IMPACTS OF SEWAGE SLUDGE CODISPOSAL ON WASTE DEGREDATION IN AEROBIC AND ANAEROBIC BIOREACTORS

by

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ABSTRACT

Due to economical, endustrial and technological developments, waste and wastewater sludge productions are increasing gradually all over the world. This situation constitutes to the problem of sustainable solid waste and sludge disposal. Especially diffuculties in finding new landfill spaces, high cost of wastewater sludge management and environmental concerns are forced to find a new method for disposal. Although there are many methods such as incineration, surface disposal and composting, codisposing of wastewater sludge with MSW in the aerobic and anaerobic bioreactors are most used and economical way.

Therefore, the objective of this study was to obtain information about sludge codisposal with MSW in the aerobic and anaerobic bioreactors in order to evaluate and determine an efficient and cost-effective landfill management system by utilizing two stage laboratory experiments. These stages include, the lab-scale reactor set-up in order to determine optimum solid waste to sludge ratios by using anaerobic bioreactors and then pilot scale reactor set-up in order to understand the effect of sludge addition on waste stabilization process by using aerobic and anaerobic bioreactors.

The results of this study showed that in the lab-scale reactors, codisposal of wastewater sludges with MSW is a very promising method. This method accelerates waste stabilization rates in the anaerobic bioreactors while the energy potential of the sludge is used beneficial. In the pilot scale reactors this study showed that, aeration is a feasible way to treat leachate, fastens stabilization rate and provides rapid removal of organics and nitrogen. On the other hand, CH_4 production in the anaerobic was more promising.

ÖZET

Dünya üzerindeki katı atık ve atık su arıtma tesisi çamuru üretimi ekonomik, endüstriyel ve teknolojik gelişmelere bağlı olarak gitgide artmaktadır. Bu durum, sürdürülebilir katı atık ve çamur depolama problemini oluşturmaktadır. Özellikle, düzenli depolama alanı bulma sorunları, atık su arıtma tesisi çamurunun işleme maliyetinin yüksek olması ve çevresel endişeler yeni bir bertaraf metodu bulma zorunluluğunu ortaya koymuştur. Artıma çamurlarının bertarafı için yakma, yüzeysel depolama ve kompostlama gibi birçok alternatif olmasına rağmen, arıtma çamurlarının katı atıklarla birlikte aerobik ve anaerobik biyoreaktörlerde birleşik tasfiyesi en çok kullanılan ve en ekonomik metotdur.

Bundan dolayı bu çalışma, verimli ve uygun maliyetli bir depolama sistemi belirlemek ve değerlendirmek için iki aşamalı laboratuvar deneyleri yapılarak, çamurların katı atıklarla birlikte anaerobik ve aerobik biyoreaktörlerde bileşik tasfiyesi ile ilgili bilgi edinmeyi amaçlamaktadır. Bu aşamalar optimum katı atık ve arıtma çamuru oranını belirlemek için anaerobik biyoreaktörlerin kullanıldığı laboratuvar ölçekli ve sonrasında çamur eklentisinin katı atık stabilizasyon prosesi üzerindeki etkilerini belirlemek için aerobik ve anaerobik biyoreaktörlerin kullanıldığı pilot ölçekli aşamaları içermektedir.

Çalışmanın sonucunda, laboratuvar ölçekli reaktörlerde çamurların katı atıklarla birlikte bileşik tasfiyesinin umut vaadeden bir metot olduğu görülmüştür. Bu metot, anaerobik biyoreaktörlerde katı atık stabilizasyonunu hızlandırırken, çamurun enerji potansiyeli faydalı bir şekilde kullanılabilmektedir. Pilot ölçekli reaktörlerde ise, havalandırmanın sızıntı suyunu işlemede uygun bir metod olduğunu, stabilizasyonu hızlandırdığını ve organik ve nitrojenlerin hızlıca giderimini sağladığını ortaya koymaktadır. Bir diğer taraftan anaerobik biyoreaktörlerde CH_4 üretiminin daha verimli bir metot olduğu gözlemlenmiştir.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF SYMBOLS/ABBREVIATIONS

Symbol	**Explanation**	**Units**
MSW	Municipal Solid Waste	
CH_4	Methane	(%)
CO_2	Carbon dioxide	(%)
O_2	Oxygen	(%)
COD	Chemical Oxygen Demand	(mg/L)
VFA	Volatile Fatty Acids	(mg/L)
TKN	Total Kjeldahl Nitrogen	(mg/L-N)
NH_3-N	Ammonia Nitrogen	(mg/L)
C/N	Carbon /Nitrogen	

1. INTRODUCTION

The progress of civilization and the increase in population around the world have contributed significantly to the increase in the quantity of waste and wastewater sludge generated. This situation leads to a rise in the cost of treatment and disposal of these wastes and sludges. Besides, huge amounts of municipal solid waste (MSW) and wastewater sludge from industrial to domestic activities can pose significant threats to human health and environment. When the problems from the open dump sites are combined to these, revolutionary landfill management systems are becoming more attractive, such as bioreactor landfills.

Bioreactor landfills can provide a more controlled system. They can reduce the emission of greenhouse gases, and additionally they can provide immediate improvements to the surrounding local environment in terms of controlling odor and methane gas migration (Alvarez et al., 2000). Besides, bioreactor landfill concept increases potential for waste to energy conversion, recovers air space, and ensure sustainability and stabilization (Warith, 2002).

Like many other wastes, biosolids which is known as sewage sludge from wastewater treatment plants creates problems of disposal. Throughout the world, due to the industrial revolution, the production of sludge is gradually increasing. However, the disposal solution has not been effectively solved yet. Current disposal practices include land application, landfilling, surface disposal, incineration, and composting. Landfilling of sludge has significant advantages such as easier handling of sewage sludge, acceleration of waste stabilization rates, smaller disposal/treatment area requirements and lower capital investment when compared to other disposal techniques.

Combined disposal (codisposal) of MSW and sewage sludge within a single facility is a very promising and cost saving disposal method. Codisposal of MSW with the sludge offers many opportunities in bioreactor landfills. This can be the inexpensively and effectively solution of the problem of incremental sludge production and MSW stabilization. Besides, the energy potential of the sludge can be used beneficially in bioreactor landfills. The pathogens and high nitrogen content of sludge can be reduced by the codisposal of sludge with solid wastes while minimizing the potential problems by proper stabilization (Güneş, 2005).

Therefore, in this study, the wastewater treatment plant sludge cake and anaerobically digested sludge were directly codisposed with solid waste to investigate an alternative solution for sludge disposal by utilizing two stage laboratory experiments. In the first stage, the lab-scale reactor set-up was used; in order to determine optimum solid waste to sludge ratios. This step enables to find the most promising mixing ratio for optimum methane formation rate and stabilization process. The second step was the pilot scale bioreactor set-up. In this stage, anaerobic versus aerobic simulated bioreactors were used in order to understand the effect of sludge addition on waste stabilization process. In other words, this step's aim is to find an alternative sludge disposal option which accelerates the waste stabilization rate.

For this purpose, in the lab-scale reactor step, four 10L plexi-glass reactors were constructed and operated at 35 ^{o}C constant temperature. 1.5 kg synthetically prepared wastes and different amount of sludge cakes were codisposed together. The composition of solid waste that was loaded to each reactor was determined according to average values of solid waste composition for Istanbul region. In the pilot scale bioreactor step, 1 aerobic, 1 anaerobic with a volume of 100L bioreactors were used. The most promising result obtained from lab-scale reactor step was used in order to loading the pilot scale bioreactor step.

2. LITERATURE REVIEW

Literature review section has been divided into three main sections: (1) Modern landfill management systems, (2) Landfill stabilization and factors affecting landfill stabilization (3) Codisposal of solid waste and sludge.

2.1. Modern Landfill Management Systems

Due to increasing population, health and environmental problems and industrial revolution, effective solid waste management has become a very important issue around the world. Solid wastes are the end product of economic industrial and social activities. They have negative impacts on environment (soil, water, air pollution) and threaten human, animal and vegetative health. When these problems combined together with the improper waste disposal techniques such as open dump sites, "Integrated Solid Waste Management" systems has emerged (Erses, 2008).

The most effective part of the "Integrated Solid Waste Management Hierarchy" is the minimization of the solid waste generated; in other words, source reduction. This continues with, recycling, energy recovering and disposal. The lowest rank of the system is landfilling (disposal). However, it is most popular due to the economic considerations (Tchobanoglous et al., 1993).

According to Tchobanoglous et al. (1993), "Landfills are the physical facilities used for the disposal of residual solid wastes in the surface soils of the earth." (Tchobanoglous et al., 1993). Sanitary landfill term is used to describe the landfills as an engineering facility for the disposal of municipal solid waste and minimize their negative impacts. Today's modern sanitary landfills have some collection systems that collect biogas and leachate. In order to eliminate the potential environmental risk of leachate and gas production, two management systems are used in sanitary landfills, conventional and bioreactor landfill operation (Erses et al., 2008).

Figure 2.1. Waste Management Hierarchy (EPA, 2012).

2.1.1. Conventional Landfills

Sanitary landfills are generally operated by conventional techniques, where anaerobic conditions are created within the landfill (Erses et al., 2008). This ends up with several negative effects, such as slow stabilization rate, unstable methane gas production and uncontrolled of leachate generation which can pollute groundwater.

A conventional landfill site is an engineered waste disposal facility where solid waste is deposited in the ground. To decrease the environmental risks garbage is disposed into the cells and covered with earth fill materials. Environmental controls are unified into the engineering design of the facility to protect the human and natural environments (Warith et al., 2003). According to some researches, long term negative environmental impacts of conventional landfills can proceed centuries because of a significant portion of the solid waste remains unstabilized in conventional landfills (Erses et al., 2008, Kruempelbeck and Ehrig, 1999).

2.1.2. Bioreactor Landfills

Bioreactor landfill systems are a modification of conventional landfills with a controlled moisture addition. The aim of developing these systems is to minimize environmental impacts through optimizing waste degradation (Erses et al., 2008). Also, bioreactor landfills are operated in a controlled fashion where an in situ environment forms up and has a positive effect on microbial degradation of waste (Berge et al., 2008). The bioreactor landfill significantly increases the extent of organic waste decomposition, conversion rates and process effectiveness.

Generally, there are four advantages for employing bioreactor landfill technology comparing to conventional landfills: (1) contain and treat leachate, (2) rapidly recover air space, (3) accelerate waste stabilization and avoid long-term monitoring and maintenance and delay siting of a new landfill, and (4) make more potential benefits from increased methane generation in anaerobic bioreactor landfill (Warith, 2005). A comprasion schematic of bioreactor landfill and conventional landfill is given Figure 2.2.

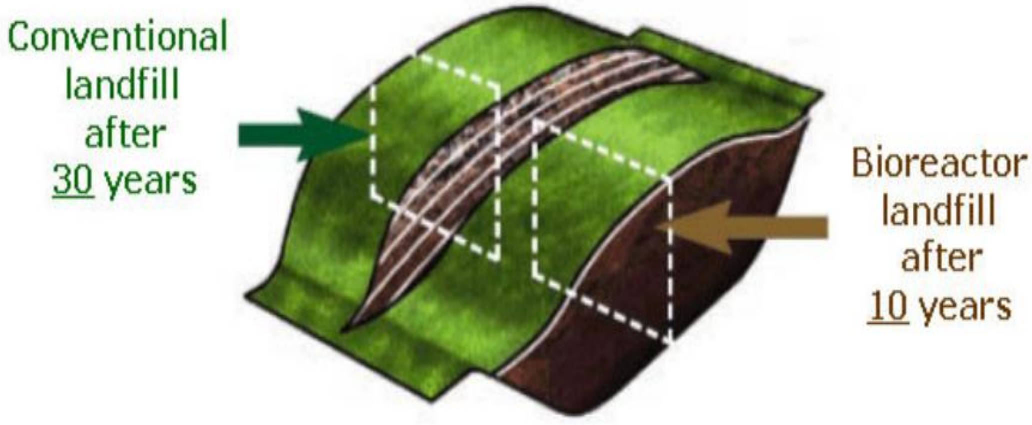


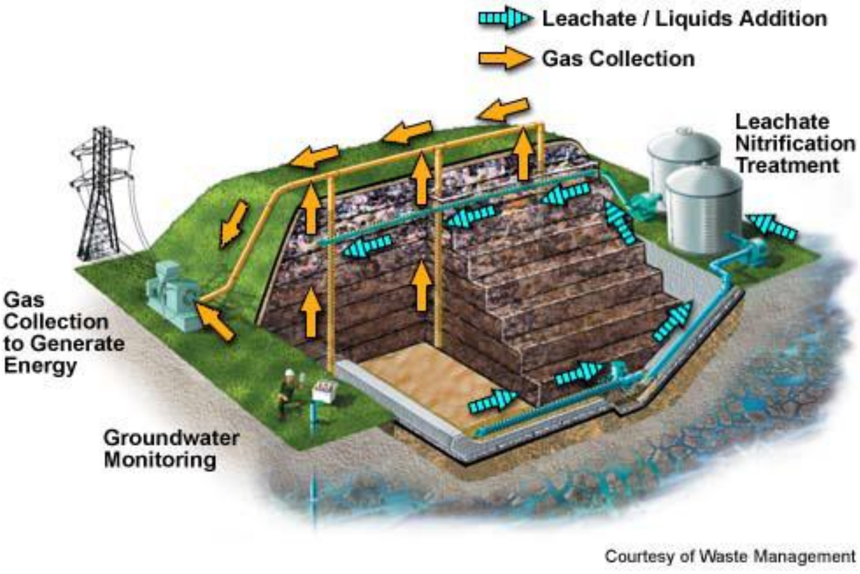
Figure 2.2. Comparison between conventional and bioreactor landfills (Warith, 2003).

The addition of moisture (leachate recirculation) to the bioreactor landfills has several advantages such as, enhancing the stabilization of waste and generation of landfill gas, uniform distribution of nutrients and microorganisms, pH buffering, dilution of inhibitory compounds, liquid storage, minimizing environmental impacts and operating cost

(Reinhart and Townsend, 1998). There are several researches concluded these results (Pohland, 1980; Townsend et al., 1996; El-Fadel et al., 1999; Onay and Pohland, 1998; San and Onay, 2001; Arrpet, 2004; Erses et al., 2008; Khatib, 2010).

While a variety of operational approaches have been tested to accelerate waste degradation, in general bioreactors can be categorized into three types or groups: anaerobic bioreactors, aerobic bioreactors, and hybrid bioreactors.

2.1.2.1. Anaerobic bioreactor landfill. Bioreactors are generally operated under anaerobic conditions where anaerobic microorganisms responsible for waste degradation. These microorganisms produce landfill gas (LFG). Landfill gas consists of primarily methane, and carbon-dioxide. These gases utilize for energy generation. In an anaerobic bioreactor landfill, recirculated leachate is used for moisture addition to obtain optimal moisture levels (Tchobanoglous et al., 1993). A representative figure of an anaerobic reactor is given in Figure 2.3.



Courtesy of Waste Management

Figure 2.3. The design and operational features of anaerobic bioreactors (Warith, 2003).

2.1.2.2. Aerobic bioreactor landfill. The addition of air to landfills for aerobic bioreactors is an innovative technology. Air is injected using vertical or horizontal wells, to promote aerobic activity and accelerate aerobic waste stabilization (Warith, 2003). Into the aerobic conditions, waste stabilizes more rapidly than anaerobic conditions. Besides, the methane gas generation and leachate production are reduced.

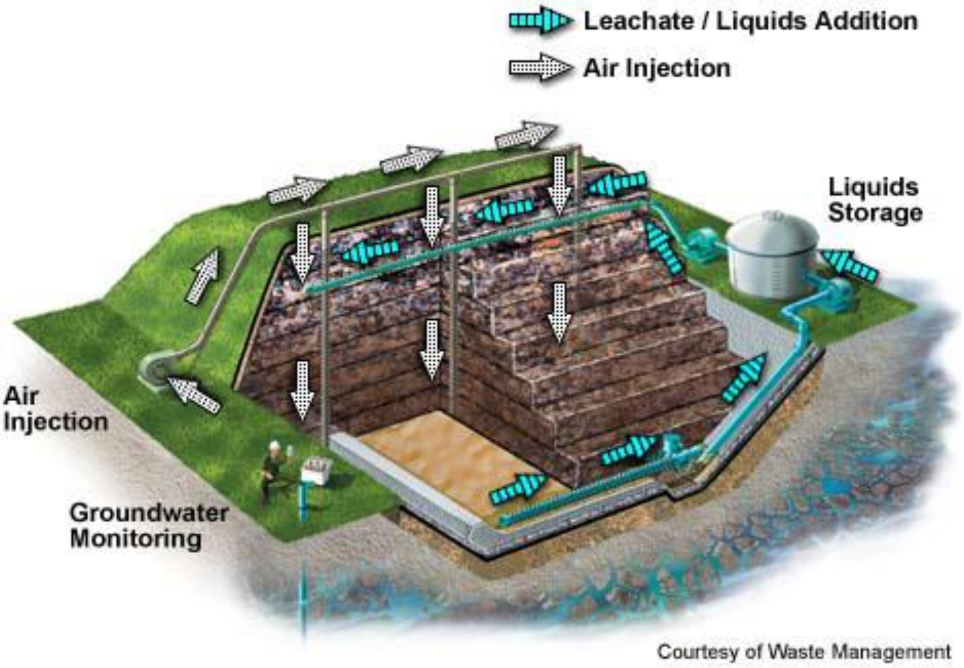


Figure 2.4. The design and operational features of aerobic bioreactors (Warith, 2003).

2.1.2.3. Hybrid (Aerobic-Anaerobic) bioreactor landfill. In the hybrid bioreactor landfill concept aerobic-anaerobic treatment is used together to rapidly degrade organics in the upper sections of the landfill. Methanogenesis in hybrid reactors starts earlier compared to aerobic landfills. Hybrid bioreactor landfill concept is known as Fukuoka Method in Japan. The air injected to the system accomplished by natural ventilation where temperature differences are used. Biodegradation and stabilization of hybrid reactors are faster than anaerobic bioreactors and they reduce the greenhouse gas emissions. Hybrid bioreactor landfill concept has received much more attention due to cost effectiveness (International Regions Benchmarking Consortium, 2011).

2.2. Landfill Stabilization

Waste degradation in landfills is the combination of chemical, physical and biological processes. In the landfill environment, these processes are: biological decomposition of organic materials by either aerobic or anaerobic processes, chemical oxidation of waste compounds, transport of gases, liquid hydraulic transport, dissolution and transport of organic and inorganic constituents, and uneven settlement (Pohland, 1993).

Waste stabilization has been defined as finding a balance between the residual contaminant load and their decreasing impacts on the environment (Erses, 2008). At the end of the stabilization or waste degradation process generally, gaseous products and leachate produces.

2.2.1. Microbiology of Anaerobic Landfill Stabilization

In the waste stabilization process several groups of bacteria take place. The balance among all microorganism groups is very essential and important factor for the efficient waste stabilization. In anaerobic landfill areas, decomposition begins with aerobic degredation. In aerobic step, microorganisms convert the organic material to carbon dioxide, water and energy. After this step anaerobic conditions occurs in the landfill area (Blackall and Silvey, 1994).

During the anaerobic conditions, organic fraction of wastes are converted to methane and carbon dioxide. This conversion includes a four-stage process; hydrolysis, acidogenesis, acetogenesis and methanogenesis (Tchobanoglous et al., 1993). Figure 2.5 illustrates stages of anaerobic digestion.

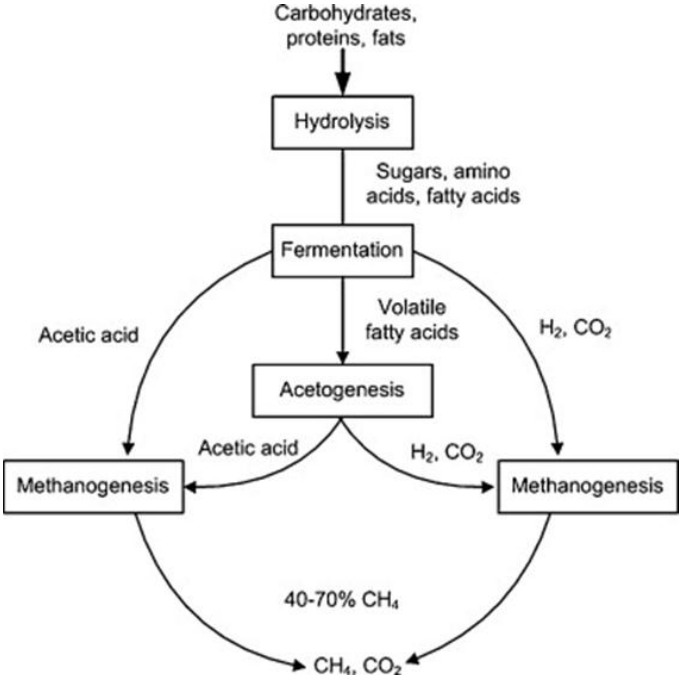


Figure 2.5. Anaerobic pathway (Li et al., 2011).

The hydrolysis is the first stage of anaerobic digestion. At this stage, complex insoluble compounds are hydrolyzed to dissolved organics, primarily sugars, alcohols, aminoacids and higher fatty acids by extracellular enzymes. Hydrolysis is the rate limiting step of anaerobic conversion of the solid waste (Demirel, 2009).

After the hydrolysis, fermentation step starts. At this stage, hydrolyzed soluble organic compounds are fermented by acid forming bacteria into volatile organic acids, carbon dioxide and hydrogen gas (Demirel, 2009).

The third stage is the oxidation step of alcohols and volatile acids to acetic acid, carbon dioxide and hydrogen. This stage is accomplished by obligate hydrogen (H_2) producing acetogenic bacteria (Christensen et al., 1989).

In the final step, acetate and hydrogen are converted to methane and carbon dioxide. It is estimated that 70% of the methane is produced from acetate (Barlaz et al., 1990).

2.2.2. Microbiology of Aerobic Landfill Stabilization

Aerobic digestion of waste is the natural biological degradation and purification process. In this process, bacteria break down and digest the waste in oxygen-rich environments. During oxidation process, pollutants are broken down into carbon dioxide (CO_2), water (H_2O), nitrates, sulphates and biomass. The microorganisms which can only survive in aerobic (presence of oxygen) conditions are known as aerobic organisms (Water/Wastewater Distance Learning Home Page, 2000).

2.2.3. Phases of Anaerobic Landfill Stabilization

There are five phases in the anaerobic landfill stabilization (Pohland, 1993). These phases are; initial adjustment, transition, acid formation, methane fermentation and final maturation. Each phase, characterized by the quality and quantity of leachate and landfill gas produced, and show the change in the microbial processes within the landfill. Figure 2.6 illustrates these phases.

Initial Adjustment Phase: this phase begins after the placement of the wastes into the landfill. Due to certain amount of air or oxygen present in the refuse, wastes are decomposed under aerobic conditions. Low moisture content limits the microbial activity in this phase (Tchobanoglous et al., 1993).

Transition Phase: this phase starts after oxygen consumption; after the anaerobic conditions starts. In this phase first leachate generation is observed and nitrate and sulfate can be used as electron acceptors. Chemical reducing conditions starts at this stage (Tchobanoglous et al., 1993).

Acid Formation Phase: the major gas product of this stage is carbon dioxide. In this stage the pH of the leachate drops because of the accumulation of volatile fatty acids and high concentration of carbon dioxide. Nutrients, nitrogen and phosphorus are released from the waste and support to biomass growth. (Pohland and Harper, 1986).

Methane Fermentation Phase: all of the products from other steps are converted to methane and carbon dioxide in this stage. The pH of leachate increases to neutral with the conversion of volatile organic acids. Oxidation-reduction potentials are at their lowest values. Ammonia nitrogen and heavy metal concentrations are reduced at this stage (Pohland and Harper, 1986).

Maturation Phase: the readily available organic compounds are converted to carbon dioxide and methane. Measurable gas productions minimized because of most of the nutrients are consumed during the methane fermentation phase. Heavy metal concentrations are still dropping in this phase (Tchobanoglous et al., 1993).

2.2.4. Phases of Aerobic Landfill Stabilization

Aerobic digestion is the most common process. By using oxygen aerators the process can be significantly aerated. The oxygen enhances the rapid biodegradation and stabilization in the aerobic landfill areas.

Aerobic treatment usually has better effluent quality than anaerobic processes. Also, because of the oxygen, it shortens monitoring time after closure. The aerobic pathway is shown in Figure 2.7. In aerobic degradation there is also, a small amount of energy releases (Pohland and Harper, 1986).

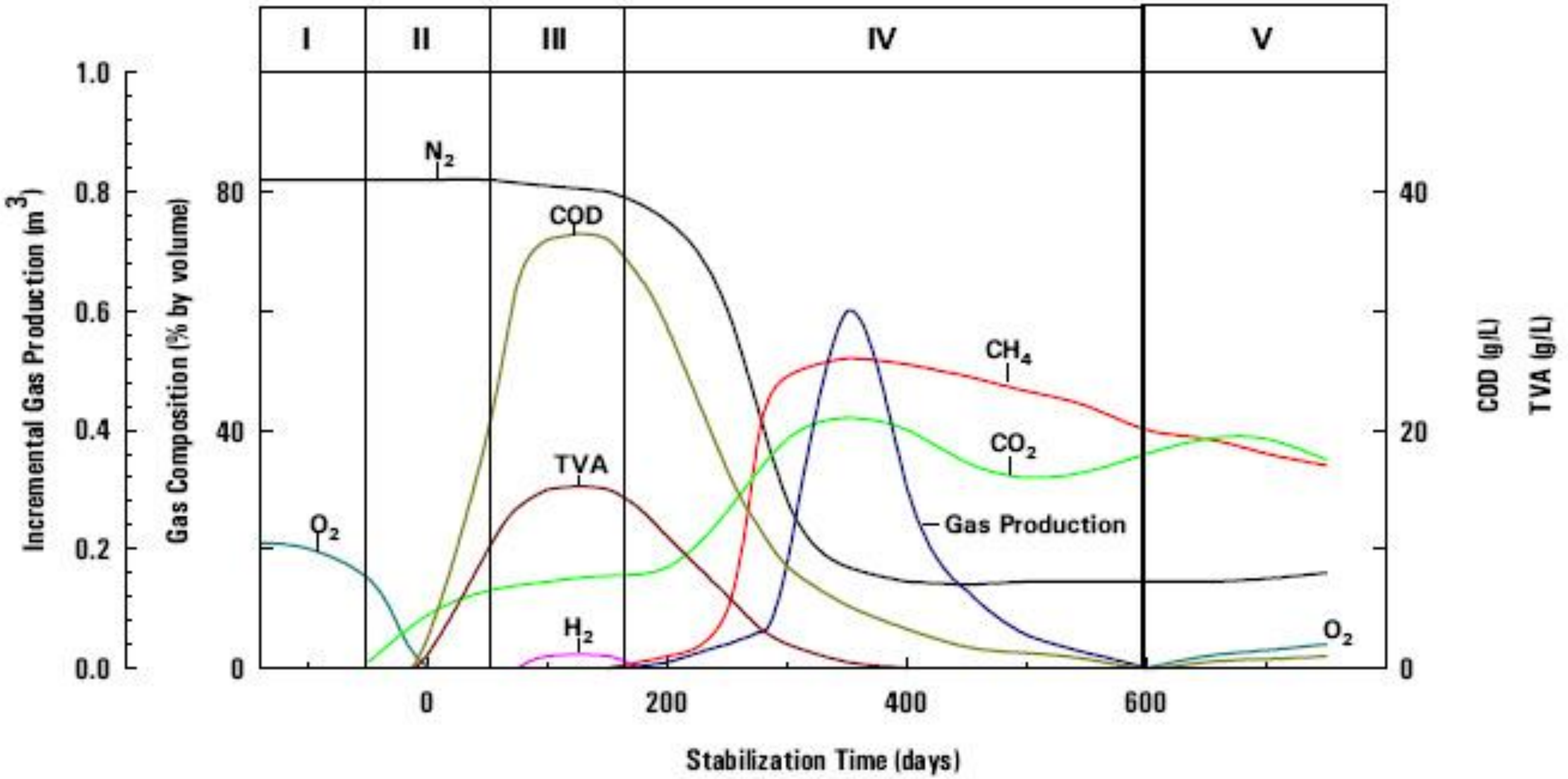


Figure 2.6. Waste stabilization phases (Onay, 1995).

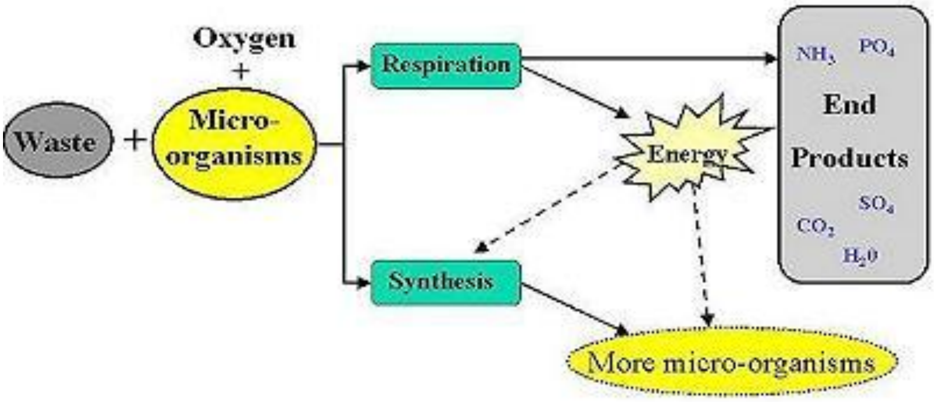


Figure 2.7. Aerobic pathway (Erses, 2008).

2.2.5. Enhancement of Anaerobic Landfill Stabilization

Stabilization process is a controlled decomposition of easily degradable organic matter. A complete stabilization process results in a significant reduction of volatile solids content, a change of an unpleasant smell, and reduction of pathogens (Taş, 2010). In order to have a complete stabilization process, it is very important to manage and operate landfill in a proper way. While the decomposition of wastes, in improperly managed landfills, there are seriously problems taking place, such as releasing of greenhouse and toxic gases and uncontrolled migration of leachate.

Stabilization of anaerobic landfill may take decade and the environment around landfills can be affected from the negative impacts of the landfill for a long time. These negative impacts can be solved by shortening the stabilization time by enhancement of anaerobic microbial processes. Therefore, enhancement of complete landfill stabilization is very important and depends upon the microbial processes. There are two reasons for enhancement of landfill stabilization. Firstly, when the organic compounds in leachate are converted to methane by microorganisms, the treatment of leachate becomes cheaper, easier and more environmental friendly. The other reason is the utilization of LFG which reduces environmental hazards and recovered for electricity generation as a renewable energy source (Meadows, 1995; Chan, 1999).

Leachate recirculation, pH control/buffer addition, nutrient addition, sludge addition, and compaction are some techniques that can promote the enhancement of anaerobic landfill stabilization (Güneş, 2005).

Leachate recirculation is the most popular technique used in the enhancement anaerobic landfill stabilization. When leachate is recirculated, biological, physical and chemical reactions will occur and stabilization becomes easier due to the more uniform environment. The recirculation of leachate provides various nutrients that are required for the growth of bacteria. Some researchers reported that, leachate recirculation increases the landfill’s moisture content (Warith et al., 2003; Vavilin et al., 2003; Kusch et al., 2009).

pH control is one of the most critical parameters in the anaerobic landfills because of the methagonic microorganism production. In the literature it is stated that, a combination of leachate recirculation and pH adjustment can minimize the inhibitory effects of acid accumulation (Vavilin, 2003). pH adjustment is achieved by buffer addition in anaerobic landfills.

2.2.6. Enhancement of Aerobic Landfill Stabilization

In aerobic landfills enhancement of stabilization can be achieved by the leachate recirculation and the addition of air to the system. Sufficient amount of moisture and air optimizes the waste degradation. As a result, aerobic operation provided significantly faster rates of waste stabilization (Stessel and Murphy, 1992).

In aerobic landfill stabilization, landfill aeration can provide short and long term avoidance of methane generation which is a greenhouse gas. Also, it reduces leachate pollution. Shortly, aeration contributes to the acceleration of organic matter degradation and to the decrease of pollutant emission into air, water, and soil. Recirculation of leachate enables to accelerate the processes occurring in waste (Slezak et al., 2010).

2.2.7. Factors Affecting Waste Stabilization in Landfills

2.2.7.1. pH and Alkalinity. pH is one of the main control parameter in anaerobic waste degradation. It effects the growth of microorganisms. Some researchers found that the optimum pH for anaerobic decomposition ranges between 6.4 and 7.6 (Anderson and Yang, 1992; Barlaz et al., 1990). The methanogenic bacteria are the most sensitive group to pH changes. Under the pH 6.2, the methanogenic bacterias cannot survive (Tchobanoglous et al., 1993).

In order to keep the pH level in desired range, addition of sufficient amount of alkalinity is very important. Alkalinity represents the ability of a system to buffer the effects of volatile and other acids. The total alkalinity of 1000-5000 mg/L as $CaCO_3$ is suitable for landfills (Tchobanoglous and Burton, 1979).

In aerobic decomposition pH is also very important. In the early stages of aerobic decomposition pH level drops due to volatile acid formation. Microorganisms use organic acids as a substrate at this stage. Aerobic reactions may occur in the pH range of 3-11, but better results are obtained in the pH range of 5-9 (Chefetz et al., 1996). Buffering does not necessary and it is not an important factor in aerobic landfills.

2.2.7.2. Temperature. Temperature is one of the most important parameter in all chemical and biological activities. In anaerobic digestion, the methane bacteria are very sensitive to temperature changes. Three temperature ranges are defined for anaerobic decomposition: psychrophilic (below 20 °C), mesophilic (20-40 °C) and thermophilic (50-70 °C) (Demirel, 2009).

In the literature, there are several researches about the optimum temperature in anaerobic bioreactors. Most of them found the optimum temperature for methane production within the range of 30 °C to 37 °C (Esteves, 1981). Also, it was reported that the methane production rate increased significantly at the temperature of 30 °C (Christensen and Kjeldsen, 1989).

Optimum temperature promote waste degradation in aerobic landfills. The optimum temperature range is 35 and 55 °C in aerobic landfills. But the temperature can raise up to 60 °C due to exotermic reactions (Güneş, 2005).

2.2.7.3. Moisture Content. In anaerobic digestion process, moisture content is one of the important limiting factors affecting the LFG production and waste degradation. Besides, moisture in the bioreactors provides a medium for transporting nutrients and dilute inhibitors (Tchobanoglous et al., 1993).

High moisture content (60 to 80%) favors methane production yields in anaerobic landfills (Emcon Associates, 1980). The lack of moisture can significantly limit gas production rate in landfills and also, the waste can remain without any decomposition for a long time (Esteves, 1981).

Moisture content also, is very important parameters in aerobic decomposition process. There is a very close relationship between aeration and moisture content in aerobic bioreactors. If the moisture content is too low, the landfill becomes dry and aerobic decomposition stops. Adversely, if the moisture content is too high, diffusion of oxygen becomes harder and it limits the microorganism's activities. It is determined that the optimum moisture content in the aerobic landfill is about 60% (Tchobanoglous et al., 1993).

2.2.7.4 Nutrients. In both aerobic and anaerobic systems, major nutrients are phosphorus and nitrogen. The anaerobic decomposition and gas production rates are dependent upon bacteria that require macronutrients (nitrogen and phosphorus) and micronutrients (iron, nickel, cobalt, calcium, selenium etc.) and some trace organics (Esteves, 1981).

The nutrient requirement of system is described by chemical oxygen demand:nitrogen:phosphorus (COD:N:P) ratio. The optimal ratio in the literature is 100:0.44:0.08 for anaerobic processes and phosphorous is the nutrient most likely limiting the decomposition (Christensen and Kjeldsen, 1989). On the other hand, in aerobic processes, the ratio of carbon to nitrogen should be in the range of 20:1-50:1 (Erses, 2008).

2.2.7.5 Inhibitors. The presence of toxic substances or inhibitors (high concentration of ammonia nitrogen, sulphides, heavy metals, toxic organic constituents, alkali and alkaline earth metals and excess volatile organic acids) in landfills negatively affects the waste stabilization.

In the anaerobic systems methanogens are very sensitive to the inhibitors. Especially ammonia which is a source of nitrogen is inhibitory at high concentrations. The inhibitory effects of ammonia are caused by free ammonia releasing. Its inhibitory effects have been observed at about 1500-2000 mg/L especially at high pH values. Besides, the concentrations above 3000 mg/L causes termination in gas production. Table 2.1 illustrates adverse effects of ammonia nitrogen concentrations.

Table 2.1. Adverse effects of ammonia N (Sacramento Regional County Sanitation District, 2008).

Ammonia N Concentration	**Effect on Anaerobic Process**
50-200 mg/L	Beneficial
200-1,000 mg/L	No adverse effect
1,500-3,000 mg/L	Inhibitory at higher pH values (7.4-7.6)
Above 3,000 mg/L	Toxic

Heavy metals are generally the constituents of industrial activities. Minimum amount of trace heavy metals are necessary for the microbial population. However, at higher concentrations it is inhibitory to the system.

Depending on the system pH, the heavy metals can be precipitated with sulfide, carbonate or hydroxide. Esteves (1981) reported the list of heavy metals according to the order of its decreasing toxicity; Ni>Ca>Pb>Cr>Zn. (Esteves, 1981). Inhibitory levels of some heavy metals to the microorganisms are given in the Table 2.2.

Table 2.2. Toxic concentrations of some heavy metals (Guneş, 2005).

Metal	**Toxic Concentration (mg/L)**
Copper	300
Nickel	800
Zinc	350
Chromium	200

Above limit concentration, alkali and alkaline earth metal salts such as Na, K, Ca and Mg have also adverse effects on landfills. Table 2.3 summarizes the inhibitory concentrations of earth metals.

Table 2.3. Stimulatory and inhibitory concentrations of the earth metals (Christensen, 1989).

Salts	**Inhibatory Concentrations (mg/L)**	
	Moderate	**Strong**
Sodium	3500-5500	8000
Potassium	2500-4500	12000
Calcium	2500-4500	80000
Magnesium	1000-1500	3000

In anaerobic reactors, sulfate is reduced to sulfide by the sulfate reducing bacteria. In anaerobic landfills, there are two different stages of inhibition exist as a result of sulfate reduction. First one is due to microorganism competition for common organic and inorganic substrates which has a negative effect of methane production. Second one is the inhibition resulting from the toxicity of sulfide to various bacteria groups (Chen, 2007). Sulfide is inhibitory at higher concentrations (1500 mg/L or higher) in anaerobic digestion process.

Volatile organic acids may also inhibit the methanogenic microbial growth. Propionic acid is the most toxic volatile fatty acid (Pohland, 1993). The control methods for inhibitory substances are;

- Removal of toxic materials from the waste
- Dilution of toxic materials
- Formation of insoluble complexes
- Antagonizing toxicity with another material
- pH adjustment (Esteves, 1981).

2.2.7.6 Aeration. In aerobic landfills in order to protect the system from anaerobic conditions and adjust temperature and moisture content of mass, proper amount of air must be injected to the system. The aeration rate varies from landfill to landfill depending upon the nature and quantity of the waste (Erses, 2007). The aeration rates for solid waste that have been reported in the literature is given in the Table 2.4.

Table 2.4. Literature review of aeration rates in aerobic landfills (Erses, 2007).

Reference	**Aeration Rates**
Borglin et al., 2004	0.04 L/min/kg waste
Ishigaki et al., 2003	0.8 L/min/kg waste
Cossu et al., 2003	0.17 L/min/kg waste
Kim and Yang, 2002	0.003 L/min/kg waste
Smith et al., 2000	0.0002 L/min/kg waste
Hanashima, 1999	4.2 L/min/m3 waste
Bernreuter and Stessel, 1999	0.5 L/min/kg waste
Keener et al., 1997a,b	0.35-0.97 L/min/kg waste

Aeration plays a critical role in the aerobic landfills. It controls the temperature and moisture content of the landfill. Moreover, in order to prevent oxygen using microorganisms, aeration is very important for removing carbon dioxide from the landfill.

2.3. Codisposal of Sludge

2.3.1. Sludge Management and Production in Turkey

Due to large volumes of municipal wastewater treatment the production of sludge is increasing every year around the world. Significant efforts have been spent to develop a new technology to treat and handle the wastewater sludge. For the long-term and sustainable management of the sludge, new solutions have not been developed yet and these new solutions must be environmental friendly, economically viable and socially acceptable (Wang et al., 2008; Metcalf and Eddy, 1991).

Based on the 2004 statistical data of TUIK, Turkey produced 580,000 tons of sewage sludge/year by 2010 (Table 2.5). Besides, according to TUIK, the ratio of population served by wastewater treatment plants to the total population is given 37%. Assuming solids production as 60 g/P/d, the amount of sewage sludge can be estimated as 1600 tons/day. Based on population equivalent, the produced sludge amounts as kg/PE.year and m^3/PE.year are given in Table 3.1 according to the applied treatment processes (Habitat, 2008).

In another study, ENVEST Planners estimated that Turkey's sludge production is approximately 275,000 tons/year in 2005 (Envest Planners, 2005). A comprehensive study on management of municipal or industrial wastewater sludge for Turkey is not available. Thus, it is not possible to determine the exact quantity and characteristics of the sludge that is being generated (Wurtz et al., 2007).

Sludge quality depends on the composition of the incoming water and the applied method to that water (Güneş, 2005). Sewage sludge has been used in agriculture, land filled, or incinerated. (Wang et al., 2008). Wastewater sludge contains nutrients and energy. That energy can be used beneficially. Thus, an alternative method to sewage sludge can be waste to energy conversion processes. Landfilling of sludge can be an alternative method to produce energy.

Landfilling of sludge has significant advantages such as easier handling of sewage sludge, acceleration of waste stabilization rates, smaller disposal/treatment area requirements and lower capital investment when compared to other disposal techniques. The major disadvantages associated with landfilling of sewage sludge include odor emission, landfill slope instability, and overload of leachate treatment system. Besides, landfilling of sludge concentrates metals in organic landfills and poses long term risks due to C and N stocks. There are several ways landfilling of sludge. These are (Reinhart and Townsend, 2008):

- Codisposal with MSW
- Pit or Trench Burial
- Mixing with Additives
- Mix with Additives and Use as Daily Cover
- Spread in Thin Layers on the Working Face and Cover with MSW.

Managing biosolids in ways that protect public health and the environment require diverse knowledge and skills. It also requires strong organizations capable of operating facilities and systems continually, with a high degree of accuracy and quality control (Habitat, 2008).

Table 2.5. Estimated sewage sludge production and population of reporting countries (Habitat, 2008).

Country	**Estimated Sewage Sludge Production (10^3 dry metric tons/year)**	**Population**
Brazil	372	118 078 000
China	2 966 000	1 313 974 000
Turkey	580	70 414 000
Slovakia	55	5 439 000
Hungary	120	9 981 000
Japan	2 000 000	127 464 000
Canada	550	33 100 000
Italy	1 000 000	58 134 000
Norway	86,5	4 611 000
Czech Republic	200	10 235 000
USA	6 514 000	298 444 000
Portugal	236,7	10 606 000
Germany	2 000 000	82 422 000
United Kingdom	1 500 000	60 609 000
Slovenia	57	2 010 000
Finland	150	5 231 000
Netherlands	1 500 000	16 491 000

2.3.2. Codisposal of Solid Waste and Sludge in Bioreactor Landfills

One means of achieving a more integrated approach to waste management is the combined disposal of MSW and sewage sludge within a single facility. Combined disposal can be defined as the mixing of sludge with solid wastes prior to disposal in a landfill and also known as codisposal, that offers potential cost savings over the development of separate facilities for different types of waste, and would involve the development of a single site. An integrated process for consultation, planning applications and a single program of transport, site selection, and characterization can be provided by means of codisposal (King et al., 2002; Güneş, 2005). The pathogens and high nitrogen content of sludge can be reduced by the codisposal of sludge with solid wastes while minimizing the potential problems by proper stabilization (Güneş, 2005).

Codisposal benefits include dilution of potential toxic compounds, improved balance nutrients, synergistic effect of microorganisms, increased load of biodegradable organic matter so that the biogas yield is enhanced. Additional advantages include hygienic stabilization and increased digestion rate (Ağdağ et al., 2002; Mata et al., 2002). Potential benefits that are associated with an increased rate of decomposition include (Röhrs, 1998):

“1. A shortened acid-producing phase before methanogenesis. This reduces the period during which acidic leachate containing high COD, metal and acid concentrations is formed.

2. Concentrations of parameters such as COD and iron reduce with an increase in leachate pH (COD reduces as a result of acid utilization by methanogens). By reducing the duration of the acid phase, the risk of groundwater contamination is lowered and ultimate leachate treatment costs are reduced.

3. Gas production is concentrated over a shorter period. The total volume of gas production remains unaltered, but by shortening the period over which the peak in the gas release occurs, the extraction of methane is economically more viable. Landfill sites can be reused earlier, since the gas potential is exhausted sooner than at slowly decomposing landfill sites.

4. Settlement of the waste bulk occurs sooner than at landfills where degradation is slower. A greater volume of waste can thus ultimately be contained at the landfill site.”

The studies on codisposal of MSW and sewage sludge in a bioreactor landfill have both positive and negative effects. It is shown that anaerobically digested sewage sludge can serve as a seed to microorganisms as well as source of nitrogen, phosphorous, and other nutrients (Warith et al., 2005). Besides, some of the studies reported that the addition of anaerobically digested sludge in the reactor provided the most enhanced methane production; it results in enhanced stabilization of combined waste and improved quality of leachate and it is favourite seedings for microorganisms.

Many researchers have studied the impact of codisposal of sludge on bioreactor landfills. Many of them found that codisposal of sludge increased gas production; it is a good source of moisture and improved quality of leachate. One of them (Güleç et al., 2000) reported that pH of leachate ranged from 7.0 to 8.5 compared to sharp drop in pH levels to the acidic range in the no sludge addition reactors. This can be explained by the buffer capacity of sludge.

Filibelli and co-workers (2010) used 8.5 L lab-scale anaerobic bioreactors for codisposal in their study. At the end of the study they have found that, anaerobic digestion is an effective method for sludge stabilization and it leads to decrease of the organic content of sludge (Filibelli et al., 2010).

Blakey and others (1997) found that codisposal of MSW and sludge improves leachate quality and methane yield in their field study (Blakey et al., 1997). Several researchers obtained the similar results in their laboratory studies: (Edelmann et al., 1999; Bujoczek, 2001).

Kinman and Nutini (1987) reported that the addition of anaerobically digested sludge in the reactor provided the most enhanced methane production. They evaluated techniques to enhance landfill gas production from landfills. Some of the techniques proposed, including leachate recirculation, nutrient addition, anaerobic sludge addition and buffer addition, sludge addition was found to give the highest methane yield. (Kinman et al., 1987) Several researchers obtained the similar results (Leuschner, 1989; Craft and Blakey, 1998; Gulec et al., 2000).

Gomez (2006) codisposed primary sludge with organic (only fruit and vegetables) part of the MSW. At the end of the study they have found codisposal of organic solid wastes is more promising method and enhance the methane yield. This reactor performs better than the others.

In other study, Sosnowski et al. (2003) observed codisposal of MSW with sludge protects environment and saves energy considering to the other methods.

Barlaz (2006) observed that the addition of anaerobic sewage sludge to fresh refuse did not show any effect on methane production. He concluded that sludge addition to fresh refuse led to the accumulation of carboxylic acid and decreases in the leachate pH. (Barlaz, 2006). Cossu and co-workers, concluded that the addition of alkaline sludge in the MSW resulted in enhanced methane production. He also found, nutrient addition was responsible for shortening of acidogenic phase (Kommilis,1999).

Another research by Güneş (2005), found that disposing sludge on a bioreactor landfill is an effective way for stabilization of the solid wastes. Also he found that after the 55th day of the experiment adding yeast to reactors increase the gas production gradually.

On the other hand, Sponza and co-workers (2004) used industrial sludge on their reactors. They found that at the end of the anaerobic toxicity assay (ATA) test, there is significant toxicity in industrial sludge-added reactors under anaerobic conditions since decreases in methane gas productions was observed. On the 98th day of the experiment methane production started to decrease (Sponza et al., 2004).

In contrast to Sponza, Çınar and co-workers (2002) investigated cumulative gas production in their reactors. They found that the gas quantity of the sewage sludge contained reactor was higher than only solid waste reactors. This was happened because of sewage sludge’s higher organic content (Çınar et al., 2002).

Valencia and co-workers (2008) reported that codisposal of septic tank sludge had a positive effect on the municipal solid waste (MSW) stabilisation process in Bioreactor Landfill simulators. It has two direct benefits, including the safe and environmentally sound disposal of septic tank sludge and an improvement of the overall performance of the Bioreactor Landfill by increasing moisture retention and supplying a more acclimatised bacterial population (Valencia et al., 2008).

Conversely, Leuschner (1989) reported that septic tank sludge was a poor source of microbial inoculum. The reactor that received septic tank sludge experienced a pH depression due to the accumulation of volatile organic acids in leachate, causing low methane production. Therefore, the addition of sludge from a septic tank had a negative effect on methane production and leachate quality (Cinar et al., 2002).

3. MATERIALS AND METHODS

The objective of this study is to investigate the impact of aerobic and anaerobic digestion processes on municipal solid waste codisposed with sewage sludge by utilizing two stage laboratory experiments. These stages include, the lab-scale reactor set-up in order to determine optimum solid waste to sludge ratios and anaerobic versus aerobic simulated bioreactors to understand the effect of sludge addition on waste stabilization.

3.1. Reactor Experiments

The wastes that are loaded to the pilot and lab-scale reactors prepared synthetically. According to literature, synthetically prepared wastes accelerate the biodegradation and the stabilization processes in the bioreactors. The composition of solid waste that was loaded to each reactor is given in Table 3.1. These values were determined according to average values of solid waste composition for Istanbul Region.

Table 3.1. Solid waste composition (Istaç Annual Report, 2009; Sezgin et al., 2003; Ministry of Environment Annual Report, 2008).

Material	**%**
Organic	54
Paper	10,5
Yard	5
Plastic	7
Glass	6
Metal	3
Electronic	0,5
Other	6
Inert	8
Total	100

For both lab-scale and pilot systems, the wastewater treatment plant sludge cake was taken from Paşaköy Biological Wastewater Treatment Plant. In order to accelerate and enhance the stabilization process, 50 grams of anaerobic seed sludge was added to the lab-scale and 3000 grams of anaerobic seed sludge was added to the pilot scale system. Seed sludge was taken from Fritolay. A physical and chemical solid waste and sludge characterization were performed before the reactor operations.

Paşaköy from where sludge cake is taken from is an advanced biological treatment plant designed to treat domestic wastewater from 2,500,000 person. It is located in the Ömerli area with a capacity of 500,000 m^3/day. The sludge cake used in the experiments was taken from the thickener unit, centrifuge outlet (ISKI, 2012).

3.2. Configuration of the Lab Scale Reactors

A schematic of the simulated anaerobic landfill bioreactors for sludge codisposal processes were constructed in the laboratory presented in Figure 3.1.

In order to simulate the anaerobic codisposal process occurring in landfills, five 10-liter plexiglass reactors with a length of 30 cm, and a diameter of 20 cm were constructed. The reactors operated in a constant temperature water bath an average temperature of 32 ^{o}C to simulate mesophilic temperature range. The temperature was arranged by an automatic heat controller and the water bath was prepared with a base of 100x30 cm and with a height of 40 cm.

The reactors were equipped with a sloped outlet near the bottom to collect leachate samples and three ports at the top; one for to measure the gas, other for sample the gas generated the other for the adding liquid if necessary.

The generated gas was collected and measured by the inverted cylinder method. A 0.5 L cylinder was inverted and placed into a 1 L cylinder which was filled with confining solution (acidic) to minimize the dissolution of CO_2 and CH_4 in the liquid. A sulfuric acid and water solution (pH<2) was prepared (Lagerkvist et al., 1993). The amount of gas generated in the reactors was indicated by the volume occupied in the inner cylinder.

After the loading, the reactors were checked for leaks by pressurizing the system with nitrogen gas. A coating of silicon was applied to all connections and joints to ensure that the units are water and gas tight. Then, the reactors remained pressurized for 24 hours in order to observe any leakage. Leak test was conducted until no leakage was observed.

3.2.1 Loading of the Lab-Scale Reactors

Five different reactors with the different sludge:waste ratios were used in this experiment. In order to determine the effects of sludge codisposal in anaerobic bioreactors, the Control 1 Reactor was loaded only with municipal solid waste and Control 2 Reactor was loaded only with sludge. The other three reactors were loaded according to the references showed in Table 3.2.

Table 3.2. References used to load the reactors.

Reactor	**References**
1:4	EPA Sludge Manual, 1979; Güneş, 2005
1:7	EPA Sludge Manual, 1979; Çınar et al., 2002; Habiba, 2009
1:10	EPA Sludge Manual, 1979

The lab-scale reactors were loaded with shredded and compacted solid waste mixture of 1.5 kg and different amounts of sludge cake. For this experiment 10 kg of solid waste was prepared and 5 kg and 1 kg of sludge cake and seed sludge was taken from the plants, respectively. After preparing the solid waste mixture, this mixture was put into the plastic bag and stored at the cold room at 4 ^{o}C until the reactor loading and performing experiments. The same procedure was followed after taking the samples from the plants for the sludge cake and the seed sludge.

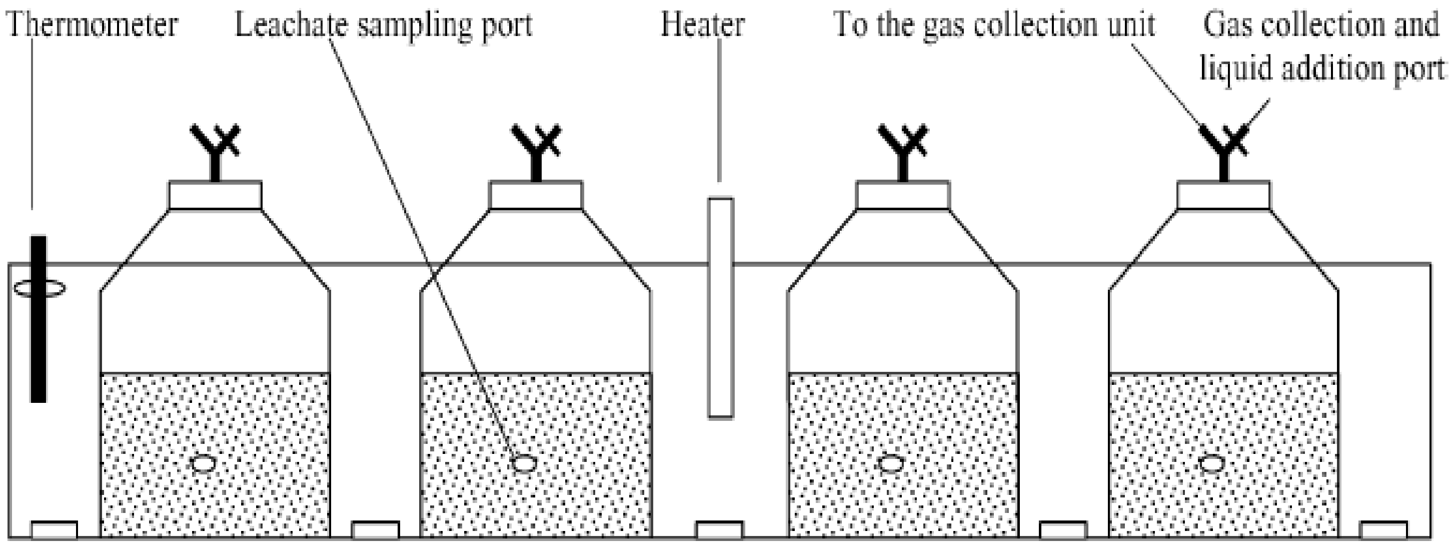


Figure 3.1. Configuration of the lab-scale reactor system.

The moisture content of the reactors was adjusted to %85. The volume of the reactors and the amount of water required to obtain the selected moisture content were considered, it was concluded that 1.5 kg of waste should be loaded to each reactor. Also, loading 1.5 kg of waste is advantageous for mixing the reactors. If the reactors were totally full, mixing of the reactors wouldn’t be suitable. Mixing of the reactors was provided by shaking manually. In this study, no stirring apparatus was used because of the use of a large aquarium full of water for temperature control.

The amount of water to be added to each reactor was calculated by using the moisture contents of sludge and solid waste. For the system, wastewater treatment plant sludge cake was disposed together with the municipal solid waste. The loading conditions of the reactors are given in Table 3.3.

Table 3.3. Loading conditions of the lab-scale reactors.

Reactor Name	**Sludge: Waste Ratio**	**Moisture Content %**	**Solid Waste-Added Wet (g)**	**Sludge Cake-Added Wet (g)**	**Seed Sludge-Added Wet (g)**	**Water Added (g)**	**Density (kg/m^3)**
Control 1	1:0	75	-	1500	25	2291	818
Control 2	0:1	75	1500	-	25	2415	47.8
Reactor1	1:4	75	1500	375	25	1867	692.5
Reactor2	1:7	75	1500	214.3	25	2102	592.35
Reactor3	1:10	75	1500	150	25	2195	516.78

On the day of the loading of the reactors: the stored solid waste and sludge was taken out of cold room to ambient temperature, then the reactors were loaded according to the values given in Table 3.2.

3.3. Configuration of the Pilot Scale Reactors

In the pilot scale experiment one aerobic, one anaerobic 100L bioreactors with a length of 1 m and with a diameter of 35 cm were used. A schematic of these simulated landfill bioreactors that was used in the laboratory are shown in Figure 3.2.

The bioreactors was operated at 32 °C in a temperature controlled room. The reactors were equipped with three ports; one port which is located at the bottom was used to drain and sampling the leachate while other two ports was used to collect gas samples and to add liquid.

The leachate was recirculated once a week through the collection tank by the help of pump. An ISMATEC S460 MINI pump was used to deliver leachate collected in the plastic container to the reactors. The gas production was measured by means of a wet gas meter.

After placement of solid waste into the reactors, a coating of silicon was applied to all connections and joints to ensure that the units are water and gas tight. In the anaerobic reactor the system was purged with nitrogen gas for providing anaerobic conditions and remove oxygen. Aerobic conditions was provided by air that will be supplied from a compressor at a flow rate of 2.2 L/min. This flow rate was arranged according to literature reviews. Erses (2007), used 0.11 L/min/kg flow rate in her study (Erses, 2007). Also, Cossu and his coworkers (2003), used 0.17 L/min/kg flow rate in their studies (Cossu et al., 2003).

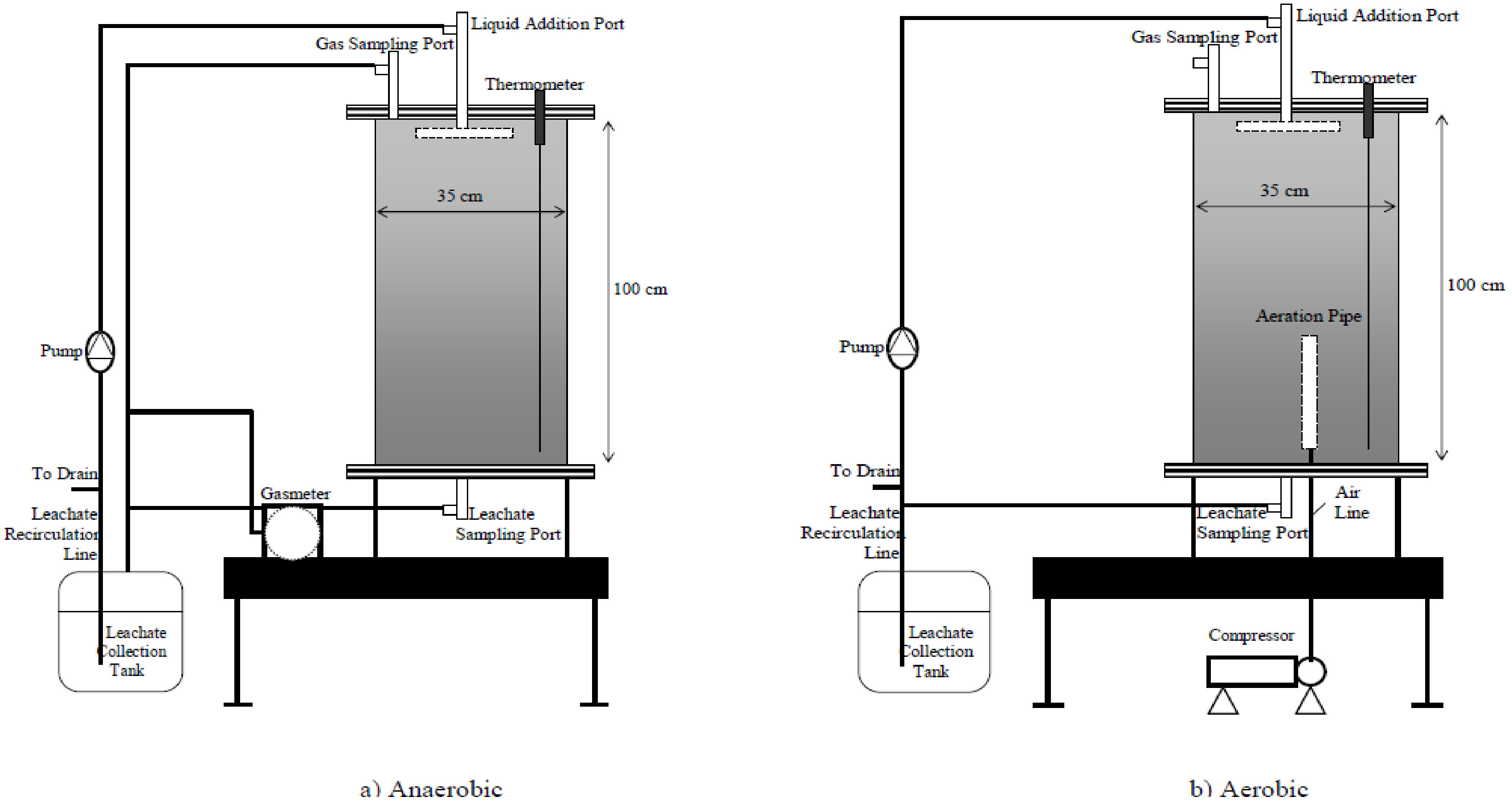


Figure 3.2. The design and operational features of pilot scale reactors.

3.3.1. Loading of the Pilot Scale Reactors

Bioreactors were filled with the same ratio of sludge:waste determined from the first stage Lab-scale experiment reactor results. The moisture content of the pilot scale reactors were 55-60%. The amount of water to be added to each reactor was calculated by using the moisture contents of sludge and solid waste.

The pilot scale reactors were loaded with shredded and compacted solid waste and sludge mixture of 19.5 kg. For this experiment 35 kg of solid waste was prepared and 20 kg and 7 kg and 4 kg of sludge cake and seed sludge were taken from the plants, respectively. In aerobic bioreactor no seed sludge was used. After preparing the solid waste mixture, this mixture was put the plastic bag and stored at the cold room at 4 ^{o}C until the reactor loading and performing experiments. The same procedure was followed after taking the samples from the plants for the sludge cake and the seed sludge.

Table 3.4. Loading conditions of the pilot scale reactors.

Reactor Name	**Sludge: Waste Ratio**	**Moisture Content, (%)**	**Solid Waste-Added Wet (kg)**	**Sludge Cake-Added Wet (kg)**	**Seed Sludge-Added Wet (kg)**	**Density (kg/m^3)**
Reactor 1 (Anaerobic)	1:4	60	16	4	3	847.5
Reactor 2 (Aerobic)	1:4	55	16	4	-	872.5

3.4. Analysis of the Leachate from the Reactors

Leachate samples collected from reactors were analyzed to better understanding the degree of waste stabilization and the effects of sludge codisposal. The leachate samples were analyzed for pH, ORP, COD, alkalinity, ammonia nitrogen, conductivity, salinity and heavy metals. All these analyses were performed according to “Standart Methods for the

Examination of Water and Wastewaters” (Clesceri et al., 1989). The frequency and methods of these analyses are given Table 3.5.

Table 3.5. Methods and frequency of leachate analyses.

Parameter	Method	Frequency
pH	Examination # 2730.B	3/week
Oxidation Reduction Potential (ORP)	Examination # 2580	3/week
Chemical Oxidation Demand (COD)	Examination # 5220 D	Weekly
Alkalinity	Examination # 2320.B	Weekly
Ammonia- nitrogen (NH_3-N)	Examination # 4500- NH_3 E	Twice in a month
Total Kjheldal Nitrogen (TKN)	4500 Method (APHA, AWWA, WEF-1998)	Weekly
Volatile Fatty Acids (VFA)	Gas Chromatograph	Weekly
Heavy Metals	ASTM 3010	Once in a month

pH and ORP are physical-chemical parameters. They were monitored for the indication of acid-base conditions and for oxidation-reduction (Clesceri et al., 1989). pH of the leachate samples was measured by WTW pH-320 and the ORP was measured by a probe attached to a model SAS20 pH meter.

COD is produced in landfills as a result of waste degradation. COD is a very important parameter and can used for the determination of the organic strength of leachate. In this study, COD analysis was performed by the dichromate closed reflux method. In these experiments, samples were prepared by pipetting them into COD vials with digestion solution (potassium dichromate and sulfuric acid solution). These vials were kept at 150 ^{o}C in Hach COD Digester for 2 hours. After this step, Hach DR/3 Spectrometer was used in order to determine the absorbances. The COD results were obtained by calibration curve prepared by a standard KHP solution.

Alkalinity represents a capability of a system to buffer the effects of volatile acids. In shortly, it is a measure of system buffer capacity. Alkalinity is affecting the pH and it is necessary to maintain a stable pH in the digester for optimal biological activity. Alkalinity in leachate is a result of carbonates, bicarbonates, borates, ammonia, organic bases, sulfides and phosphates concentrations (Yılmazer, 2009). In this study, alkalinity was monitored according to the method 2320B. Samples were measured by the titration method.

Nitrogen is one of the major nutrient for biological process. Most of the nitrogen available in landfill bioreactors is in the form of ammonia nitrogen. Ammonia nitrogen is the most significant long-term component in the leachate (Christensen et al., 1998). In this study, ammonia nitrogen was determined by distillation method followed by titrimetric method. For the distillation step, Gerhardt Vapodest Distillation Appartus was used. The samples were buffered to pH 9.5 and then distilled into the boric acid solution. Then, the samples titrated with the standart sulfuric acid solution (0.2 N) for ammonia determination. TKN is the sum of the ammonia nitrogen and organic nitrogen. In this study, digestion, distillation, and titration methods were used in order to determine TKN. HACH digester was used in the digestion process.

VFAs are the main products of the biological degradation of the organic constituents of the wastes. It is an indication of the acid formation phase. In this study, gas chromatograph method was used in order to determine VFA concentrations. Gas Chromatograph HP 5890 appartus was used.

Heavy metals are the most important pollutants in the landfill leachate. The analyses of heavy metals were performed by Perkin Elmer Inductively Coupled Plasma (ICP-OES) after the acid digestion step.

3.5. Analysis of the Gas from the Reactors

The gas generated in the reactors was monitored to better understand and identify the degree of waste stabilization. The volume of daily gas production and the gas composition were monitored throughout the study. The gas production was measured daily by observing the displacement of the acid solution at every gas collection unit in lab-scale bioreactors and by wet gas meter in pilot scale bioreactors.

In this study the gas composition analyses were performed by Gas Chromatograph HP 6850. In anaerobic reactors it is very important to observe CH_4 concentrations. The percentage of methane and carbon dioxide in the sampled gas was determined by using a gas chromatograph (GC). The operational temperature of injection port, the oven and detector were 50, 80, 80 ^{o}C, respectively. In aerobic bioreactor gas composition was not performed. Methods and the frequency of the parameters are given in the Table 3.6.

Table 3.6. Methods and frequency of gas analyses.

Parameter	**Method**	**Frequency**
Gas Production	Inverted Cylinder (Wet Gas Meter)	Every Day
Gas Composition	Gas Chromatograph (GC)	Once in a week

Daily gas production was measured in the anaerobic reactors by recording the amount of gas produced in 24 hours. Whenever gas readings were not read on one day, the result of reading was divided into the days that no reading was done.

Cumulative gas production was calculated by summing all of the observed daily gas production. Therefore, the cumulative gas production changed according to daily gas production data.

4. RESULTS AND DISCUSSION

4.1. Initial Waste Analysis

Preliminary analyses on the waste mixture provided data required to characterize the landfill materials. The results of the solid waste and sludge analysis are given in this section.

4.1.1. pH, Moisture Content, TKN, COD Analyses

In order to determine the amounts of water to be added each reactor, the moisture content of solid waste, sludge cake and seed sludge was performed at the beginning of the experiments. pH was determined for having an observation to buffer capacity of the solid waste and the sludge types. Also, TKN and COD values were determined for the suitability of the landfill materials used in the reactors.

Results of these analyses showed that the materials used for this experiment were convenient. For the lab-scale bioreactor they are given in Table 4.1 and for the pilot scale bioreactors they are given in Table 4.2.

Table 4.1. Solid waste, sludge cake and seed sludge analysis for the lab-scale setup.

Sample	**pH**	**Moisture Content (%)**	**TKN (mg/L)**	**COD (mg/L)**	**Density (kg/m^3)**
Solid Waste	6.05	34.44	50495.049	30223	360.35
Sludge Cake	7.96	26.505	60362.17	15544.85	720
Seed Sludge	7.26	93.71	15544.85	23839.45	572.2

Table 4.2. Solid waste, sludge cake and seed sludge analysis for the pilot scale setup.

Sample	**pH**	**Moisture Content (%)**	**TKN (mg/L)**	**COD (mg/L)**	**Density (kg/m^3)**
Solid Waste	6.18	35.12	51094.04	31093	287.5
Sludge Cake	8.01	25.52	59963.17	16134.12	757
Seed Sludge	7.35	93.89	15401.68	23909.5	498

4.1.2. Heavy Metal Analysis

The heavy metal analyses of the solid waste samples were performed in order to detect possible heavy metal inhibition in the systems and are given in Table 4.3 for the lab-scale and Table 4.4 for the pilot scale bioreactors. Heavy metals can be inhibitory to microbial life. In order to understand the suitability of the solid waste, seed sludge and sludge cake and showing the acceptable limit values it is very important to determine heavy metal concentrations and compare these results to heavy metal limit values in European Council Directives. When heavy metal concentrations in the solid waste, sludge cake and seed sludge compared to limit values are given in European Council Directives it was found that heavy metal concentration in the solid waste and different sludge types were lower than the limit values. As a result, throughout the experimental study no heavy metal inhibition was observed.

Table 4.3. Heavy metal concentration in the samples for the lab-scale bioreactors.

Sample	**Cr (mg/kg)**	**Mn (mg/kg)**	**Fe (mg/kg)**	**Ni (mg/kg)**	**Cu (mg/kg)**	**Zn (mg/kg)**	**Cd (mg/kg)**	**Pb (mg/kg)**
Seed Sludge	44.20	175.43	12054	43.78	43.08	824.34	0.59	9.78
Sludge Cake	388.5	497.21	1263	176.83	326.76	891.91	1.79	40.71
Solid Waste	10.12	76.44	4843.2	8.89	13.55	30.05	0.39	4.37

Table 4.4. Heavy metal concentration in the samples for the pilot scale bioreactors.

Sample	**Cr (mg/kg)**	**Mn (mg/kg)**	**Fe (mg/kg)**	**Ni (mg/kg)**	**Cu (mg/kg)**	**Zn (mg/kg)**	**Cd (mg/kg)**	**Pb (mg/kg)**
Seed Sludge	44.73	172.73	12001	43.78	42.19	824.1	0.99	9.03
Sludge Cake	396.5	502.19	1436	174.89	356.73	839.14	1.805	42.021
Solid Waste	14.15	73.37	4193.4	8.84	12.507	32	0.25	3.35

4.1.3. Elemental Analysis

The ultimate analysis involving the determination of the percent C (carbon), H (hydrogen), O (oxygen) and C/N were used to characterize organic composition in the waste mixture. It was also used to define the mix of waste materials and to ensure nutrient availability for biological conversion. In this study, the organic composition results showed that nutrients availability were enough for biological conversion. The results of the elemental analysis are given in the Table 4.5 for lab-scale and Table 4.6 for pilot scale .

Table 4.5. Elemental analysis results for the lab-scale bioreactors.

Sample	**C (%)**	**H (%)**	**O (%)**	**C/N**
Solid Waste	39.11	5.535	7.65	5.112
Sludge Cake	32.54	4.84	3.66	8.89
Seed Sludge	34.65	4.72	3.12	6.02

Table 4.6. Elemental analysis results for the pilot scale bioreactors.

Sample	C (%)	H (%)	O (%)	C/N
Solid Waste	38.17	5.19	7.46	5.42
Sludge Cake	31.98	5.03	3.19	8.96
Seed Sludge	34.51	4.2	3.12	6.17

4.2. Lab Scale Leachate Analysis

4.2.1. pH

While the degradation of the wastes, pH value creates a proper environment for the anaerobic bacteria. Providing optimum pH is very important in order to achieve complete stabilization. The pH of the system connected to the relationship of volatile organics and alkalinity. In order to enhance the growth of methanogens, the pH value in the anaerobic digestion reactor is normally in the range of 6.5-8.0. In the early stages of anaerobic digestion due to acid formation, pH value is lower (5.0-6.0) (Tchobanoglous et al., 1993).

The pH values of the leachates from five reactors are given in Figure 4.1. It was observed that initial pH values of the Reactors were 6.75 for Control 1, 5.08 for Control 2, 4.89 for Reactor 1, 4.86 for Reactor 2 and 4.82 for Reactor 3, respectively. All reactors were acidic at the beginning period of the experiment due to the acid formation phase.

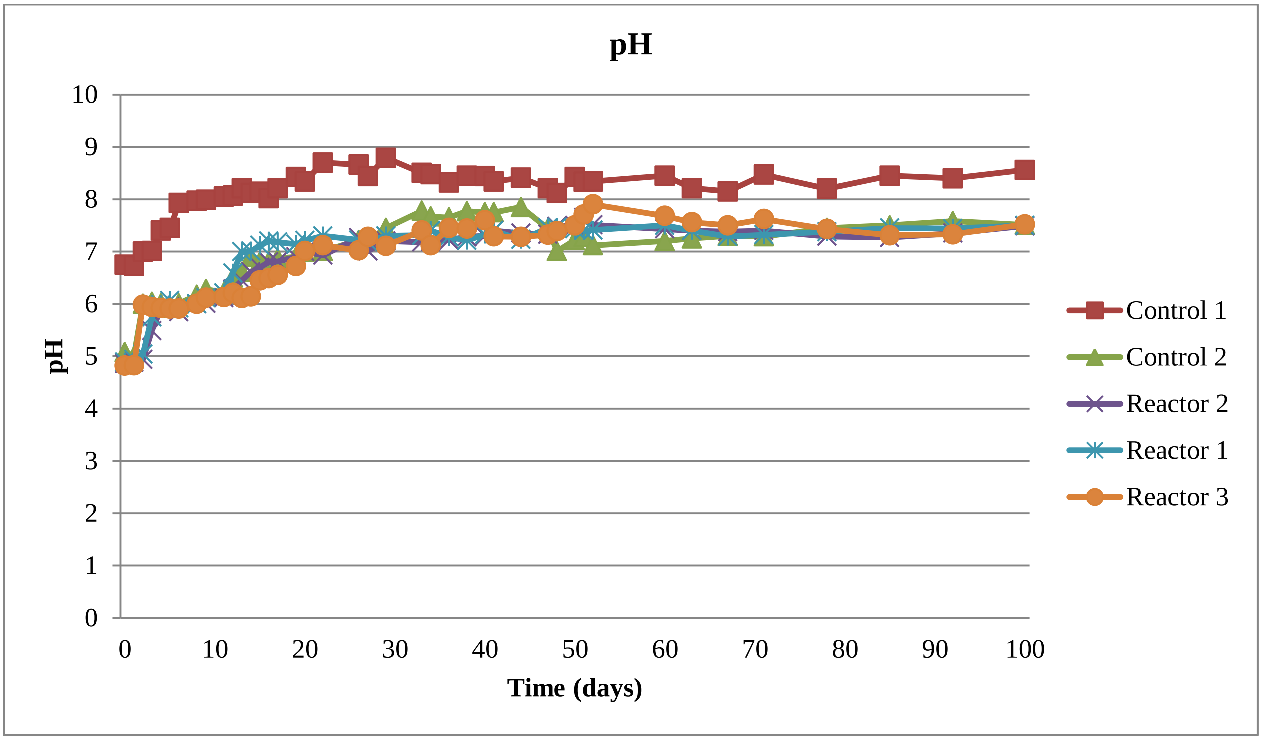


Figure 4.1. pH values of the lab scale reactors.

Each of the reactors exhibited a similar trend throughout the experiment. After the Day 20, all four reactors except Control 1 Reactor, attained the neutral pH values and after this day pH values of the reactors fluctuated between 7.0-8.0. These increases in the pH values were directly related to the conversion of VFAs into methane and carbon dioxide by the methanogenic activity where methane production started to increase after Day 20. In the Control Reactor 1 optimum pH values were observed at the beginning of the experiment. This was confirmed with the methane production in this reactor.

Barlaz and co-workers (1990) confirmed that acids accumulate and pH decreases due to the methanogenic activity. In this study, highest sludge addition reactor showed more balanced pH due to sludge addition speeds up methanogenic activity.

4.2.2. ORP

ORP of a system shows the oxidation-reduction potential. It measures oxidizing or reducing condition states of a system. The ORP is particularly important in defining the chemical characteristics of the landfill environment. The ORP values of the leachates from four reactors are given in Figure 4.2.

Negative ORP values should be measured in anaerobic processes. It is an evidence methanogenic activity. By the depletion of the oxygen and changes in the stabilization mechanisms, negative ORP values were observed in theexperiment. Also, a negative ORP value is indicative of microbially mediated reduction of sulfate to sulfide and nitrate to ammonium.

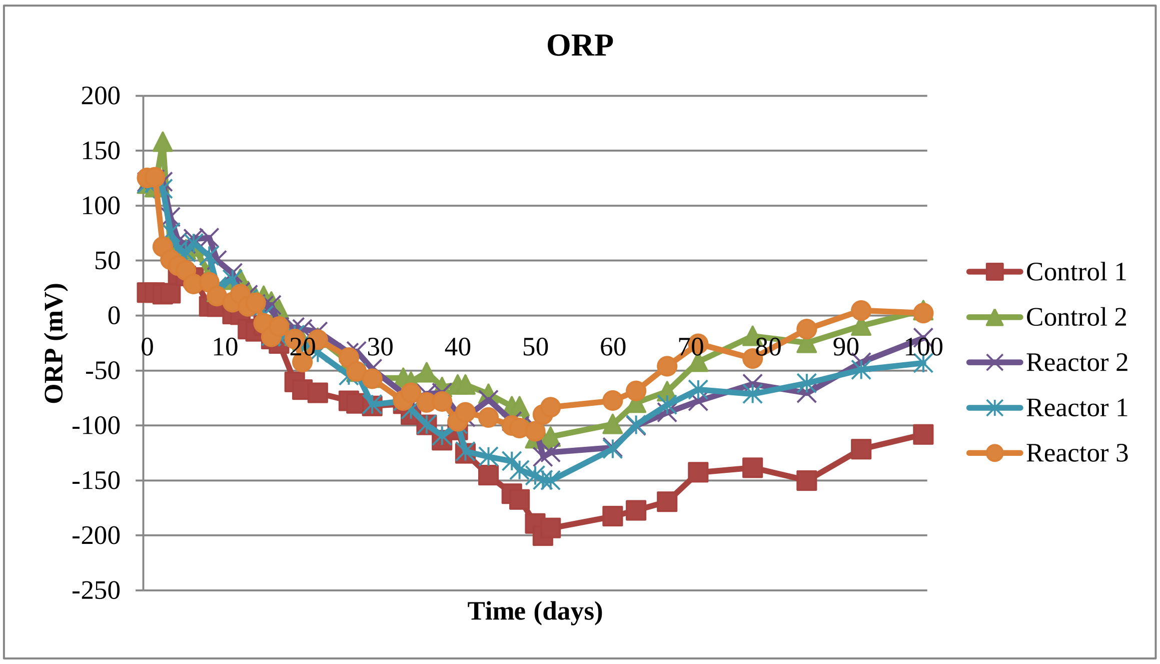


Figure 4.2. ORP values of the lab scale reactors.

At the beginning of the experiment ORP value was very close, except the Control 1 reactor. Initial ORP values in the reactors were 20.75 mV for Control 1, 120 mV for Control 2, 120.2 mV for Reactor 1, 121.5 mV for Reactor 2 and 125.2 mV for Reactor 3. After Day 3, a sharp decrease was observed in each reactor. After this day, Control 1 Reactor was reached optimum ORP values to produce methanogenic bacteria.

After Day 27, leachate ORP values in the Reactor 1, 2, 3 and Control 2 Reactor had reached to the negative values. Negative values indicated that reducing conditions were being established. The lowest ORP value was observed in the Control 1 Reactor (- 200.5 mV) and Reactor 2 (-124.5 mV) follows this reactor. However, all of the reactor's ORP value was very close to each other during the experiment except the Control 1 reactor.

Until Day 78, each of the reactor showed similar trend and fluctuated between -5 and -200 mV. After this day, the ORP values started to increase. During this period COD removal rate and gas production were very low during this period.

According to the study conducted by Erses (2008), the leachate sample taken from anaerobic reactor indicated ORP values decreased to –379 mV on the methanogenesis phase. On the other hand, Pohland (1980) and Christensen and Kjeldsen (1989) found optimum leachate ORP values in the anaerobic reactors can be decreased until to -300 mV. These results showed that the measurements of leachate ORP values in the reactors were higher than the literature. A mistake in measurement or a defect in ORP probe could cause this situation.

4.2.3. COD

Chemical oxygen demand (COD) is the amount of oxygen required to stabilize organic matter determined by a using strong oxidant. Leachate COD is produced as a result of waste degradation and can provide evidence regarding the progression or inhibition of landfill stabilization processes (Droste, 1987). Leachate COD concentrations for the reactors are shown in Figure 4.3.

The initial leachate COD concentrations of Control 1, 2 and Reactors 1, 2, 3 were 17295 mg/L, 22649 mg/L, 22139 mg/L, 21245 mg/L and 22393 mg/L, respectively. Sludge addition of the reactors resulted in COD increasing in the reactors. In the acid formation phase, until Days between 18-25 of the experiment, COD concentrations of the Control 1, 2 and Reactors 1, 2, 3 suddenly increased to 54665 mg/L, 40334 mg/L, 62190 mg/L, 60594 mg/L and 41897 mg/L, respectively. This was due to, the rapid decomposition of complex organics from solid waste and confirmed with the increase in VFA concentrations of the reactors.

After the methanogenic conditions started to take place in the reactors, COD concentrations decreased due to conversion of biodegradable products. At the end of the experiment COD concentrations of the reactors were 1234 mg/L for the Control 1 Reactor, 765 mg/L for the Control 2 Reactor, 906 mg/L for the Reactor 1, 1203 mg/L for the Reactor 2 and 1302 mg/L for the Reactor 3.

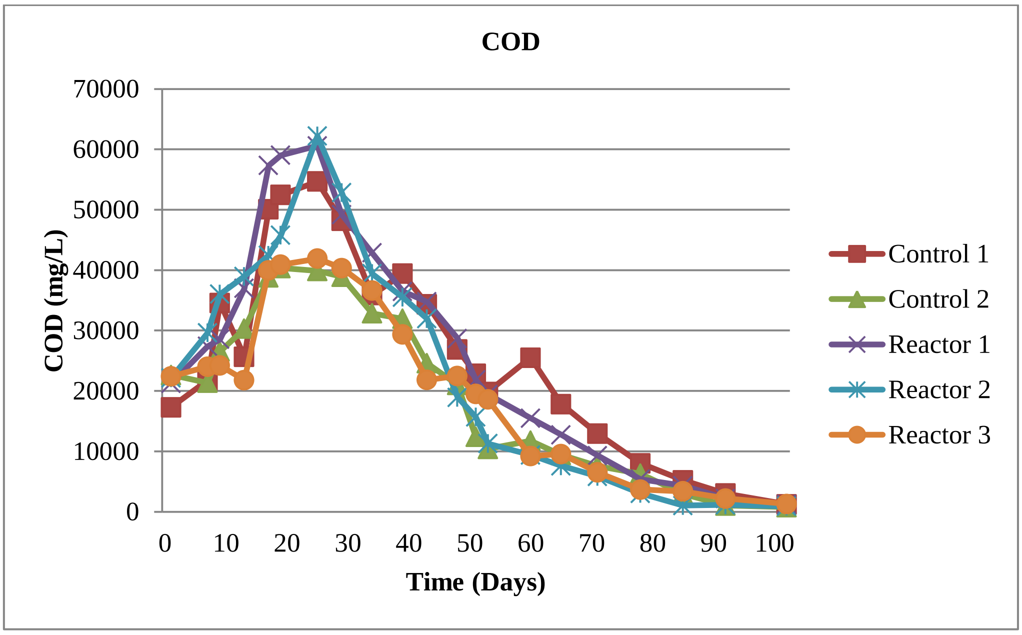


Figure 4.3. COD concentrations in the lab scale reactors.

The removal efficiency of COD concentrations in the reactors were 92.8% for Control 1, 95.4% for Control 2, 95.9% for Reactor 1, 94.3% for Reactor 2 and 94.2% for Reactor 3. Althought, Reactor 1 had the highest amount of sludge added and the highest COD concentrations in leachate, it was observed that stabilization process had the same efficiency as the other reactors. These results can be explained with that sludge addition speeded up the stabilization process and showed that higher amount of sludge can be codisposed with MSW in anaerobic bioreactors which can be a positive solution for sludge disposal.

4.2.4. Alkalinity

Alkalinity represents the buffering capacity of a system. In a system which has a fairly stable pH, alkalinity measures the ability of water bodies to neutralize acids and bases. Leachate contains compounds, such as bicarbonates, carbonates, hydroxides, borates, ammonia, organic bases, sulfides and phosphates. Alkalinity is very important to keep the pH desirable levels (Addy et al., 2004).

Figure 4.4 represents the alkalinity concentrations of the reactors. The initial alkalinity values of the reactors were 4994 mg/L as $CaCO_3$ for Control 1, 9889 mg/L as $CaCO_3$ for Control 2, 6653 mg/L as $CaCO_3$ for Reactor 1, 8271 mg/L as $CaCO_3$ for Reactor 2 and 7911 mg/L as $CaCO_3$ for Reactor 3.

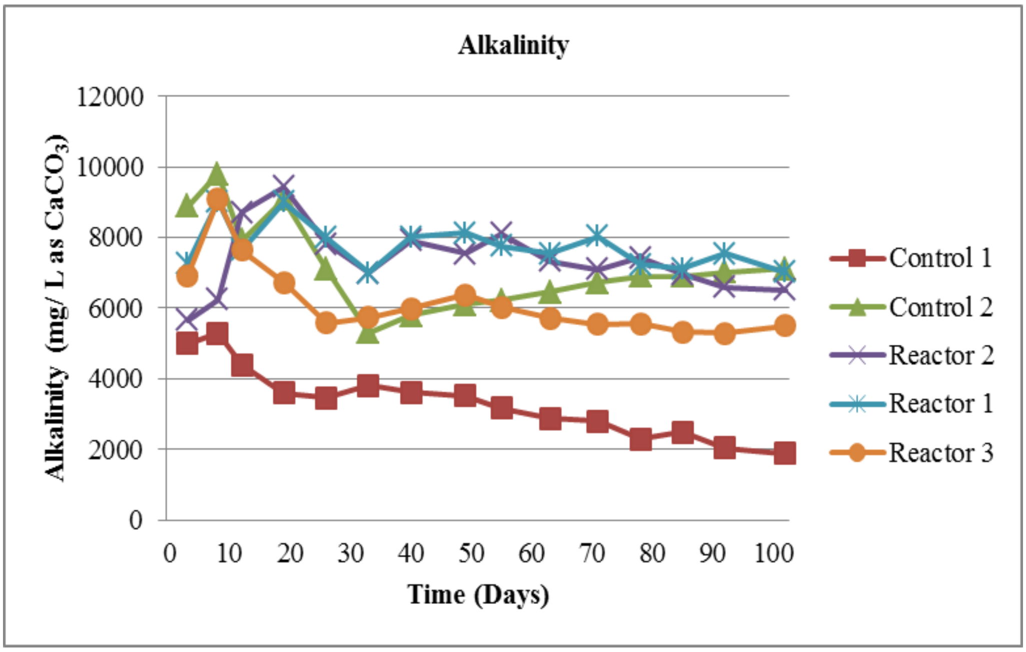


Figure 4.4. Alkalinity concentrations in the lab scale reactors.

At the beginning of the experiment Control 2, Reactor 1, 2 and 3 had higher amount of alkalinity due to high VFA concentrations and low pH value during acid formation phase took place. After transition phase, pH values increased and total alkalinity concentrations decreased. This was due to methanogens using the TVFAs as substrate to convert it to the methane.

After the transition phase, the total alkalinity in the Reactor 1, Reactor 2 and Reactor 3 was decreased to 7500 mg/L, 8010 mg/L and 6497 mg/L, respectively.

For the Control 1 Reactor, the total alkalinity was 4994 mg/L at the beginning and 1869 mg/L at the end of the experiment. The alkalinity concentration of this reactor was lower than the other four reactors, which can not be explained with current findings.

4.2.5. Ammonia Nitrogen and TKN

Nitrogen is one of the essential nutrients and it is found that in many forms including ammonia, organic nitrogen and nitrate. Raw wastewater nitrogen is normally present in the organic nitrogen and ammonia forms, with small quantities of the nitrite and nitrate forms (Pennsylvania Department of Environmental Protection, 2012). Total khejdal nitrogen (TKN) in the leachate is the sum of the organic nitrogen and ammonia nitrogen.

Ammonia is very essential for anaerobic digestion. Also, it could be toxic compound. High concentrations of NH_3-N are very toxic in anaerobic bioreactors. In the literature it was stated that concentrations between 200 and 1500 mg/L have shown to have no adverse effects on anaerobic process, concentrations ranging from 1500 to 3000 mg/L were shown to have inhibitory effects at higher pH levels, and concentrations above 3000 mg/L were very toxic (Pohland, 1987). Also, NH_3-N is a significant long-term pollution problem in the anaerobic landfills.

Figures 4.5 and 4.6 represent the results of the ammonia nitrogen and TKN, respectively. Leachate TKN and ammonia nitrogen concentrations were very similar. The initial ammonia nitrogen concentrations in the reactors are 1757 mg/L for the Control 1 Reactor, 392 mg/L for the Control 2 Reactor, 1028 mg/L for the Reactor 1, 745.9 mg/L for the Reactor 2 and 507.9 mg/L for the Reactor 3. The initial TKN concentrations for Control 1, Control 2 Reactor, Reactor 1, Reactor 2 and Reactor 3 were 2710 mg/L, 700 mg/L, 1930 mg/L, 1040 mg/L and 790 mg/L, respectively. The reactors which had higher sludge addition had higher amount of nitrogen.

For the first days of the experiment an increase was observed both of TKN and NH_3-N concentrations due to the release of ammonia in the acid phase. After the Day 30, where methanogens appeared in the system, TKN concentrations started to decrease. Normally, organic nitrogen concentrations in the anaerobic bioreactors are transforming to ammonia nitrogen in the methane fermentation phase. In this experiment NH_3-N concentrations were decreased due to biological assimilation to ammonia. In other words, ammonia nitrogen was used by microorganisms during the active waste stabilization.

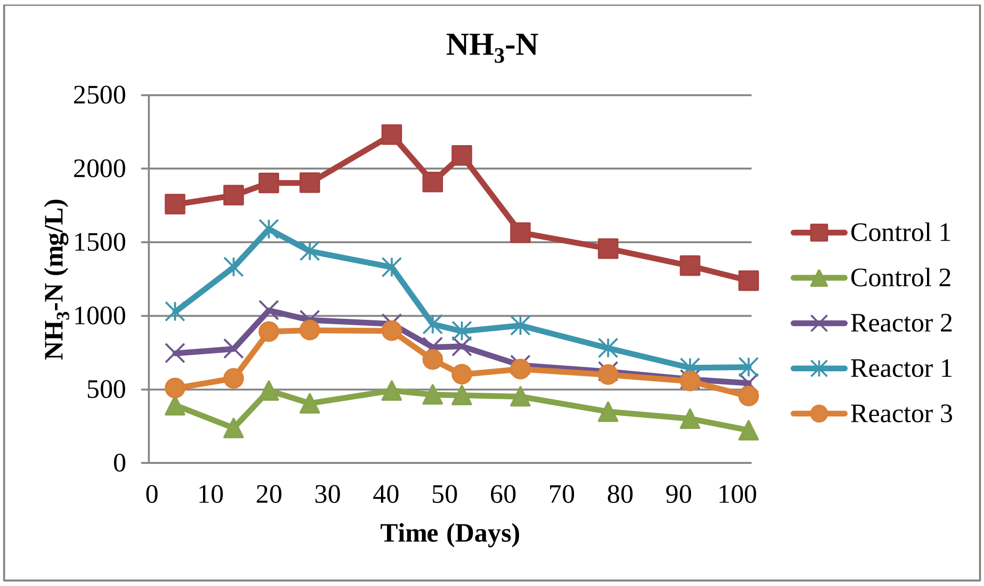


Figure 4.5. Ammonia Nitrogen concentrations in the lab scale reactors.

At the end of the study ammonia nitrogen and TKN concentrations were 1238.5 and 2130 mg/L for the Control 1 Reactor, 222 and 440 mg/L for the Control 2 Reactor, 651 and 1220 mg/L for the Reactor 1, 542 and 660 mg/L for the Reactor 2 and 454.3 and 530 mg/L for the Reactor 3, respectively. As it can be seen from the figures, there was no significant change in the nitrogen. Also, the utilization rates were higher in the reactors which had higher sludge addition.

The NH_3-N:TKN ratio of leachate indicates the biodegradation state of landfill and the ratio increases to about 1. The NH_3-N:TKN ratio was calculated in the range of 0.5-0.82 which is normal for this experiment.

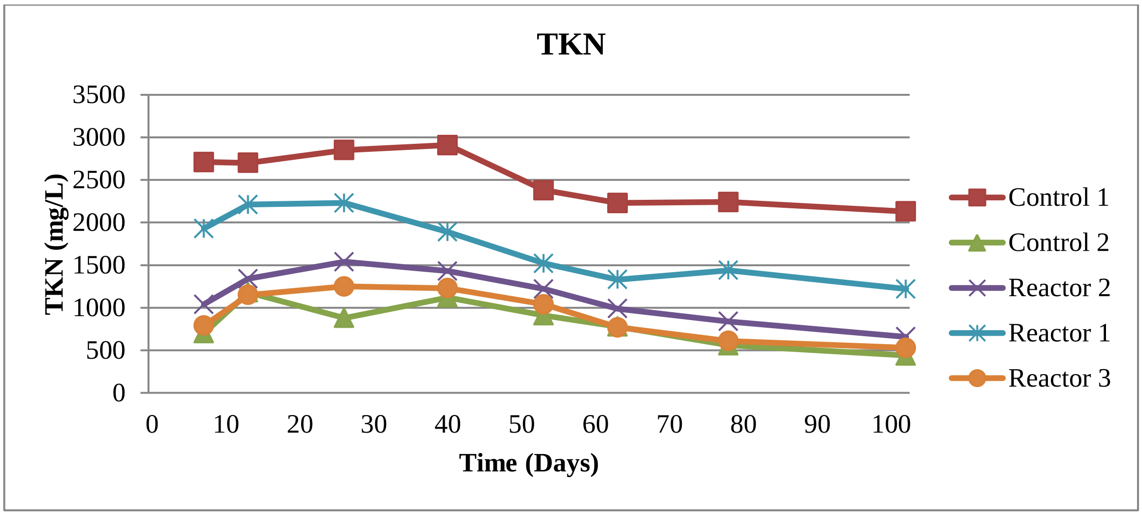


Figure 4.6. TKN concentrations in the lab scale reactors.

4.2.6. Volatile Fatty Acids

One of the most important end products of the acid fermentation process is volatile fatty acids. After the acid fermentation phase, these substances are converted into the CH_4 and CO_2. The total volatile fatty acids (TVFAs) represent the sum of the individual volatile fatty acids (proponic, valeric, caproic, etc.). Leachate total volatile fatty acid concentrations expressed as acetic acid for the reactors were given in Figure 4.7.

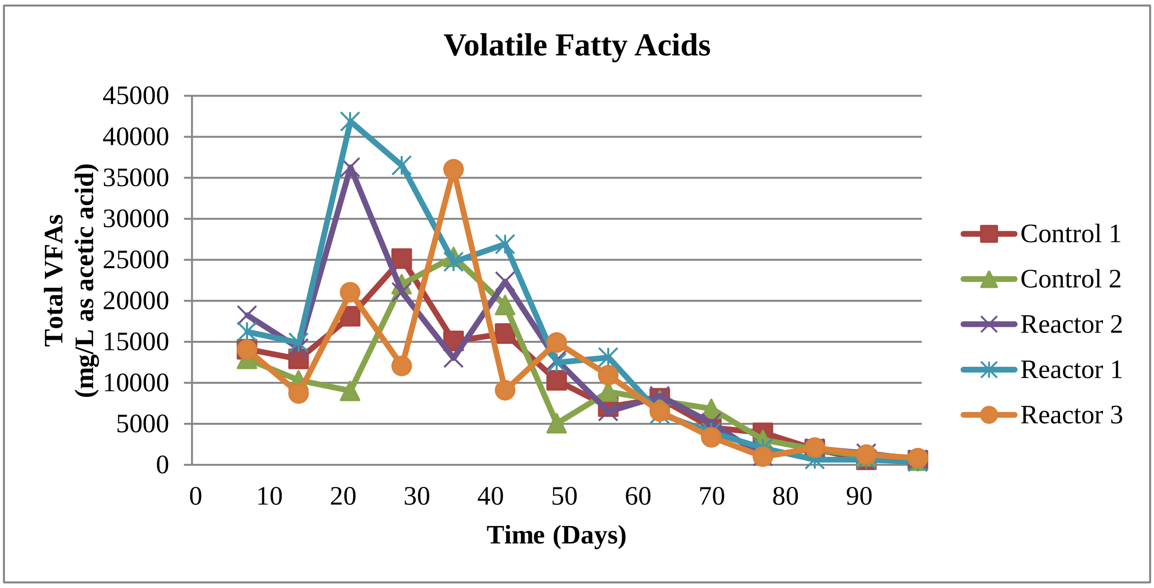


Figure 4.7. Total VFAs concentrations in the lab scale reactors.

At the beginning of the experiment high TVFA concentrations were observed in the reactors due to the initial acid fermentation phase. The initial TVFAs concentrations in the Control 1,2 and Reactor 1, Reactor 2 and Reactor 3 were 14090 mg/L, 12894 mg/L, 16209 mg/L, 18220 mg/L and 14032 mg/L, respectively. As it can be seen, reactors with higher sludge content had higher amount of TVFAs concentrations. This observation was also confirmed with the higher COD concentrations and higher methane content of these reactors.

During Day 21 and 28 of the experiment, TVFA concentrations were reached the highest value in the Control 1, Control 2, Reactor 1, Reactor 2 and Reactor 3 which were 25128 mg/L, 25331 mg/L, 41859 mg/L, 36273 mg/L and 21029 mg/L, respectively. After this day, TVFA concentrations were stared to rapidly decrease due to VFAs conversion to methane and carbon dioxide. This situation was also confirmed by decrease in the COD concentrations and by an increase at the methane production rate at the reactors. At the end of the experiment TVFA concentrations were 542 mg/L for Conrol 1 Reactor, 500.1mg/L for Conrol 2 Reactor, 338.2 mg/L for Reactor 1, 662.3 mg/L for Reactor 2 and 804.3 mg/L for Reactor 3.

4.2.7. Heavy Metals

Heavy metals are the important pollution source in the landfill area. In this study heavy metal concentrations were measured rarely due to the use of MSW. The change in leachate heavy metal concentrations from the reactors are given in the Tables 4.7, 4.8 and 4.9.

At the beginning of the experiment, heavy metal concentrations were higher in Control 1 and Reactor 1 due to the higher heavy metal content of the sludge. Throughout the study, no significant changes were observed in the heavy metal concentrations. Heavy metal concentrations did not inhibit the biological activity in the reactors.

Table 4.7. Heavy metal concentrations (Day of 40).

Reactor	Cr (mg/L)	Mn (mg/L)	Fe (mg/L)	Ni (mg/L)	Cu (mg/L)	Zn (mg/L)
Reactor 1	0.84	14.35	192.29	2.05	1.38	4.13
Reactor 2	1.36	11.3	238.58	3.1	1.97	5.16
Reactor 3	0.57	8.69	185.12	1.09	0.88	4.22
Control 1	5.06	6.24	163.02	2.83	15.27	12.32
Control 2	0.13	10.47	167.17	1.16	1.25	3.51

Table 4.8. Heavy metal concentrations (Day of 70).

Reactor	Cr (mg/L)	Mn (mg/L)	Fe (mg/L)	Ni (mg/L)	Cu (mg/L)	Zn (mg/L)
Reactor 1	0.77	13.99	190.14	1.98	1.08	4.01
Reactor 2	1.12	10.8	236.28	2.8	1.67	4.96
Reactor 3	0.58	8.54	182.1	1	0.81	3.98
Control 1	4.92	6.02	161.01	2.03	12.36	12.12
Control 2	0.09	10.12	166.8	1.11	1.21	3.46

Table 4.9. Heavy metal concentrations (Day of 100).

Reactor	Cr (mg/L)	Mn (mg/L)	Fe (mg/L)	Ni (mg/L)	Cu (mg/L)	Zn (mg/L)
Reactor 1	0.71	13.86	187.9	1.67	0.98	3.94
Reactor 2	0.8	9.98	231.5	2.5	1.01	4.12
Reactor 3	0.49	8.41	180.4	0.91	0.82	3.67
Control 1	4.86	5.87	158.5	1.97	11.96	11.98
Control 2	0.1	10.01	164.6	1.11	1.1	3.01

4.3. Pilot Scale Leachate Analysis

4.3.1. pH

The change in leachate pH from the reactors is given in Figure 4.8. Initial pH values of the Reactors were 6.43 for the Reactor 1 and 6.66 for the Reactor 2. Both of the reactors were acidic conditions at the beginning of the experiment. After loading the pH value of Reactor 1 started to decrease and became acidic until the Day 16, after that day it started to increase. Two stages (acidic and methanogenic) of pH in the Reactor 1 was observed during the experimental period. Until Day 55, in the anaerobic reactor (Reactor 1) acidic values stayed due to the accumulation of organic acids. After Day 55, methanogenic conditions took place in Reactor 1 and it was measured between 7.0 and 7.5 until the end. This was confirmed by gas composition and CH_4 production.

At the Reactor 2, after the Day 30, neutral conditions were occurred and pH value of this reactor increased and fluctuated between 8.0 and 9.1 until the end of the experiment.

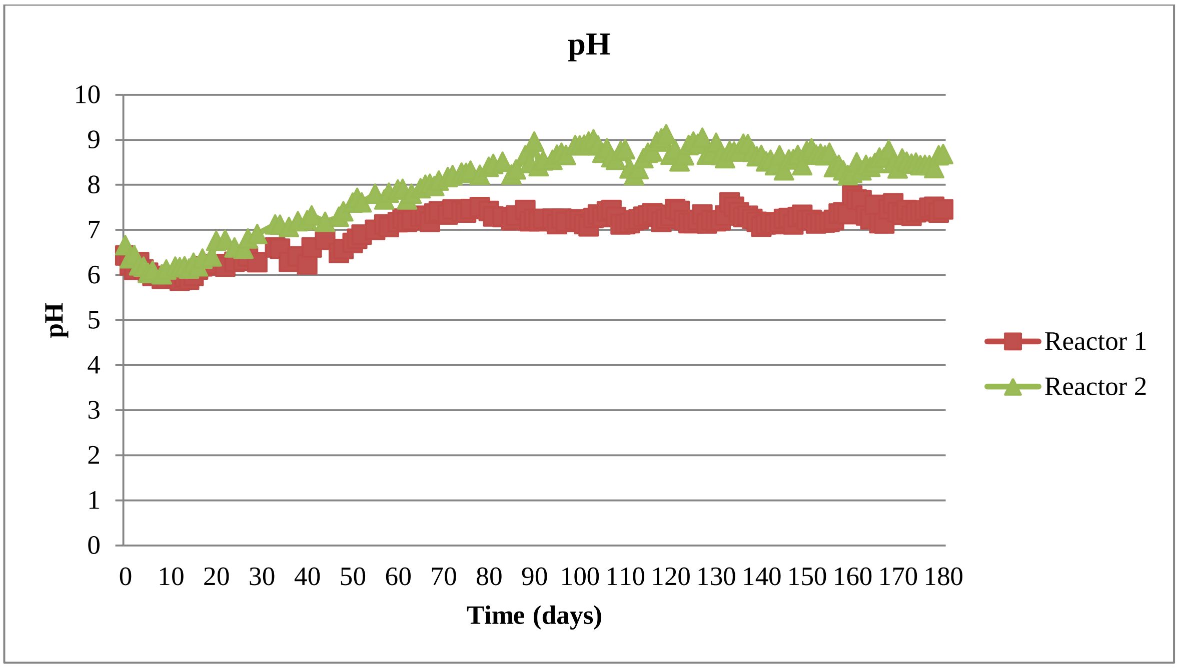


Figure 4.8. pH values of the pilot scale reactors.

As it can be seen from the Figure 4.8, the pH of aerobic reactor was higher than the pH from anaerobic reactor due to CO_2 stripping air flow.

4.3.2. ORP

The change in leachate ORP values from the reactors is given in Figure 4.9. Initial ORP values of the Reactors were around 33 mV for the Reactor 1 and 29 mV for the Reactor 2. Until Day 22, ORP values of the Reactor 2 were decreased and reached to -61.5 mV due to rapid organic degradation. After Day 22, the ORP values for Reactor 2 increased. Positive values were observed in this reactor and fluctuated between +100 and +250 mV.

Until Day 57, positive ORP values observed in the Reactor 1. After Day 57, the ORP values of this reactor was measured between -100 and -300 mV. Negative ORP values should be observed in the anaerobic reactors for methanogenesis. Some researchers found that an optimum ORP requirement which generally ranges from -100 to -300 mV (Pohland et al., 1993).

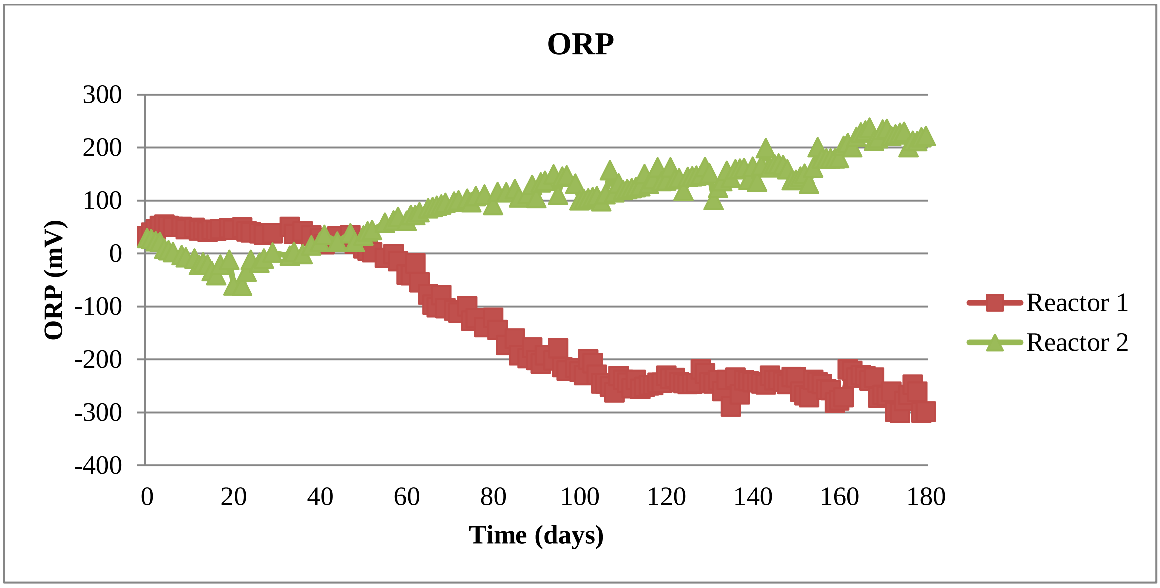


Figure 4.9. ORP values of the pilot scale reactors.

4.3.3. COD

The change in leachate COD values from the reactors is given in Figure 4.10. Initial COD values of the Reactors were 25678 mg/L for the Reactor 1 and 21487 mg/L for the Reactor 2.

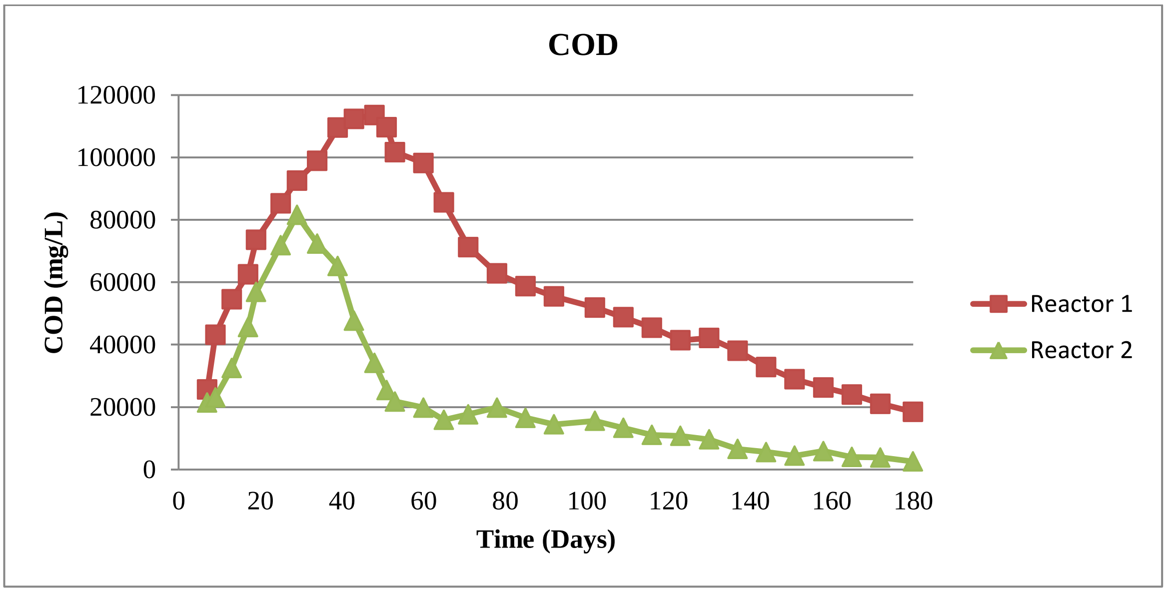


Figure 4.10. COD concentrations in the pilot scale reactors.

COD concentrations of the Reactor 1 increased from 25678 mg/L to 113578 mg/L due to rapid release and hydrolysis of complex organics from solid waste to the leachate. This was confirmed with the decrease in pH. After the methagonic conditions provided, COD value of the Reactor 1 rapidly began to decrease and reached 18432 mg/L at the end of the experiment.

Until the pH was neutralized in Reactor 2, COD value was increased from 21487 mg/L to 81576 mg/L. After Day 30, a rapid decrease was observed in this reactor. The COD concentrations of the Reactor 2 was reached to 1092 mg/L at the end of the experiment.

In aerobic reactor, decrease in COD values were faster than the anaerobic bioreactor due to aeration enabling significantly faster biodegradation of organic matter. The results were very similar to Erses et al., (2008) where COD value of the anaerobic reactor decreased from 38022 mg/L to 900 mg/L at the end of the 433th day and at the aerobic reactor decreased from 17932 mg/L to 1596 mg/L at the end of the 77th day. The removal COD rates of the reactors were 28.21 % for Reactor 1 and 88.11% for Reactor 2.

4.3.4. Alkalinity

The change in leachate alkalinity values from the reactors is given in Figure 4.11. Initial alkalinity values of the Reactors were 4178 mg/L as $CaCO_3$ for the Reactor 1 and 3478 mg/L as $CaCO_3$ for the Reactor 2.

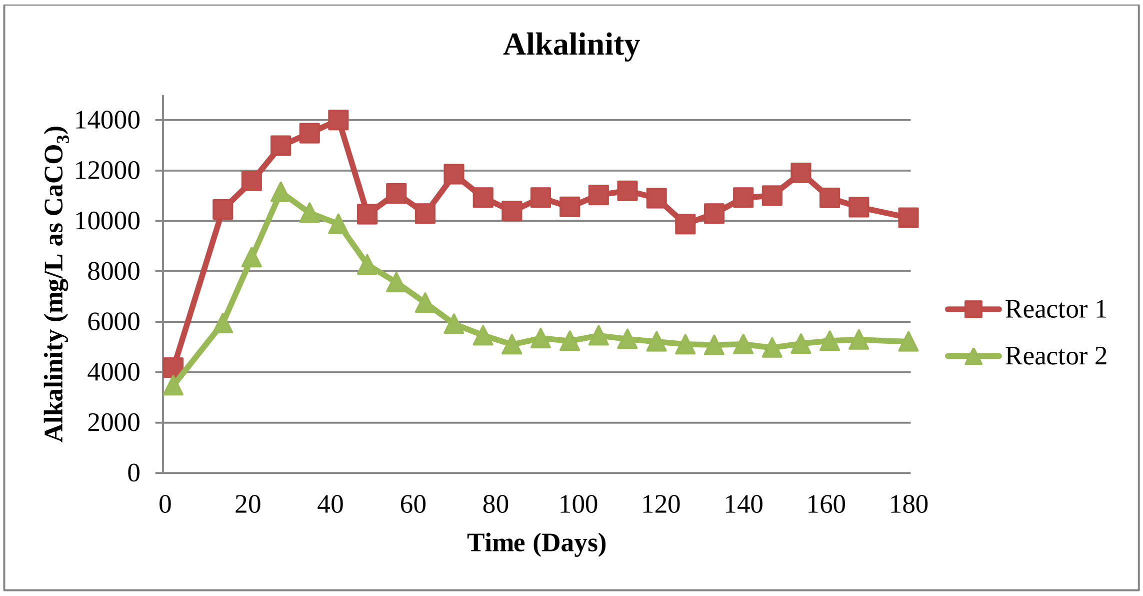


Figure 4.11. Alkalinity concentrations in the pilot scale reactors.

As can be seen from the figure, both reactors showed similar trend. For the first days at the experiment, in Reactor 1, leachate alkalinity increased due to the high VFA concentrations and low pH values where acidogenic conditions took place. In the later stages, pH values increased and total alkalinity concentrations started to decrease because since VFA was transformed to methane. Total alkalinity of the anaerobic reactor decreased to 10123 mg/L as $CaCO_3$ at end of the experiment.

Until the 29th day of the experiment, alkalinity concentrations in the Reactor 2 increased to its highest value (11145 mg/L as $CaCO_3$) due to the rapid biodegradation. At the end of the experiment the alkalinity concentration in Reactor 2 decreased to 5210 mg/L as $CaCO_3$.

4.3.5. Ammonia Nitrogen and Total Kjeldahl Nitrogen

The change in leachate ammonia nitrogen and TKN values from the reactors are given in Figures 4.12 and 4.13, respectively. High concentrations of NH3-N are very toxic in anaerobic bioreactors.

In the literature it was stated that concentrations between 200 and 1500 mg/L have shown to have no adverse effects on anaerobic process, concentrations ranging from 1500 to 3000 mg/L were shown to have inhibitory effects at higher pH levels, and concentrations above 3000 mg/L were very toxic (Pohland, 1987). In this experiment, NH3-N concentrations in the anaerobic reactor do not reach inhibitory or toxic levels.

Initial NH_3N values of the reactors were 718 mg/L for Reactor 1 and 601 mg/L for Reactor 2. The leachate NH_3-N concentrations in the reactors increased rapidly at the beginning of study and reached to 1568 mg/L for Reactor 1 and 1096 mg/L for Reactor 2 due to the degradation of organic nitrogenous compounds. After methanogenic bacteria apperad in the Reactor 1 a sharp decline was observed at TKN and NH_3 -N concentrations in this reactor. In this experiment NH_3 -N concentrations were decreased due to biological assimilation. At the end of the experiment, NH_3-N concentrations in Reactor 1 and Reactor 2 were 509 mg/L and 63 mg/L, respectively. The elimination rate of NH_3-N concentrations in Reactor 1 was very low. This was due to, ammonia is stable and the most significant long-term component in anaerobic reactors (Christensen et al., 1998).

Initial TKN concentrations of the reactors were 1190 mg/L for Reactor 1 and 896 mg/L for Reactor 2. Initial TKN and NH3N concentrations were very similar since TKN was mainly formed of ammonia-nitrogen. At the end of the study TKN concentrations of the Reactor 1 and Reactor 2 were 836 and 161 mg/L, respectively.

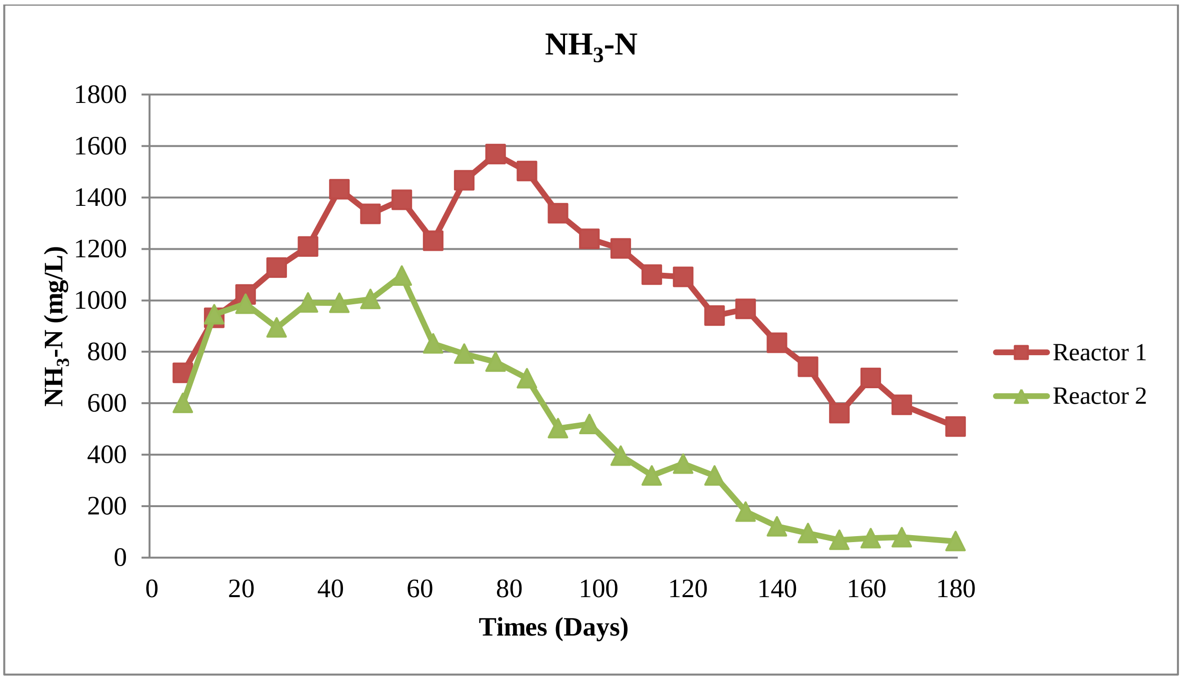


Figure 4.12. Ammonia-nitrogen concentrations in the pilot scale reactors.

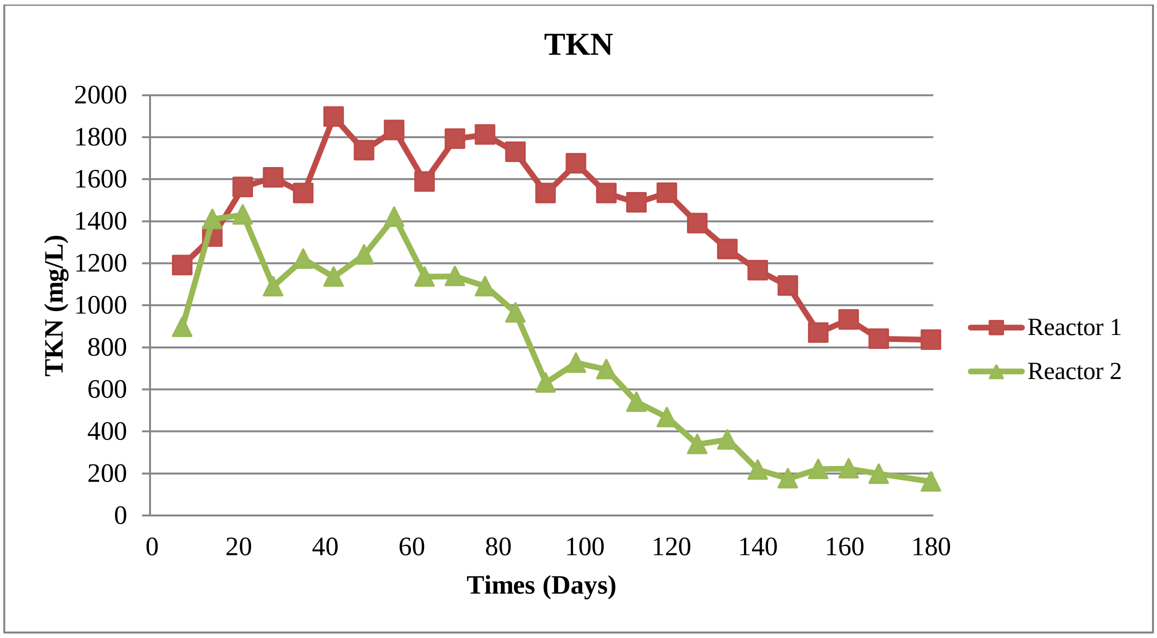


Figure 4.13. TKN concentrations in the pilot scale reactors.

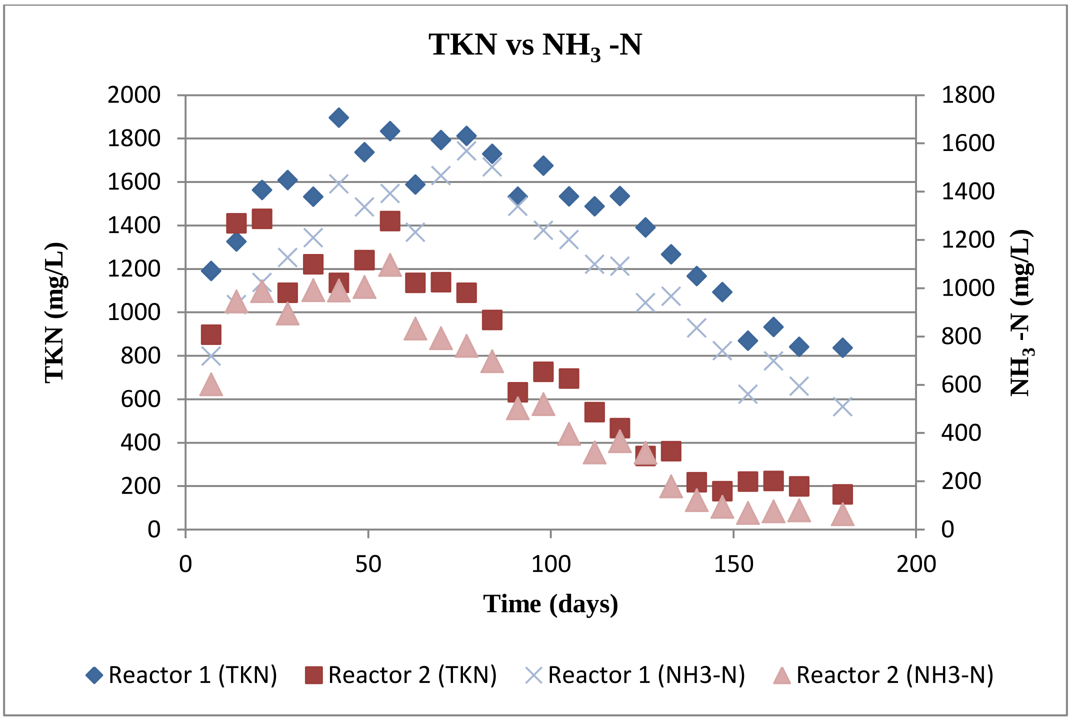


Figure 4.14. TKN and NH_3-N concentrations in the pilot scale reactors.

The change in NH_3-N and TKN concentrations with time is given in the Figure 4.14. As can be seen, NH_3-N and TKN concentrations were lower and removal efficiency were higher in the aerobic reactor throghout the experiment. Aerobic conditions NH_3 -N and TKN concentrations decreased could be attributed to nitrification process which can be confirmed with this study.

4.3.6. Volatile Fatty Acids

The change in leachate VFA concentrations from the Reactor 1 is given in Figure 4.14. Initial VFA values of the reactors were 15232 mg/L for Reactor 1 and 128 mg/L for Reactor 2.

At the beginning high TVFA concentrations were observed in Reactor 1 due toacid formation phase. At Day 56, TVFA concentrations were reached the highest value, 20335 mg/L. After this day, TVFA concentrations were stared to rapidly decrease due to VFAs were converted to methane and carbon dioxide. This situation was confirmed by a decrease in the COD concentrations and by an increase in the methane production. At the end of the experiment TVFA concentrations was 5202 mg/L. The removal rate of the VFA was 68.1% for Reactor 1.

In Reactor 2 where aerobically conditions took place, TVFA concentrations were very low due to rapid degradation of the organics. At the end of the experiment TVFA concentrations of the Reactor 2 was 32 mg/L.

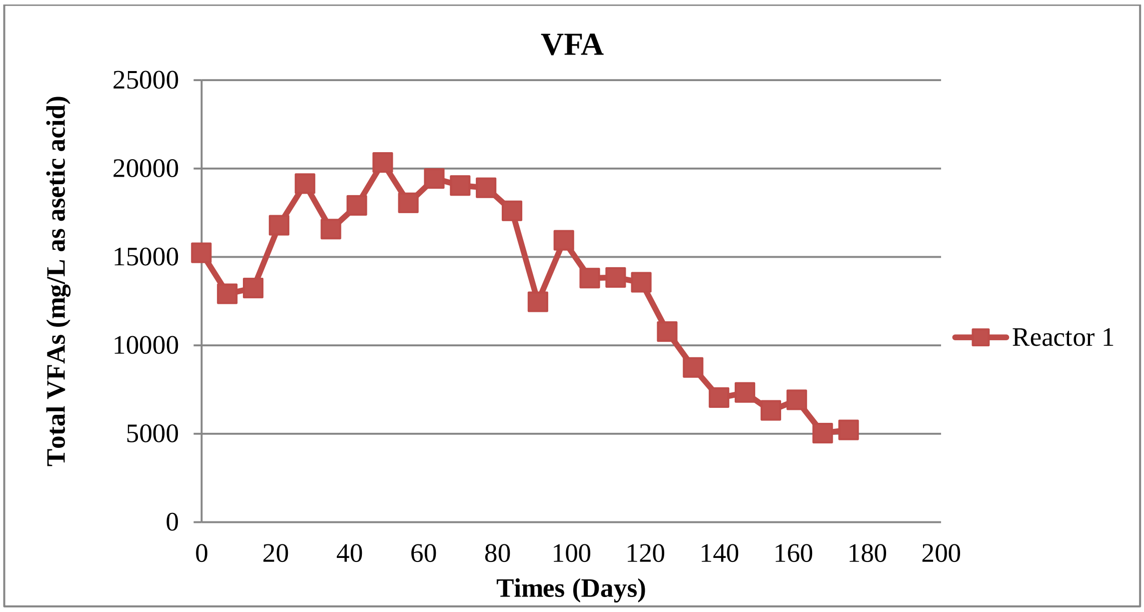


Figure 4.15. VFA concentrations in the pilot scale Reactor 1.

4.3.7. Heavy Metals

Heavy metals are important pollutants in the leachate. In this study heavy metal concentrations were measured rarely due to domestic wastes were used. The change in leachate heavy metal concentrations from the reactors are given in Tables 4.10 - 4.16.

At the beginning of the experiment, heavy metal concentrations were higher in Reactor 1 due to seed sludge. In this study, no significant changes were observed in the heavy metal concentrations. Heavy metal concentrations did not affect to the biological activity in the reactors.

Table 4.10 The heavy metal concentrations at day 30.

Reactor	**Cr (mg/L)**	**Mn (mg/L)**	**Fe (mg/L)**	**Ni (mg/L)**	**Cu (mg/L)**	**Zn (mg/L)**
Reactor 1	1.58	19.35	267.98	0.96	1.99	1.66
Reactor 2	0.54	5.32	52.29	0.14	1.38	2.13

Table 4.11 The heavy metal concentrations at day 60.

Reactor	**Cr (mg/L)**	**Mn (mg/L)**	**Fe (mg/L)**	**Ni (mg/L)**	**Cu (mg/L)**	**Zn (mg/L)**
Reactor 1	1.12	10.8	232.24	2.8	2.07	4.69
Reactor 2	0.77	5.99	50.14	1.98	1.41	2.01

Table 4.12 The heavy metal concentrations at day 90.

Reactor	**Cr (mg/L)**	**Mn (mg/L)**	**Fe (mg/L)**	**Ni (mg/L)**	**Cu (mg/L)**	**Zn (mg/L)**
Reactor 1	0.8	9.98	231.5	2.5	1.91	4.58
Reactor 2	0.51	5.84	48.9	1.67	1.62	2.44

Table 4.13 The heavy metal concentrations at day 120.

Reactor	**Cr (mg/L)**	**Mn (mg/L)**	**Fe (mg/L)**	**Ni (mg/L)**	**Cu (mg/L)**	**Zn (mg/L)**
Reactor 1	0.36	8.73	218.56	2.1	1.67	4.26
Reactor 2	0.18	4.75	43.25	1.05	1.35	2.13

Table 4.14 The heavy metal concentrations at day 150.

Reactor	**Cr (mg/L)**	**Mn (mg/L)**	**Fe (mg/L)**	**Ni (mg/L)**	**Cu (mg/L)**	**Zn (mg/L)**
Reactor 1	0.42	7.81	196.24	1.98	1.08	4.01
Reactor 2	0.17	4.99	40.16	0.89	1.42	1.82

Table 4.15 The heavy metal concentrations at day 180.

Reactor	**Cr (mg/L)**	**Mn (mg/L)**	**Fe (mg/L)**	**Ni (mg/L)**	**Cu (mg/L)**	**Zn (mg/L)**
Reactor 1	0.38	6.98	181.5	1.5	0.91	3.18
Reactor 2	0.09	4.46	37.9	0.87	0.98	1.94

4.4. Gas Analysis

Gas analyses and its compositions are the most important indicators which show progression of the anaerobic bioreactors. The final products of aerobic decomposition are water and CO_2 while hydrogen, methane, various organic acids and carbon dioxide are the final products of anaerobic decomposition (My Agriculture Informantion Bank, 2011). In this experiment, no gas measurement was done in aerobic reactor due to organic matter was completely hydrolyzed at the beginning of the study.

4.4.1. Gas Production

Gas production in the reactors were calculated by inverted cylinder method in the lab scale reactors; by wet gas meter in the pilot scale reactor.

4.4.2. Gas Production in the Lab Scale Reactors

The results of daily and cumulative gas production rates in the laboratory scale reactor are given in the Figure 4.15 and Figure 4.16, respectively.

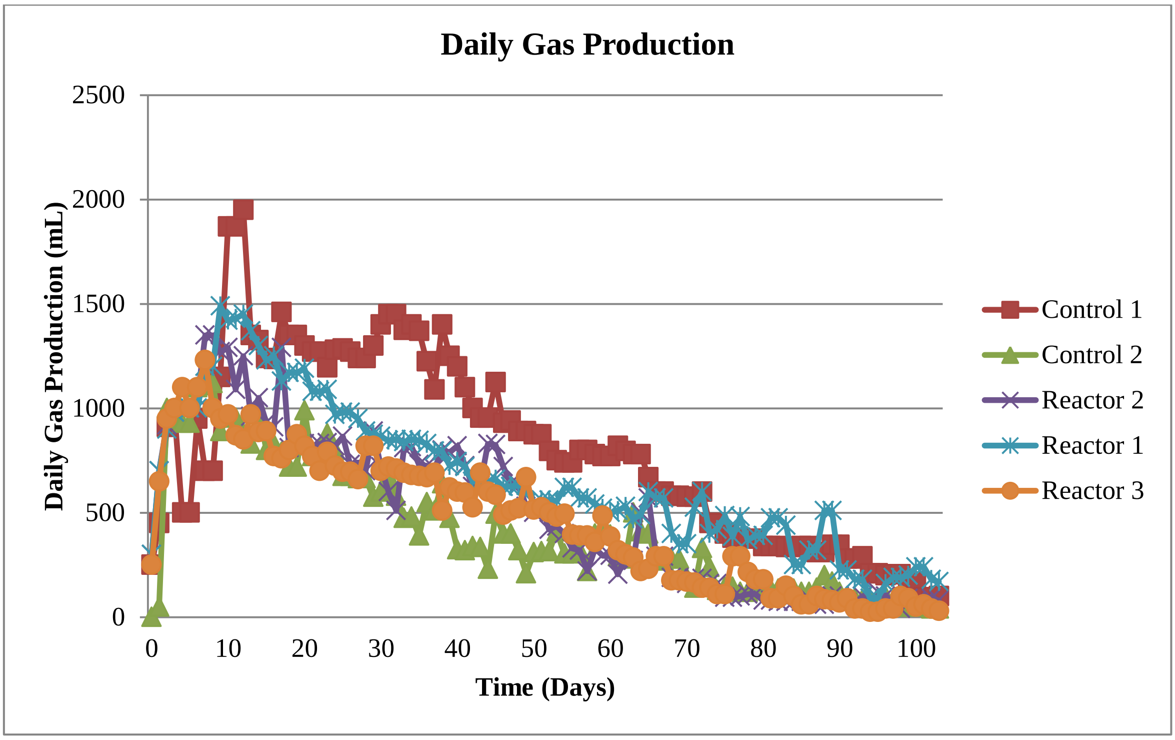


Figure 4.16. Daily gas production from the lab scale reactors.

The initial gas production rates in the Control 1, Control 2, Reactor 1, Reactor 2 and Reactor 3 were 250, 0, 300, 300, 250 ml, respectively. As it can be seen from the figure at highest sludge added reactor, observed highest gas production rates throughout the experiment. The daily gas production rate in the Reactor 1 reached its highest level (1450 ml/day) on Day 12, due to rapid hydrolyses of the organics. This was confirmed with the increased values of the concentration of TVFAs and COD. The same period observed in the other reactors; for the Control 1 reactor 1950 ml/day at Day 12, Control 2 reactor 1120 ml/day at Day 8, for the Reactor 2 1350 ml/day at Day 7 and for the Reactor 3 1230 ml/day at Day 7.

A sharp decline began between the Day 20 and Day 25 in the reactors. At this time of period COD concentrations started to decrease. The stabilization process started after this period in each of the reactors. At the end of the experiment the daily gas production rates in the Control 1, Control 2, Reactor 1, and Reactor 3 were 100, 40, 170, 80, 30 ml respectively.

The cumulative gas productions of the Control 1 Control 2, Reactor 1, Reactor 2 and Reactor 3 were 83535, 43805, 69155, 52735, 48565 mL respectively. For the first days of the experiment a sudden increase was observed in the reactors because of the hydrolyses of the organics. Especially, sludge added the reactors this observation was higher. After the stabilization process started the slope of the cumulative gas production figure was on the decrease. This is because the daily gas production rates of the reactors were decreased.

For the reactors, each of them showed the similar trend and the reactors had higher COD and sludge addition showed higher gas production rate throughout the experiement. Control 1 reactor which has the 1:0 sludge to waste ratio reached highest daily gas production (1950 ml/day) and cumulative gas production (83535 ml) rates. Reactor 1 which has the 1:4 sludge to waste ratio follows this reactor with the daily gas production of (1450 ml/day) and cumulative gas production of (69155 ml) rates. Control reactor which has no sludge addition showed the lowest daily gas production (1120 ml/day) and cumulative gas production (43805 ml) rates. Therefore, these results may be concluded that the sludge addition was enhancing decomposition and gives the most promising gas production rates.

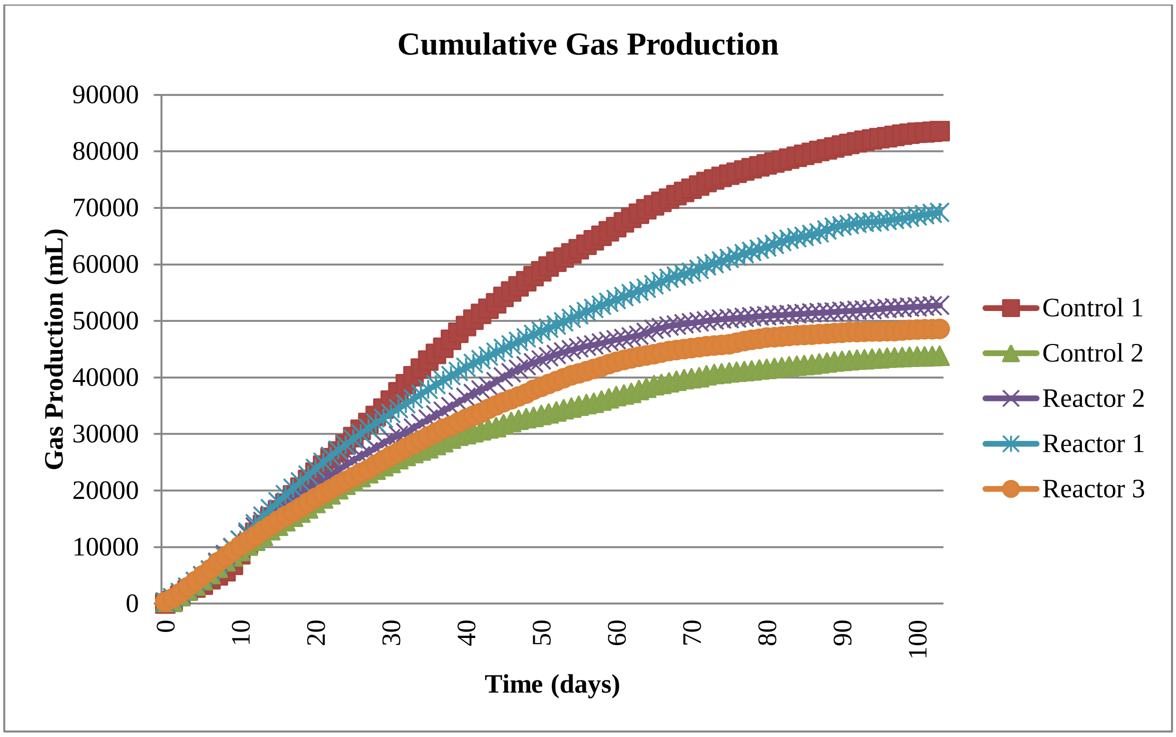


Figure 4.17. Cumulative gas production in the lab scale reactors.

4.4.3. Gas Production in the Pilot Scale Reactor

The results of daily and cumulative gas production in the pilot scale reactor are given in Figures 4.17 and 4.18, respectively. Initial gas production was observed on Day 2 of the experiment. After Day 16, daily gas production started to increase and on Day 28, daily gas production increased to its highest value of 24.9 L/day. After Day 108, a slight decrease of gas production was observed at this reactor. This was confirmed with the decrease in VFA and COD concentrations. The cumulative gas production of the Reactor 1 reached 2930.48 L at the end of the experiment.

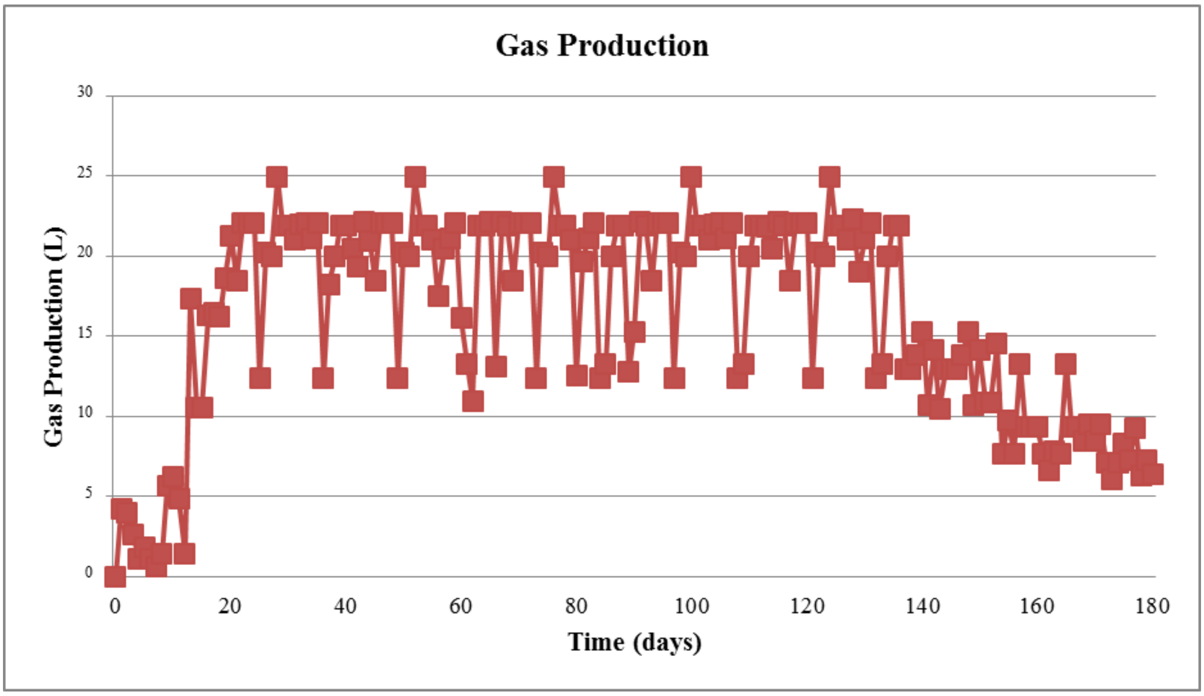


Figure 4.18. Daily gas production in the pilot scale reactor.

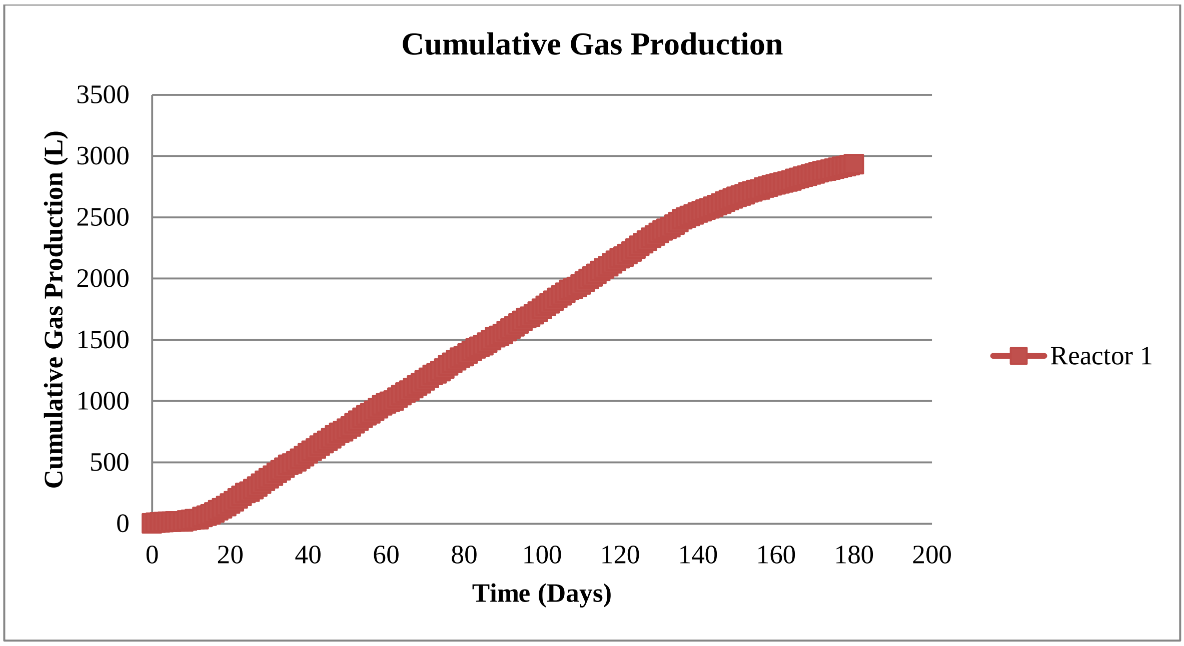


Figure 4.19. Cumulative gas production in the pilot scale reactor.

4.4.4. Gas Composition

Methane and carbon dioxide are the major gases produced under anaerobic conditions and present at approximately 55–65% and 35–45%, respectively, during the stable methanogenic phase (Christensen et al., 1996).

4.4.5. Gas Composition in the Lab Scale Reactors

The gas compositions of the Control 1, Control 2, Reactor 1, Reactor 2 and Reactor 3 are given in the Figure 4.19, 4.20, 4.21 and 4.22, 4.23 respectively. Due to nitrogen was purged in the each reactors for oxygen replacement; nitrogen content was very high at the beginning of the experiment.

For the first 21 days, no methane was observed in the reactors headspace because reactors were into the acid fermentation phase. After the 21th methane started to appear and reached its highest value between Days 77 -84.

It was observed that the percentage of methane in the reactors were fluctuated between 0-52.9%. Oxygen percentages of the Reactor 1,2,3 were between 1-4% for the first 7 days than these values were fluctuated between 0-1.5%. At the Control 2 Reactor, between Day 56 and 63, a sudden oxgen entering was happened. This was resulted in alkalinity increasing in the system.

Highest CH_4 concentrations of the Control 1, Control 2, Reactor 1, Reactor 2 and Reactor 3 were 52.9%, 45.5%, 47.9%, 44.9% and 45.9%, respectively. As can be seen, highest amount of sludge added reactors were produced more methane concentrations.

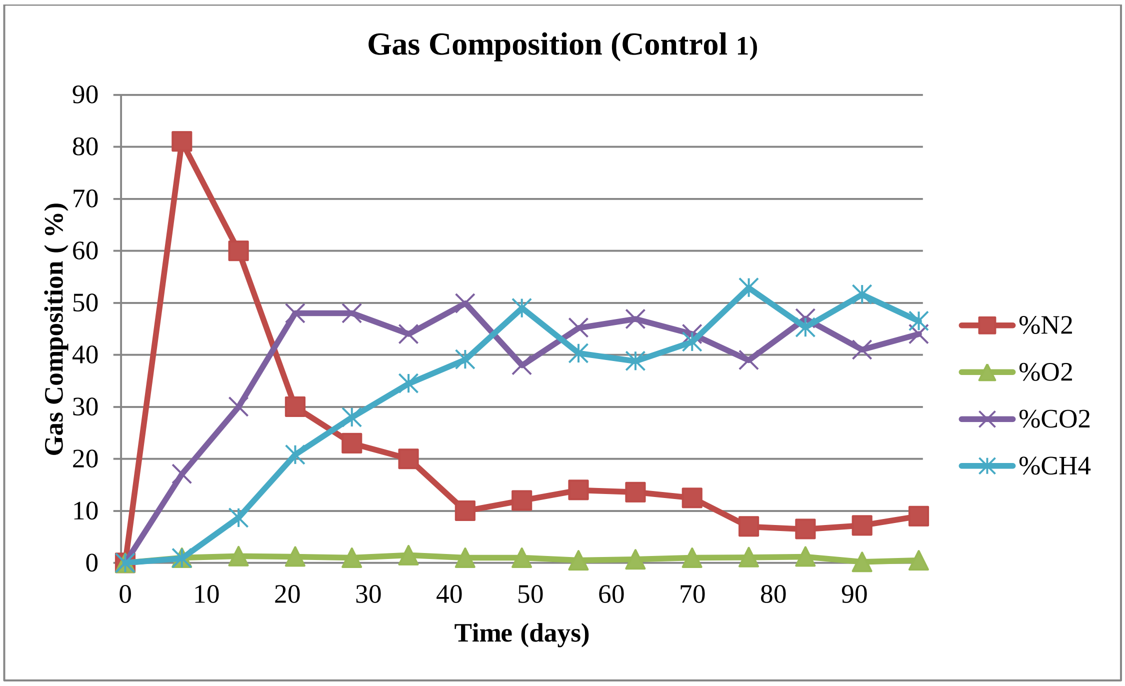


Figure 4.20. Gas composition in the Control 1 Reactor.

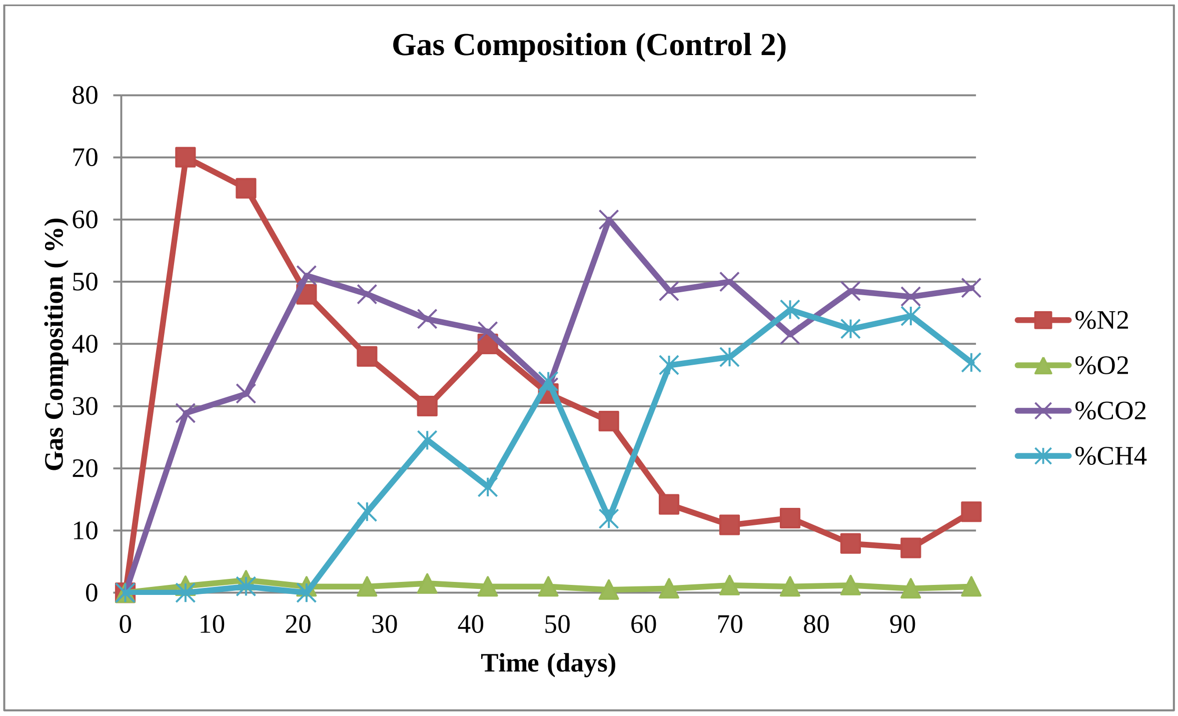


Figure 4.21. Gas composition in the Control 2 Reactor.

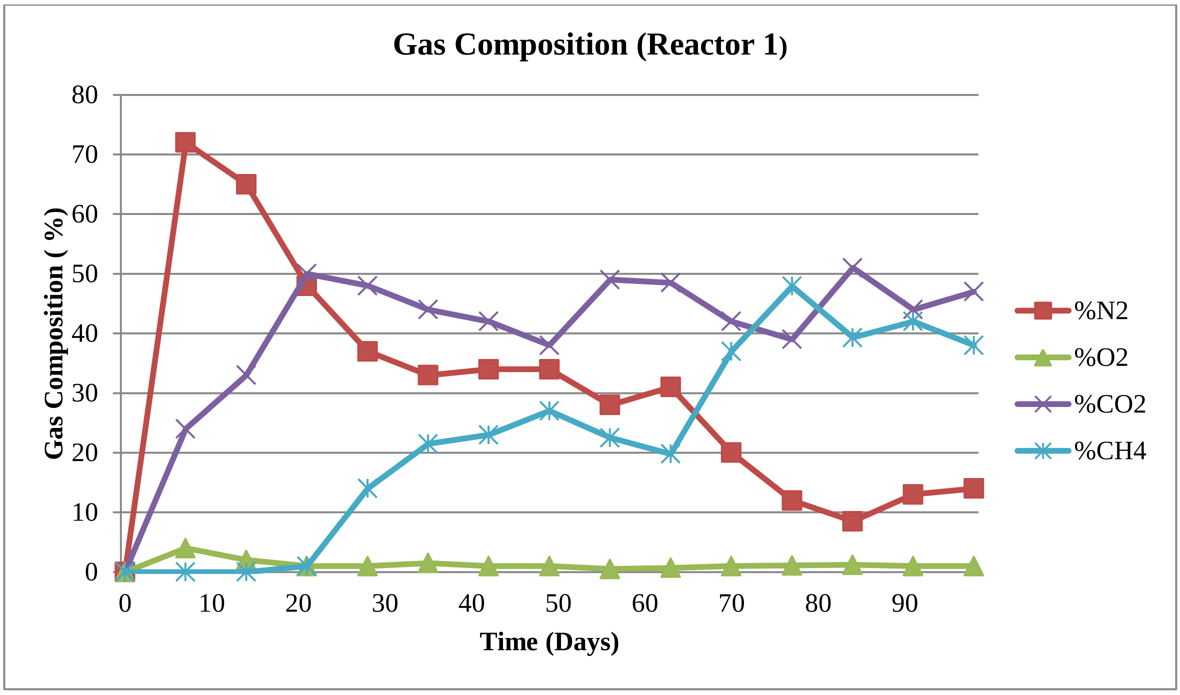


Figure 4.22. Gas composition in the Reactor 1.

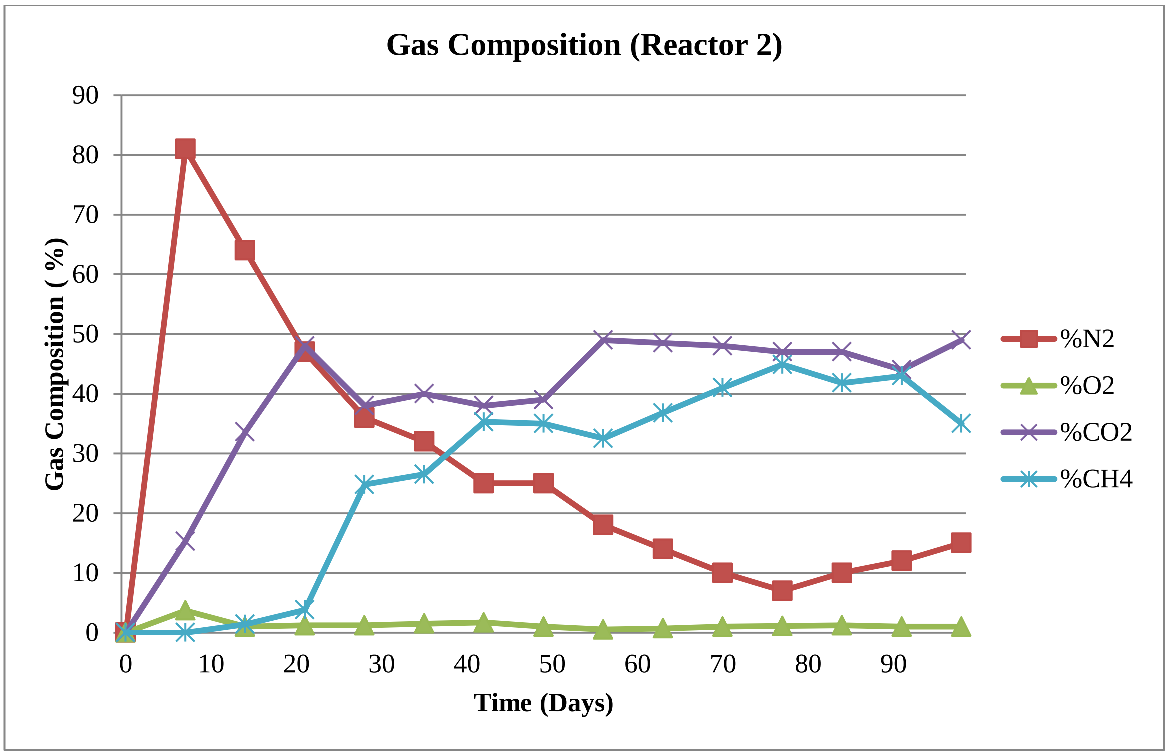


Figure 4.23. Gas composition in the Reactor 2.

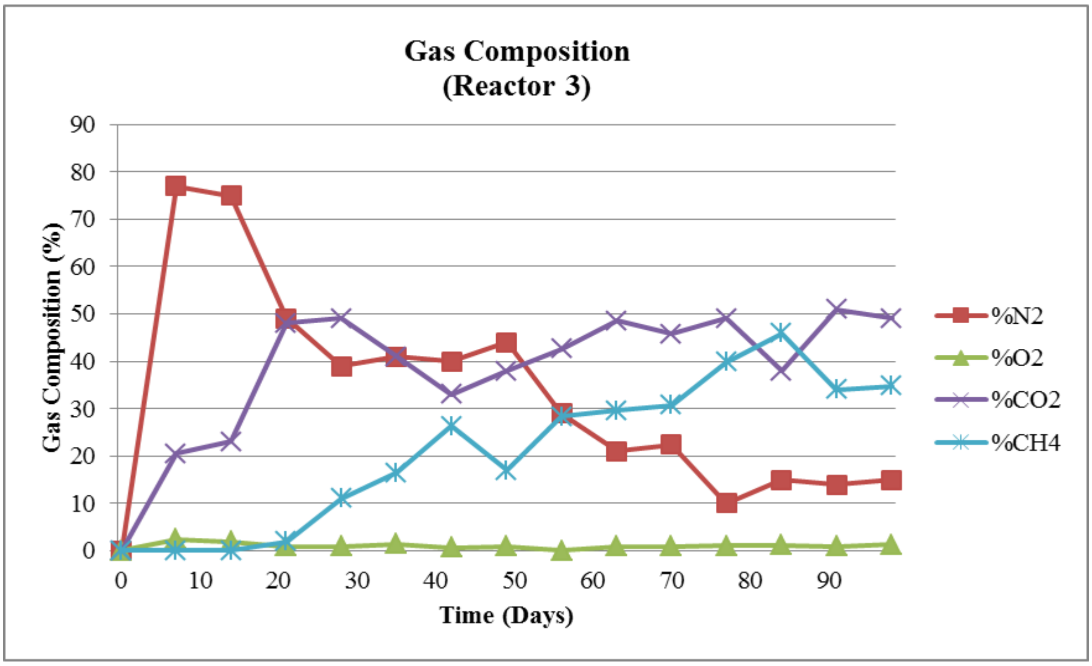


Figure 4.24 Gas Composition in the Reactor 3

4.4.6. Gas Composition in the Pilot Scale Reactors

The gas composition of the Reactor 1 is given in the Figure 4.24. Due to nitrogen was purged to reactors for oxygen replacement, nitrogen content was very high at the beginning of the experiment.

For the first 70 day of the experiment, methane gas percentage that was observed in the reactor was small (between 0 and 6.3%) because reactors were into the acid fermentation and transition phases. After Day 70, methane started to increase and reached to its highest value between the days 91-119. The highest percentage of the methane in the reactor was recorded as 60.89%. This high CH_4 concentrations in the gas along with the decrease in leachate COD, and VFA concentrations demonstrated the onset of methanogenic activities.

It was observed that the percentage of methane in the reactor was fluctuated between 50-60%. Oxygen percentage in the reactor was between 1-3.7% for the first 7 days than these values were fluctuated between 1-4%. These low O_2 concentrations did not affect anaerobic waste decomposition in the reactor.

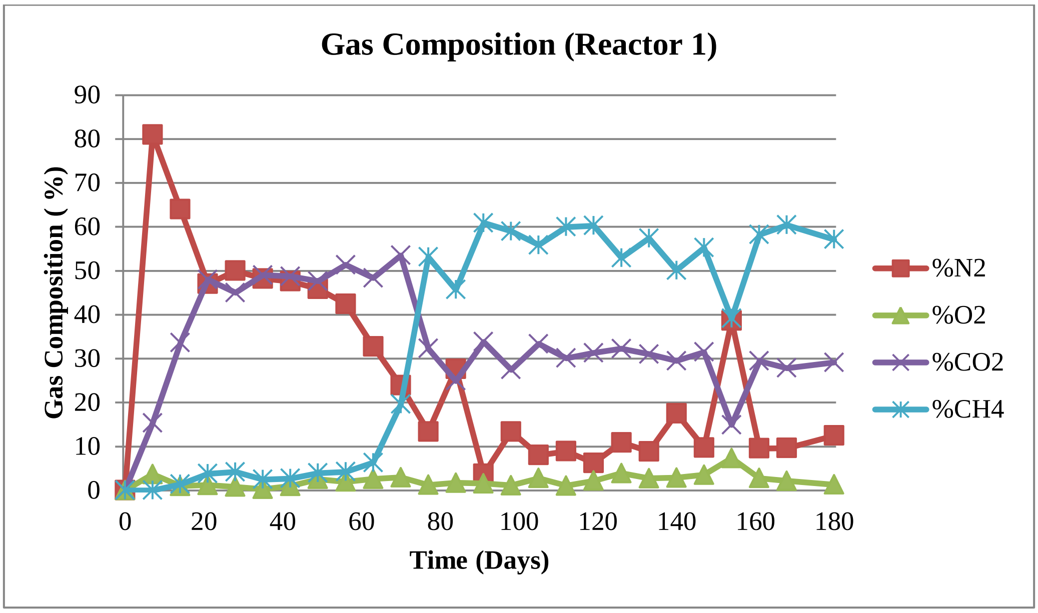


Figure 4.25 Gas Composition in the Reactor 1

4.4.7. Comparasion of the Findings

<u>4.4.7.1. Lab-Scale Reactors:</u> The results of lab scale experiments clearly demonstrated that the Reactor 1(1:4) is more suitable than other reactors due to high CH_4 content and daily and cumulative gas production. Moreover, total volatile fatty acid, heavy metals, COD and TKN removal were high in this reactor. The comparison of these values is given in Table 4.16.

Table 4.16. Comparing results of the lab scale experiments.

Experiment/Reactor	**Control 1**	**Control 2**	**Reactor 1**	**Reactor 2**	**Reactor 3**
COD Removal (%)	92.8	95.4	95.9	94.3	94.2
TKN Removal (%)	21.2	37.1	36.8	36.5	32.9
Cumulative Gas (mL)	83535	43805	69155	52735	48565
Cumulative CH_4(mL)	24841.2	6087.3	10829.5	9572.7	6093

4.4.7.2. Pilot Scale Reactors: At the end of the second stage experiments it was observed that the COD and TKN removal in aerobic reactor (Reactor 2) was more promising than the anaerobic reactor. The comprasion of these values is shown in Table 4.17. Especially, pollutant concentrations were lower in Reactor 2 and stabilization process was faster in this reactor. This can be explained by the oxidation provided by oxygen. On the other hand, CH_4 production in Reactor 1 was found high and promising for energy recovery.

Table 4.17. Comparing results of the pilot scale reactors.

Experiment/Reactor	**Anaerobic**	**Aerobic**
COD Removal (%)	28.21	88.11
TKN Removal (%)	29.74	82.03

5. SUMMARY AND CONCLUSIONS

As a result of the increase in population and industrilization, sewage sludge production increases and becomes a very big problem around the world. To solve this situation, researchers looking for an alternative method for sludge disposal. One of the economical way to disposing sewage sludge is codisposal with MSW. With this purpose this study was done to investigate the impact of aerobic and anaerobic digestion processes on municipal solid waste codisposed with sewage sludge by utilizing two stage laboratory experiments. These stages include, the lab-scale reactor set-up in order to determine optimum solid waste to sludge ratios and then anaerobic versus aerobic simulated bioreactors to understand the effect of sludge addition on waste stabilization process.

In accordance with objectives of the study, first stage lab scale experiments findings can be summarized as follows:

- Leachate pH values in the reactors were reached neutral values and were fluctuated between 7.0-8.0. This value was optimum for the methanogenic bacteria.
- Leachate ORP values in the reactors reached to the negative values after Day 27. The low ORP values were indicated for the onset of methanogenic conditions.
- Sludge addition of the reactors resulted in COD increase in the reactors. At the end of the experiment, althought Reactor 1 was reached higher values than the other reactors, stabilization process had the same efficiency as the other reactors. These results showed that codisposal can be a positive solution for sludge disposal. The removal rates of COD concentrations were 92.8% for Control 1, 95.4% for Control 2, 95.9% for Reactor 1, 94.3% for Reactor 2 and 94.2% for Reactor 3.
- Alkalinity concentrations in all reactors throughout the experimental study were observed to be sufficient to buffer the possible effects of the volatile fatty acids released as a result of decomposition of the waste.
- NH_3-N and TKN removal efficiency of the reactors were 29.5% and 21.2% for Control 1, 30.8% and 37.1% for Control 2, 27.4% and 36.8% for Reactor 1, 36.6% and 36.5% for Reactor 2 and 10.5% and 32.9% for Reactor 3, respectively. There were no significant changes in the nitrogen concentrations were observed in the

reactors. This can be showed codisposal has no negative effect on nitrogen concentrations.

- The highest VFA concentrations were observed at the Reactor 1. This may explained with, sludge addition speeded up the VFA concentrations in Reactor 1.
- Control 2 Reactor reached highest daily gas and cumulative production rates (1950 mL/ day and 83535 mL, respectively). Reactor 1 which has the 1:4 sludge to waste ratio followed this reactor with daily gas production (1450 mL/day) and cumulative gas production (69155 mL) rates. Control reactor which has no sludge addition showed the lowest daily gas production (1120 mL/day) and cumulative gas production (43805 mL) rates Highest CH_4 concentrations in Control 1, Control 2, Reactor 1, Reactor 2 and Reactor 3 were 52.9%, 45.5%, 47.9%, 44.9% and 45.9%, respectively. As can be seen, sludge codisposal has positive effects on gas and methane production.

Second stage pilot scale experiments findings can be summarized as follows:

- Leachate pH values in the reactors were nearly same at the beginning of the experiment. Thereafter, leachate pH value in the areobic reactor started to increase very rapidly and flactuated between 8-9; while in the anaerobic bioreactor this value was near the neutral. Neutral pH values were observed due to, in the aerobic reactor pH increases by air.
- Leachate ORP values in the anaerobic reactor reached negative values and flactuating between -100 to -300 mV while ORP values of the aerobic reactor flactuated between +100 to +500 mV. Leachate ORP value in Reactor 2 began to be negative throughout the first 22 days due to rapid organic degradation.
- In aerobic reactor, decrease in COD values was faster than the anaerobic bioreactor due to aeration and it fastens biodegradation of organic matter. COD removal rates in the reactors were 28.21% in the anaerobic reactor and 88.11% in the aerobic reactor.
- Alkalinity concentrations in all reactors were sufficient enough to buffer the VFAs.
- Initially, nearly same NH_3-N and TKN concentrations were observed in the reactors. NH_3-N and TKN removal efficiency in the reactors were 29% and 29.74%

in Reactor 1 and 89.5% and 82.03% in Reactor 2. Results showed that the removal of NH_3-N and TKN concentrations were higher in Reactor 2, because of ammonia nitrogen probably oxidized to nitrite and nitrate by nitrifying bacteria. In the anaerobic reactor NH_3-N removed due to ammonia nitrogen was used by microorganisms during the active waste stabilization.

- Cumulative gas production in Reactor 1 was 2930.48 L at the end of the experiment. The highest percentage of the methane in the reactor was recorded as 60.89% . At the end of the experiment CH_4 concentrations in Reactor 1 was 57.1%. These values showed that sludge codisposal in bioreactors is a very promising option.

The results of the experiments clearly showed that sludge codisposal is an important alternative method to sewage sludge disposal. Landfilling of sludge has significant advantages such as easier handling, acceleration of waste stabilization rates, smaller disposal/treatment area requirements and lower capital investment when compared to other disposal techniques. Also, landfilling of sludge with MSW is an alternative method to produce energy. Especially, in the anaerobic bioreactors sludge codisposal improved methane yield and the COD and TKN removals. In this lab scale experiment, higher sludge content reactor (Reactor 1), showed clearly these results. On the other hand, at the end of the second stage experiments it can be observed that according to the COD and TKN removal rates aerobic reactor (Reactor 2) was more promising than anaerobic reactor. However, due to the high CH_4 production in Reactor 1, an alternative option for energy production can be investigated by a more detailed feasibility studies.

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