞ (m)	V_{rp0} (mm/day)
1	1.29	1.52	0.33	4.8
2	1.36	1.52	0.39	4.7
3	1.36	1.48	0.31	6.0

Table 6.4 shows that there is not much difference among the parameters obtained in the three lysimeters. The final settlements determined by the model adjustment cannot be considered the "ultimate final settlement" since an acceleration of the settlements was noticed, as mentioned before, and may change the final settlement. The same applies to the initial settlement rates, which are somewhat different from the observed initial settlement rates.

Unfortunately, there are very few studies with data presented in the general literature about time-dependent settlements, mainly due to the difficulties to conduct the experiment over a long period of time. One of the very few studies conducted is presented by Rao et al. (1977). The authors also used a mixture of materials to simulate the average composition of MSW in the United States, and utilized a 61-cm (2-ft) diameter consolidometer to study the long-term settlement of MSW under different densities and relative increments of load applied ($\Delta\sigma/\sigma$). The study suggested that the relative increment of load applied is possibly the parameter that can most affect the long-term settlement of MSW.

Figures 6.28 to 6.33 present the application of the Hyperbolic model to the study of Rao et al. (1977) for a density of 2 kN/m^3 and stresses between 68 kPa and 1,550 kPa, and for relative increment of loads ($\Delta\sigma/\sigma$) between 0.54 and 0.98. It must be noticed that only the linear part of the data, (typically above 1,000 min) was considered. Table 6.5 presents the parameters of the Hyperbolic model determined for the study of Rao et al. (1977).

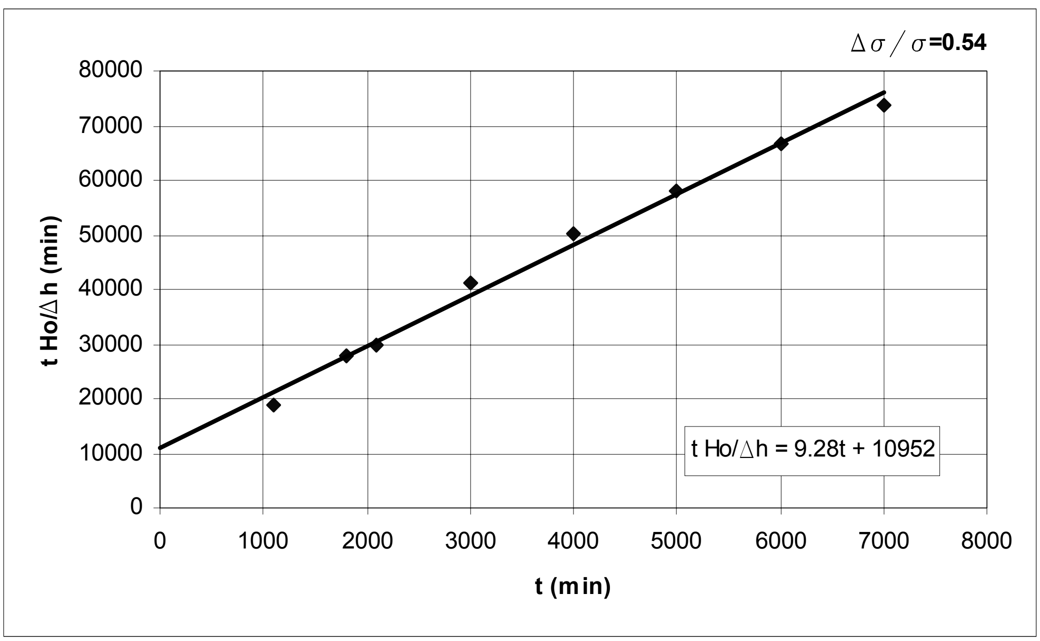

Figure 6.28 – Application of Hyperbolic Model to the Study of Rao et al. (1977) for Relative Increment of Load of 0.54

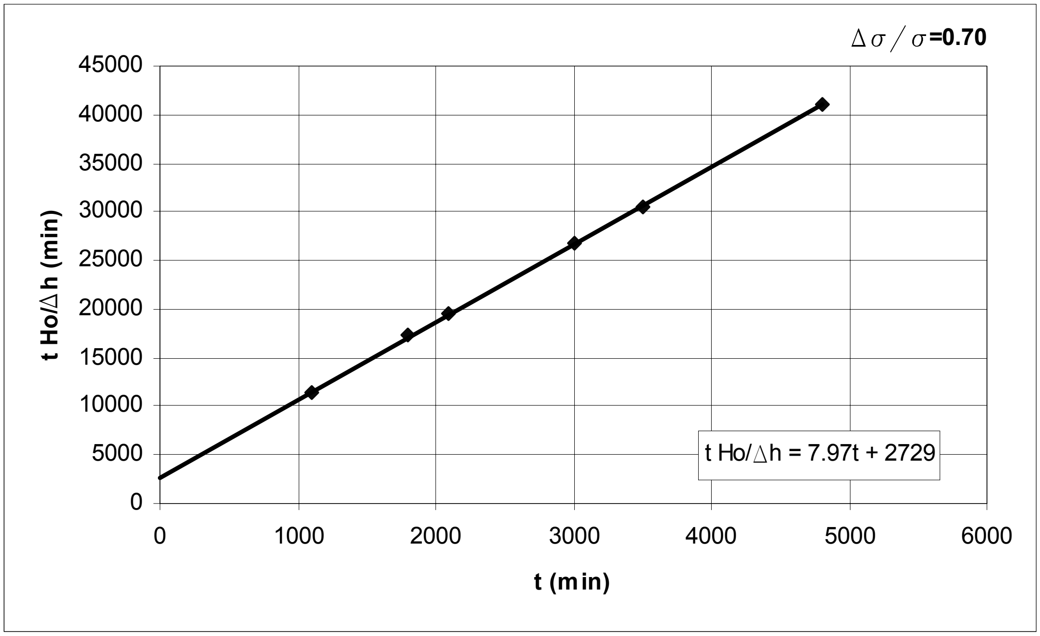

Figure 6.29 – Application of Hyperbolic Model to the Study of Rao et al. (1977) for Relative Increment of Load of 0.70

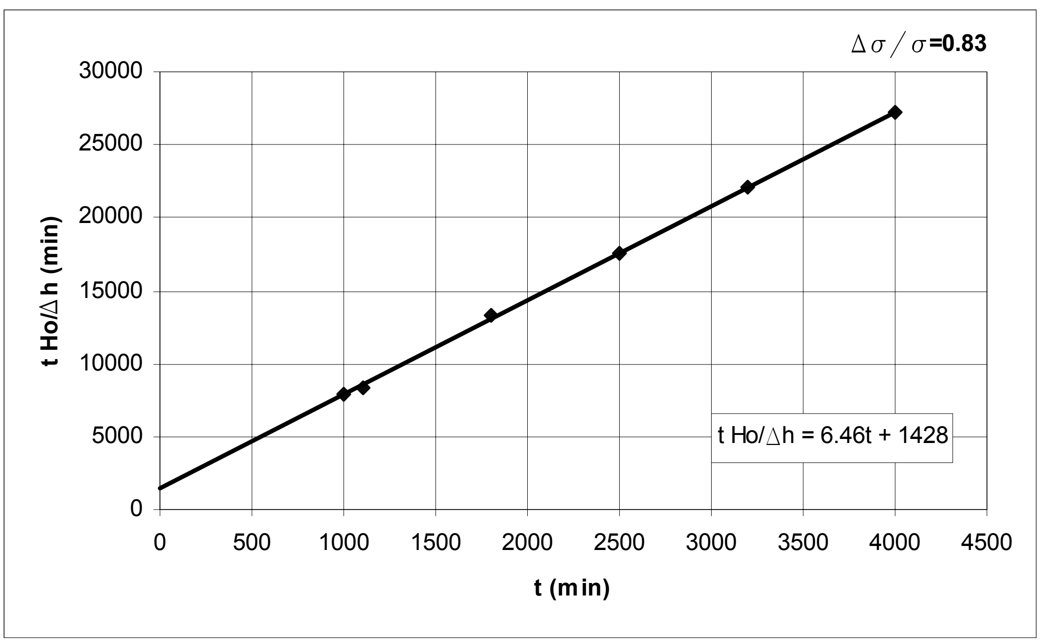

Figure 6.30 – Application of Hyperbolic Model to the Study of Rao et al. (1977) for Relative Increment of Load of 0.83

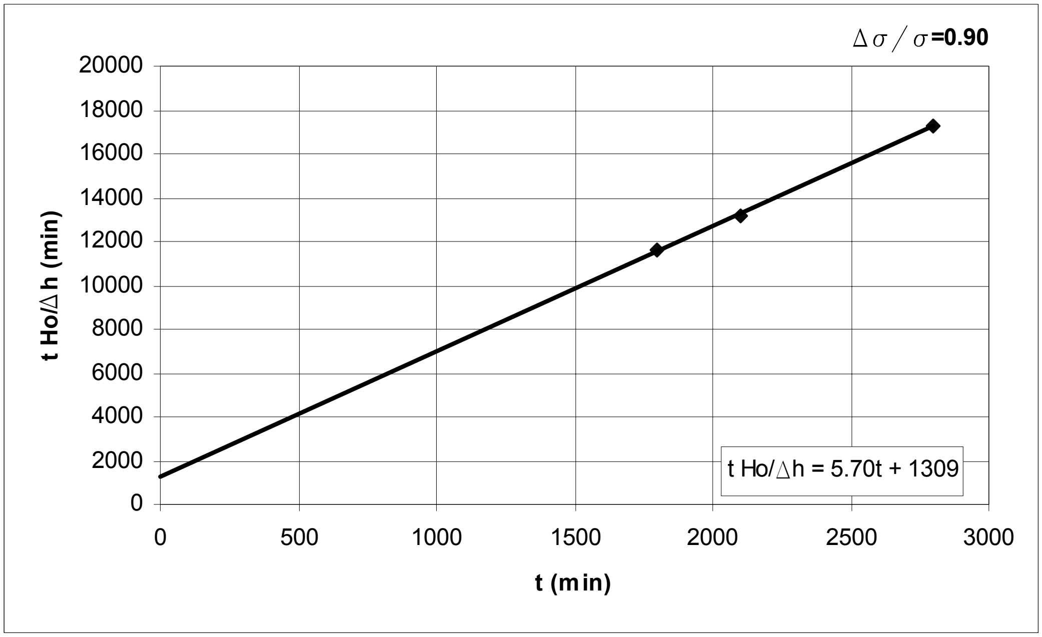

Figure 6.31 – Application of Hyperbolic Model to the Study of Rao et al. (1977) for Relative Increment of Load of 0.90

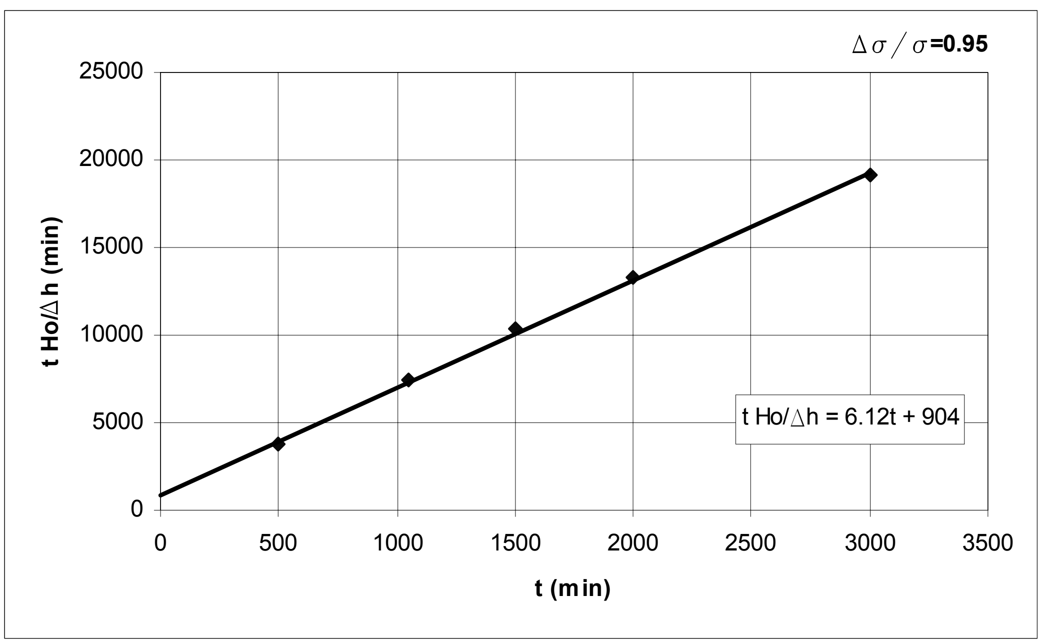

Figure 6.32 – Application of Hyperbolic Model to the Study of Rao et al. (1977) for Relative Increment of Load of 0.95

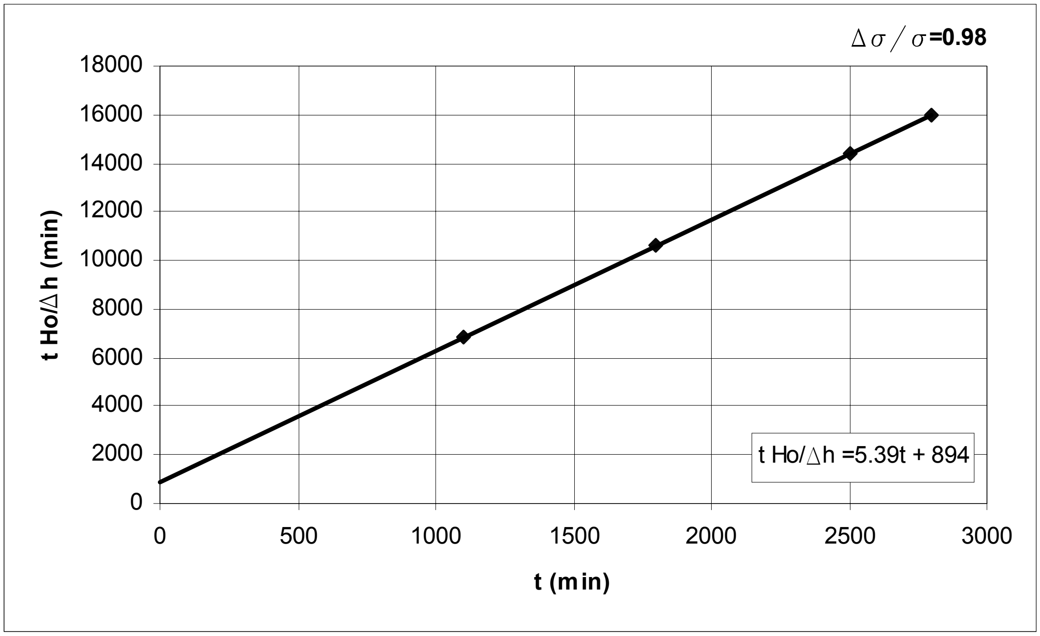

Figure 6.33 – Application of Hyperbolic Model to the Study of Rao et al. (1977) for Relative Increment of Load of 0.98

Table 6.5 – Hyperbolic Model Parameters for the Study of Rao et al. 91977)

$\Delta\sigma/\sigma$	**Initial Unit Weight (kN/m^3)**	**Final Stress $\sigma+\Delta\sigma$ (kPa)**	**ΔH_∞ /Ho**	**V_{rp0} (mm/day)**
0.54	2.0	68	0.108	15.8
0.70	2.0	116	0.125	57.0
0.83	2.0	212	0.155	94.8
0.90	2.0	403	0.175	75.9
0.95	2.0	786	0.163	109.9
0.98	2.0	1552	0.185	93.4

A trend can be noticed that the relative final settlement and initial settlement rate increases with the increasing of the relative applied stress. Figures 6.34 and 6.35 present the graphs of the parameters of the Hyperbolic model as function of the relative applied stress for the tests studied.

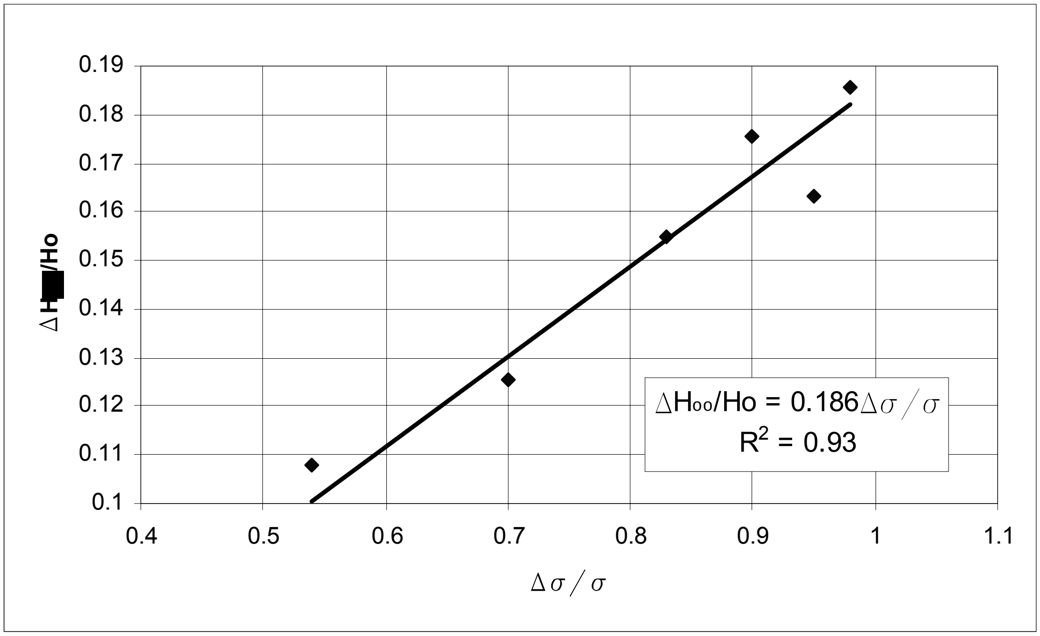

Figure 6.34 – Graph of Relative Final Settlement as a Function of Relative Stress Applied

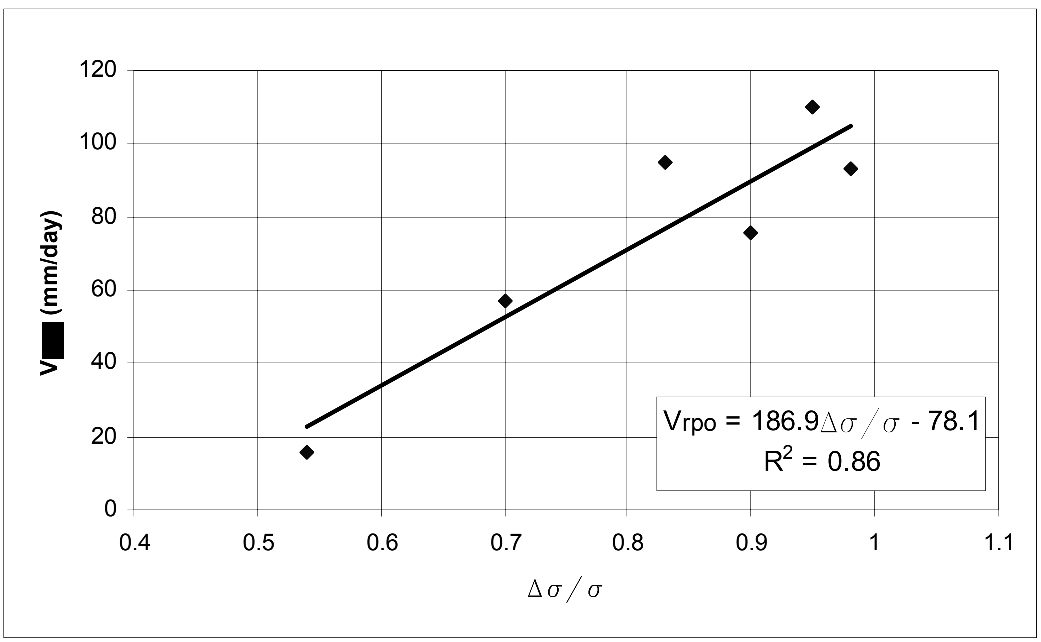

Figure 6.35 – Graph of Initial Settlement Rate as a Function of Relative Stress Applied

It must be noticed that the adjusted curves here are only an exercise suggested to verify how the parameters can vary with the relative stress applied. It is logical that minimal and maximum values for the parameters may exist in other ranges of relative stress applied and do not necessarily follow the relationship presented. Another important factor to be considered is that at the beginning of measurements, the curve adjusted by the Hyperbolic model does not fit the first points very well. Since the tests conducted by Rao et al. (1977) had a duration of less than 5 days, the quality of the parameters obtained is unknown.

From the lysimeters, the values of maximum relative settlement obtained were 0.17, 0.20, 0.15, which correspond approximately to the maximum values obtained for the study of Rao et al. (1977) when the Hyperbolic model was applied. The relative stress applied to the waste in the lysimeters was greater than 8, while in the study of Rao et al. (1977) this value is lower than 1. The initial settlement rate parameters determined for the lysimeters are much lower than those determined using the study of Rao et al. (1977), as well. It must be noticed that the values of the initial settlement rate obtained for the study of Rao et al. (1977) are very high and do not follow what should be expected in practice.

6.2.3. New MSW Compressibility Model

6.2.3.1. The Hyperbolic Rheological Model

The proposed model can be understood as a rheological model with the immediate settlement represented by a spring and the slow, time-dependent deformation by a dashpot and another spring linked in parallel and in series, as shown in Figure 6.36. The rheological model is similar to the Gibson and Lo Rheological model, but instead of an exponential function, hyperbolic expressions are applied.

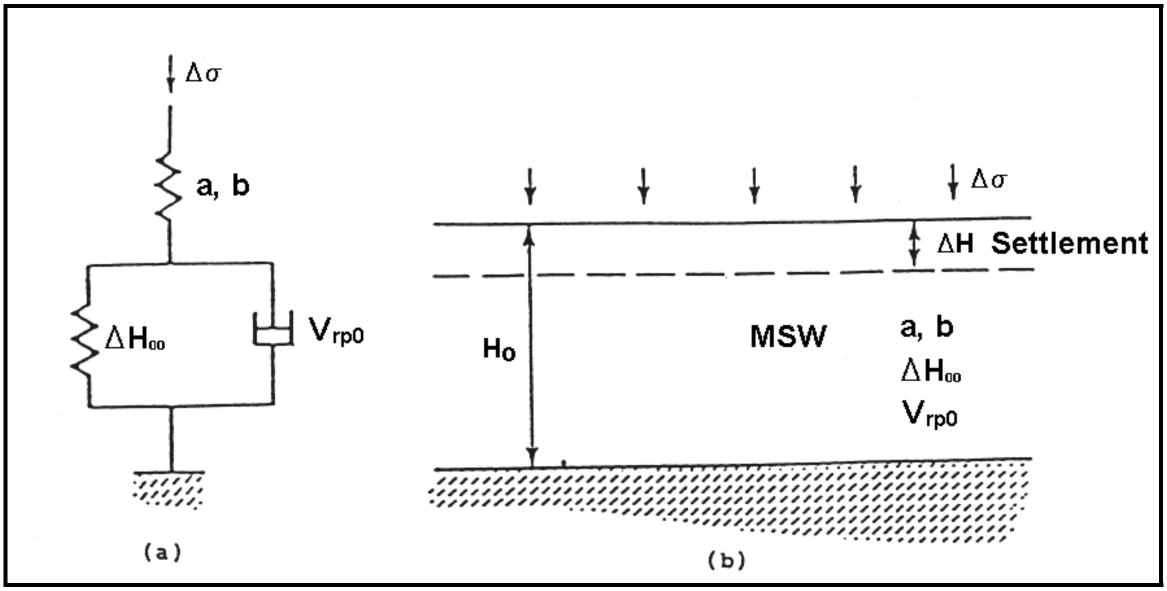

Figure 6.36 – Representation of the Hyperbolic Rheological Model

Whenever there is a load increment (for example, the construction of new cells), the spring deforms and is immediately compressed. The second spring compression arrangement is retarded due to the dashpot, and the loading applied to the spring is progressively transferred to the dashpot. At the end of the settlements, all loading will be absorbed only by the springs and the dashpot will not sustain any part of the total applied loading.

As shown in the previous sections, the formula for the immediate and time-dependent settlements is:

$$\Delta H_i = \frac{H_e \cdot \sigma_v \cdot (a \cdot \sigma_{ve} + b)}{[(a-1)\cdot \sigma_{ve} + b)] \cdot (a \cdot \sigma_v + b)} - \frac{\sigma_{ve} \cdot H_e}{(a-1)\cdot \sigma_{ve} + b} \quad \text{……..………..…………(6.9)}$$

for the immediate settlement, where:

ΔH_i … immediate settlement,

H_e … initial thickness of waste,

σ_{ve} ... initial total vertical stress at the middle of the layer,

σ_v … final total vertical stress at the middle of the layer,

a, b …parameters of the model.

and

$$\Delta H_{td} = \frac{t}{\frac{1}{v_{rp0}} + \frac{t}{\Delta h_{\infty}}} \quad \text{……(6.10)}$$

is the expression for the time-dependent settlement, where:

ΔH_{td} … time-dependent settlement,

V_{rp0} … initial settlement rate,

Δh_{∞} … final settlement.

Thus, the total settlement (ΔH) corresponds to:

$$\Delta H = \Delta H_i + \Delta H_{td} \quad \text{……(6.11)}$$

For immediate settlement, the parameter "a" can be physically understood as a "non-linear spring coefficient", and the lower the value, the softer the material. For the same material (as MSW), the lower the density, the lower parameter "a" is. An expression relating parameter "a" to the initial unit weight was found to be:

$$a = 0.11 \cdot \gamma + 0.97 \quad \text{……(6.12)}$$

where:

γ in kN/m^3, "a" dimensionless.

Parameter “b” controls the initial variation stress-strain (at stress = 0), and the lower the initial unit weight, the lower its value. An expression relating parameter “b” to the initial unit weight was:

$$b = 18.2 \cdot e^{0.22 \cdot \gamma} \qquad \text{………………………………………………………...…..(6.12)}$$

where:

γ in kN/m^3, “b” in kPa.

For the time-dependent settlement, it was noticed that short-term laboratory experiments were not able to accurately determine the parameters for the model presented, although it appears that they vary with the relative stress applied ($\Delta\sigma/\sigma$). Long-term tests as conducted with lysimeters for one year seem to be more accurate. However, since the load applied to the waste in all lysimeters was the same, it was not possible to verify the variation of the relative load applied with the parameters of the model. In addition, it can be speculated that biodegradation may not be so significant to settlements, as observed in the lysimeters and also in the general literature. However, more evidence is needed to support this statement.

6.2.3.2. Verification of the Model

To verify the applicability of the new model, an example is presented. Using the data of a landfill located in Sao Paulo, Brazil (Bandeirantes Landfill), the compressibility of one cell was evaluated. The top of the cell was regularly monitored by a concrete mark (MS-508). The sequence of construction is described below, as presented by De Abreu (2000).

The theoretical (uncompressed) height of the cell is approximately 5 m and composed of 4.5 m of compacted MSW and 0.5 m of cover soil with a unit weight of approximately 15 kN/m^3 (according to the landfill operation procedures). Initial density of the compacted waste was considered to be about 7 kN/m^3 (according to studies conducted by Marques (2001), who conducted MSW field compaction tests in the same landfill). The model compressibility parameters adopted are indicated below:

- A = 1.74 (using Equation 6.11, with initial unit weight of 7 kN/m^3),
- B = 84.9 kPa (using Equation 6.12, with initial unit weight of 7 kN/m^3),
- $\Delta h\infty/Ho$ = 0.15 (using the same parameter obtained for Lysimeter 3),
- V_{rp0} = 6 mm/day (using the same parameter obtained for Lysimeter 3).

The initial stress in a point located in the middle of the waste layer is, therefore, 15.75 kPa and, after the application of load (by the 0.5 m soil cover), the final stress is equal to 23.25 kPa.

Table 6.6 presents the results obtained with an application of the Rheological Hyperbolic method and the real values observed (Boscov and De Abreu, 2001).

Table 6.6 – Application of the Rheological Hyperbolic Model to MS-508

Time	Thickness of the Cell Calculated (m)	Real Thickness of the Cell (m)	Difference (%)
Initial Situation	-	5.00	-
Immediate Settlement (t=0)	4.76	4.80	-0.8
t= 212 days	4.34	4.10	5.9
t=701 days	4.21	3.58	17.6

From Table 6.6, it can be noticed that although there was practically no difference between the values calculated and those observed for the immediate settlements, there are considerable differences for the time-dependent settlements when the model is applied. This difference is due to the difficulties associated in obtaining the parameters for time-dependent settlement formulation for different relative loads applied, as discussed in the previous sections and probably due to the difference between the composition of Brazilian and North-American wastes that can influence the time-dependent settlements. Table 6.7 presents the results when the time-dependent parameter $\Delta h_\infty/H_o$ is changed to 0.40 (a value that gave minimum error), maintaining all other parameters the same.

Table 6.7 – Application of the Rheological Hyperbolic Model to MS-508, Using $\Delta h_\infty/H_o$ equal to 0.40

Time	Thickness of the Cell Calculated (m)	Real Thickness of the Cell (m)	Difference (%)
Initial Situation	-	5.00	-
Immediate Settlement (t=0)	4.76	4.80	-0.8
t= 212 days	4.04	4.10	-1.5
t=701 days	3.55	3.58	-0.8

Table 6.7 shows that practically no difference can be noticed between the values calculated and observed, when a higher value of relative final settlement is used.

Of course, the same principle can be applied to landfills with multiple layers (or cells). In this case, the upper layers should be considered as surcharges for the bottom layers and each layer would have its own stress-strain-time relationship. The use of a computer code (Fortran, for example) or spreadsheets could easily be utilized to calculate the settlements.

Although the use of the Rheological Hyperbolic model is promising, based on the several benefits and verifications herein presented, more studies are necessary to adequately calibrate the parameters, mainly the parameters associated with the time-dependent settlements to make it applicable to full-scale situations.

It is important to notice that the Rheological Hyperbolic model has all the features that the Technical Committee TC5 from the International Society of Soil Mechanics and Geotechnical Engineering (ISSMGE, 1997) points out to be necessary for a settlement model for MSW:

- Be dimensionally correct,
- Be defined by a few parameters,
- These parameters must have a physical meaning or at least be related to a known property,
- Be capable of separating the influence of the relevant factors in the analysis,
- And, mainly, be able to provide realistic and precise full-term predictions.

CHAPTER 7

CONCLUSIONS

7.1. Environmental Study

7.1.1. Conclusions

Based on the results obtained and the analysis conducted for this study, the following conclusions can be made:

Organic Matter Stabilization

- The use of facultative landfill bioreactor technology with recirculation of pre-treated leachate was able to stabilize the organic matter in the waste at a much higher rate than the lysimeter with straight recirculation and the conventional lysimeter.

- Values as low as 51 mg/L for COD and 11 mg/L for BOD_5 were obtained for the facultative lysimeter at the end of the period of study. COD and BOD_5 values of 396 mg/L and 120 mg/L, respectively, were obtained for the lysimeter with raw leachate recirculation. Values of 436 mg/L and 371 mg/L were obtained for the conventional lysimeter for COD and BOD_5, respectively.

- The lysimeter with straight leachate recirculation had a slightly better performance in stabilizing organic matter than the conventional landfill, when the values of COD, BOD_5, BOD-to-COD ratio, the filtered-to-total COD ratio, and the COD-to-chloride ratio are compared.

- The COD-to-chloride ratio decreased very quickly with lower values for the facultative lysimeter and the conventional lysimeter indicating that biological activity played an important role in removing COD in those lysimeters. In the conventional lysimeter, the ratio first decreased and then increased again, showing that biological activity removed the COD in the first days, but after that, slow decomposition took place and the biological activity in this lysimeter was minimum.

Nitrogen Management

- Ammonia nitrogen was removed at a much higher rate in the facultative lysimeter than in the other two. The reduction in ammonia-nitrogen concentrations was approximately 90% for the facultative lysimeter at the end of the study, while the reduction for the other two lysimeters was only equal to approximately 20%.

- Nitrate-nitrogen concentrations stayed low throughout the study. Maximum values reached 1.2 mg/L on the facultative lysimeter, while the concentrations of the treated effluent sent to the same lysimeter reached 24 mg/L.

- The ammonia-nitrogen- to-TKN ratio stayed typically between 0.3 and 0.6 throughout the study, with organic nitrogen representing about 40% to 70% of the TKN values.

- Based on the values of the several parameters studied in the analysis of the leachate and from the analysis of the gases generated in the lysimeters, it can be concluded that conversion of nitrates into nitrogen gas took place inside the facultative lysimeter due to denitrification, using the organic matter available inside the lysimeter.

Metals

Metals such iron and aluminum presented low concentration values. Iron concentrations peaked very early in the study (around days 46 to 60) and then decreased to low values, staying with low values during the remaining time of the study. Aluminum concentrations were typically lower than 0.2 mg/L throughout the study for all lysimeters. The probable reason for those low values is that metals are typically mobile at low pH values, which were not observed during the study.

Stabilization Phases

- The values of pH and TVA suggest that all three lysimeters passed through the three first decomposition phases very fast, staying at the methanogenic phase (lysimeter with straight recirculation and conventional lysimeter) and quasi-post-methanogenic phase (facultative lysimeter) after 50 days.

- After 90 days it was also noticed that the facultative lysimeter leachate was visually clearer than the other two with significant transparency and without color.

Gas Composition

- Methanogenesis was established in the lysimeter with straight recirculation and in the conventional lysimeter. Much greater fractions of methane and carbon dioxide were observed when compared to other gases in these two lysimeters than in the facultative lysimeter.

- Raw leachate recirculation was beneficial in generating better quality gas to energy recovery, and agreed with other studies presented in the general literature. The fraction of methane generated was initially almost the same, around 20% to 30% in the lysimeter with straight recirculation and in the conventional lysimeter. At the end of the study, it increased to 40% for the lysimeter with straight recirculation and remained at the same levels for the conventional lysimeter.

- The results demonstrate that the facultative lysimeter was able to operate in both conditions: denitrification (when treated leachate was added to the lysimeter) and methanogenic conditions (when treated leachate was not added to the lysimeter). The achievement of methanogenic conditions in the lysimeter during a certain period of time supports that the lysimeter had microbiological activity under anaerobic or semi-aerobic conditions.

- A tentative procedure of achieving denitrification and methanogenesis at the same time was tried at the end of the study by reducing the rate of treated effluent applied to the facultative lysimeter. An increase in the methane fraction was noticed to occur at a level not observed when treated effluent was added to the lysimeter, therefore verifying that both conditions can co-exist in the lysimeter.

Leachate Production

As expected, the bioreactor lysimeters produced much more leachate than the conventional lysimeter, since leachate production is directly related to the quantity of liquids that enter the landfill.

7.1.2. Applicability of the Study to Full-Scale Landfills

Based on the results of this study, the facultative bioreactor landfill is a promising technology for waste treatment and disposal. The stabilization of the waste mass is achieved more quickly than in conventional landfills. In addition, it also offers a mechanism for removing ammonia, which can be the major component that limits the potential to safely discharge leachate to the environment.

Although it can be a successful technology for full-scale landfills, there are still several points that must be addressed such as:

- Leachate produced. A larger quantity of leachate is produced by the facultative landfill when compared to conventional landfills. This can increase the hydraulic loading to the landfill liner and increase the rate of groundwater pollution by leachate. To avoid this, a special leachate collection system can be designed to capture the excessive leachate. However, the use of a double-composite liner with a leachate detection system instead of a single-composite liner in bioreactor landfills is recommended by Lee and Lee (1994). It must be noticed, however, that in the facultative landfill, the liner will be subjected to significant quantities of leachate in a short period of time, theoretically, as opposed to conventional landfills with lower quantities during a longer period of time. A specified designed life of 20 years for geosynthetic membranes may not provide adequate protection for conventional landfills with stabilization periods of decades (Reinhart and Townsend, 1998).

In addition, cover systems must be reviewed since seepage through the slopes can occur for facultative landfills due to the addition of liquids to the waste mass.

However, it must be noticed that a lower quantity of liquids (treated leachate, in the case of the facultative landfill) can possibly be added to the waste and have the same benefits as large quantities. In this case, further investigation studying the effects of the rate of treated leachate added in waste stabilization must be addressed. Nevertheless, this study demonstrates that for the same quantities of liquids added, the facultative lysimeter had much better performance than the lysimeter with raw leachate recirculation.

- <u>Geotechnical stability considerations</u>, such as the increase in internal pore pressures due to the addition of liquids and a decrease in strength of the components of the waste, can result in flatter slope requirements than in conventional landfills, therefore making less available space for landfilling.

- <u>Injection systems</u>. The efficiency of treated leachate distribution and waste moisture absorption may vary with the device used to inject liquids into the waste mass. Some of the methods used in full-scale bioreactor landfills can also be used in facultative landfills, such as vertical injection wells and horizontal infiltration devices. A combination of methods can also be used for good distribution in all parts of the landfill.

- <u>Plastic bags</u>. The utilization of plastic bags by households to dispose their domestic waste sometimes can “seal” the waste from receiving the beneficial effects of treated leachate addition.

- Energy recovery. Since lower quantities of methane are generated in the facultative landfill, the gases cannot be used to recover energy.

- Costs. It is necessary to evaluate the costs (including monitoring costs) comparing conventional landfills to landfills with leachate recirculation and to facultative landfills. The cost of the benefits of having a cleaner technology, while minimizing liability concerns and risks to the environment, must also be taken into consideration.

Another relatively new technology in waste management is the reclamation of landfills (the recovery and reuse of components of waste after stabilization). The reclamation of landfills can be understood as the natural extension of bioreactor landfills. In facultative landfills, the stabilization periods are reduced and, therefore, the recovery of not only recyclable materials, but also valuable landfill space, can be accomplished more quickly.

7.1.3. Suggestions for Additional Studies

Some of the additional studies suggested as an extension of this study and necessary for further development of the technique are:

- Verification of the influence of the recirculation rate of treated leachate in the waste stabilization rates.

- Verification of the influence of other parameters, such as waste composition, and waste permeability (or density) in the waste stabilization rates in facultative landfills.

- Evaluation of techniques for distribution of leachate. In addition, numerical models for moisture transport prediction can also be studied.

- Evaluation of pore-pressures and MSW strength in the design of facultative landfills.

- Study of the economic feasibility of facultative landfills.

- Quantitative study of the parameters of the leachate treatment plant, including retention time, and the oxygen added (flow and pressure).

7.2. Geotechnical Study

7.2.1. Conclusions

Based on the results obtained and analysis effectuated for the study of MSW compressibility, some conclusions can be drawn:

- The use of classical Soil Mechanics formulation to describe the MSW compressibility is not adequate, primarily due to the fact that the settlement mechanisms between soils and MSW are different. In addition, even when the model is understood as an empirically adjusted model, it still presents several deficiencies.

- A division of MSW settlements into two categories, immediate and time-dependent, is more understandable than the traditional division of primary and secondary compressions.

- The application of load to the waste inside the lysimeters produced immediate relative settlements (or strains) of 15.2%, 10.3%, and 8.0% for the facultative lysimeter, for the lysimeter with raw leachate recirculation, and for the conventional lysimeter, respectively. The same load was applied to the three lysimeters.

- Not much difference was noticed among the lysimeters for the time-dependent settlement at the end of the study. Relative settlements of 12.5%, 14.8%, and 13.3% were observed for the facultative lysimeter, for the lysimeter with raw leachate recirculation, and for the conventional lysimeter, respectively.

- It was noticed that both the immediate and the time-dependent settlements can be modeled according to a hyperbolic law. The MSW time-dependent settlement was observed to follow a hyperbolic function by Ling et al. (1998). Using compressibility tests performed in this study and in other studies, it was noticed that the immediate settlement also can be modeled as a hyperbolic function on pressure.

- A Rheological Hyperbolic model was proposed to model and predict the settlements on landfills. The model can be understood by the representation of the immediate settlement by a spring and the time-dependent settlement by a dashpot and another spring, both linked in parallel. Both springs are non-linear.

- Although the use of the Rheological Hyperbolic model is promising due to the several benefits and verifications herein presented, more studies are necessary to adequately calibrate the parameters associated with the time-dependent settlements to make it applicable to full-scale situations.

7.2.2. Applicability of the Study to Full-Scale Landfills

The successful application of the Rheological Hyperbolic model to full-scale landfills is dependent on two factors, as presented below:

- Use of adequate model parameters corresponding to the characteristics of the landfill.

- The model considers that a uniform, infinite load is applied to the top of the waste. This consideration is valid for the center parts of a landfill, but not for the edges. In the case of the application of finite loads, solutions such as the Boussinesq theory can be coupled to the model. However, it must be noticed that the application of the Boussinesq theory to MSW is questionable since it is based on the theory of elasticity.

7.2.3. Suggestions for Additional Studies

- Calibration of the immediate settlement parameters and evaluation of the influence of waste composition, initial unit weight, and moisture content that can affect their values.

- Calibration of the time-dependent settlement parameters and evaluation of the influence of loading, waste composition, initial unit weight, and moisture content that can affect their values. The tests should be conducted with lysimeters or pilot-scale cells over a period of time that exceeds one year.

- Verification of the influence of biodegradation in the development of time-dependent settlements, according to several waste compositions.

- Verification of the influence of leachate and gas pressures in the model.

- Development of software or analytic solutions to apply the model to successive layers.

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APPENDIX A

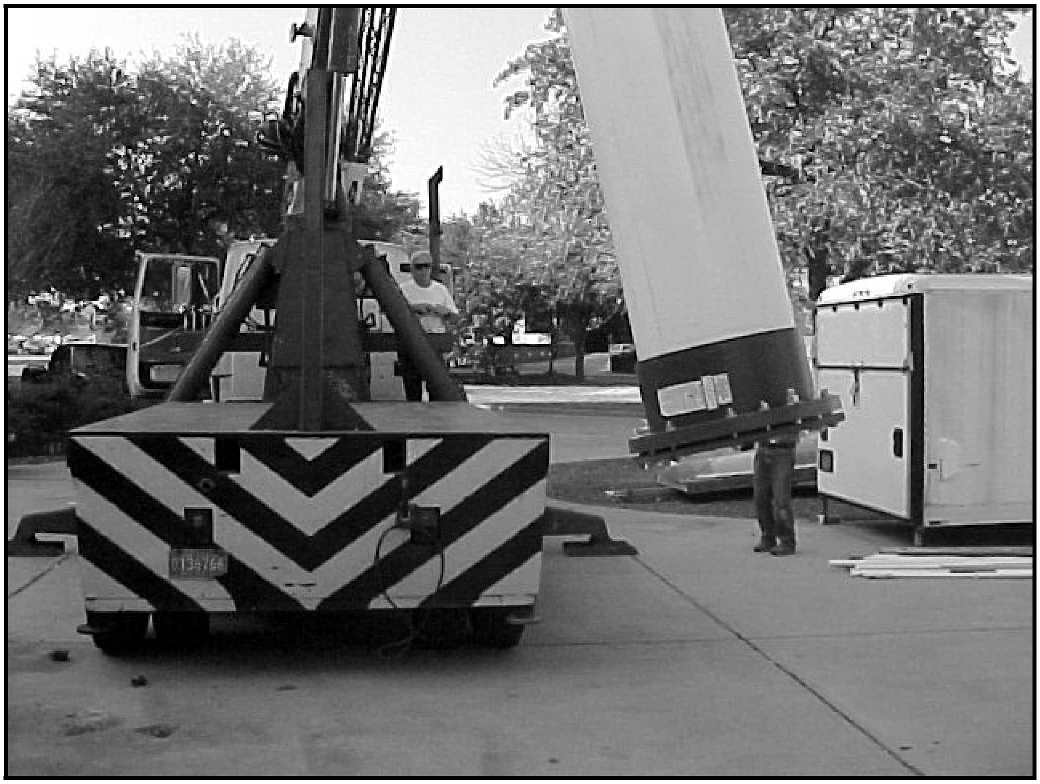
PICTURES OF PILOT PLANT CONSTRUCTION AND SET-UP

Picture 1 – Lysimeters at Pilot Plant

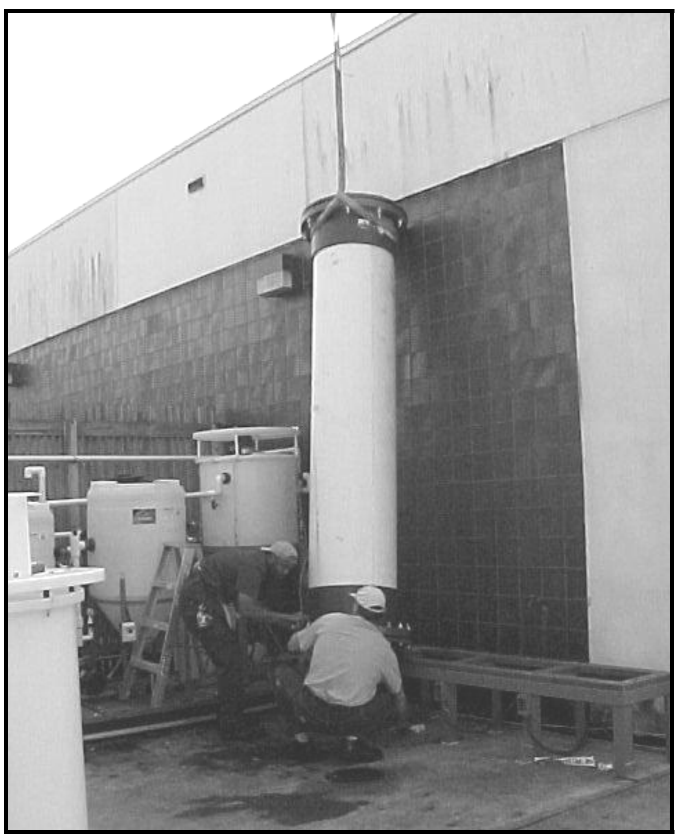
Picture 2 – Leachate Treatment Plant

Picture 3 – Plant Parts Prior to Shipping

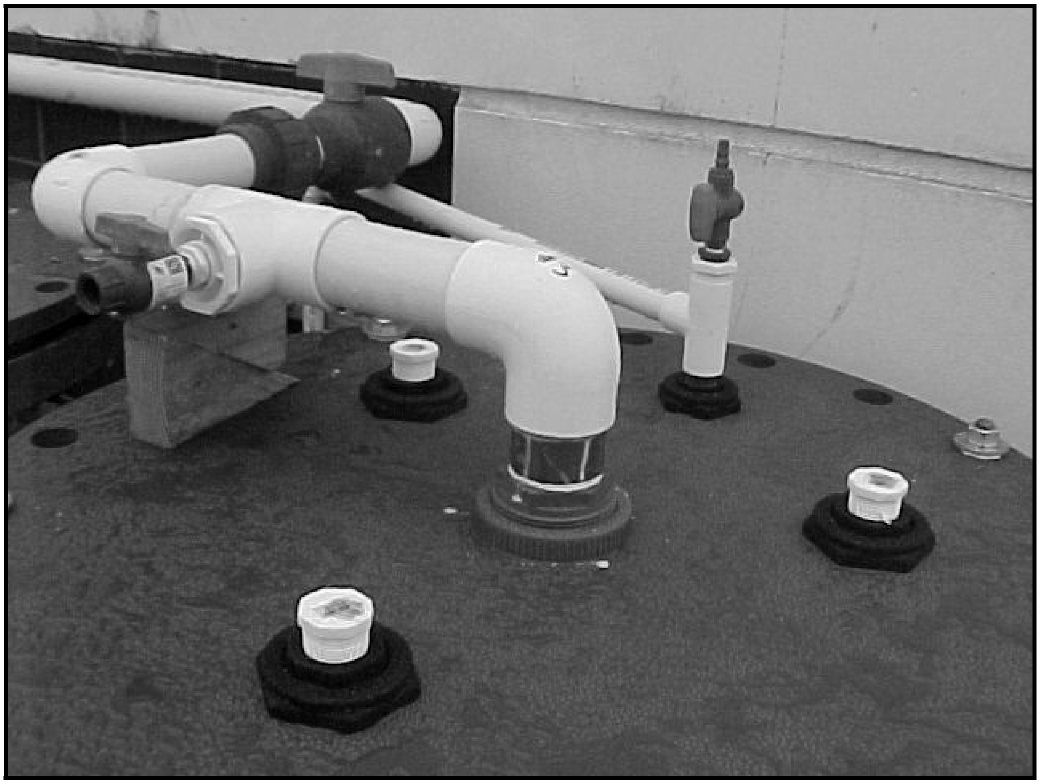
Picture 4 – Lysimeters Prior to Assembly at Pilot Plant



Picture 5 – Moving Lysimeter to Final Position with a Crane-Truck



Picture 6 – Placing Lysimeter at Final Position



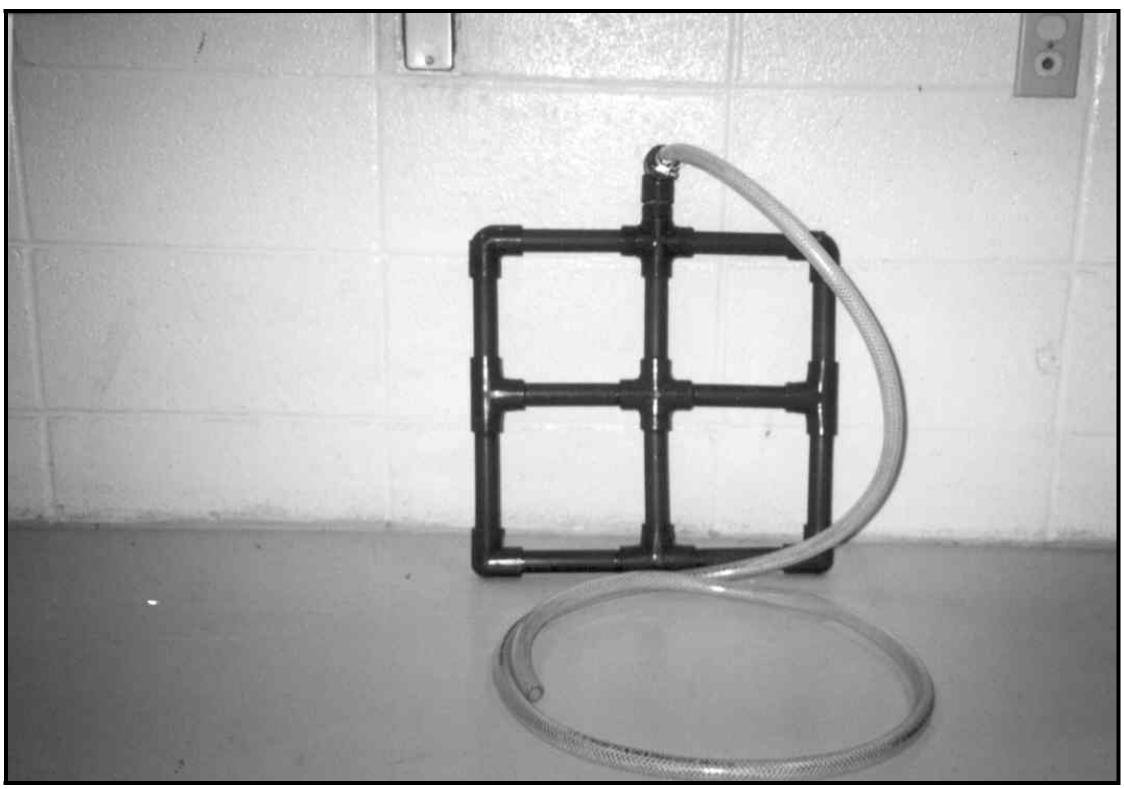
Picture 7 – Top Cover of the Lysimeter

Picture 8 – Arrangement for Filling Lysimeters

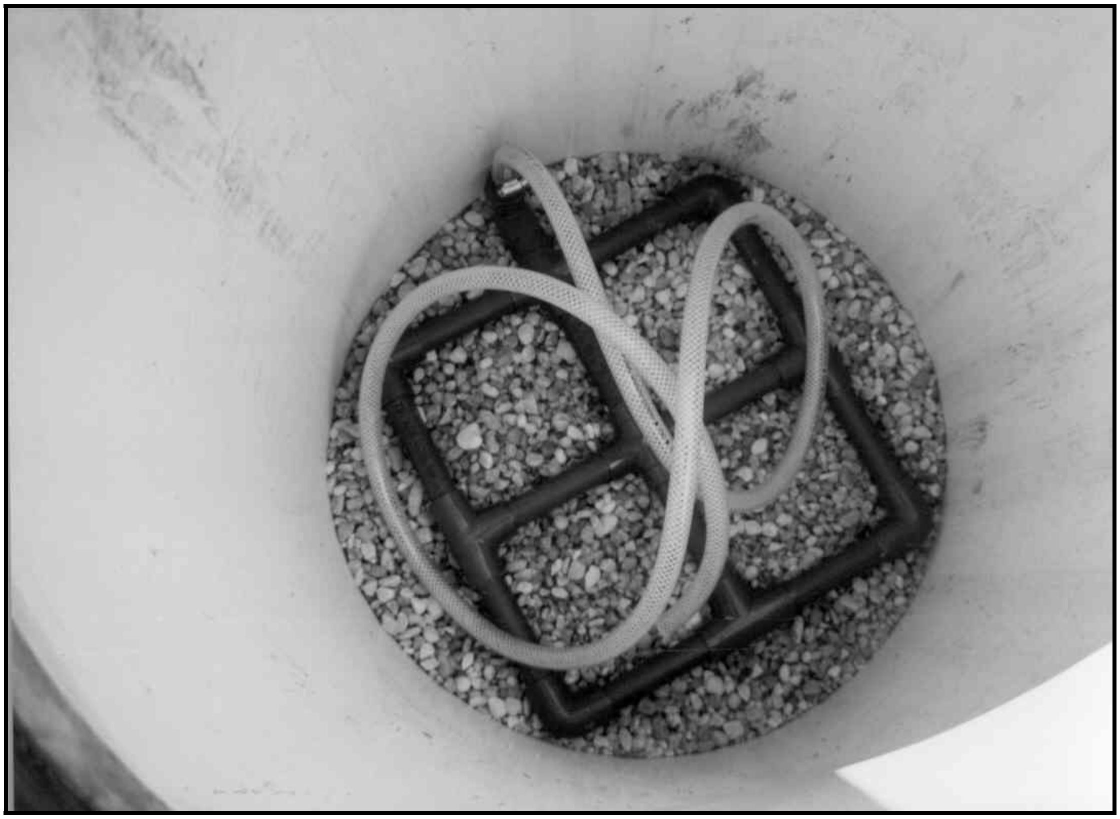
Picture 9 – Waste Bags Containing Synthetic MSW

Picture 10 – Filling Lysimeter with Synthetic MSW

Picture 11 – Synthetic MSW Inside Lysimeter after Manual Compaction

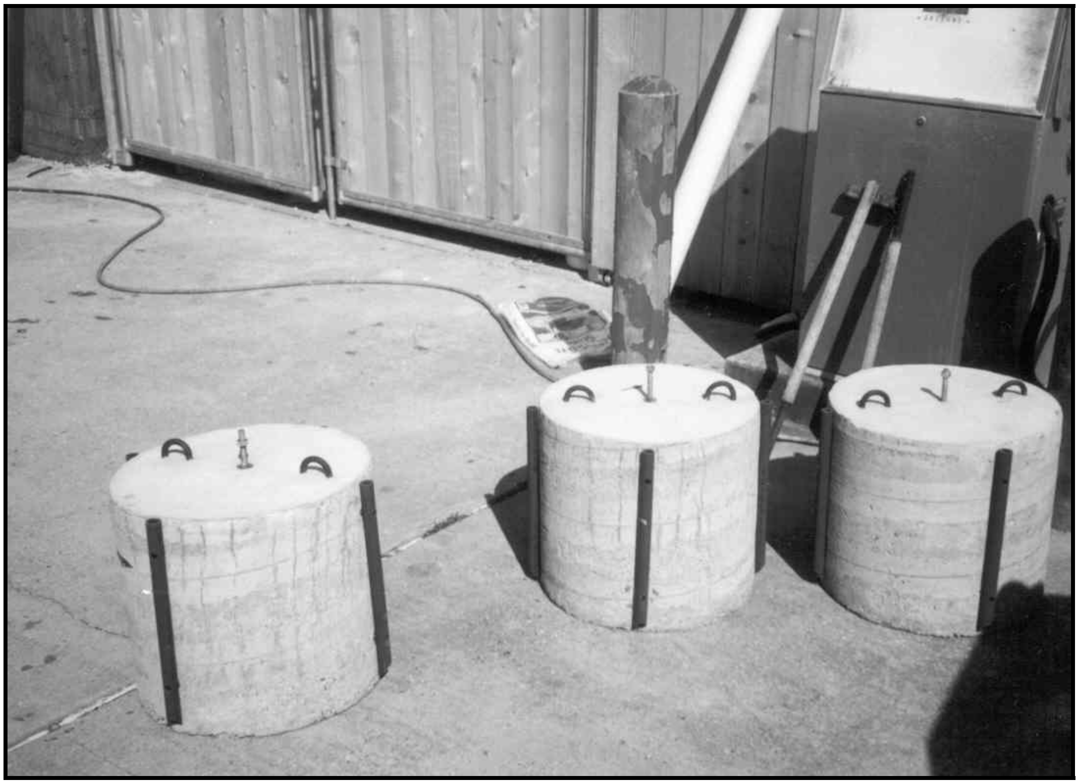


Picture 12 – PVC Manifold for Recirculation/Rainfall Simulation



Picture 13 – PVC Manifold for Recirculation/Rainfall Simulation Inside Lysimeter

Picture 14 – Lysimeter Prepared to Receive Concrete Weight (after final pea-gravel layer was placed in)

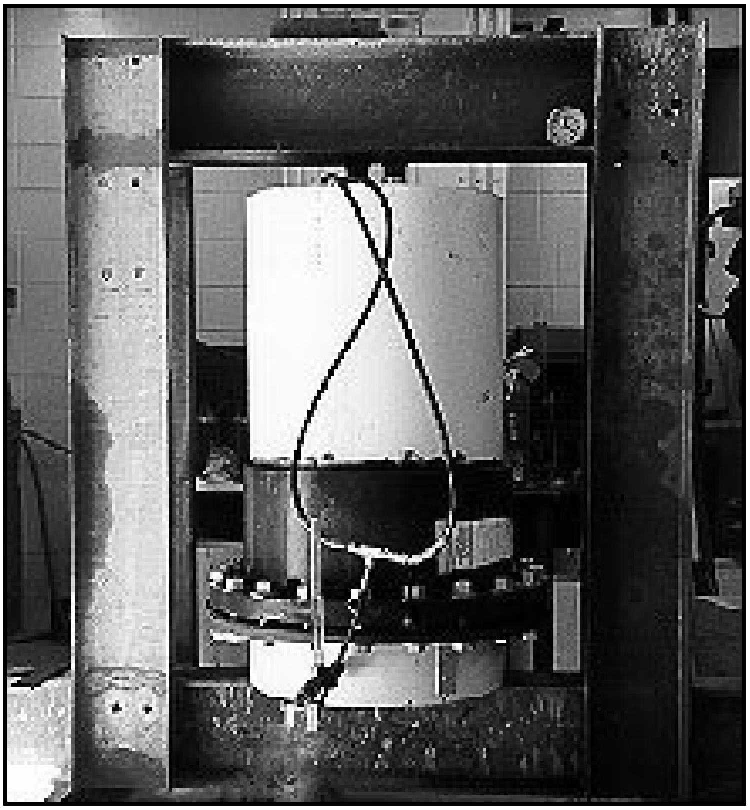


Picture 15 – Concrete Weights

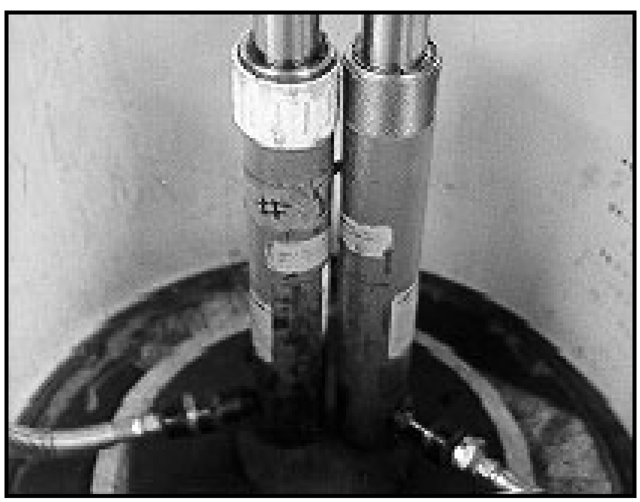
Picture 16 – Placing Concrete Weight in a Lysimeter

APPENDIX B

PICTURES OF THE COMPRESSIBILITY TESTS



Picture 1 – Test Chamber Sited on Frame



Picture 2 – Loading Jacks Inside Chamber

Picture 3 – Unloading Test Chamber

VITA

Mr. Ricardo Coelho de Abreu was born in 1970 and grew up in Sao Paulo, Brazil. He received a Bachelor of Science in Civil Engineering in 1994 from the Universidade de Sao Paulo, in Sao Paulo, Brazil. In 2000, Mr. de Abreu received a Master in Civil Engineering also from the Universidade de Sao Paulo with a thesis entitled “Compressibility of Waste Landfills”. From 1992 to 2001, he was employed as an Engineering Intern, Staff Engineer, and Project Engineer in three Brazilian geotechnical companies: Vector Projetos Integrados, Do Val & Silveira Engenharia Consultiva, and Cepollina Engenheiros Consultores, all located in Sao Paulo, Brazil. In the fall of 2001, he began his doctoral studies in civil engineering at the University of New Orleans. After graduation, Mr. de Abreu will work as a Geotechnical Engineer for Soil Testing Engineers, Inc., in Baton Rouge, Louisiana. He is married to Reni A. de Abreu.

DOCTORAL DISSERTATION REPORT

CANDIDATE: Ricardo C. de Abreu

PROGRAM OF STUDY: Engineering and Applied Sciences

TITLE OF DISSERTATION: Facultative Bioreactor Landfill: An Environmental and Geotechnical Study

APPROVED:

Kenneth L. McManis

Major Professor & Chair

Robert C. Cashner

Dean of the Graduate School

EXAMINATION COMMITTEE:

Dr. Gordon Boutwell

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Dr. Mysore Nataraj

Dr. Ronald Stoessel

DATE OF EXAMINATION: July 7, 2003