EFFECTS OF DIFFERENT RATIOS OF TREATMENT PLANT SLUDGE ON THE STABILIZATION OF SOLID WASTES AT ANAEROBIC AND AEROBIC LANDFILL AREAS

by
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ABSTRACT

Sanitary landfilling is one of the solid waste disposal methods. Sanitary landfills have created problems related to biogas emissions, generation of leachate, and long times required for the area recovery.

Aerobic landfilling is a new solid waste disposal method. The supply of proper air flow and leachate recycle to an aerobic landfill accelerates the decomposition of organic materials.

In this study, the reactors were loaded with solid wastes mixed with different types and ratios of sludge in order to find out the stabilization of solid wastes under anaerobic and anaerobic conditions. Moreover, after 55th day of the research, both aerobic and anaerobic decomposition rates were tried to enhance by using yeast as additive.

The results of the study suggested that disposal of sludges mixed to the solid wastes was an effective technique for the stabilization of the solid wastes. At the end of the analyses, the highest ratio 1/4 was found as the optimum sludge ratio for the stabilization of the solid wastes in both the aerobic and the anaerobic reactors.

Moreover, the yeast solution addition was an effective technique for the enhancement of biodegradation. Among the different reactor types, the aerobic reactors were more effective than the anaerobic ones.

ÖZET

Katı atık bertaraf metodlarından biri de düzenli depolama alanlarıdır. Düzenli depolama alanları, biyogaz ve toksik çöp sızıntı suyu üretmesinin yanında, bu alanların uzun sürelerde rehabilite edilmesine neden olan bir çok problem yaratmaktadır.

Aerobik katı atık depolama alanları yeni bir atık bertaraf yöntemidir. Aerobik katı atık depolama alanlarında sağlanan uygun miktardaki hava akışı çöp sızıntı suyu devri organik maddelerin parçalanmasını artırmaktadır.

Bu çalışmada deney reaktörleri farklı türlerde ve oranlarda karıştırılmış atıksu çamuru ile katı atıkların hem anaerobik hem de aerobik stabilizasyonu belirlenmeye çalışılmıştır. Ayrıca deneyin 55. gününden sonra, reaktörlere maya eklenerek, aerobik ve anaerobik parçalanma hızlandırılmaya çalışılmıştır.

Bu çalışma sonunda, farklı türlerde ve oranlarda atıksu çamuru ile evsel içerikli çöplerle bertarafı etkin bir katı atık stabilizasyon tekniği olduğu belirlenmiştir. Analizlerin sonunda, 1:4 oranına sahip çamurun aerobik ve anaerobik reaktörlerde en fazla katı atık stabilizasyonu verimi sağlayan çamur oranı olduğu saptanmıştır.

Ayrıca, maya eklemenin de biyolojik parçalanmayı artıran etkin bir teknik olduğu saptanmıştır. Reaktör tipleri arasında ise aerobik reaktörlerin anaerobik reaktörlerden verim olarak üstün olduğu gözlenmiştir.

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Symbol	**Explanation**	**Units used**
COD	Chemical Oxygen Demand	(mg/L)
MSW	Municipal solid waste	
ORP	Oxygen Reduction Potential	(mV)
TKN	Total Kjeldahl Nitrogen	(mg/L)

1. INTRODUCTION

The population growth is one of the significant problems faced in all over the world. Increase in the population, change in the life standards, and consequently industrialization and urbanization result in the generation of increasing in the amount and the diversity of solid wastes. This leads to a rise in the cost of treatment and disposal of these wastes. The interest in alternative municipal solid waste (MSW) management strategies will likely intensify as the production of MSW continues to increase while the disposal capacity via traditional landfilling is diminishing (Stroot et al., 2001). The difficulty of finding new proper landfill spaces is a very serious problem, especially in the densely populated areas (Nagao et al., 2003).

Public concern regarding the environmental issues played an important role in searching for the new and alternative methods for the elimination and management of wastes. In the economically developing countries, the lack of finance, technology and qualified personnel prevent the implementation of acceptable solid waste treatment and disposal options.

With respect to the other alternatives such as incineration and open dumping, disposal in sanitary landfills remains as the major disposal method for the solid wastes. The flexibility and relative simplicity of the technology of sanitary landfilling are well-suited for the developing countries as a means of managing the disposal of wastes. Sanitary landfilling controls the harmful effects of solid wastes placed on the land.

The solid wastes in sanitary landfill areas are decomposed by physical, chemical and biological processes under anaerobic conditions (Cossu et al., 1997). If the anaerobic decomposition in sanitary landfill areas is not controlled properly, large amount of toxic leachate and biogas will be generated. This causes unfavorable impacts on the environment and risks to human health both during the operation and after the closure of the landfill areas.

On the other hand, if the anaerobic decomposition processes are operated properly, biogas generated in sanitary landfills can be used as an important energy source. For that purpose, leachate recirculation, sludge addition, buffer, yeast, and nutrient addition are used to enhance the anaerobic decomposition (Özkaya et al., 2004)

Biogas can be defined as a gas mixture produced by the fermentation of organic wastes under the anaerobic conditions. Biogas consists of 60-70 per cent methane, 20-30 per cent carbon dioxide, and trace amounts of hydrogen sulfide and other gases. Methane is a valuable end product because of its energy content and the methanogens are responsible for the methane production processes. However, it is difficult to study with the methanogens because of their sensitivity to oxygen and to the other environmental parameters (Metcalf and Eddy, 1991).

As the landfill layers used for the impermeability will deteriorate in the environment by time, toxic leachate will reach the groundwater and pollute it. Also, the slow stabilization of the wastes under anaerobic conditions results in a decrease in the settlement potential of sanitary landfills. These adverse impacts resulting from the sanitary landfills are very significant especially for the economically developing countries which can not meet the requirements of the regulations for the design and operation of the landfill areas because of their tight budgets (Read et al., 2000).

Many studies have been conducted to overcome the adverse impacts associated with the sanitary landfill areas recently. In minimizing these impacts originated from the anaerobic decomposition, minimization of the leachate generation and controlling of biogas generation in landfills are becoming more attractive in research areas.

Aerobic landfilling especially for the decomposition of organic fractions of solid wastes has gained more importance among the other solid waste management alternatives. The possibility of reducing the volume and weight of solid wastes easily made the aerobic landfills a favorable alternative. Recycling of the leachate and injection of air into the wastes result the decomposition of organic fraction in a shorter time compared to the anaerobic conditions. Consequently, the methane gas generation and leachate to be treated are reduced (Peter, 1990, Read et al., 2000).

Sludge is a common by-product of the water and wastewater treatment plants as well as the many industrial and commercial activities. Landfilling is one of the disposal methods used for the disposal of the sludge. Improper landfilling of sludge created potential environmental hazards including the generation of odor, methane gas, and more leachate (Öztürk et al., 1999). The options for the disposal of sludge depend on the water content, organic and metal contents of the sludge (Irene et al., 1995).

Stabilization of treatment plant sludges in a beneficial way is necessary from the points of economical, environmental and health issues. Co-disposal of MSW with sludge offers many opportunities in both anaerobic and aerobic landfilling. Co-disposal can be defined as the mixing of sludge with solid wastes prior to disposal in a landfill. Sludge and solid wastes are disposed together in many sanitary landfills. The pathogens and high nitrogen content of sludge can be reduced by the co-disposal of sludge with solid wastes while minimizing the potential problems by proper stabilization in aerobic and anaerobic landfills (January et al., 2004).

The main objectives of this research are to compare the stabilization of the solid wastes by mixing different ratios of sludge and to investigate the effects of the yeast on the decomposition rates of organic materials under anaerobic and aerobic conditions. The results obtained from both aerobic and anaerobic reactors were compared to find the most effective landfilling technique.

2. LITERATURE REVIEW

2.1. Solid Waste Management

The rapid developments of some cities in Turkey in the past decades created environmental problems that pose a serious threat to the environment. Solid waste is one of these significant problems that create a great threat to the environment and public health.

Solid wastes comprise all the wastes arising from human and animal activities that are normally solid and are discarded as useless or unwanted (Tchobanoglous et al., 1993). Solid wastes are a by-product of economic and social activities. Thus, the solid waste content consists of different materials such as organic waste, recyclable material, some kind of metals and industrial wastes. As an example, the composition of solid wastes in major cities of Turkey is given in Table 2.1.

The main component of the solid wastes generated in economically developing countries is the organic wastes which is mostly the food remains having moisture content as high as 80 per cent. This value is around 25-40 per cent in the solid wastes of the developed countries. The high moisture content results in differences in the duration of the degradation periods of the solid wastes. This situation consequently cause to some variations in the compositions and amounts of leachate and biogas generated (Kocasoy, 2000).

While the average solid waste generation was 0.6-0.7 kg per capita per day in 1990; today this amount reaches approximately to one kg per capita per day in Turkey. It is estimated that the annual amount of solid wastes generated in the whole country is almost 23 million tones per year. 76.84 per cent of this amount is dumped in open dumps, 1.65 per cent is disposed in rivers, seas, lakes, 1.94 per cent is burned in open areas, 0.5 per cent is composted; 3.66 per cent is used as filling material and 12.99 per cent is disposed in sanitary landfills (Kirkitsos et al., 2002, Kocasoy, 2000).

In most of the economically developing countries, like Turkey, municipal budgets are not sufficient for the construction of proper sanitary landfill areas, therefore, open dumping is the preferred method for the MSW disposal by most of the municipalities. The disposal of municipal solid wastes to the open dump areas are made randomly. All types of waste materials like treatment plant sludge, chemical wastes and debris are also dumped into these municipal solid waste disposal areas. Solid wastes are dumped into these places without taking any precaution. They are neither compacted nor covered. The open dumps do not have any leachate or methane collection systems and they continuously pollute the environment while causing significant problems for the environment, public health and the welfare of the population (Kocasoy, 2000).

In contrast to the open dumps, in a modern sanitary landfill, refuse spread in thin layers, each of which is compacted and covered before the next spread. Moreover, biogas and leachate produced in landfills were collected by collection systems. In Turkey, there are eleven sanitary landfills currently functioning. Moreover, many new sanitary landfills have been under construction. There are almost 2915 open dumps in Turkey. However, most of the open dump areas have been closed and rehabilitated (http://www.cevreorman .gov.tr/atik_01.html).

Table 2.1. Municipal solid waste composition in major cities of Turkey (Metin et al., 2002)

Parameter	**Bursa (Bursa Municipality, 2000)**	**İstanbul (İstanbul Municipality, 2000)**	**İzmir (İzmir Municipality, 2000)**	**Adana (JICA, 2000)**	**Mersin (JICA, 2000)**
population, (person)	1,958,529	9,198,809	3,114,859	1,682,483	1,508,232
organic, (%)	53.1	43	46	64.4	63
recyclable, (%)	36.4	33	31	25.2	29.4
paper/board, (%)	18.4	7.8	12	14.8	18.42
plastics, (%)	11.6	14.2	12	5.92	6.69
metal, (%)	3	5.8	3	1.4	1.25
glass, (%)	3.4	6.2	4	3.08	3.08
others, (%)	10.5	23.1	23	11.4	7.6

2.2. Sludge Management

Throughout the world, the treatment of the water and wastewater treatment plants cause generation of sludge in large volumes. Sludge quantities increase continuously, but the options for the sludge disposal are limited because of the stringent regulations enacted to protect the environment. Thus, treatment and disposal of the sludge is one of the significant problems of any water and wastewater treatment process (Matthews, 1998).

Sludge consists of a high content of organic and inorganic compounds. Its solid content is usually between one to five per cent in a liquid. In water treatment plants, sludge is mostly produced in the preliminary and primary treatment, sedimentation, and filtration units. Aerobic treatment plants produce more sludge than the anaerobic treatment plants. The wide variety of incoming wastewater and available treatment technologies determine the volume and the characteristics of wastewater treatment plant sludges. So, sludge quality depends on the composition of the incoming wastewater and the applied treatment methods.

Organic fraction of sludge is the main reason for its putrescibility. On the other hand, the presences of inorganic components of sludges cause another adverse effect on the environment. Therefore, stabilization of wastewater sludges is inevitable from the point of economical, environmental, and health reasons. Sludge stabilization prevents generation of the offensive odors, and reduces pathogens and water content. There are several techniques for sludge disposal such as land application, surface disposal, landfilling, incineration, ocean disposal, and others (Sosnowski et al., 2003).

When these techniques are evaluated, land application strictly depends on the sludge quality and its metal contents. Incineration is neither efficient nor cheap. Ocean dumping is banned by many countries. Landfilling of sludge is a common method and is considered as an acceptable practice in the world. Different countries are taking a variety of approaches to the treatment, disposal and reuse of sludges depending on the quantity and quality of the sludges produced as well as the regulations, and prevailing public attitudes (Read et al., 2000).

2.3. Cost of the Management of the Solid Waste and Sludge

The majorities of generated solid wastes end up in some sort of land disposal facility, either sanitary landfill or open dumps. In the sanitary landfill areas, slow stabilization of the wastes, landfill gases production, generation of harmful leachate, and long times required for the area recovery are the major drawbacks. The operation cost of landfill area is in the range of 18-67 per cent and approximately 25 per cent of the whole budget allocated for the solid waste management (Kocasoy, 2000).

Wastewater treatment plants generate different types and volumes of sludge. The treatment of this sludge is necessary before disposing of it. However, environmental protection policies are making the disposal and use (as biosolids) more difficult (Matthews, 1998). There are many sludge disposal techniques such as incineration, anaerobic digestion, landfilling, composting, etc., but most of these disposal techniques depend on sludge qualities. Therefore, the disposal of sludge is difficult and expensive, often requiring over 50 per cent of the operating budget for the wastewater treatment plant (Vesilind et al., 2001).

2.4. Classification of Landfill Areas

Landfills are used for the isolation of the disposed wastes from the environment to bring the wastes to an unharmful condition to the environment and public health. Generally, landfills can be classified as anaerobic, aerobic, and semi-aerobic landfills. The biological, chemical, and physical processes play significant roles in the landfills. Biological processes in landfills may occur either aerobically or anaerobically.

Over the last decades, there has been a considerable increase in the utilization of the anaerobic landfills for the disposal of the solid wastes. In the anaerobic landfills, the organic portion of the wastes is broken down by the anaerobic microorganisms in the absence of free oxygen. Biogases are generated in landfills through the anaerobic decomposition of the organic fraction of the solid wastes. If a significant amount of methane is present, it may be explosive. In addition to the methane, toxic leachate is also produced in landfill areas.

In contrast to the anaerobic landfill, the biological decomposition takes place in the presence of free oxygen in the aerobic landfills. The final products are water, carbon dioxide, and new biomass. Aerobic landfills are safer and better methods for the land disposal than the anaerobic landfills since waste is stabilized much more rapidly. Moreover, the generation of leachate and landfill gasses is hindered at aerobic landfills (Hudgins and Harper, 1997). Semi-aerobic landfills can be defined as a combination of aerobic and anaerobic landfills. Biological processes in semi-aerobic landfills occur both at the aerobic and anaerobic conditions simultaneously.

2.5. Anaerobic Sanitary Landfills

In recent times, understanding and controlling of the stabilization processes in landfills are essential to prevent problems generated from the landfill areas. Physical, chemical and microbial processes occur in the landfills simultaneously. These processes can be summarized as the decay of organic matters biologically, chemical oxidation of waste constituents, dissolution and transportation of constituents, diffusion and transportation of generated gases, hydraulic liquid transport and settlement (Pohland et al., 1993). The decomposition processes produce solid, liquid and gaseous by-products such as leachate and biogases.

2.5.1. Microbiology of Anaerobic Landfill Stabilization

Landfill is an extremely variable and heterogeneous environment. There are many microorganisms in the landfilled wastes. In the landfill areas, the solid waste decomposition begins aerobically and than continues in the anaerobic conditions. Aerobic organisms convert the organic fraction of wastes to carbon dioxide, water, and energy. However, later the remaining organic fraction of wastes is converted to carbon dioxide, ammonia, hydrogen sulfide and methane by the anaerobic organisms.

The important phases of the anaerobic decomposition of the solid wastes are hydrolysis, acidogenesis, acetogenesis and methanogenesis. A schematic presentation of anaerobic methane fermentation is shown in the Figure 2.1 (Christensen and Kjeldsen, 1989).

The hydrolysis process is the first step to initiate the decomposition of the solid waste in the landfill stabilization. Reducing complex organic matters to their soluble components via extra cellular enzymes allows their soluble components transport across the cell membrane. The hydrolysis products of complex organics such as carbohydrates, proteins and lipids are soluble sugars, amino acids, fatty acids and glycerol.

When hydrolysis does not occur, the bacteria can not use complex organic matters in their metabolisms. Hydrolysis is performed by hydrolytic and fermentative bacteria (Farquhar and Rovers, 1997). It was stated that hydrolysis can be the rate limiting step of anaerobic conversion of solid wastes (San, 1999). On the other hand, it was reported that the hydrolysis probably can not be a rate limiting step in decomposition (Farquhar and Rovers, 1997).

After the hydrolysis, the acid forming bacteria begin to convert soluble compounds (sugars, amino acids, fatty acids and glycerol) to acetic, propionic, butyric acid, alcohols, ammonia, hydrogen and carbon dioxide during the acidogenesis phase. Temperature, hydrogen ion concentration (pH), alkalinity and the composition of wastes affect the efficiency of this phase directly.

In the third stage, an obligate group of hydrogen producing bacteria converts the products from the second stage to acetate, hydrogen, and carbon dioxide. Since, the acidogenesis phase products can not be used by the methanogenic bacteria directly, the activity of acetogens is considered as the rate-limiting step of the soluble part of the anaerobic digestion. Hydrogen and carbon dioxide are produced during the acetogenesis phase and consumed by the hydrogen-consuming bacteria to form methane (Gujer and Zehnder, 1983).

In the final stage, the methanogenic bacteria can utilize only a limited number of substrates, including formate, methanol, methylamine, hydrogen, carbon dioxide and acetate. It is estimated that 70 per cent of the methane is produced from acetate (Barlaz et al., 1990).

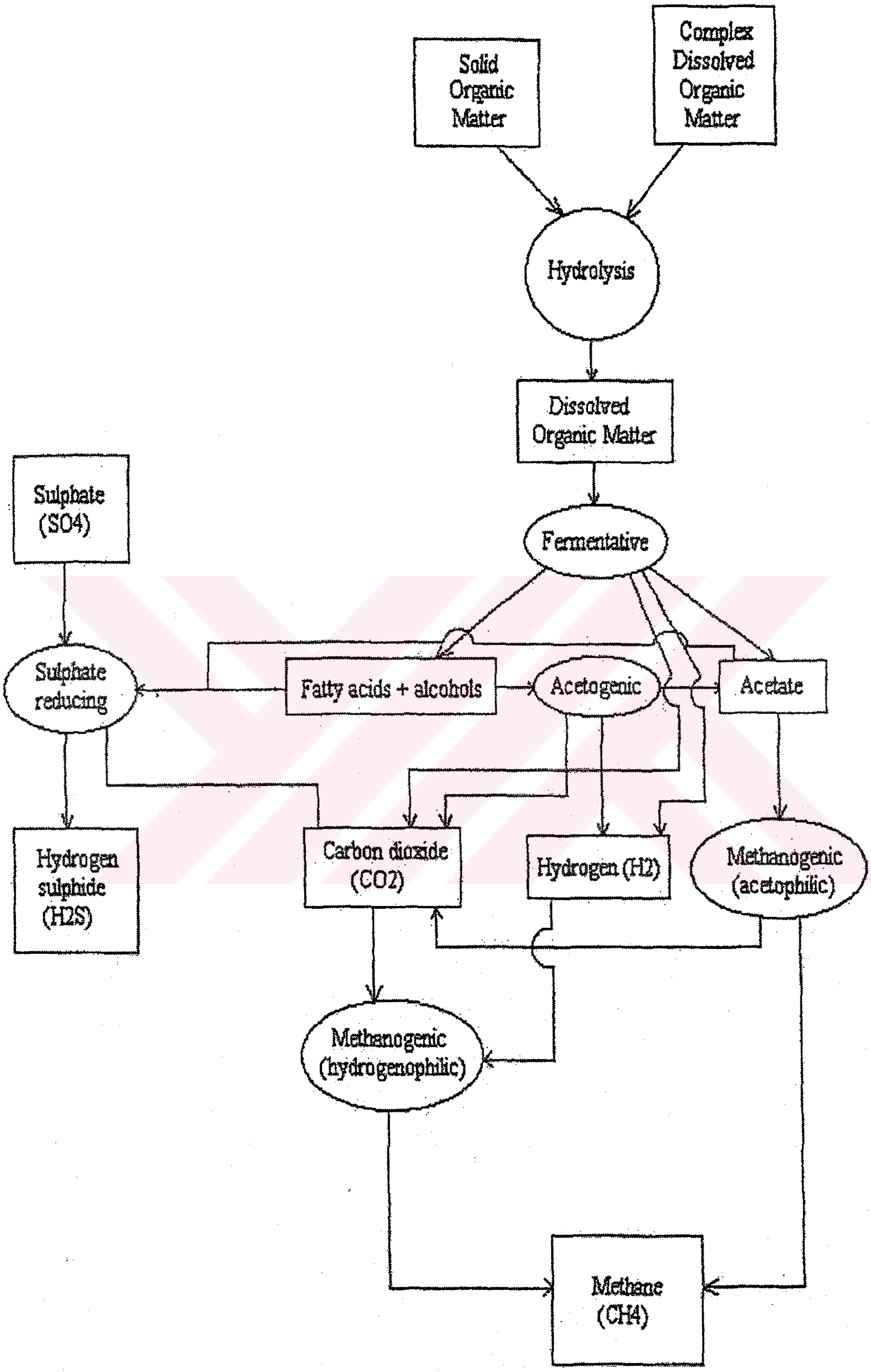


Figure 2.1. Major bacterial groups and substrates involved in the methane generating ecosystem (Christensen and Kjeldsen, 1989).

Methanogen are any of a group of one-celled organisms that obtain energy from the metabolic production of methane gas. Methane is basically produced from carbon dioxide and hydrogen. The methanogenic bacteria are most active in the pH range of 6.8 to 7.4 (Barlaz et al., 1990). The methanogenic bacteria control the pH of the ecosystem by the consumption of acetate. They regulate the flow of electrons by the consumption of hydrogen. Moreover, the methanogenic bacteria create thermodynamically favorable conditions for the catabolism of alcohols and acids and excrete organic growth factors, including vitamins and amino acids that are used by other heterotrophic bacteria in the ecosystem. If carboxylic acids and hydrogen accumulate, the pH of the system decreases, thus inhibiting methanogenesis (Barlaz et al., 1990). Some of the important methanogenic processes are given in the below equations:

$$4H_2 + CO_2 \longrightarrow CH_4 + 2H_2O \qquad (2.1)$$

$$CH_3COOH \longrightarrow CH_4 + CO_2 \qquad (2.2)$$

$$4HCOOH \longrightarrow CH_4 + 3CO_2 + 2H_2O \qquad (2.3)$$

$$4CH_3OH + H_2 \longrightarrow 3CH_4 + CO_2 + 2H_2O \qquad (2.4)$$

$$4CO + 2H_2O \longrightarrow CH_4 + 3CO_2 \qquad (2.5)$$

Sulphate exists in landfill areas in variable forms. Desulfovibrio and Desulfotomaculum are the main groups of sulfate reducing bacteria in landfills. The sulfate reducing bacteria is similar to the methanogeneous in many ways. The sulfate reducing bacteria convert hydrogen, acetic acid and higher volatile fatty acids during the sulphate reduction (Christensen and Kjeldsen, 1989).

High biologic activities of sulphate reducing bacteria cause some adverse effects on the anaerobic systems. The amount of organics available for methane production may be decreased because of sulfate reducing bacteria. It was reported that the sulphate reducing bacteria can live in wider pH ranges than the methanogenic bacteria. The sulphate reducing bacteria can survive when pH is less than 5 to 9 (Christensen and Kjeldsen, 1989).

2.5.2. Phases of Anaerobic Landfill Stabilization

There are five phases in the anaerobic landfill stabilization. Figure 2.2. illustrates the changes in the gas and leachate composition during the five phases of the landfill stabilization.

2.5.2.1. Initial Adjustment Phase. The organic decomposition of MSW in landfills begins just after the placement of the wastes into the landfill. The solid waste degradation occurs under the aerobic conditions as a certain amount of air is trapped within the landfill (Tchobanoglous et al., 1993, Farquhar and Rovers, 1997).

In this phase, carbohydrates are converted mainly to carbon dioxide and water, and proteins are reduced to amino acids. Moreover, fats are hydrolyzed to fatty acids and glycerol. This phase is characterised by the depletion of oxygen and nitrogen and the production of large quantities of carbon dioxide because of the aerobic respiration of the microorganisms (Swarbrick, 1995). Exothermic microbial decomposition processes cause to an elevation of temperature in the landfills. Moreover, moisture begins to accumulate and first changes in environmental parameters are observed. Leachate production is not observed at this phase.

2.5.2.2. Transition. After the consumption of oxygen in the refuse, the decomposition of MSW continues under the anaerobic conditions. As the landfills become anaerobic, nitrate and sulfate can be used as electron acceptors (Tchobanoglous et al., 1993). Moreover, highly chemical reducing conditions begin to develop in landfills. Therefore, carbon dioxide production is increased during this phase (Swarbrick, 1995).

The activity of acid-fermentation bacteria causes generation of volatile fatty acids and carbon dioxide. pH of the leachate begins to decrease because of the acidic environment. On the other hand, the concentration of the inorganic ions such as chloride, calcium, sodium increases. Methane production does not begin at this phase. On the other hand, first leachate generation is observed at this stage as a result of accumulation of moisture inside the landfill (Swarbrick, 1995).

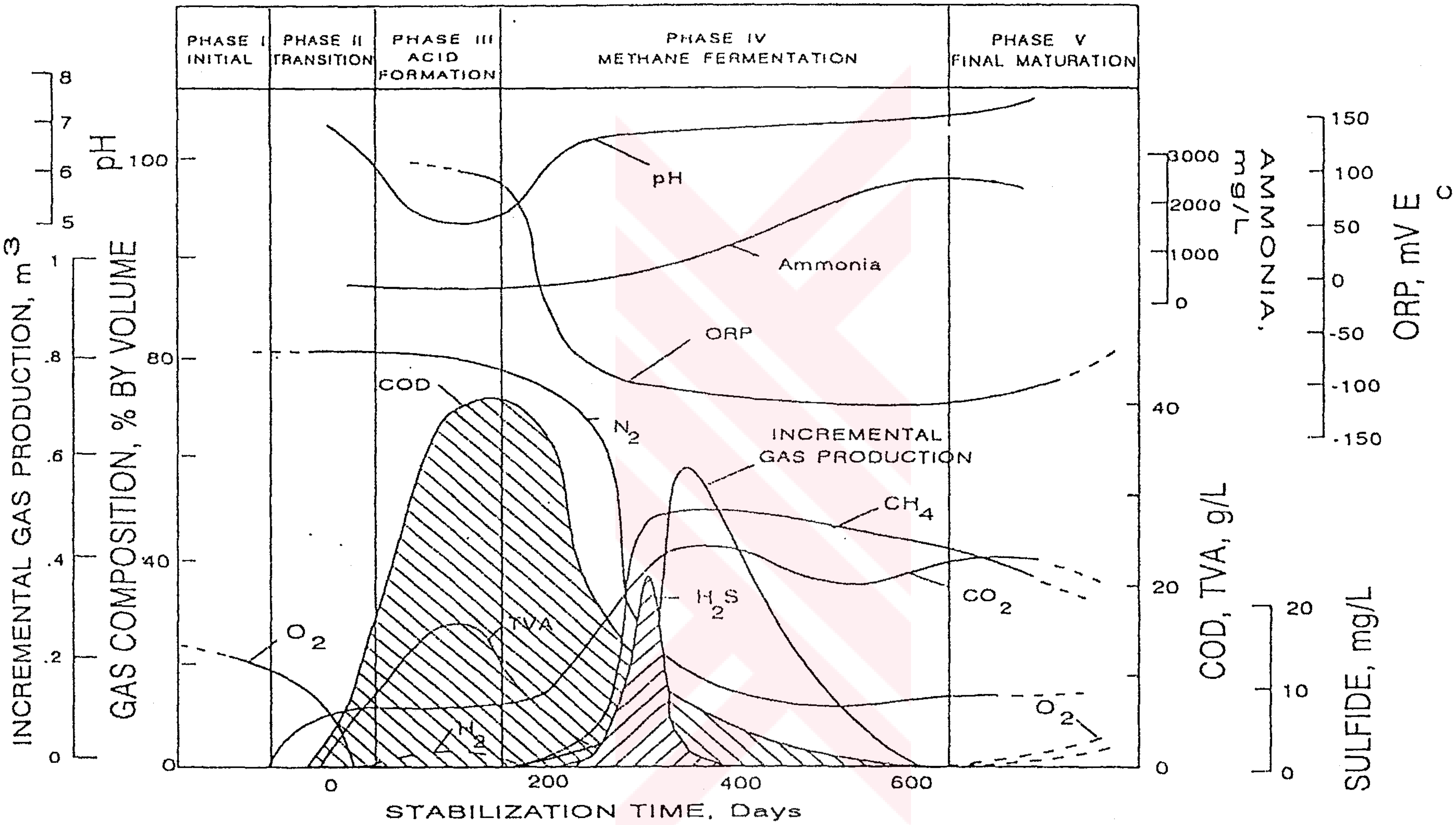


Figure 2.2. The main changes in leachate and gas composition during the phases of landfill stabilization (San, 1999).

2.5.2.3. Acid Phase. Although the methane concentration increases, the concentration of carbon dioxide is the major gas generated during the Phase III. The pH of the leachate continues to drop because of the accumulation of volatile fatty acids and high concentration of CO_2. The chemical oxygen demand (COD) can be reached to the maximum level because of the dissolution of organic acids in the leachate.

2.5.2.4. Methane Fermentation Phase. In Phase IV, the methanogenic bacteria become dominant. The methanogenic bacteria are very sensitive to oxygen and pH concentrations. Therefore, they should have optimum environmental conditions in order to have activity with high efficiency.

At this phase, the composition of the gases changes significantly. Landfill gas composition in the range of 50 to 70 per cent methane and 30 to 50 per cent carbon dioxide can be expected in this phase. Little amount of nitrogen and hydrogen may also be present (Farquhar and Rovers, 1997).

2.5.2.5. Maturation Phase. After the depletion of the biodegradable organic materials in the Phase IV, only refractory waste fractions remain in the landfilled waste. The generation of landfill gases decreases as a result of the insufficient amount of nutrients and substrate which are removed in the previous phases. The principal gases evolved in the Phase V are CH_4 and CO_2. During the maturation phase, the leachate will often contain humic and fulvic acids, which are difficult to decompose biologically (Tchobanoglous et al., 1993).

The solid waste stabilization phases will vary by the distribution of wastes in the landfill, the availability of moisture and nutrients, and the concentration of toxic materials. In order to detect the intensity and period of each phase of landfill stabilization, leachate and gas generated from the landfill environment should be monitored by physical and chemical parameters. These parameters are pH, oxygen reduction potential, chemical oxygen demand, biochemical oxygen demand, volatile acids, nitrogen and phosphorus. Moreover, alkalinity, heavy metal concentration, conductivity, chloride, nitrates, sulphates and microorganisms should also be monitored during the stabilization processes. Also, daily gas production and its composition should be measured.

2.5.3. Factors Affecting Anaerobic Landfill Stabilization

2.5.3.1. pH and Alkalinity. It is a common knowledge that pH is one of the important parameters to control the anaerobic waste degradation. The optimum pH range for the anaerobic decomposition varies between 6.8 and 7.4 (Barlaz et al., 1990). It was stated that the pH of aqueous environment should range from 6.5-7.5 (Metcalf and Eddy, 1991). However, the process can progress successfully in the pH range of 6.0-8.0.

High values for volatile fatty acids in the anaerobic digestion contribute to the reductions in pH and accumulation of volatile fatty acids. In this situation, methane production will decrease or may eventually cease. The methanogenic bacteria are the most sensitive group to pH changes. The drop of pH below 6.2 may be toxic to the methanogenic bacteria and the methane bacteria can not function below this point (Metcalf and Eddy, 1991).

Sufficient bicarbonate alkalinity should be present in order to neutralize the volatile fatty acids that might accumulate in order to be able to keep the pH in required range. The buffer capacity is defined in terms of alkalinity. An alkalinity in excess of 2000 mg/L as calcium carbonate is considered as optimum (Farquhar and Rovers, 1997). The alkalinity provides resistance to a change in pH, buffers the system, and stabilizes the gas production process (Albükrek, 1998).

2.5.3.2. Temperature. Temperature is a very significant parameter in all biological and chemical processes. The maintenance of appropriate temperature values within a landfill affects the landfill stabilization efficiency and the type of the bacteria that may be dominant in the landfill.

Temperature has an important effect on the survival and growth of the microorganisms. Changes in temperature, both increases and decreases, may adversely affect the digestion performance. A sudden temperature change causes a simultaneous increase in the concentration of all the volatile fatty acids. The extent of the impact depends on the factors such as the magnitude of the temperature change applied, the exposure time and the bacterial composition of wastes (Ahn and Forster, 2002).

The methanogenic bacteria are very sensitive to the temperature changes. The effects of the temperature on the specific activity of the anaerobic microorganisms have been described in three different temperature ranges.

Typical temperature ranges for the bacteria in each of these categories are:

- psychrophilic range (10-30^{0}C, optimum temperature is 15°C),
- mesophilic range (20-50^{0}C, optimum temperature is 35°C),
- thermophilic range (45-75^{0}C, optimum temperature is 55°C).

According to Tchobanoglous (1993), temperatures below the optimum typically has a more significant effect on the bacterial growth rate than temperatures above the optimum. It was reported that the methane production rate increased significantly such as 100 times more at 40°C than at 20°C and 30°C (Christensen and Kjeldsen, 1989). The methane production during the anaerobic decomposition is possible at temperatures ranging from 5°C to 55°C, but maximum production rates occurred at specific temperatures within each of the three ranges. The optimal temperature is 37°C for the mesophilic range (Farquhar and Rovers, 1997). Methanogenesis is an exothermic reaction causing the increase in the temperature of the landfill area.

2.5.3.3. Moisture Content. Moisture content is one of the limiting factors affecting the landfill gas production and the waste degradation. The major moisture sources in the landfills are precipitation, ground waters, decomposition of wastes, sludge and fresh refuse additions. For the establishment of the anaerobic microbial populations in the landfills, sufficient moisture must be supplied. If sufficient moisture is not available, anaerobic decomposition will decrease or may eventually cease. Moisture promotes the transportation of the organic substrates and nutrients and dilutes potential metabolic inhibitors leading to an increase in the methane generation along the landfills. After the leachate recycle, the indigenous microorganisms utilize the moisture and nutrients provided to consume the organic fractions of the waste at a higher rate (Purcell, 2000).

When the moisture drops below the 55 per cent level, or lower, microbial activity is significantly inhibited. It ceases at 12 per cent completely. Therefore, decomposition can be expected to proceed very slowly in the landfills located in arid regions (Barlaz et al., 1990).

High moisture content, in the range of 60 per cent to 80 per cent favours maximum methane production. In reality, the moisture content of solid waste at the time of placement is approximately 25-40 per cent. Therefore lack of moisture significantly limits the gas production rate in a sanitary landfill. However, an increase in the moisture content of the solid wastes alone will not significantly increase the gas production. The addition of nutrients, addition of inoculums of micro organisms and buffer for pH control in combination with moisture result in the enhancement of the methane production (Buivid et al., 1981).

2.5.3.4. Nutrients. Phosphorus and nitrogen are called as the major nutrients in anaerobic digestion processes. The growth rate of microorganisms depends on the composition of the wastes, which has micro and macro nutrients. Solid wastes contain enough nutrients for the microorganisms. The C/N ratio should be in the range of 20-30 for the optimum anaerobic decomposition. A high C/N ratio in landfill results in the less methane production and total bioconversion takes place over a period of many years (Buivid et al., 1981).

The optimum ratios for the organic matter expressed as chemical oxygen demand, nitrogen and phosphorus (COD: N: P) are 100:0.44:0.08 used to describe the nutrient requirement of the ecosystem. Generally, nitrogen and phosphorus are enough for the microorganisms in the landfill areas. However, it can be insufficient in the landfill areas because of heterogeneity of the waste and landfill area. Phosphorus is the limiting nutrient for the anaerobic decomposition processes (Christensen and Kjeldsen, 1989).

Sodium, calcium, potassium, magnesium, chloride, and sulfate ions which are known as micro-nutrients are necessary for anaerobic digestion processes.

2.5.3.5. Inhibitors. The presence of toxic or inhibitory materials such as the alkaline and alkaline-earth metals, heavy metals, ammonia nitrogen, sulphide, organic compounds and volatile organic acids in ecosystem affect the microbial processes adversely. The methanogenesis are very sensitive to these materials (Christensen and Kjeldsen, 1989).

Toxic levels of hazardous materials depend on the features of substances and vary from one substance to another.

Sulphides, volatile acids, salts, ammonia, alkali and alkaline earth metals are some of the inhibitors (Speece, 1996).

Although soluble sulphide concentrations of 50 to 100 mg/L do not show any inhibition in anaerobic digestion process, its higher concentrations (1500 mg/L or much) have toxic effects. In general, sulphides are more toxic for the methanogens because of the generation of hydrogen sulphide (H_2S) which inhibits the activity of the methanogens. On the other hand, sulphide can be used effectively to remove heavy metals from solution in anaerobic digestion process (Speece, 1996).

Inhibitions of volatile acids depend on pH. When the pH value decreases, the concentration of unionized volatile acids (UVA) increases (Barredo and Evison, 1991).

Toxic effect of salts in anaerobic digestion process is related to the cation portion of the salts. Table 2.2. gives the inhibitory concentration levels of salts.

Table 2.2. Inhibitory concentration of the salts (Olcay, 2001)

Salts	Inhibitory Concentrations (mg/L)	
	Moderate	Strong
sodium	3500-5500	8000
potassium	2500-4500	12000
calcium	2500-4500	80000
magnesium	1000-1500	3000

Even though, ammonia is used as a buffer, high ammonia concentrations have a toxic effect in anaerobic process. The inhibitory effects of ammonium are caused by free ammonia with increasing pH (Christensen and Kjeldsen, 1989). Ammonia concentration of 300 mg/L at pH value of 7.1 inhibits methanogenic microorganisms. Free ammonia concentrations are higher than 150 mg/L show a toxic effect at elevated pH levels (Angelidaki and Ahring, 1994).

Heavy metals have a high toxic capacity and cause inhibition in anaerobic digestion processes. Soluble heavy metals have an inhibition effect more than insoluble ones. Sulphide can prevent heavy metal toxicity. Inhibitory concentration levels of heavy metals are illustrated in Table 2.3.

Table 2.3. Reported values of toxic concentrations of heavy metals in anaerobic digestion (Speece, 1996)

Metal	Toxic Concentration (mg/L)
copper	300
nickel	800
zinc	350
chromium	200

2.5.3.6. Input Solid Waste Characteristics. The composition of the solid waste is the significant parameter in the waste decomposition (Lay and Noike, 1998). The wide variety of landfill components that can be broken down biologically depend heavily on the organic material of solid wastes. The physical changes of the solid wastes in landfills occur by compression, dissolution, and sorption. These physical reactions change particle size, surface area of solid wastes, and settlement potential of landfills. Particle size and surface area of solid wastes influence the decomposition rates and the amount of the adsorption. The shredding of the wastes is found to increase the rate of decomposition and leads quickly to methane production (San, 1999).

2.5.4. Enhancement of Anaerobic Landfill Stabilization

Some environmental problems such as release of greenhouse and toxic trace gases, and migration of combustible gases may occur because of improperly managed and operated landfills. Stabilization of the landfills takes very long time. Thus, enhancement of landfill stabilization is necessary in order to accelerate the degradation and stabilization processes.

Microbial processes in landfills enhance the landfill stabilization. When the organic compounds in leachate are converted to the methane by methanogen bacteria, leachate treatment can be much cheaper and easier. Enhancement of gas production with higher concentration of methane can also be achieved. This generated gas may be used for electricity or heat. Moreover, the landfill rehabilitation will be easier and settling process will be occurred in a shorter time period.

Leachate recirculation, pH control, buffer addition, nutrient addition, sludge addition, and compaction are the techniques that promote decomposition, shorten the degradation time and typically maximize the methane production.

Leachate recirculation is the most popular applied technique. It provides enhancement of the methane production because of the increasing microbial activity in the landfills. It was reported that leachate recirculation increased the moisture and nutrient distribution and maintained conditions suitable for degradation in landfill areas (Chan et al., 2002). pH is an important parameter affecting methanogenic degradation of the landfilled wastes. Thus, pH adjustment and pH control can be provided by adding solid buffer materials such as lime or calcium carbonate. The addition of nitrogen and phosphorus as major nutrients enhances the biodegradation. Rapid landfill stabilization can be achieved by sludge addition (Sosnowski, 2003). Also compaction may be used as an affective technique in the degradation process. Higher compaction causes a higher moisture content in a landfill area. This high moisture content is a requirement for the microorganisms.

2.6. Aerobic Sanitary Landfill

The decomposition of the organic fraction of solid wastes is very important in aerobic landfills. Aerobic processes occur naturally in the environment. The controlled injection of moisture and air into the waste mass through pipes create aerobic conditions in landfills.

Aerobic landfill processes and composting operations are similar with each other (Leikam et al., 1997). The solid waste landfill area is used as a bioreactor. Biologic processes play an important role in the process of decomposition and stabilization in landfill areas. It was stated that organic materials found in MSW can be degraded in significantly short time frames by the introduction of air and moisture at the proper rate (Murphy and Stessel, 1992). In aerobic landfills, aerobe bacteria convert the organic materials to mostly carbon dioxide and water. Also, organic fraction of MSW can be usefully integrated to obtain a higher quality final compost product and better overall process performance.

Before the air injection, the moisture content of solid wastes must be between 50 to 70 per cent by weight. Otherwise, the solid wastes may be dry because of air. Long time requirement for the stabilization of the anaerobic landfills, treatment of generated leachate, and postclosure costs increase the expenses of anaerobic landfills in comparison with the the aerobic landfills. The aerobic landfills prevent these adverse effects of the anaerobic landfills. Moreover, aerobic landfills improve the landfill stabilization.

Aerobic biodegradition occurs rapidly because of proper air and moisture content. In aerobic conditions, the following benefits have been observed at aerobic bioreactor landfills (Hudgins and Harper, 1997):

- more rapid waste and leachate stabilization,
- landfill airspace savings (increased rate of landfill settlement),
- reduction of methane generation by 50-90 per cent,
- capability of reducing leachate volumes,
- reduces long-term maintenance and monitoring costs,
- potential for landfill mining and sustainability,
- reduction of environmental liabilities,
- average costs are less than landfill gas collection systems.

2.6.1. Microbiology of Aerobic Landfill Stabilization

Organic materials in sanitary landfills create a lot of threats for the environment such as generation of toxic leachate and landfill gases (Read et al., 2000). Much of the landfilled wastes can be biodegradable. In reality, however, most landfilled wastes remain 30-40 years after disposal. These conditions at those sites are often not favorable for biodegradation. Such conditions include various factors; such as low moisture, low oxygen concentration, and high heterogeneity of materials. Further, many waste components are non-biodegradable or very slow to degrade. These types of landfills release pollutants to the groundwater and atmosphere. In several respects, aerobic landfilling prevents the effects of organic materials disposed in landfills.

The proper air injection and liquid are pumped into the waste mass in order to create aerobic conditions in landfills. The oxygen in the air enhances the rapid biodegradation of organic materials. During the oxidative process, electrons are removed from the substrate and passed via the electron transport chain to a terminal electron acceptor (TEA). For the aerobic organisms, the TEA is oxygen. For the anaerobic organisms, the TEA is a combined form of oxygen such as an organic metabolite like CO_2, NO_3^- or SO_4^{-2}, or an oxidized metal, e.g., Fe^{+3} (http://www.biology.arizona.edu/biochemistry/problem_sets/metabolism/14t.html).

The addition of water (or leachate) promotes the transportation of organic substrates and nutrients in landfill areas. It dilutes potential metabolic inhibitors and leads to decrease generated temperature along landfills. By this way, the decomposition of organics is accelerated.

Accelerating stabilization by the aerobic method creates rapid settlement and reduces the organic constituents in leachate, partially removes some salts and metals, greatly eliminates offsite disposal of leachate, and shortens monitoring time after closure (Borglin et al., 2004). Stabilization of the mass at both types of bioreactors occurs more rapidly than in dry tombing, but the aerobic method promises nearly complete settlement in far less time.

The digestion of sewage sludge and solid wastes together is a useable technique (Tchobanoglous et al., 1993). On the other hand, the anaerobic co-digestion of sewage sludge with organic fraction of municipal solid wastes seems to be especially attractive (Hamzawi et al., 1998). The feasibility of anaerobic codigestion of waste activated sludge and a simulated organic fraction of municipal solid wastes was examined by Poggi-Varaldo and Oleszkiewicz (1992). The advantages of the co-disposal include:

- decrease in leachate and offensive odours,
- improve balance of nutrients,
- synergistic effects of microorganisms,
- increase load of biodegradable organic matter,
- better biogas yield.

When the process occurs under the thermophilic conditions, the process provide much hygienic stabilization. Hovever, digestion rate of the process decreases.

2.6.2. Factors Affecting Aerobic Landfill Stabilization

The major environmental factors that affect the aerobic processes are moisture, aeration, temperature, and pH (Purcell, 2000). The environmental factors determine the rate and extent of decomposition. A deficiency in any factor would limit rate and extent of the aerobic decomposition. These factors can not be analyzed independently in practice since they are interrelated to each other.

2.6.2.1. Moisture. Moisture content is one of the important parameters in aerobic processes. There is a close relationship between the moisture content and aeration in an aerobic process. The moisture content of biologic environment determines the dominant species and their growth abilities. Bacteria grow at high moisture contents, whereas fungi grow at lower moisture contents. Thus, moisture content should be optimal for all dominant species. Provision of this optimum condition is difficult, because, while the moisture content may enhance the activity of some species, it may inhibit others at the same time.

Excess water is not preferable in landfills. Excess water causes plugging of the spaces between the particles of the materials. In this case, the pores of the materials are filled by water and the oxygen transfer in the landfill is hindered. In practice, moisture content should be closely monitored and adjusted. If the moisture content exceeds 60 per cent, aeration is hindered, nutrients are leached out, biological decomposition is decreased, and the odor from anaerobic decomposition is emitted (Tchobanoglous et al., 1993).

2.6.2.2. Aeration. The proper air must be injected in landfills in order to prevent anaerobic conditions and adjust temperature and moisture content of the mass. The major sources of air required by the microorganisms come from the air injection and the voids between the solid particles. Diffusion of air into the aerobic landfill is relatively minor in terms of providing the microbial oxygen demand. The aeration rate of an aerobic landfill depends on the nature and quantity of waste. Thus, the supply of proper aeration rate for landfills is very difficult.

Aeration plays a critical role in landfills. The removing of excess moisture from the wet materials and controlling of the landfill temperature are made by aeration. Moreover, aeration is necessary to remove carbon dioxide in landfills. Precipitation of water vapor decreases aeration and carbon dioxide dissolved in the water with a risk of generating anaerobic conditions (Jakobsen, 1994; Haug, 1993).

Oxygen must be supplied to the aerobic landfills in order to support microbial activity. Proper aeration promotes the conversion of carbohydrates into CO_2 and water. Moisture is necessary for the biological activity; extreme conditions in moisture removal may result in impeded biological activity. The controlled aeration rates enable the control of heat transfer mechanisms (Haug, 1993).

2.6.2.3. Temperature. The temperature is an important parameter in microbial and chemical activities Optimum temperature may promote the waste degradation in the aerobic landfills. It was stated that the optimum temperatures in the aerobic landfills were between 35 ^{0}C and 55 °C (Hudgins and Harper, 1997).

Landfills temperatures are changed by air and liquid injection. Excess temperature may disrupt or cease biological activities in the landfills. Aerobic reactions are exothermic reactions that increase the temperatures of the environment. The excess temperature results in breaking down of active microbial cells.

Large quantities of leachate can be evaporated because of the generated heat. It was reported the leachate volume reductions of 86 per cent and 50 per cent at two aerobic bioreactor landfills (Hudgins and Harper, 1997). Waste temperatures are controlled by changing the rate of air and liquid addition. Waste temperatures are maintained in the optimal range, and only enough air is injected into waste to support aerobic biodegradation.

2.6.2.4. pH. pH is also important in the aerobic landfills. The pH level usually drops at the starting of the aerobic landfills. The pH drop is the result of the formation of volatile fatty acids. When the acids are utilized by microorganisms as substrates, the pH level begins to rise. 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). It was stated that the degradation rate of organic matter in the pH-controlled experiments was faster than that without (Nakasaki et al., 1993).

At the early stages of decomposition, the pH values may drop because of the acid formation from the simple organics. Microorganisms use organic acids as substrate. However, the production of organic acids drops the pH of the landfills. The subsequent rise afterwards is the indication of the acid utilization. Buffering is not necessary and could even have adverse consequences. (Diaz et al., 1996). Excess buffering in high ph values causes vaporization of ammonia.

2.7. Yeasts

Environment is a discontinuous and heterogeneous surrounding that contains large number of diverse organisms. Environmental organisms can affect many aspects of life and are easily transported between environments. Organisms in the environment are diverse in origin and ubiquitous.

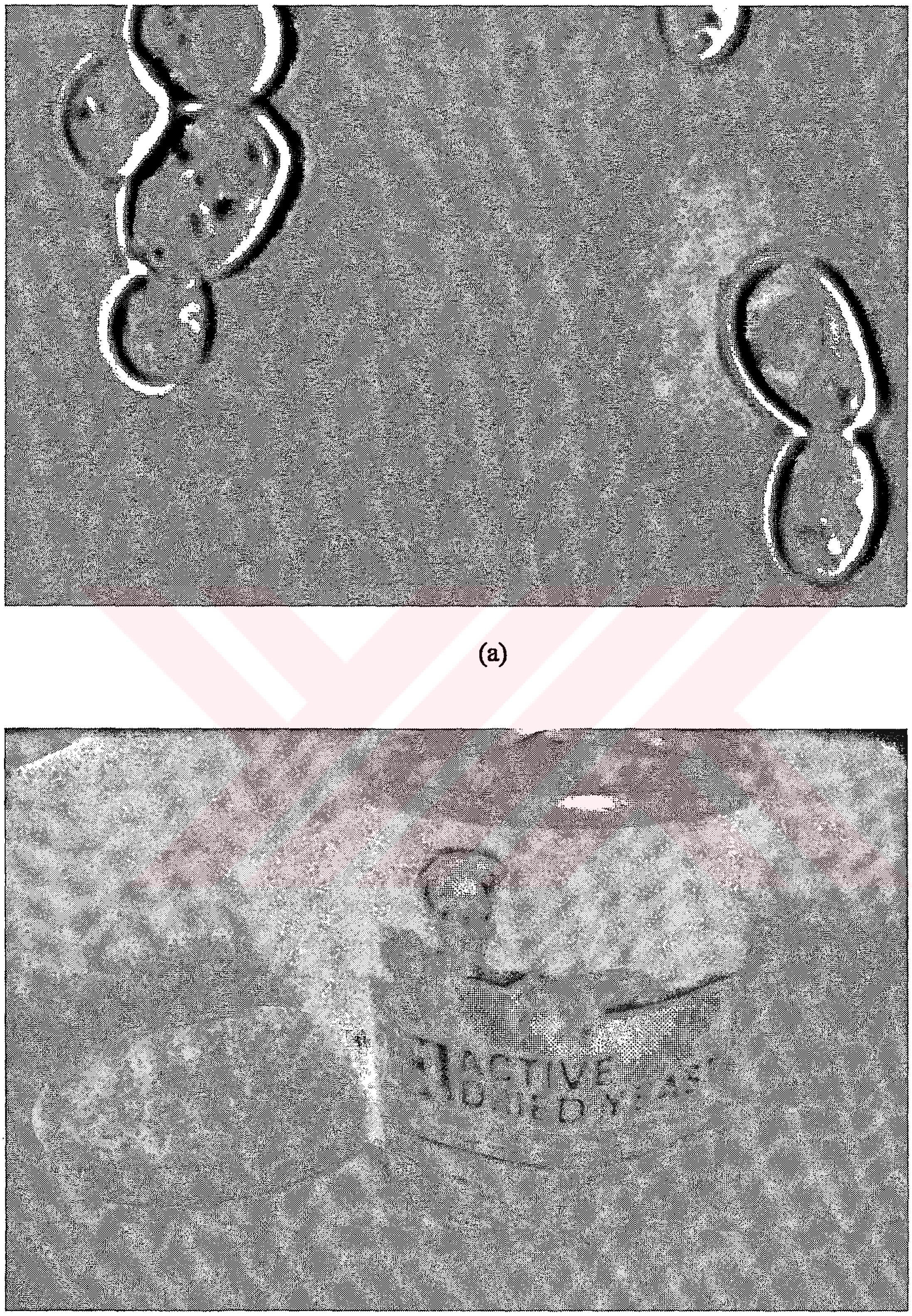
Yeasts, which are unicellular fungi, critically affect the cycling of organics as well as the bioremediation. They can be classified into 40 genera, with about 350 species. Yeasts can metabolize organic materials anaerobically and they are less numerous than aerobic mycelium-forming fungi. Generally, yeasts are found at populations of up to 10^3 per gram of soil. Yeasts multiply from a single cell by budding (eg *Saccharomyces*), by direct division (eg. *Schizosaccharomyces*), or by growing as simple irregular filaments (mycelium). In sexual reproduction, most yeasts form asci, which contain up to eight haploid ascospores. These ascospores may fuse with adjoining nuclei and multiply through vegetative division or, as with certain yeasts, fuse with other ascospores.

From the point of cost of energy and economic competition, the usage of microorganisms in the industries has gained a significant attention (Olcay and Kocasoy, 2004). Yeasts can be used in many industrial purposes. Generally, they can grow in low pH and in low moisture areas.

2.7.1. *Saccharomyces Cerevisiae*

The most well-known and commercially significant type of yeasts are the strains of *Saccharomyces cerevisiae*. These organisms are used to ferment the sugars of rice, wheat, barley, and corn to produce alcoholic beverages and utilize in the baking industry to expand, and raise dough. *Saccharomyces cerevisiae* is commonly used as baker's yeast and for some types of fermentation. Yeast is often taken as a vitamin supplement because it is 50 per cent protein and a rich source of B vitamins, niacin, and folic acid. The picture of yeast used in the experiment is given in Figure 2.3 a and Figure 2.3 b.

Saccharomyces cerevisiae give responses to some taxonomic tests. In codes for responses to tests `+` represents positive, `-` represents negative, `+,-` represents both positive and negative`, and `D` represents the positive response to taxonomic tests but these positive responses delay longer than one week (Barnet et al., 1990). The responses of *Saccharomyces cerevisiae* to some taxonomic tests are given Tables 2.5, 2.6, 2.7, and 2.8.



(a)

(b)

Figure 2.3. The picture of yeast used (http://www.micro.msb.le.ac.uk/video/Scerevisiae. .html).

Table 2.4. Semi-aerobic fermentation tests (Barnet et al., 1990)

Compounds	Response
F1 D-glucose	+
F2 D-galactose	+,-
F3 maltose	+,-
F4 methane α-D-glucoside	+,-
F5 sucrose	+,-
F6 α,α-trehalose	+,-
F7 melibiose	+,-
F8 lactose	-
F9 cellobiose	-
F10 melezitose	+,-
F11 raffinose	+,-
F12 inulin	-
F13 starch	+,-
F14 D-xylose	-

Table 2.5. Aerobic utilization and growth tests (Tests N1 to N9 list compounds as sole 6 sources of nitrogen) (Barnet et al., 1990)

Compounds	Response
N1 nitrate	-
N2 nitrite	-
N3 ethylamine	-
N4 L-lysine	-
N5 cadaverine	-
N6 creatine	-
N7 creatinine	-
N8 glucosamine	-
N9 imidazole	-

Table 2.6. Aerobic growth test V1 to V10, T1 to T5 and O1 to O5, D-glucose is the sole carbon source (Barnet et al., 1990)

Compounds	**Response**
V1 w/o vitamins	+,-
V2 w/o myo-inositol	+,-
V3 w/o pantothenate	+,-
V4 w/o biotin	+,-
V5 w/o thiamin	+,-
V6 w/o biotin and thiamin	+,-
V7 w/o pyridoxine	+,-
V8 w/o pyridoxine and thiamin	+,-
V9 w/o niacin	+,-
V10 w/o paba	+,-
T1 at 25 °C	+
T2 at 30 °C	+
T3 at 35 °C	+,-
T4 at 37 °C	+,-
T5 at 40 °C	+,-
O1 0.01% cycloheximide	-
O2 0.1% cycloheximide	-
O3 1% acetic acid	-
O4 50% d-glucose	+,-
O5 60% d-glucose	-

Table 2.7. Tests C1 to C44 list compounds as sole sources of carbon (Barnet et al., 1990)

Compounds	**Response**
C1 D-glucose	+
C2 D-galactose	+,-
C3 L-sorbose	-
C4 D-glucosamine	-
C5 D-ribose	-
C6 D-xylose	-
C7 L-arabinose	-
C8 D-arabinose	-
C9 L-rhamnose	-
C10 sucrose	+,-
C11 maltose	+,-
C12 α,α-trehalose	+,-
C13 me α-d-glucoside	+,-
C14 cellobiose	-
C15 salicin	-
C16 arbutin	-
C17 melibiose	+,-
C18 lactose	-
C19 raffinose	+,-
C20 melezitose	+,-
C21 inulin	-
C22 starch	+,-
C23 glycerol	+,-
C24 erythritol	-
C25 ribitol	-
C26 xylitol	-
C27 l-arabinitol	-
C28 D-glucitol	-,D
C29 D-Mannitol	-,D
C30 galacticol	-
C31 myo-inositol	-
C32 D-Glucono-1,5-lactone	-,D
C33 2-ket0-D-gluconate	-
C34 2-ket0-D-gluconate	-
C35 D-gluconate	-
C36 D-glucuronate	-
C37 D-galacturonate	-
C38 DL-lactate	+,-
C39 succinate	-,D
C40 citrate	-
C41 methanol	-
C42 ethanol	+,-
C43 propane 1,2 diol	-
C44 butane 2,3 diol	-

3. MATERIALS AND METHODS

Throughout the research, the stabilization of the solid wastes by mixing different ratios and types of sludge and the effects of the yeast addition on the decomposition rates of organic materials under aerobic and anaerobic conditions are detected. The details of the research conducted are given in the following sections.

3.1. Experimental Set-up

3.1.1. Equipment Used

The equipments used in the research are given below:

- four anaerobic reactors,
- four aerobic reactors,
- two aquariums,
- eight gas holders,
- four heaters,
- eight air diffusers,
- four air compressors,
- two pumps,
- leachate collection container,
- four plastic meshes,
- plastic pipes,
- plastic and fabric cover materials,
- pH meter,
- ORP meter,
- oven,
- spectrophotometer,
- atomic absorption spectrophotometer,
- COD reactor,
- gas chromatography,

- gas tighted syringes,
- mechanical stirrer,
- TKN digestion apparatus.

3.1.2. Experimental Set-up of the Anaerobic Reactors

The experimental set-up consisted of four anaerobic reactors; eight gas holders, an aquarium, and a heating system. Four reactors were used to simulate anaerobic landfill. The schematic view of anaerobic reactors and connected gas holders is given in Figure 3.1.

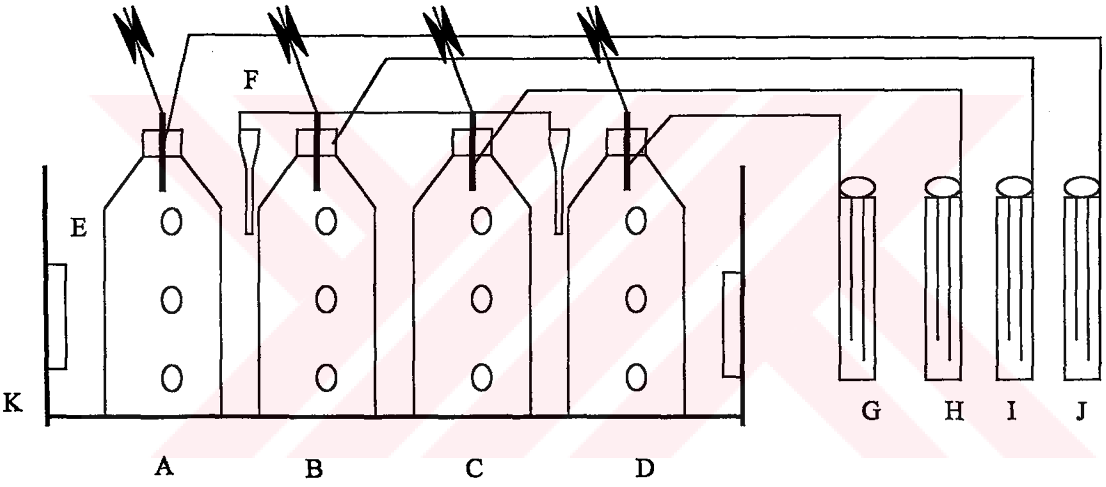


Figure 3.1. General view of anaerobic reactors and gas holders.

A : Reactor 1
B : Reactor 2
C : Reactor 3
D : Reactor 4
E : Air compressor

F : Heaters
G : Gas holder 4
H : Gas holder 3
I : Gas holder 2
J : Gas holder 1

K : Aquarium

<u>3.1.2.1. Anaerobic Reactors.</u> Anaerobic reactors used in this study were 40 cm in height with a radius of 40 cm. In the set-up, anaerobic reactors that were identical in terms of shape and material were used. Each plexyglass reactor had 12.5 L usable volume. Each reactor had V-shaped connection attached on it. One of these connections carried the generated gas of the reactor to the gas holder, while the other connection was used for taking the gas samples and feeding. The dimensions of anaerobic reactors are given in Figures 3.2, respectively.

There were three outlets for the sampling of leachate on each reactor. However, only the bottom outlet of the anaerobic reactors was used for taking leachate samples. The top and middle outlets were not used. For the prevention of the liquid and gas escape in the anaerobic reactors, silicon and isolation paste were applied to all around the edges of the reactors, junction points and faucets.

The anaerobic reactors were filled with water for detection of leakage. Moreover, the pressurized nitrogen gas was applied to all anaerobic reactors for 24 hours in order to detect any leakage. During this period, no leakage was detected at the anaerobic reactors. This test was also conducted for the aerobic reactors by filling water but no leakage was observed in the aerobic reactors either.

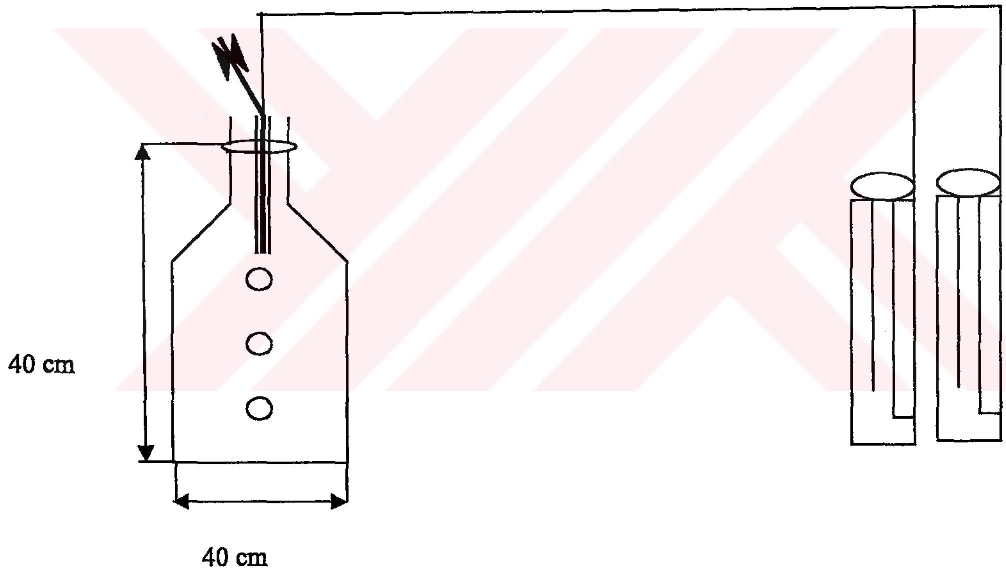


Figure 3.2. The dimensions of the anaerobic reactors.

3.1.2.2. Gas Holders of the Anaerobic Reactors. The biogas generated in the anaerobic reactors were collected and measured in the plastic gas holders. Each reactor had two gas holders with a volume of 0.5 L. The gas holders were inverted and placed into 1L glass cylinders filled with an acid solution. The pH of the acid solution was very low ($pH<2$) and the acid solution was used to hinder dissolution of carbon dioxide and methane in water. The transfer of generated biogases from the anaerobic reactors to the gas holders was provided with six mm diameter plastic pipes. The picture of the system is given at Figure 3.3.

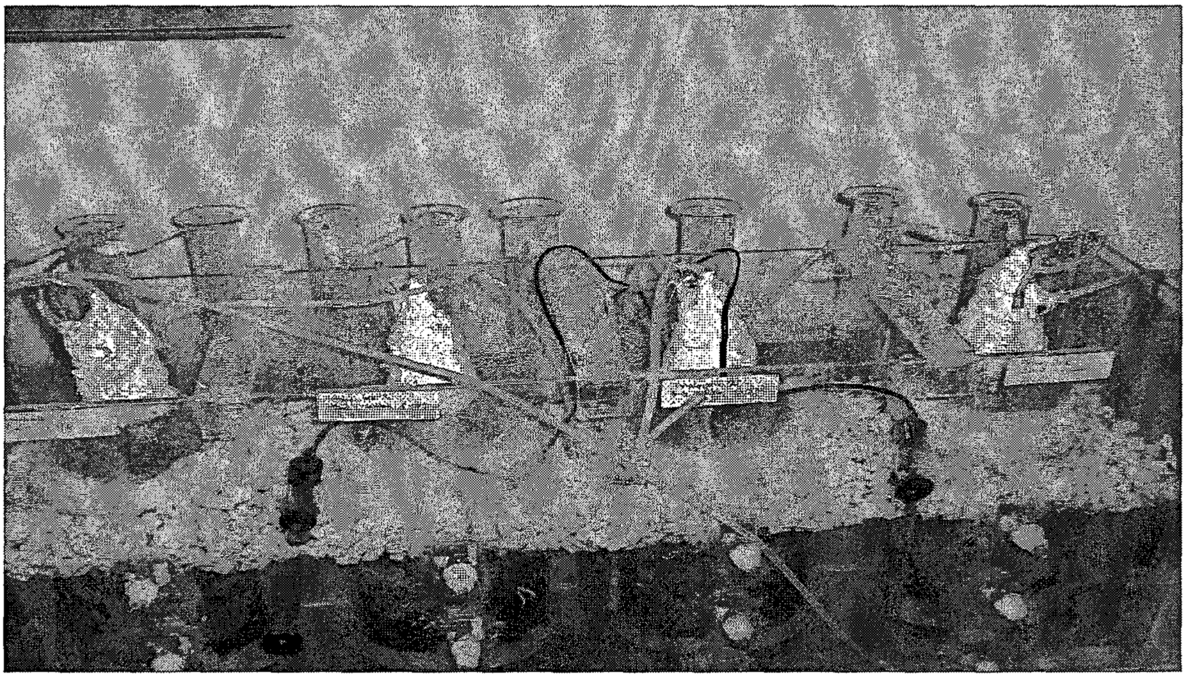


Figure 3.3. The picture of the gas holders and aquarium containing anaerobic reactors.

3.1.2.3. Aquarium of the Anaerobic Reactors. Four anaerobic reactors, three air diffusers, and two heaters were placed in a plexyglass aquarium filled with water. The aquarium basement is 130x45 cm and its height is 50 cm. It was also made from plexyglass.

There were 12 outlets on the aquarium. However, only four bottom outlets of the aquarium were connected to the bottom outlets of the anaerobic reactors. The top and middle outlets were not used.

3.1.2.4. Heating System. Temperature is one of the important control parameters in especially biological and chemical processes. Thus, it must be adjusted for getting high efficiencies. In order to supply optimum temperature for the microorganisms, the water in the aquarium was heated up to 34 ^{0}C by the two identical heaters. The picture of the heater is given in Figure 3.4. The air diffusers circulated water in order to distribute heat into the aquarium. The air was supplied by the air compressor. Styrofoam particles were used to cover the surface of the water to prevent the evaporation of water in the aquarium.

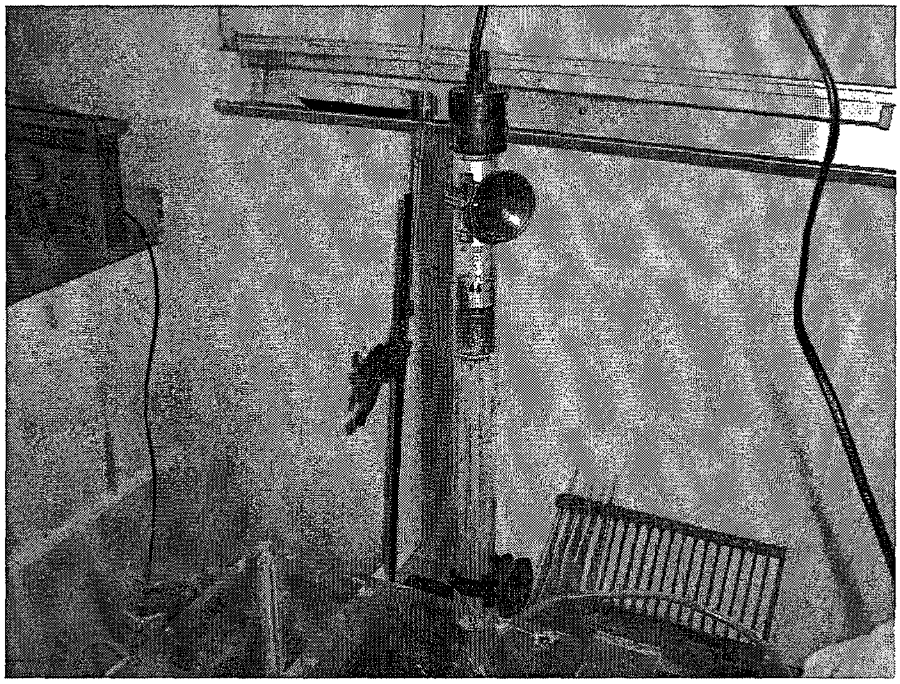


Figure 3.4. The picture of the heater.

3.1.3. Experimental Set-up of the Aerobic Reactors

The experimental set-up, simulating the aerobic landfilling, consists of four identical plexyglass aerobic reactors, four air compressors and distributor pipes, two pumps, a leachate container, and four air diffusers, an aquarium, and a heating system. The schematic view of aerobic reactors and connected air compressors is given in Figure 3.5.

3.1.3.1. Aerobic Reactors. Aerobic reactors were 35 cm in height with a radius of 40 cm. The dimension of the aerobic reactors is given in Figure 3.6. Each of the aerobic reactors had 12.5 L usable volume. A plastic mesh was placed inside the each container to catch the wastes and to facilitate the collecting of the generated leachate. After placing the plastic mesh, leachate samples were then taken under the plastic mesh by the pump. To aid the thermal isolation and to eliminate the effects of photosynthesis, top of the reactors were covered a plastic material and the aquarium was encased by a fabric cover material. The aquarium was operated at 34 °C. The leachate generated from the aerobic reactors was recycled back to the aerobic reactors depending on the moisture contents of the wastes in the reactors. The picture of the aerobic reactors is given in Figure 3.7.

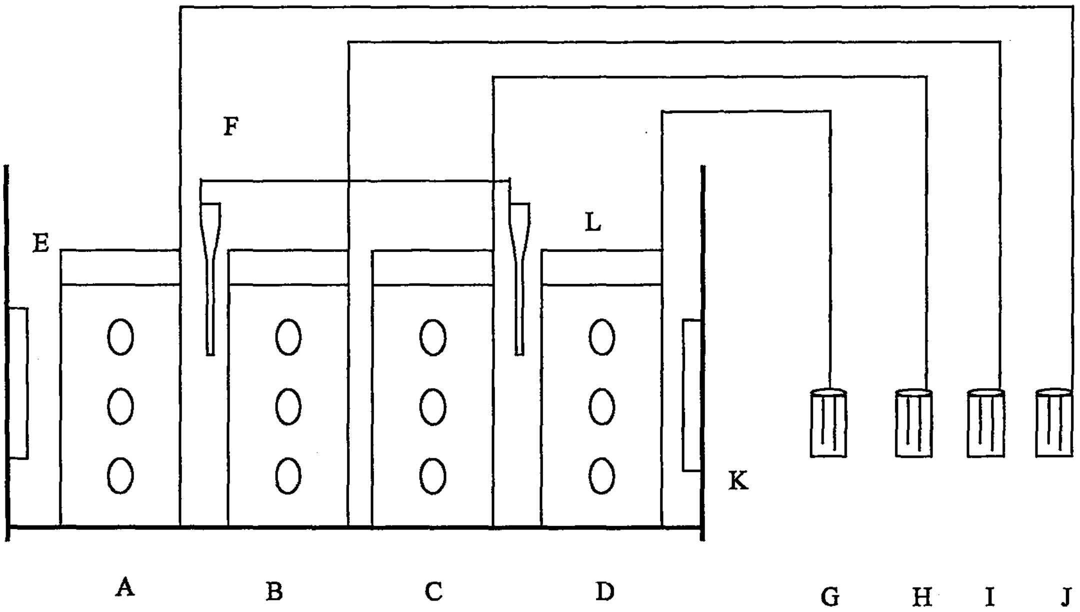
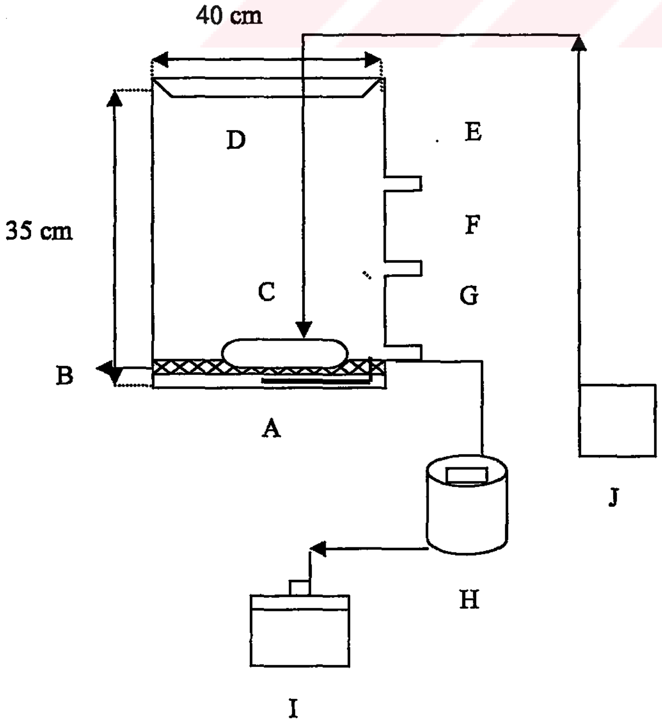


Figure 3.5. General view of the aerobic reactors and air compressors.

A : Reactor 1
B : Reactor 2
C : Reactor 3
D : Reactor 4
E : Air diffuser
F : Heaters
G : Air compressor 1
H : Air compressor 2
I : Air compressor 3
J : Air compressor 4
K : Aquarium
L : Plastic cover



A : Plexyglass reactor
B : Plastic mesh
C . Air diffuser
D : Plastic cover
E,F,G : sampling ports
H : Pump
I : Leachate collection container
J : Air compressor

Figure 3.6. The dimension of the aerobic reactor and system tools.

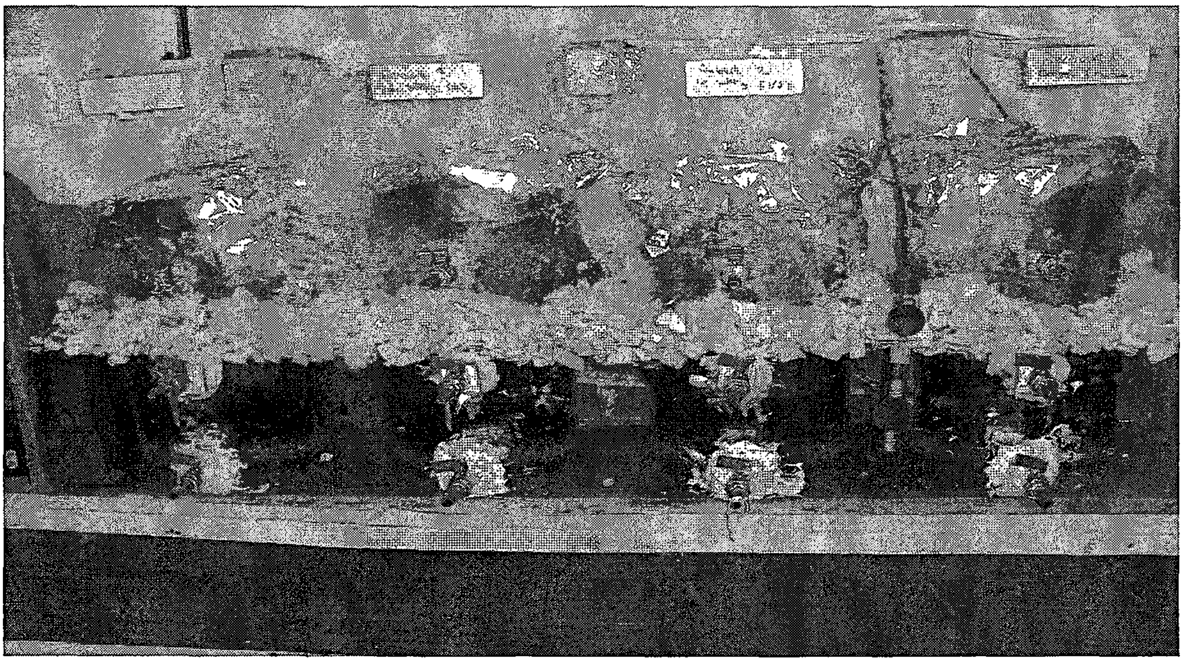
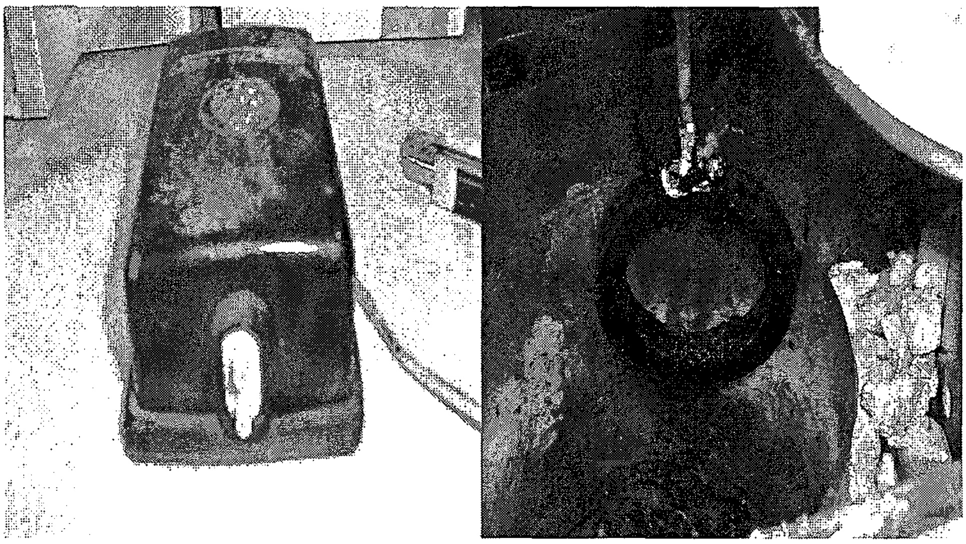


Figure 3.7. Picture of the aquarium containing the aerobic reactors.

3.1.3.2. Air Compressors and Air Diffusers. Four compressors were used to provide the required air flow for the maintenance of the aerobic conditions in the reactors. Each compressor produced 2 L air/minute. Air diffusers were used to distribute the generated air gently to the surface areas of the aerobic reactors. The pictures of the air compressor are given in Figure 3.8 a. and the picture of the air diffusers are given in Figure 3.8 b.



(a) (b)

Figure 3.8. The picture of the air supplier (a) and the picture of the air diffusers (b).

3.1.3.3. Pumps and Leachate Container. Two identical pumps were used to pump the generated leachate in the aerobic reactors. The picture of the pump is given in Figure 3.9. Top outlets of the pumps were connected to the bottom outlets of the aerobic reactor. Also, the bottom outlets of the pumps were connected to the leachate container into, where the leachate was collected. Depending on the moisture content of the aerobic reactors, leachate was recirculated to the reactors to supply the optimum moisture content. After the recirculation, excess leachate was discarded.

Figure 3.9. Picture of the pump.

3.2. Materials Used

The materials used for the experimental set-up are given below:

- raw sludge from a municipal and industrial treatment plant,
- municipal solid wastes from the Kemerburgaz Sanitary Landfill Area,
- yeasts,
- different chemicals for the analysis.

3.2.1. Solid Waste Composition

Municipal solid wastes used in the research were taken from the Kemerburgaz Sanitary Landfill Area located at the European Side of İstanbul. The landfill has been operated since 1995. 6000 tones of municipal solid waste are landfilled in the area every day. The municipal solid wastes have been disposed at a part of 25 hectares. Moreover, the landfill has equipped with gas extraction wells from the beginning of the operation.

The main components of the municipal solid wastes disposed at the Kemerburgaz Sanitary Landfill Area are given in Table 3.1. (Kocasoy, 2003; Metin et al., 2002). Solid wastes taken from different parts of Kemerburgaz Sanitary Landfill Area were put into six plastic bags and then taken to the laboratory. After the arrival to the laboratory, the solid wastes were prepared to conduct the moisture content and solid waste analyses. Characteristics of the solid wastes were determined before they were filled into the reactors. The results of the characterization analyses of the solid wastes used in the research are given in the Table 3.2. As it can be seen from Table 3.2, the main part of the wastes was composed of organic materials with a percentage of 52 per cent.

The moisture analyses were performed for the municipal solid waste samples. The samples were weighed and then kept at 105^0C for 24 hours in the oven. The samples were then reweighed to find the loss in weight. The remaining parts were measured as dry weights of the samples. Determination of moisture content is vital for the correct calculations (Kocasoy, 1996).

The moisture content of the solid wastes was found as 61.2 per cent. The results of mositure content analyses of the solid waste samples used in the aerobic and anaerobic reactors were represented in Table 3.2. The results were used to calculate the required amount of water to be added to each reactor in order to adjust the moisture contents properly.

The heavy metal analyses of the solid waste samples were also performed at the laboratory of İzaydaş. By this way, the heavy metal concentrations of the solid samples were detected. The results of heavy metal analyses of the solid waste samples used in the aerobic and anaerobic reactors were summarized in Table 3.3. The results of the heavy metal analyses of the solid samples were lower than the disposal limits of them. Limit values for the analyses were taken from the Turkish Hazardous Waste Regulation (The Ministry of Environment and Foresty, 1991).

After the analyses, the bulky items such as big cans and nylon bags were separated from the wastes before filling them into the reactors. Compaction was not applied to the wastes while the reactors were filled. Anaerobic reactors were sealed for better anaerobic conditions, while the solid wastes in aerobic rectors were left uncovered for exposing to the natural conditions.

Table 3.1. Solid waste components of Kemerburgaz Sanitary Landfill Area (Kocasoy, 2003; Metin et al., 2002).

Parameters	**WHO-UNEP**	**CH2M-HILL**	**Arıkan**
organic material, %	60.6	45	48
paper, %	8.8	14.5	8.4
plastic, %	3.1	9.5	11.0
glass, %	0.7	3.8	4.6
textile, %	3.1	5.6	2.9
metals, %	1.5	2.2	2.3
ash, %	4.6	15.0	13.2
others, %	6.9	4.4	6.3
baby diapers, %	-	-	3.2

Table 3.2. Composition of the solid waste used in the reactors

Component	**Reactors**
organic materials, %	52
paper and cardboard, %	8.2
plastics and nylon, %	9.4
glass, %	4.2
textile, %	4.3
metals, %	2.5
baby diapers, %	4.7
others, %	14.7
moisture, %	61.2

Table 3.3. Heavy metal concentrations of the solid waste samples filled into the reactors

Parameter	**Reactor (mg/L)**	**Limit Values (mg/L) (The Ministry of Environment and Foresty, 1991).**
Cu	0.0347	2-10
Ni	0.0224	0.4-2
Pb	0.0871	0.4-2
Cd	<0.0134	0.1-0.5
Cr	0.0845	0.1-0.5
Zn	0.0748	2-10
As	0.0005	-
Hg	<0.0009	1.5

3.2.2. Sludge Compositions

The sludge used in the research was taken from the Industrial and Domestic Wastewater Treatment Plant in İzmit. The plant is operated as activated sludge process and it treats about 35000 m^3 wastewater daily (İzaydaş, 2003). The sludges used in the experiments were taken from the different units of the treatment plant such as after primary clarifier tank, secondary clarifier tank, and belt press as given in Figure 3.10.

The moisture analyses were performed for the treatment plant sludge samples as shown in Table 3.4. While the moisture content of the primary clarifier sludge was 91.6 per cent, the moisture content of secondary clarifier sludge was found as 90.9 per cent, and the belt press moisture content was 80.2 per cent.

To determine the heavy metal concentrations in the sludge samples, the heavy metal analyses of the sludge samples were performed. The results of heavy metal analyses of sludge samples used in the reactors are given in Table 3.5.

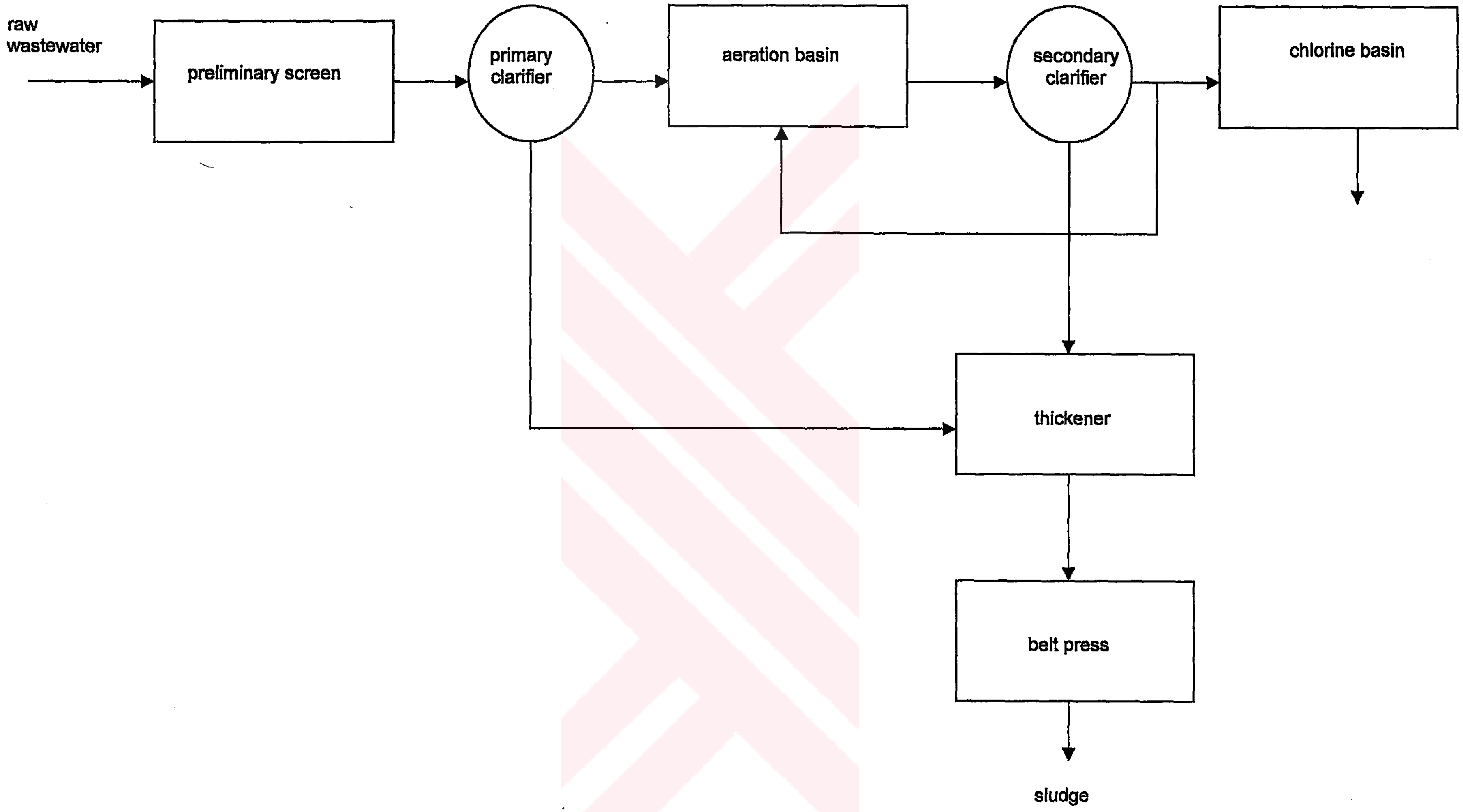


Figure 3.10. İzmit Industrial and Domestic Wastewater Treatment Plant flow chart.

The results of the heavy metal analyses of the sludge samples were lower than the inhibition effects of them on biological systems. Limit values for the analyses were taken from the Turkish Hazardous Waste Regulation (The Ministry of Environment and Foresty, 1995).

Table 3.4. Moisture content of the sludge samples used in the study

Reactors	**Primary Clarifier Sludge (%)**	**Secondary Clarifier Sludge (%)**	**Belt Press Sludge (%)**
reactor 1	-	-	-
reactor 2	91.6	-	-
reactor 3	-	90.9	-
reactor 4	-	-	80.2

Table 3.5. Heavy metal concentrations of the sludge samples

Parameter	**Primary Clarifier (mg/L)**	**Secondary Clarifier (mg/L)**	**Belt Press (mg/L)**	**Limit Values (mg/L)**
Cu	0.054	0.120	0.094	2-10
Ni	0.052	0.056	0.084	0.4-2
Pb	0.188	0.195	0.175	0.4-2
Cd	0.024	0.035	0.029	0.1-0.5
Cr	0.041	0.084	0.037	0.1-0.5
Zn	0.321	0.376	0.323	2-10
Fe	3.648	0.648	5.597	-

3.2.3. Yeast Used in the Research

In order to improve the decomposition of organics and shorten the degradation period of the processes, yeast stock solution was utilized as additive after the 55th day of the experimental study. Because of the suitability of physical and chemical conditions inside the reactors and for the survival of the yeast species, *Saccharomyces cerevisiae* were used in the yeast solution.

Preparation of yeast stock solution of *Saccharomyces cerevisiae,* 20 g of yeast was dissolved in one L of aqueous feed solution. The composition of the aqueous feed solution is given in Table 3.6. In order to enhance the reproduction of the solution, it was mixed by a magnetic stirrer at 35 ^{0}C (at a rate of 100 rpm) for 72 hours (Olcay, 2001).

Table 3.6. Composition of aqueous feed solution (Barnett et al., 1989)

No	**Compound**	**Concentration (mg/L)**
1	$CaCl_2$	17
2	K_2HPO_4	3000
3	$(NH_4)_2SO_4$	3350
4	glucose	20000

It was stated that 200 mg/L of yeast stock solution was optimal to improve the efficiency of anaerobic decomposition process after the acclimization period (Olcay *et al.*, 2004). However, there was no information about the effects of yeasts on the decomposition of organics in aerobic reactors. Both the aerobic and the anaerobic reactors (Reactor 2, Reactor 3, and Reactor 4) were loaded by the amount of 25 mL of the yeast stock solution. The yeast stock concentration of these reactors was taken as 200 mg/L.

3.2.4. Determination of the Reactor Loading Rates

Before the loading of the reactors, the moisture analyses of the solid and sludge samples were performed. The moisture content of the solid wastes loaded was 61.2 per cent. The primary clarifier sludge moisture content was 91.6 per cent, the moisture content of secondary clarifier sludge was found as 90.9 per cent, and the belt press moisture content was 80.2 per cent. After determination of the solid content of sludges, different ratios of sludge and solid waste were mixed as proposed in US EPA, Sludge Manual (US EPA, 1978). The Control reactor was only loaded with the solid wastes. The primary sludge and solid waste were mixed at 1/7 ratio in the Reactor 2. The Reactor 3 was also loaded with secondary settling sludge and solid waste at the ratio of 1/7. Finally, the Reactor 4 was loaded with belt-press sludge and solid wastes at the ratio of 1/4. These ratios are summarized in Table 3.7. and the loading ratios for the reactors are presented in Table 3.8. The moisture contents of the aerobic reactors were adjusted 65 per cent, while the moisture contents of the anaerobic reactors were adjusted 75 per cent.

Table 3.7. Ratios of the sludge to the solid waste given in US EPA Sludge Manual (EPA, 1978)

Method	**Bulking Agent**	**Solid Content of Sludge, (%)**	**Bulking Ratio**
sludge/refuse mixture	refuse	3-10	7 ton refuse / 1 ton sludge
sludge/refuse mixture	refuse	10-17	6 ton refuse / 1 ton sludge
sludge/refuse mixture	refuse	17-20	4 ton refuse / 1 ton sludge

3.3. Analytical Methods

The generated leachate in all the reactors was measured to understand and to monitor the degree of waste stabilization. The main parameters analyzed for the leachate samples were pH, oxygen reduction potential (ORP), chemical oxygen demand (COD), alkalinity, orthophosphate, sulfate, total Kjeldahl nitrogen (TKN), chloride, and then samples were preserved for the metal analyses. In addition to leachate analysis, the gases generated in the anaerobic reactors were analyzed. The parameters analyzed for the gas samples were daily gas production and gas content of the each anaerobic reactor.

Table 3.8. Loading conditions of the reactors

Reactor	**Sludge Type**	**Sludge to Waste Ratio**	**Moisture Content (%)**	**Solid Waste Added (Wet, g)**	**Water Added (mL)**
aerobic reactor 1	no sludge	-	65	1750	190
aerobic reactor 2	primary clarifier sludge	1/7	65	1750	-
aerobic reactor 3	secondary clarifier sludge	1/7	65	1750	-
aerobic reactor 4	belt press sludge	1/4	65	1750	-
anaerobic reactor 1	no sludge	-	75	1750	966
anaerobic reactor 2	primary clarifier sludge	1/7	75	1750	787
anaerobic reactor 3	secondary clarifier sludge	1/7	75	1750	771
anaerobic reactor 4	belt press sludge	1/4	75	1750	873.25

All these analyses were performed according to Standard Methods for the Examination of Water and Wastewaters (1992). The experimental parameters, analyze methods, and their schedules are given Table 3.9.

pH analyses. The hydrogen ion concentration (pH) is an important parameter for the bacterial life. The pH of the leachate samples were measured and recorded for each sample. Readings were recorded and plotted on diagrams against the time. The pH measurement was made every two days during the research.

Table 3.9. Experimental parameters and their schedules

Measurement	**Procedure**	**Frequency**
pH	examination#2730.B	every two days
oxygen reduction potential	examination#2580.B	every two days
chemical oxygen demand	examination#5220.D	twice a week
alkalinity	examination#2320.B	twice a week
orthophosphate	HACH Phosver 3 method	once a week
sulfate	HACH Sulfaver 4 method	once a week
total kjeldahl nitrogen	HACH digestion method	once a week
chloride	examination#4500.B	once a week
heavy metals analyses	HACH digestion method	twice a month
gas chromatography	Gas Chromatography	once a week
moisture	examination#208.A	twice a month

ORP analyses. The ORP values of the reactors demonstrate the oxygen reduction potential of the reactors. Positive ORP values indicate the aerobic conditions; while the negative ORP values indicate the anaerobic conditions. Like the pH measurements, ORP values were plotted on diagrams against the time and were measured at every two days.

COD analyses. COD analyses are important in determining the organic strength of the leachate. Dichromate Closed Reflux Method was used to measure the COD value of the leachate samples. COD analyses were performed twice a week.

Alkalinity. Alkalinity was measured volumetrically by titration with 0.02 N H_2SO_4 and is reported in terms of equivalent $CaCO_3$. Alkalinity measurements were made twice a week.

Orthophosphate analyses. Orthophosphate analyses were conducted by using the PhosvaVer 3 method and measured once a week.

Sulfate analyses. SulfaVer 4 method were used for sufate measurements of the samples. Sulfate was analysed once a week.

TKN analyses. TKN of the leachate was measured with HACH digestion method.TKN was measured once a week.

Chloride analyses. Chloride was monitored by using the Argentometric Titration Method. The samples were titrated by silver nitrate. Chloride was measured once a week.

Heavy metals analyses. The heavy metals were also monitored by Perkin Elmer Atomic Absorption Spectrophotometer. Atomic Absorption Spectroscopy uses the absorption of light to measure the concentration of gas-phase atoms. Before the analyses, each sample was digested with concentrated H_2SO_4 and H_2O_2. The heavy metal analyses were performed twice a month.

Gas chromatography. The gas samples were analyzed by Hewlett Packard 6850 Gas Chromatograph. The gas samples of the anaerobic reactors were taken by one µL gas tighted syringe. Then, the gas samples were injected to the gas chromatograph. After ten minutes, the characteristics of the gas samples were recorded. The content of gas analyses was analyzed once a week.

Moisture analyses. The moisture contents of the aerobic reactors were analyzed twice a month.

3.4. Experimental Procedure

At the beginning of the research, the aerobic and anaerobic experimental systems were set-up. The experimental set-up consisted of four aerobic reactors that were connected to the air supplier giving air to the reactors and four anaerobic reactors that were connected to the gas holders collecting the generated biogas inside the reactors. The reactors were placed in two aquariums.

All reactors had three outlets for the sampling of leachate. However, only the bottom outlet of the reactors was used for taking leachate samples. The top and middle outlets were not used. For the prevention of the liquid and gas escape in the reactors, silica and isolation paste were applied to all around the edges of the reactors, junction points and faucets. Moreover, water and pressurized nitrogen gas were applied to all anaerobic reactors for detecting any leakage. This test was also conducted for the aerobic reactors. However, no leakage was observed in the all reactors.

Solid wastes were taken from different parts of Kemerburgaz Sanitary Landfill Area. After the arrival to the laboratory, the moisture content and solid waste analyses were performed.

The treatment plant sludges, including primary settling sludge, secondary settling sludge, and belt-press sludge were supplied from İzmit. Like the solid samples, the moisture content and heavy metal analyses were performed. The moisture content of the primary, secondary clarifier, and the belt-pres sludge were found as 91.6, 90.9, and 80.2 per cent respectively.

After the determination of characteristics of the solid and sludge samples, the reactors were loaded in the ratios according to the values given at the EPA sludge manual (US EPA, 1978). No compaction was applied after the reactors were filled with the solid wastes and treatment plant sludges. The amount of water to be added to each reactor was calculated by using the moisture contents of sludge and solid wastes and presented in Table 3.8.

In the first stage of the study, the solid waste stabilization rates were accelerated by mixing municipal solid wastes with different ratios of treatment plant sludge. 1750 g of solid wastes were mixed with different ratios of the treatment plant sludges. The Control Reactor was only loaded with the solid wastes. The primary sludge and solid waste were mixed at 1/7 ratio in the Reactor 2. The Reactor 3 was filled with secondary settling sludge and solid waste at the ratio of 1/7. Finally, the Reactor 4 was loaded with belt-press sludge and solid wastes at the ratio of 1/4. The first stage was conducted for 55 days.

The water in the aquariums was heated up to 34 ^{0}C by the two identical heaters. The air diffusers circulated water in order to distribute heat evenly in the aquarium. Moreover, air compressors produced 2 L air/minute and distributed the generated air in the aerobic reactors. The air compressors worked 5 minutes every day. Approximately 50 mL leachate samples were taken in each aerobic reactor for the leachate analyses. Therefore, 50 mL water was added in each aerobic reactor every week. On the other hand, 100 mL water was added in each anaerobic reactor every month.

The moisture contents of solid wastes of the aerobic reactors were determined by drying samples at 104 ^{0}C for 24 hours. The samples were reweighed after they were cooled in the desiccators. After the determination of the moisture contents of the solid samples, depending on the moisture content, the recycling of leachate to the aerobic reactors was made at every two weeks. At the 15th day of the research, moisture contents of the aerobic reactors were in the range of 46-53 per cent. These moisture contents were sufficient to the microorganisms, so leachate was not discarded. However, at the 45th day of the experiment, moisture contents of the aerobic reactors decreased to 40 per cent. Therefore, 100 mL water was added in each aerobic reactor. Moreover, during the first stage, buffering was not made because of sufficient alkalinity concentrations of the reactors.

Many parameters were analyzed during the first stage of the research. pH, and ORP was measured every two days. COD was analyzed twice a week; on the other hand, alkalinity, orthophosphate, sulfate, TKN, and chloride were measured once a week. The heavy metal analyzes were made twice a month. Also, the gases generated in the anaerobic reactors were analyzed once a week.

In the second stage of the research, the yeast (*Saccharomyces cerevisiae*) stock solution was used to accelerate the rate of the anaerobic and aerobic decomposition. After the 55th day, 200 mg/L yeast stock solution was added to the Reactors 2, 3, and 4. Yeast solution was added to the three anaerobic and aerobic reactors to enhance the biological decomposition.

At the 75th day of the research, moisture contents of the aerobic reactors were lower than 40 per cent. Therefore, 100 mL water was added in each aerobic reactor again. At the end of the research, the moisture contents of the aerobic reactors were in the range of 45-

60 per cent after the yeast addition. These moisture contents were sufficient to the microorganisms, so leachate was not discarded.

Throughout the research, settlements of the wastes in all reactors were also observed. The settlements in the aerobic reactors began at the 9th day of the research. However, the settlements in the anaerobic reactors were observed after the 45th day of the research. The settlement in the aerobic reactors occurred faster than the anaerobic ones.

The study was conducted for 163 days. Analyses of the leachate generated in the anaerobic and aerobic reactors were performed from the discharge faucets of the each reactor and the quality of the leachate was measured. The schedules of analyses of pH, ORP, COD, alkalinity, orthophosphate, sulfate, TKN, chloride, and metal analyses were same as the first stage of the research. Also, the generated gases generated in the anaerobic reactors were analyzed at the end of the research.

4. RESULTS AND DISCUSSION

4.1. Analyses of Leachate Generated from the Anaerobic Reactors

The main purpose of this research was to detect effects of different ratios and types of treatment plant sludges mixed to the solid wastes under anaerobic and aerobic conditions. Another purpose of this research was to detect effects of yeast addition on the decomposition of solid waste. At the beginning of the first stage, the reactors (Reactors 2, 3, and 4) were filled with the solid wastes having the same compositions (Table 3.2.), and different types and ratios of sludges (Table 3.8.). The Reactor 1 was used as a control reactor. The Reactor 1 was loaded with the same solid wastes (Table 3.2.), but no sludge was loaded.

The first stage was continued until the 55th day of the research. COD values of leachate of the anaerobic reactors (Reactors 1, 2, 3, and 4) reached to 55800 mg/L, 61500 mg/L, 59500 mg/L, and 64500 mg/L respectively. The high values of COD concentrations of leachate of the anaerobic reactors resulted in the decomposition of organic materials in the absence of oxygen. The COD removal in the anaerobic reactors could not be detected because of the accumulation of the volatile acids. Moreover, the methane contents of the anaerobic reactors were in the range of 12.3-16.1 per cent during the first stage of the research. The low methane contents in the anaerobic reactors resulted in the accumulation of volatile fatty acids in the reactors.

The second stage was started at the 55th day of the research and continued until the end of the research. After the yeast solution addition to the Reactors 2, 3, and 4, the results were sufficient to explain the benefits of yeast addition. At the end of the research, COD values of leachate of the anaerobic reactors (Reactors 1, 2, 3, and 4) decreased 38000 mg/L, 30100 mg/L, 34500 mg/L, and 25500 mg/L respectively.The reactors with yeast addition had lower COD values than the Control Reactor without sludge and no yeast addition.

4.1.1. pH of Leachate of the Anaerobic Reactors

As it can be seen from the Figure 4.1, the initial pH values of leachate generated from the anaerobic reactors (Reactors 1, 2, 3, and 4) are 6.18, 5.79, 5.38, and 5.09, respectively. Then, the pH values of leachate of the anaerobic reactors changed between 5.03 and 6.08 during the first 28 days of the research. These fluctuations were result in the variable conditions in the reactors. After the 28th day, the pH fluctuations decreased and the pH values of leachate of the anaerobic reactors were low during this period. The formation and accumulation of the volatile fatty acids in the anaerobic reactors were the main reason for the reduction in pH. When the pH values of leachate of the anaerobic reactors were compared with each other, it might be seen that this value as the highest (6.18) in Reactor 1 (Control Reactor), while it is the lowest (5.03) in Reactor 4. The pH values of leachate of the Reactor 2 and 3 are between the pH values of leachate of the Reactor 1 and 4. According to the results, the reactors containing sludge had the lower pH values than the Control Reactor without sludge.

After the 55th day of the research, yeast solution was added to the Reactors 2, 3, and 4 except the Control Reactor in order to enhance the decomposition rates of the organic wastes. After the addition of yeast solution, low pH values were observed in leachate of the Reactors 2, 3, and 4 until the 94th day. The pH values of leachate of the Reactors 2, 3, and 4 are in the range of 5.44 to 6.06. All experimental results revealed that yeast solution addition resulted in decrease in the pH and ORP values whereas an increase in the COD removal efficiencies at the same time period. On the other hand, after the 94th day, a slight increase in the pH values was observed in leachate of the Reactor 2, 3, and 4 until the end of the study. The increase in pH in the anaerobic reactors might be explained by the conversion of the volatile fatty acids by methanogens.

When the pH of leachate of the Control Reactor was measured, it was found that, the pH value of leachate of the Control Reactor was lower than the other reactor for the same period. However, at the end of the 115th day, a slight increase in the pH values was detected in the Control Reactor. This lower pH value can be explained by the slow stabilization rate in the Control Reactor as there was no sludge or yeast addition applied to this reactor.

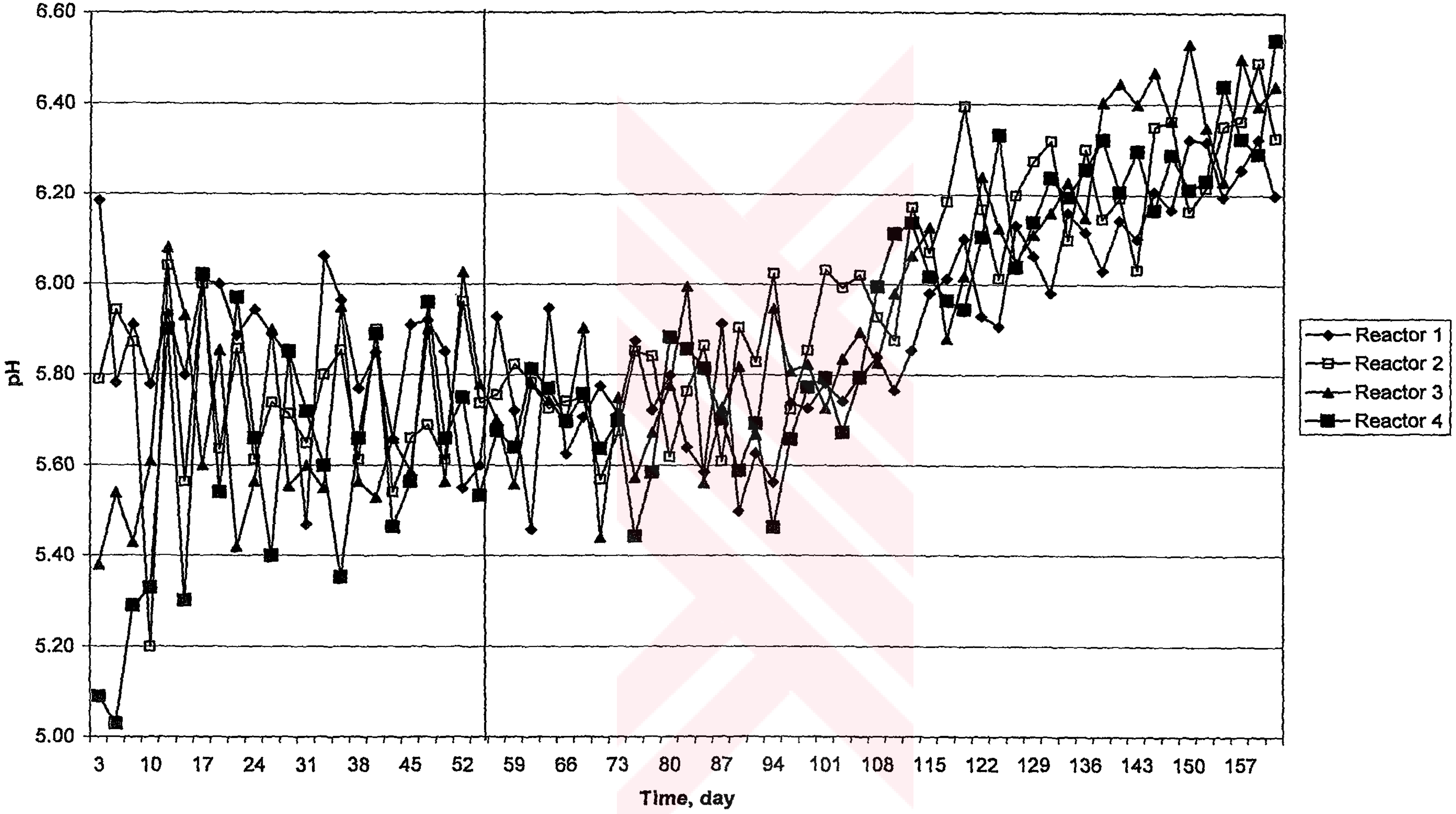


Figure 4.1. pH values of the leachate generated from the anaerobic reactors.

4.1.2. Oxygen Reduction Potential of Leachate of the Anaerobic Reactors

Oxygen reduction potential (ORP) is an indicator of the reducing or oxidizing strength of the leachate. A positive potential value indicates that leachate has an oxidizing effect which means the aerobic conditions exist. In contrast to a positive potential, a negative ORP value shows that leachate has a reducing effect indicating the anaerobic conditions. The ORP measurement of leachate of the all reactors began at the 3rd day of the study and it was monitored till the end of the study. The ORP data is given in Figure 4.2.

At the 3rd day of the research, ORP values of leachate of the anaerobic reactors (Reactors 1, 2, 3, and 4) are 20 mV, 10.7 mV, 8.4 mV, 5.8 mV, respectively. The inital ORP values of leachate of the anaerobic reactors are positive indicating the aerobic conditions in all reactors. When the presence of oxygen was depleted by the aerobic microorganisms, the negative ORP values were observed in all reactors until the end of the first stage. However, the Control Reactor had positive ORP values between the 15th-17th day of the study. It might be arisen from the experimental errors or gas leakage.

After the yeast addition, because of the highly reducing conditions in the reactors, the ORP values of leachate of the anaerobic reactors were decreased. At the beginning of the second period, the ORP values of leachate of the Reactors 1, 2, 3, and 4 were -50.2 mV, -61.4 mV, -34 mV, -44.4 mV and reached to -80.9 mV, -92 mV, and -50 mV, -124.8 mV until the end of the research, respectively. These low negative ORP values and low pH values showed the establishment of the anaerobic environment in the anaerobic reactors.

The ORP values were measured continuously. However, because of the closing down of the laboratory and some technical problems, between the 119th-129th days of the research, the ORP measurements could not be accomplished. After the 129th day, the ORP values of leachate of the reactors were continued to be measured. When the reactors were compared with each other, it was seen that the Reactor 4 was arrived to the anaerobic conditions much more quickly and had low ORP values throughout the research. After the yeast addition, the reactors with sludge had a high decomposition rate. While, the Control Reactor without sludge and yeast addition had a low decomposition efficiency.

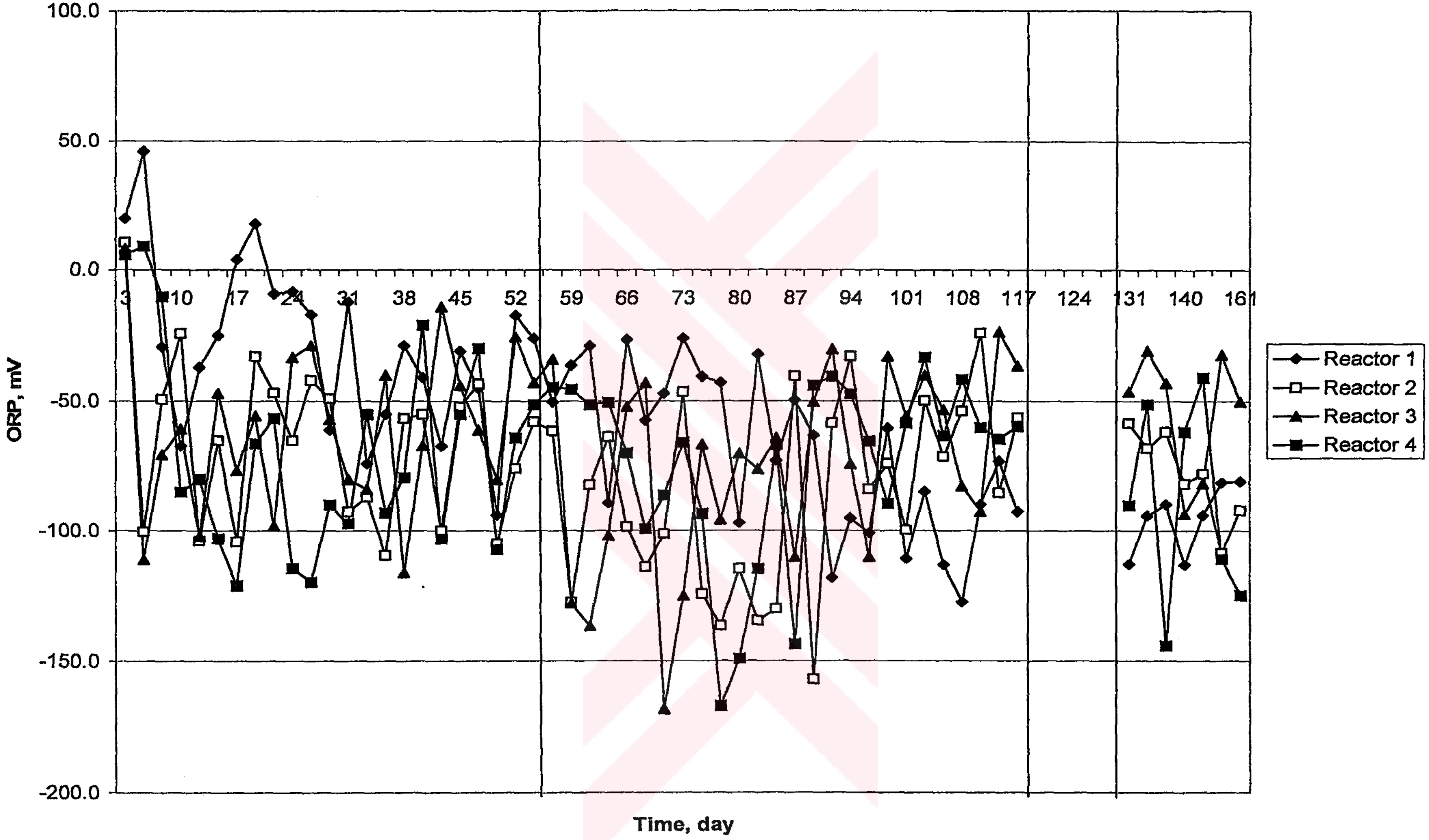


Figure 4.2. ORP values of leachate of the anaerobic reactors.

4.1.3. Chemical Oxygen Demand of Leachate of the Anaerobic Reactors

COD measurements started at the 3rd day of the research like the ORP measurements. The COD of leachate generated from the anaerobic reactors are presented in Figure 4.3.

During the first stage, it was observed that the COD values of leachate of the four anaerobic rectors showed an increasing trend because of the accumulation of volatile fatty acids. The initial COD values in the Reactors 1, 2, 3, and 4 were 41600 mg/L, 47700 mg/L, 50582 mg/L, and 51738 mg/L, respectively. The differences in the initial COD concentrations might be arisen from the loading of different type and amount of sludges. The reactors with sludge (Reactors 2, 3, and 4) had high COD concentrations until the end of the first stage. At the end of the first stage, the COD values in the Reactors 2, 3, and 4 were 56800 mg/L, 60300 mg/L, and 63700 mg/L, respectively. However, the Control Reactor without any sludge had lower COD values. The COD value of leachate of the Control Reactor was 55600 mg/L at the end of the first stage. The COD concentrations of leachate of the Reactor 4 were much higher than the COD concentrations of leachate of other three reactors. It might be concluded that loading belt-press sludge resulted in high decomposition rate of organic materials, as the belt-press sludge consisted of the mixture of primary settling and waste activated sludge. The high COD values and the low pH values of the leachate indicated the acidogenic phase in all the reactors. Any COD removal was not observed in the anaerobic reactors during this stage.

After the addition of yeast solution, no significant changes in the COD values of leachate of the Reactors 2, 3 and 4 were observed between the 56th and 77th day of the study. The values are in the range of 53700 to 64511 mg/L. After the 77th day, the COD values of leachate of these reactors decreased sharply. Adaptation of yeast to the environment might be the reason for this sharp decline. On the other hand, the COD values of leachate of the Control Reactor increased until the 87th day, and then started to decrease at the end of the research. At the end of the research, the COD removal efficiencies for the Reactors 1, 2, 3, and 4 were found as 40, 60, 55, and 66 per cent, respectively. It might be seen that the reactors with sludge after the yeast addition had higher COD removal efficiencies. However, the Control Reactor without sludge and no yeast addition had a lower COD removal efficiency (40 per cent) in comparison with the others. It might

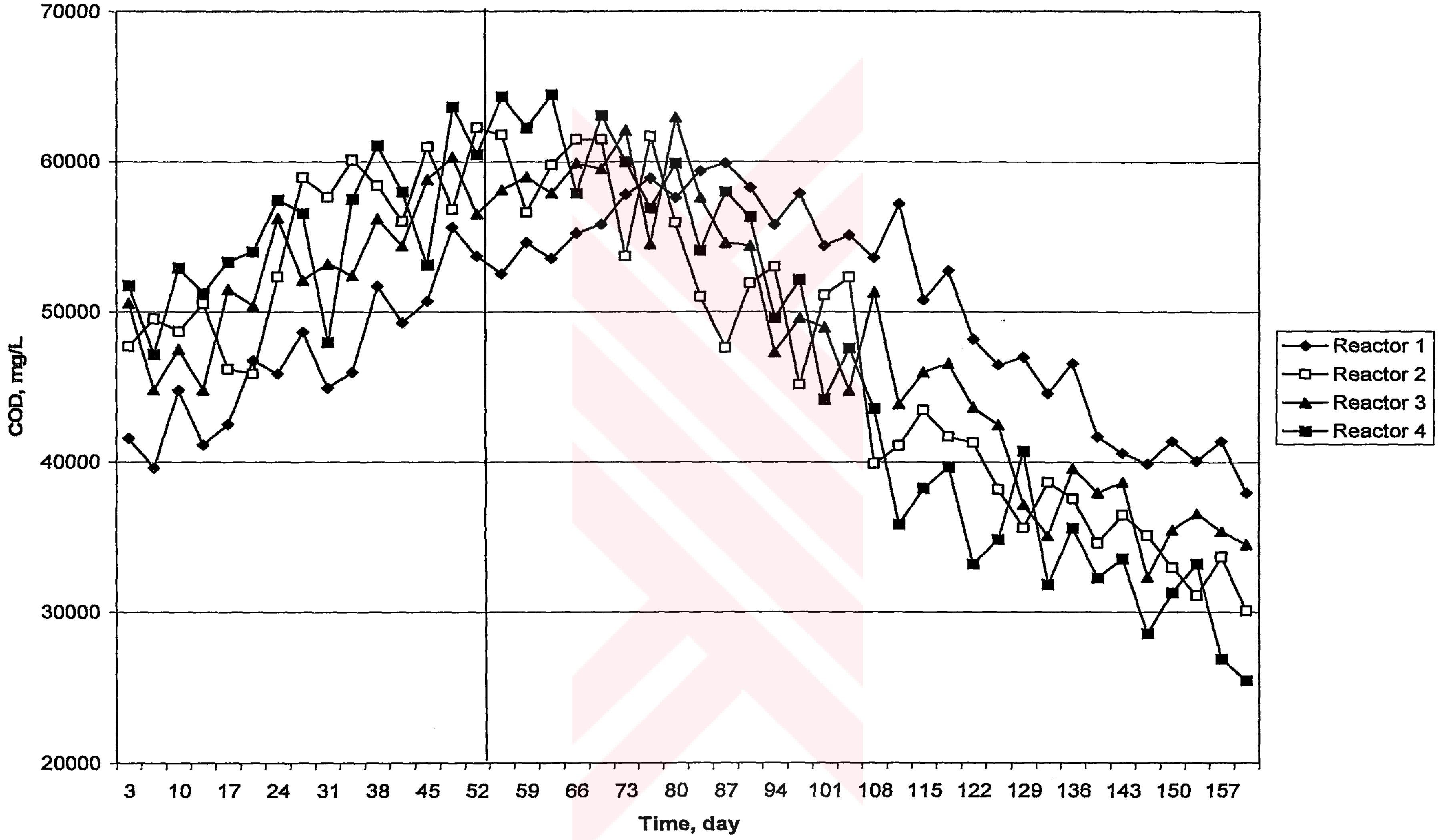


Figure 4.3. COD values of leachate of the anaerobic reactors.

be concluded that the addition of yeast improved the decomposition of organics and reduced the COD concentrations of leachate of the reactors.

4.1.4. Alkalinity of Leachate of the Anaerobic Reactors

The alkalinity in wastewater resists to change in pH. Alkalinity results from the presence of the hydroxides, carbonates and bicarbonates of elements such as calcium, magnesium, sodium, potassium or ammonia. Of these, calcium and magnesium bicarbonates are most common. The alkalinity data of leachate of the anaerobic reactors are given in Figure 4.4. for each reactor.

At the 3rd day of the research, the alkalinity values of leachate of the anaerobic reactors (Reactors 1, 2, 3, and 4) are 1720 mg/L, 1560 mg/L, 1864 mg/L, and 2040 mg/L as $CaCO_3$ respectively. Then, alkalinity values of leachate of anaerobic reactors changed in the range of 1730-3965 mg/L as $CaCO_3$ during the first stage. These alkalinity values of leachate of the anaerobic reactors were sufficient to buffer the reactors. The decomposition of organic materials contributed to the presence of high alkalinity concentrations of the reactors. It was desired that alkalinity should be at least 2000 mg/L as $CaCO_3$. Buffer addition was not made in this stage.

After the addition of the yeast, the alkalinity values of leachate of the Reactors 2, 3, and 4 exhibeted a stationary trend from the 55th to 79th day. The alkalinity values of leachate of the anaerobic reactors (Reactors 2, 3, and 4) changed in the range of 2460-3475 mg/L as $CaCO_3$. It might be resulted from the accumulation of the volatile fatty acids in the reactors. At the same period, alkalinity of leachate of the Control Reactor had higher alkalinity values than the others. The alkalinity values of leachate of the Control Reactor changed in the range of 2884-3717 mg/L as $CaCO_3$ between the 55th and 79th days. Between the 79th and 160th days, the alkalinity values of leachate of the reactors were fluctuated. This fluctuation might be resulted from the changing in the volatile acid concentrations of the reactors as it was supported by the pH data and the COD removal data. The alkalinity of leachate of the Control Reactor had lower alkalinity concentration when compared with the other reactors during the same period.

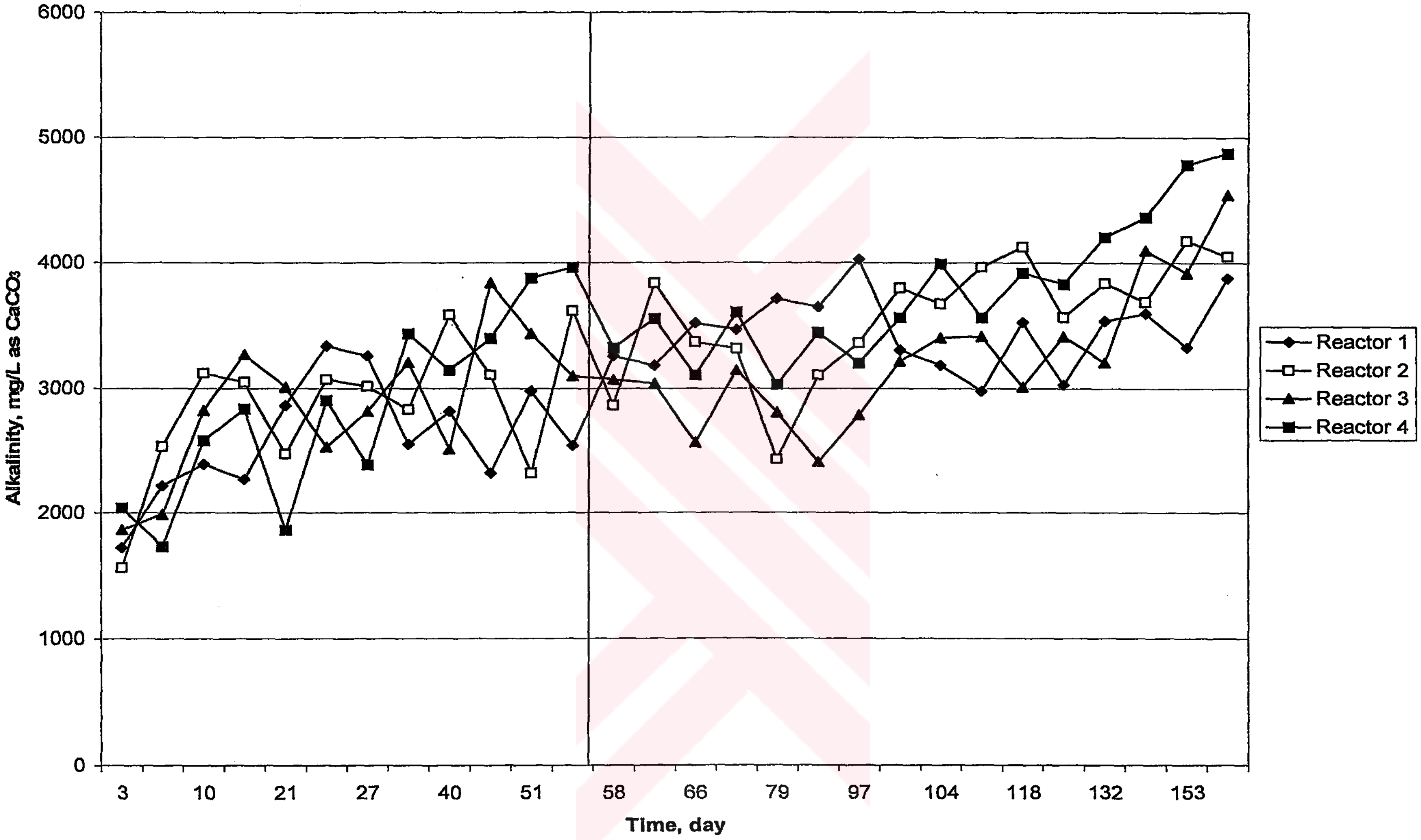


Figure 4.4. Alkalinity values of leachate generated from the anaerobic reactors.

At the end of the study, the alkalinity values of leachate of the Reactors 1, 2, 3, and 4 were 3880 mg/L, 4052 mg/L, 4545 mg/L, and 4880 mg/L as $CaCO_3$, respectively. As it could be seen that the reactors having yeast addition had higher alkalinity content, while the Control Reactor without sludge and yeast addition had low alkalinity content. This might be explained by the rate of the anaerobic decomposition in the reactors. In addition to the alkalinity value, the anaerobic conditions were established in the Control Reactor later than the others because of the fatty acids accumulation and their slow utilization.

4.1.5. Orthophosphate Concentrations of Leachate of the Anaerobic Reactors

The orthophosphate determination is an important parameter in environmental engineering practices. The available phosphorous for the microorganisms was measured as orthophosphate. The orthophosphate of leachate generated from the each anaerobic reactor were given Figure 4.5.

The initial orthophosphate concentrations of leachate of the Reactors 1, 2, 3, and 4 are 214 mg/L, 230mg/L, 208 mg/L, and 195 mg/L, respectively. The initial orthophosphate concentrations were very close to each other. During the first stage, there were fluctuations in the reactors in terms of phosphate concentrations. These fluctuations were attributed to the irregular microbial utilization and biological decomposition. The orthophosphate concentrations were reduced through the end of the first stage. The final orthophosphate concentrations of leachate of the Reactors 1, 2, 3, and 4 are decreased to 167 mg/L, 131 mg/L, 140 mg/L, and 106 mg/L, respectively.

After the yeast addition, orthophosphate concentrations of leachate of the Reactors 2, 3, and 4 followed a stationary trend. At the beginning of the second stage, the orthophosphate concentrations of leachate of the Reactors 2, 3, and 4 were 127 mg/L, 123 mg/L, and 103 mg/L, respectively. These data showed that the addition of yeast stock solution did not affect the reactions immediately. Adaptation time of yeast to the environment might be resulted in this stationary trend.

After the 62nd day, orthophosphate concentrations of Reactors 2, 3, and 4 followed a decreasing trend until the end of the study. On the other hand, the orthophosphate

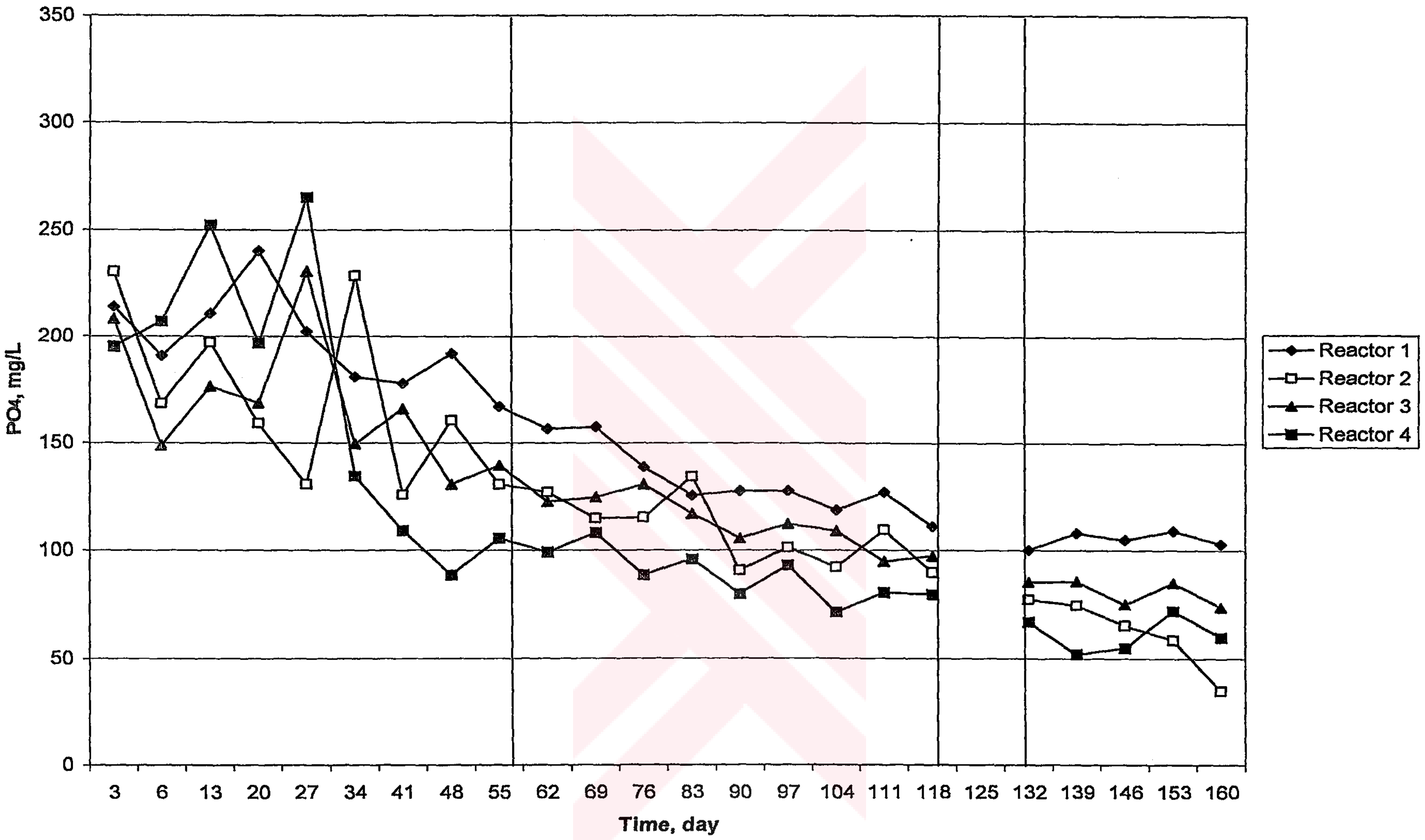


Figure 4.5. Orthophosphate concentrations of the leachate generated from the anaerobic reactors.

concentrations of leachate of the Control Reactor without sludge and yeast decreased gradually from 157 to 103 mg/L in the same period. It might be concluded that the utilization of phosphates by microorganisms reduced the orthophosphate concentrations of the Control Reactor.

At the end of the study, the phosphate removal efficiencies of the Reactors 1, 2, 3, and 4 are found as 60, 85, 68, and 76 per cent, respectively. The highest phosphate removal was observed in the Reactor 2. This high removal rate might be resulted from the efficiency of anaerobic bacteria in the reactor.

4.1.6. Sulfate Concentrations of Leachate of the Anaerobic Reactors

Sulfate can be used as an electron acceptor under the anaerobic conditions. Sulfate measurement is significant in preventing the sulphide toxicity in an anaerobic system. Sulfate might be converted to sulphide under high reducing conditions. The sulfate measurement in the reactors was started at the 3rd day of the study. The sulfate data of each reactor are given in Figure 4.6.

At the 3rd day of the research, the initial sulfate values of leachate of the Reactors 1, 2, 3, and 4 were found as 156 mg/L, 193 mg/L, 230 mg/L, 275 mg/L, respectively. The initial sulfate values of the reactors were low because of the insufficient decomposition of solid wastes in the reactors. When the biologic decomposition was increased by the microorganisms, the high sulfate concentrations were observed in the reactors until the end of the first stage. At the end of the first stage, the final sulfate values of leachate of the Reactors 1, 2, 3, and 4 were 384 mg/L, 503 mg/L, 480 mg/L, 587 mg/L, respectively. When the sulfate concentrations of leachate of the Reactor1 and Reactor 4 were compared during the first stage, it is observed that the sulfate value was the lowest (156 mg/L) and the highest (628 mg/L) in these reactors, respectively. As it is shown in Figure 4.6, the reactors with sludge had low pH and ORP values and high COD and sulfate concentrations.

After the yeast addition, sulfate concentrations of the Reactors 2, 3, and 4 followed a decreasing trend. This decreasing trend was continued till the end of the research.

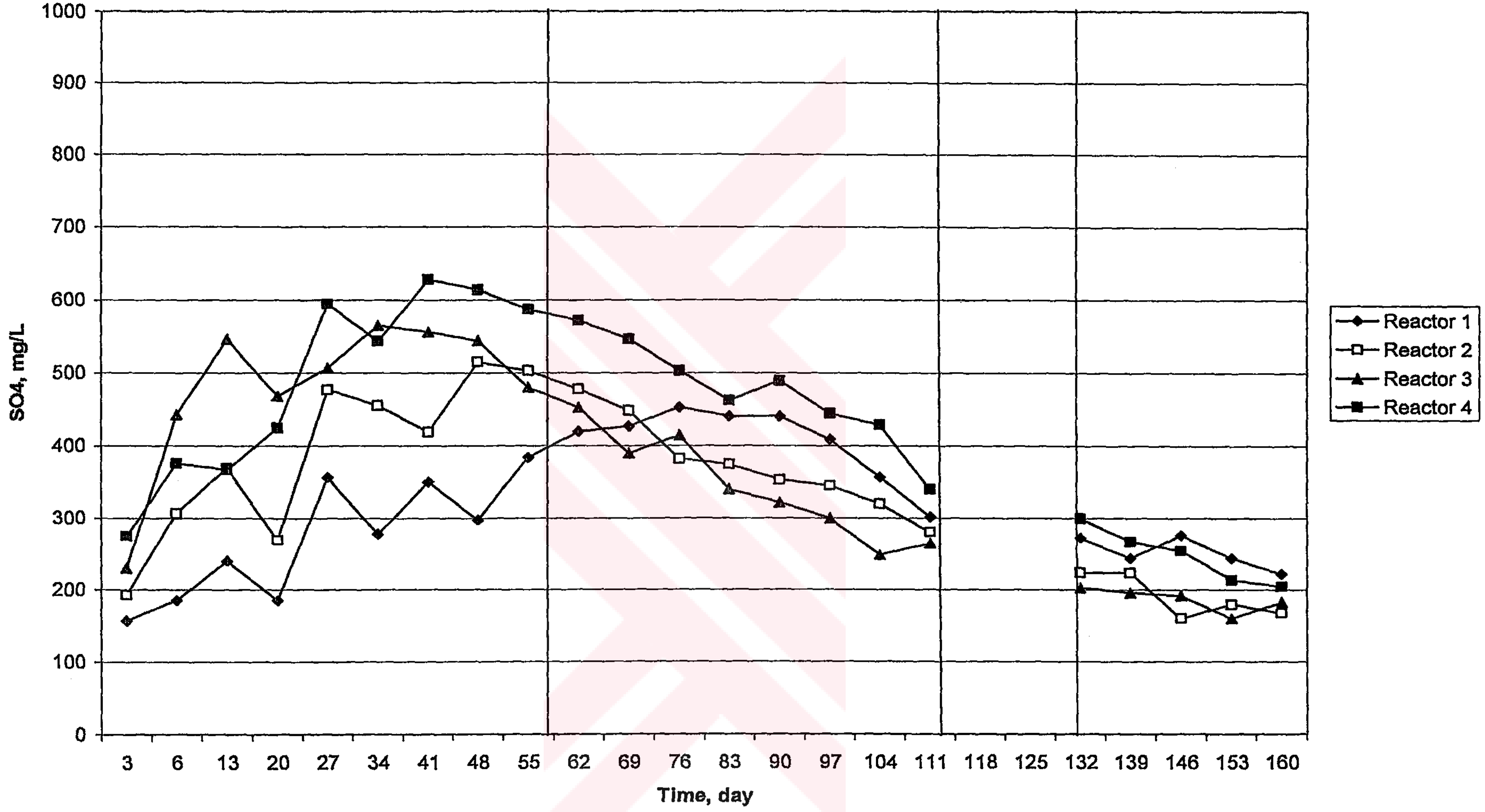


Figure 4.6. Sulfate concentrations of the leachate generated from the anaerobic reactors.

On the other hand, the sulfate concentrations of leachate of the Control Reactor increased (440 mg/L) until the 90th day, and then the sulfate concentrations decreased till the end of the research. At the beginning of the second stage, the sulfate concentrations of leachate of the Reactors 1, 2, 3, and 4 are 420 mg/L, 478 mg/L, 452 mg/L, and 572 mg/L, respectively. Whereas, the final sulfate concentrations of leachate of the Reactors 1, 2, 3, and 4 are decreased to 222 mg/L, 168 mg/L, 183 mg/L, and 205 mg/L, respectively. Throughout this stage, ORP values of the reactors ranged between -17 and -168 mV. These values showed that the ORP values were not enough to supply highly reducing conditions for methanogenic microorganisms. Thus, sulfate could not be converted to sulphide completely.

4.1.7. Total Kjeldahl Nitrogen Concentrations of Leachate of the Anaerobic Reactors

Organic nitrogen and ammonia might be determined together as "total nitrogen" or "Kjeldahl nitrogen". Nitrogen is one of the essential elements for the growth of the microorganisms. Free ammonia might inhibit anaerobic metabolisms at high concentrations. While the anaerobes might adapt to the high ammonia concentrations, large fluctuations might be harmful to the process. Free ammonia, which is much more toxic than the ammonium ion, is prevalent at high pH (Sawyer, 1978). The TKN measurement began at the 3rd day of the study. The TKN data are given in Figure 4.7.

At the 3rd day of the research, the initial TKN concentrations of leachate of the anaerobic reactors (Reactors 1, 2, 3, and 4) were 4.45 mg/L, 4.78 mg/L, 5.64 mg/L, and 5.95 mg/L, respectively. The initial TKN values of leachate of the reactors are low because of the low because of the insufficient decomposition of organic materials in all anaerobic reactors.

When the decomposition of organic materials under anaerobic conditions began, the high TKN values were observed till the end of the first stage. The maximum TKN values of leachate of the anaerobic reactors (Reactors 1, 2, 3, and 4) are 7.9 mg/L, 8.74 mg/L, 8.60 mg/L, and 8.2 mg/L, respectively.

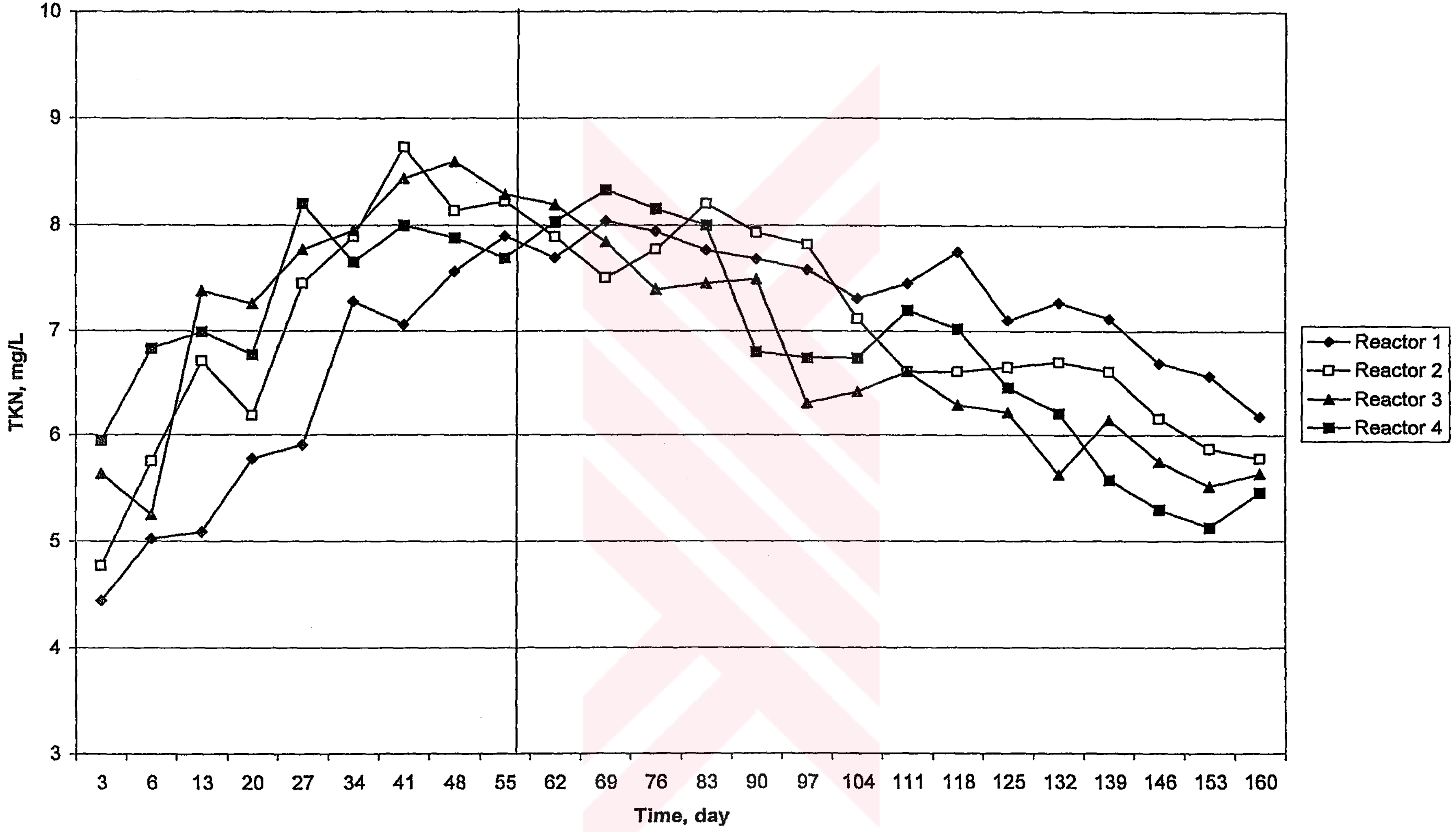


Figure 4.7. TKN values of the leachate generated from the anaerobic reactors.

Between the 55th and 83rd day of the study, no significant changes were observed in the TKN values of leachate of the reactors. The values ranged from 7.39 to 8.33 mg/L. After the 83rd day, a decreasing trend was observed for the Reactors 2, 3, and 4 till the end of the study. This decline was because of the conversion and utilization of nitrogen to new cells by the anaerobic microorganisms.

The final TKN values of leachate of the anaerobic reactors (Reactors 1, 2, 3, and 4) were found as 6.18 mg/L, 5.78 mg/L, 5.64 mg/L, and 5.46 mg/L, respectively. As it can be seen from the Figure 4.7, the Control Reactor without sludge and yeast had higher TKN concentrations than the other reactors. The anaerobic decomposition in the Control Reactor was slower than the other anaerobic reactors.

4.1.8. Chloride Concentrations of Leachate of the Anaerobic Reactors

Chloride was measured in order to estimate of the dilution and evaporation effects in this study. Chloride is a conservative and tracer component. Therefore, it resists to chemical and biological reactions. The changes in the chloride concentrations can be expressed to the dilution effects and evaporation of water (San, 1999). In the research, chloride measurement of the leachate generated from the anaerobic reactors began at the 10th day of the study and it was monitored till the end of the study. The chloride data is displayed in Figure 4.8.

At the 10th day of the research, the chloride concentrations of four reactors (Reactors 1, 2, 3, and 4) were 2089 mg/L, 1855 mg/L, 1416 mg/L, and 1682 mg/L, respectively. During the first stage, it was observed that the chloride concentrations of the leachate generated from the four rectors showed a decreasing trend because of the dilution effect. At the end of first stage, the final chloride concentrations of four reactors (Reactors 1, 2, 3, and 4) were 1393 mg/L, 1060 mg/L, 1232 mg/L, and 1185 mg/L, respectively.

After the yeast addition, chloride concentrations of leachate of the Reactors 2, 3, and 4 followed a stationary trend. At the 61st day of the research, the chloride concentrations of leachate of the Reactors 1, 2, 3, and 4 were 1312 mg/L, 1095 mg/L, 1196 mg/L, and 1202 mg/L, respectively. These data showed that the addition of yeast stock solution to the

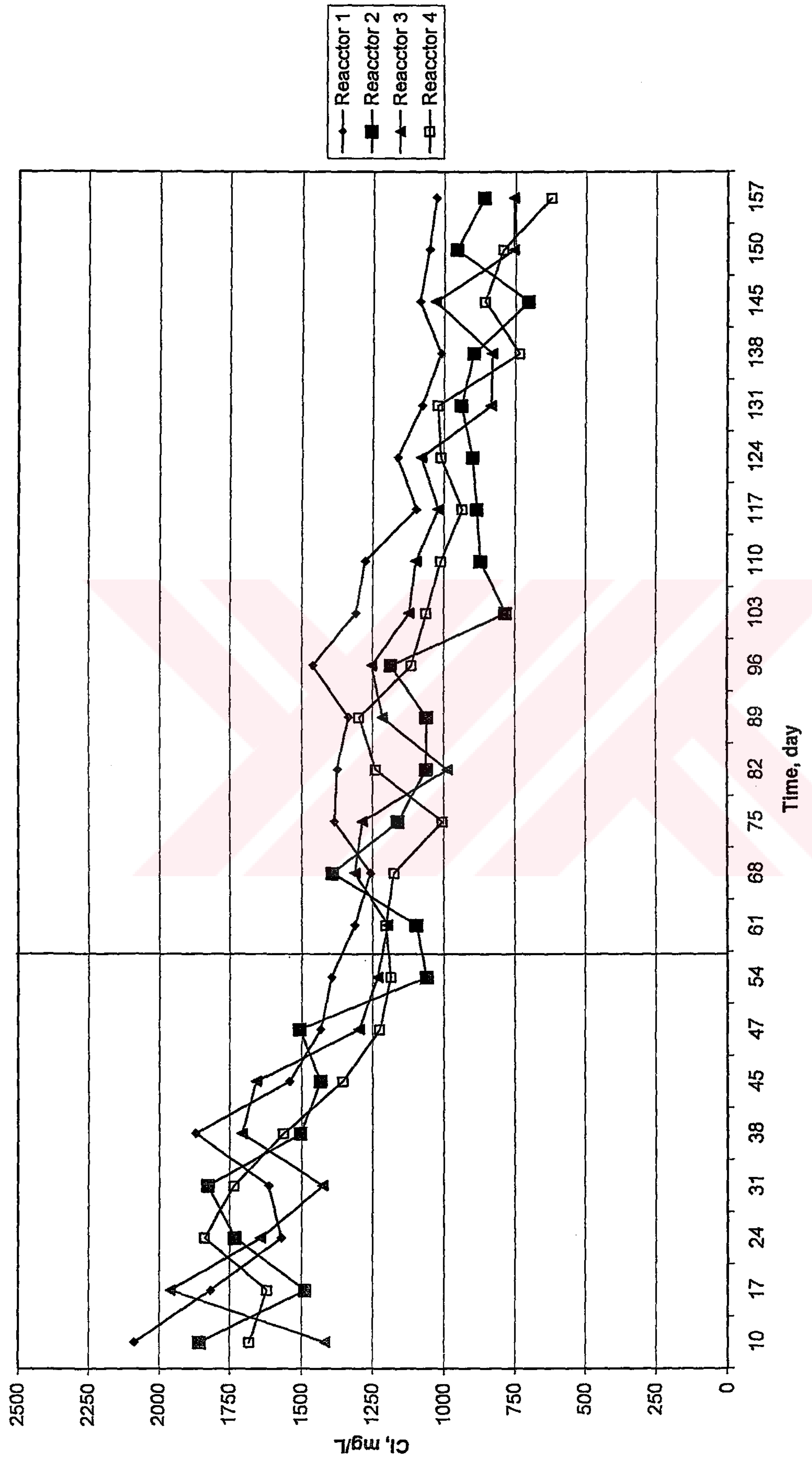


Figure 4.8. Chloride concentrations of the leachate generated from the anaerobic reactors.

Reactors 2, 3, and 4 did not affect the biologic decomposition reactions immediately. After the adaptation time of yeast to the environment, chloride concentrations of leachate of the Reactors 2, 3, and 4 followed a decreased trend until the end of the study. However, between the 75th and 96th day of the study, the chloride concentration of leachate of the Control Reactor increased slightly. After the 96th day, the chloride values of leachate of the Control Reactor decreased gradually.

At the end of the study, the chloride concentrations of leachate of the Reactors 1, 2, 3, and 4 are found as 1028 mg/L, 862 mg/L, 757 mg/L, and 623 mg/L, respectively. The reductions of chloride concentrations of leachate of all anaerobic reactors were because of dilution effects.

4.1.9. Heavy Metal Concentrations of Leachate of the Anaerobic Reactors

Heavy metals can be found in the leachate generated from the landfills and the treatment plant sludges. Heavy metals create many adverse effects to the environment. Their elimination and disposal are a significant problem because of their treatment costs. Heavy metals in landfill leachate are subject to complexation, oxidation-reduction reactions, sorption and precipitation. These processes cause attenuation of heavy metals in landfill leachate. The heavy metal concentrations of leachate generated from the anaerobic reactors are given in Figures 4.9, 4.10, 4.11, 4.12, 4.13, and 4.14.

The initial heavy metal concentrations of leachate (copper, lead, nickel, zinc, cadmium and chromium) in Reactors 1, 2, 3, and 4 were low because of the insufficent decomposition in all reactors. When the decomposition of the organic materials under the anaerobic conditions continued, high heavy metal concentrations were observed in the reactors.

During the first stage, the reactors mixed with sludge had higher heavy metal concentrations than the reactor without sludge, which means that the sludge increased the heavy metal concentrations. The high heavy metal concentrations might be explained by the addition of metal coming from the sludges and the existence of anaerobic conditions in the reactors.

After the yeast addition, the heavy metal contents of the reactors began to decrease. The addition of yeast provided highly reducing conditions in the reactors. Under the highly reducing conditions, sulfate reduced to sulfide and it precipitated with metals as insoluble sulfide compounds. The decreases of sulfate concentrations of the leachate might also confirm this reaction. On the other hand, chromium concentrations of the reactors continued to increase after the yeast addition. This state was attributed to the chromium characteristics. Chromium could precipitate with hydroxide. When the pH increased, chromium contents of leachate of the reactors decreased. In general, sulfide precipitation is an effective attenuation of metals. The other attenuation techniques are less effective than precipitation.

4.2. Gas Production and Composition of the Anaerobic Reactors

4.2.1. Daily Gas Production of the Anaerobic Reactors

The reactors had aerobic conditions during the first days of the research because of the intrusion of air during the placement of the solid wastes and sludges into the reactors.

Since the solid wastes used in the research were taken from Kemerburgaz Landfill Area which was far away from the laboratory, some of the organic components of the solid wastes were already decomposed when the samples reached to the laboratory. Therefore the volumes of the landfill gases generated were less than the expected values. The daily gas production of each reactor is given in Figure 4.15. Moreover, the daily gas production values of the anaerobic reactors are given in Appendix A.

During the first stage, the gases generated in the Reactors 1, 2, 3, and 4 are in the ranges of 0-875 mL, 0-1010 mL, 0-1120 mL, and 0-1160 mL, respectively. High gas production was observed at the beginning of the research. The reason of this high gas production might be attributed to the high decomposition rate of organic materials aerobically. When the daily gas productions of the reactors were compared to each other, the daily gas production was higher in the reactors with sludge (Reactors 2, 3, and 4) than the Control Reactor without sludge. The maximum daily gas productions of the anaerobic reactors were This might be explained by the effects of the sludge mixed with solid wastes.

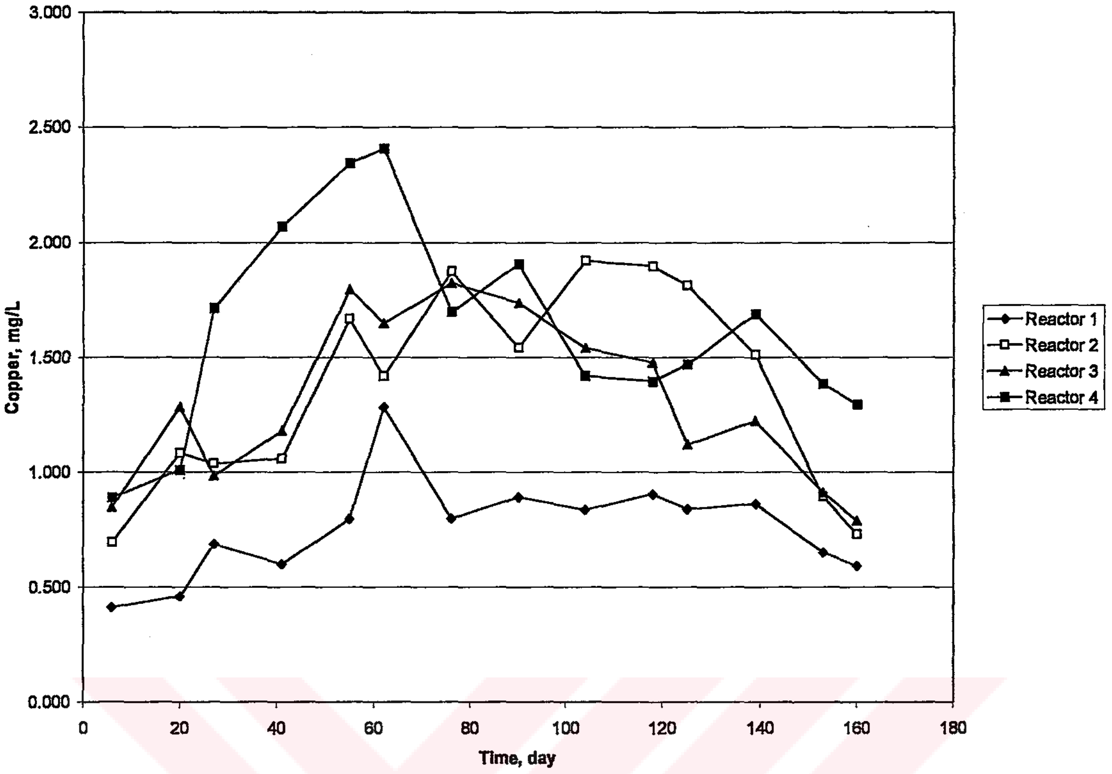


Figure 4.9. Copper concentrations of the leachate generated from the anaerobic reactors.

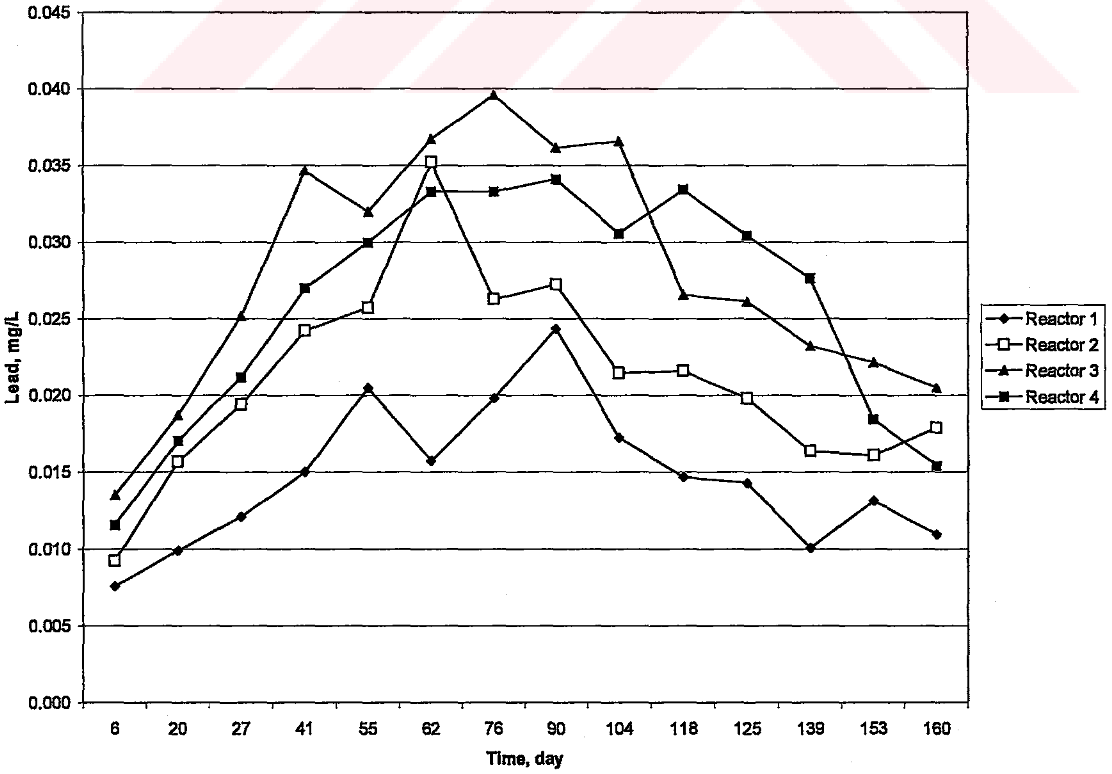


Figure 4.10. Lead concentrations of the leachate generated from the anaerobic reactors.

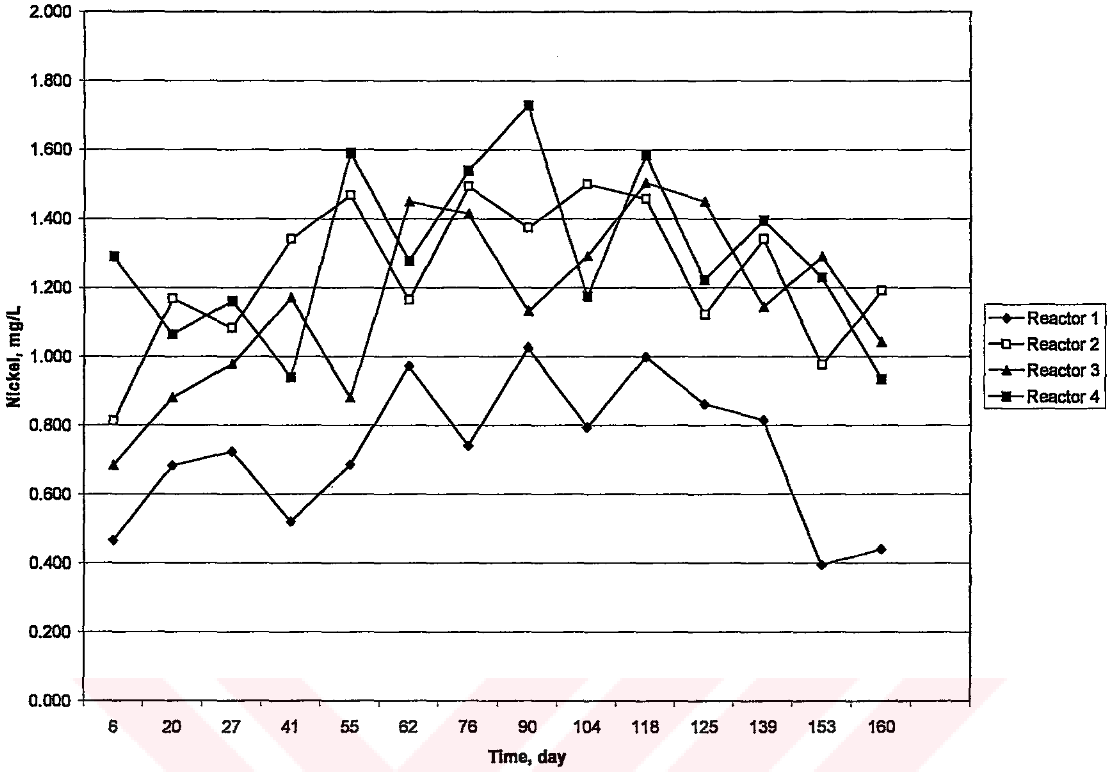


Figure 4.11. Nickel concentrations of the leachate generated from the anaerobic reactors.

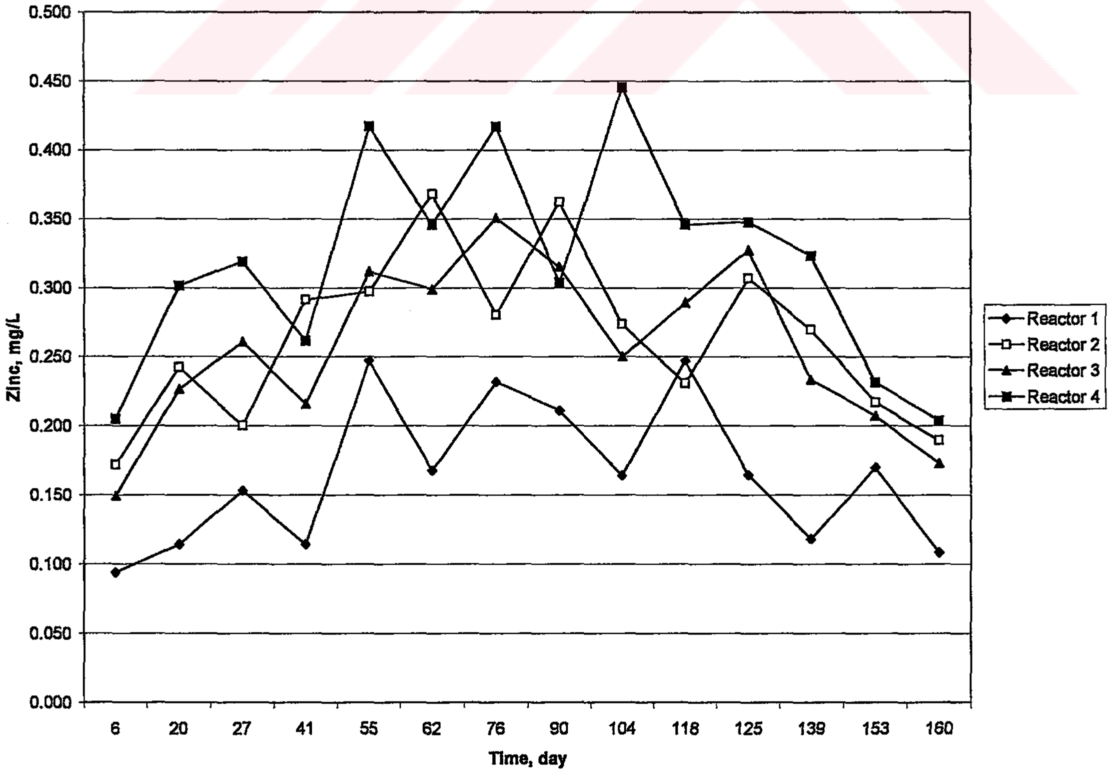


Figure 4.12. Zinc concentrations of the leachate generated from the anaerobic reactors.

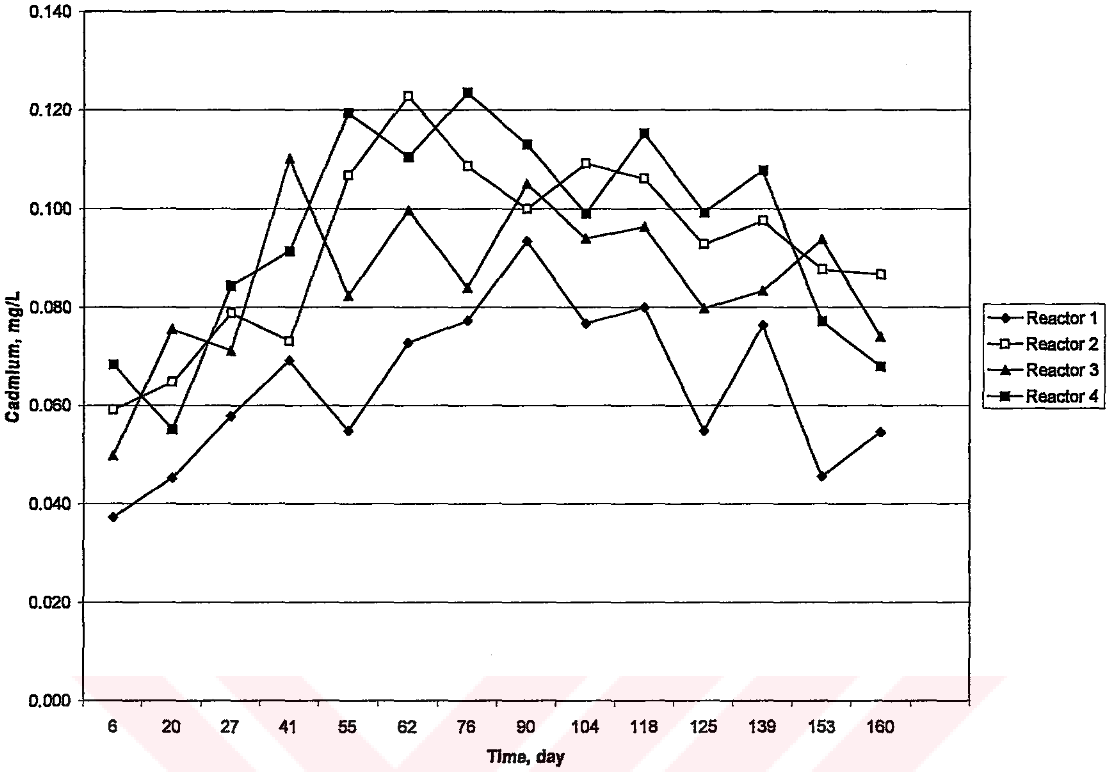


Figure 4.13. Cadmium concentrations of leachate of the anaerobic reactors.

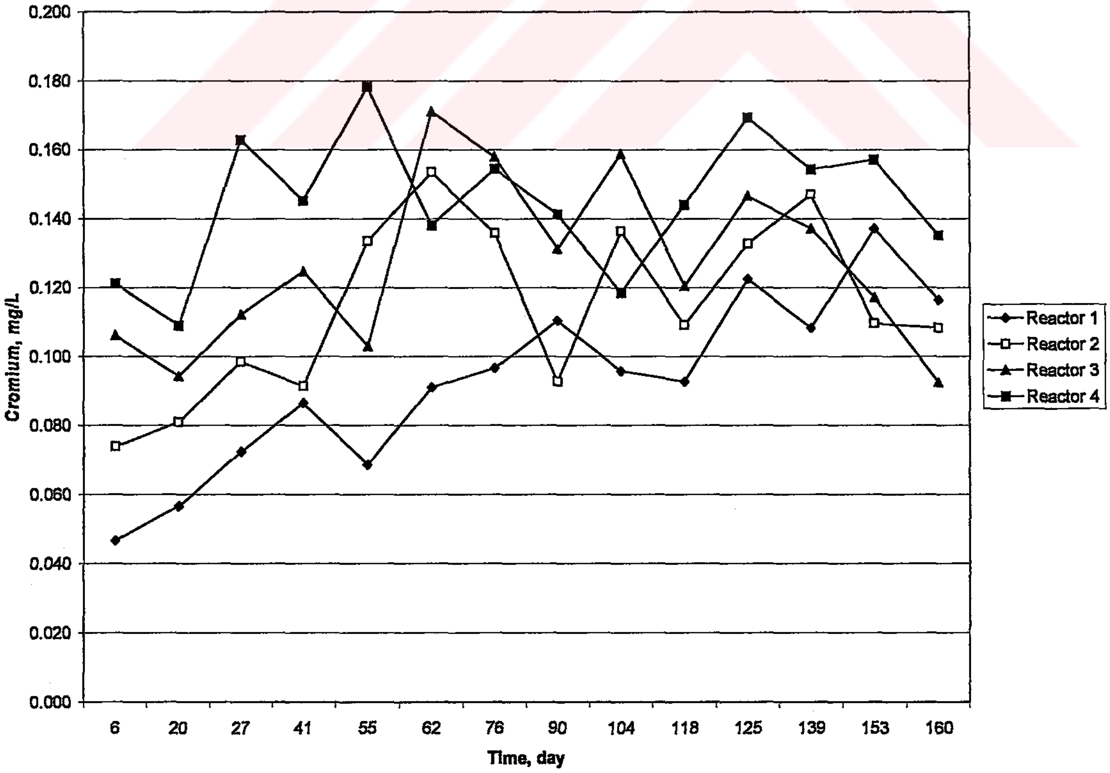


Figure 4.14. Chromium concentrations of the leachate generated from the anaerobic reactors.

Between the 55th and 65th days, the gas production of the anaerobic reactors with yeast addition followed a stationary trend. The gas generated in the Reactors 2, 3, and 4 are range of 220-486 mL, 0-509 mL, 137-454 mL, respectively. However, the daily gas production in the Control Reactor without sludge and yeast increased at the same time period. After the 65th day, the gas production in the Reactors 2, 3, and 4 also increased gradually until the 106th day. Adaptation of yeast to the environment might have promoted these gradual increases. On the other hand, the gas production of the Control Reactor did not change significantly during the same period.

Between the 106th and 159th days, the daily gas productions of the Reactors 2, 3, and 4 with yeast addition were declined. As in the other reactors, the gas production of the Control Reactor also decreased gradually during the same period. The gas production of the Reactors 1, 2, 3, and 4 are range of 0-412 mL, 0-403 mL, 0-412 mL and 0-421 mL, respectively. The final gas production values of the Reactors 1, 2, 3, and 4 were 174 mL, 258 mL, 336 ml, and 343 mL, respectively. This decline might be resulted from the insufficient biodegradable materials or presence of refractory compounds in the reactors.

4.2.2. Cumulative Gas Production of the Anaerobic Reactors

Cumulative gas production was found by sum of the daily gas production within the experimental period. Thus, it was related to the daily gas production data of the reactors. Firstly, an increasing trend was seen until the 10th day of the research. Then, a slight increasing trend was lasted at the end of the first stage. After the yeast addition, the cumulative gas production increased gradually, and at the end of the106th day of the research, a slight increasing trend was observed until the end of the study. The cumulative gas productions of each reactor in the first and second stage are presented in Figure 4.16, while the cumulative gas productions of the anaerobic reactors are given in Table 4.1.

As it is mentioned in the previous paragraph, the cumulative gas productions of the reactors were increased sharply in the first days of the research. This increase was resulted from the rapid decomposition of the organic matters. At the end of the first trend, the cumulative gas production followed an increasing trend but the slope of this trend was low.

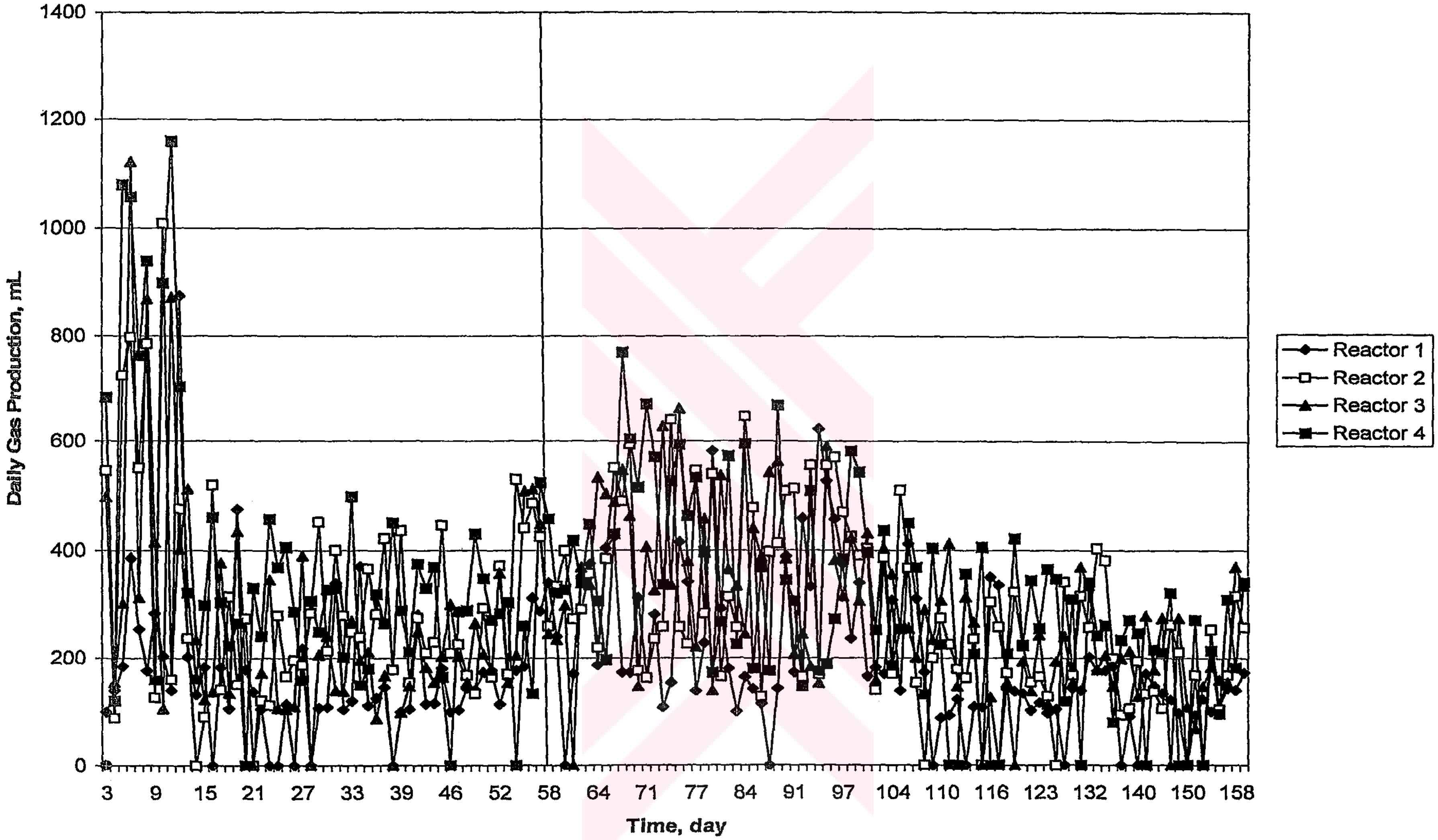


Figure 4.15. Daily gas production in the anaerobic reactors.

Table 4.1. Cumulative gas production of the anaerobic reactors

Reactor	**Cumulative Gas Production of the Reactors within the First Stage, mL**	**Cumulative Gas Production of the Reactors within the Second Stage, mL**	**TOTAL**
reactor 1	8690	17840	26530
reactor 2	15880	25290	41170
reactor 3	13500	25250	38750
reactor 4	19370	27130	46500

4.2.3. Gas Composition of the Anaerobic Reactors

Main gases generated from the landfill site are methane, ammonia, hydrogen sulfide and carbon dioxide. The conversion of carbon dioxide to methane is brought about by the cooperation of specific group microorganisms. In the research, gas composition analyses began at the 49th day because of equipment problem. The gas compositions of each reactor are presented in Figures 4.17, 4.18, 4.19, and 4.20.

At the 49th day of the research, the initial methane percentages of the Reactors 1, 2, 3, and 4 are 12.3, 15.5, 14.8, and 16.1 per cent, respectively. This low methane contents of the anaerobic reactors are because of the acidogenic conditions in the reactors. The low methane percentage in the generated gases is not high. Therefore, the percentage of methane must be raised.

After the yeast addition, the methane contents of the Reactors 2, 3, and 4 did not change immediately. But after the 67th day of the research, the methane percentage in the generated gases increased gradually. The maximum methane percentages of the Reactors1, 2, 3, and 4 are observed as 34.4, 58, 55.5, and 63.3 per cent, respectively. When the reactors with yeast were compared to the Control Reactor without yeast, it can be said that, the yeast solution affected the decomposition of organic materials as well as the quantity and the composition of the gases generated in the reactors.

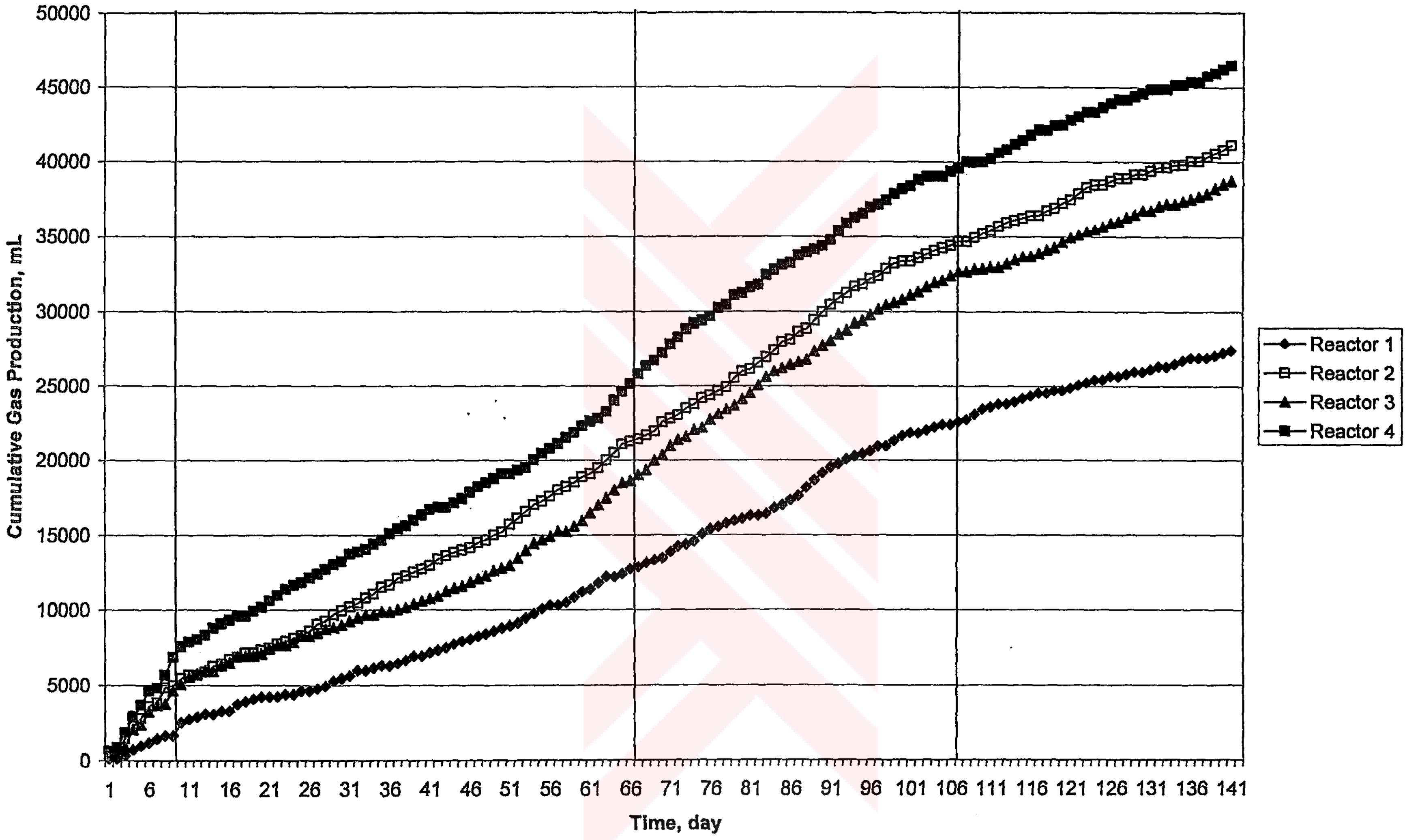


Figure 4.16. Cumulative gas production from the anaerobic reactors.

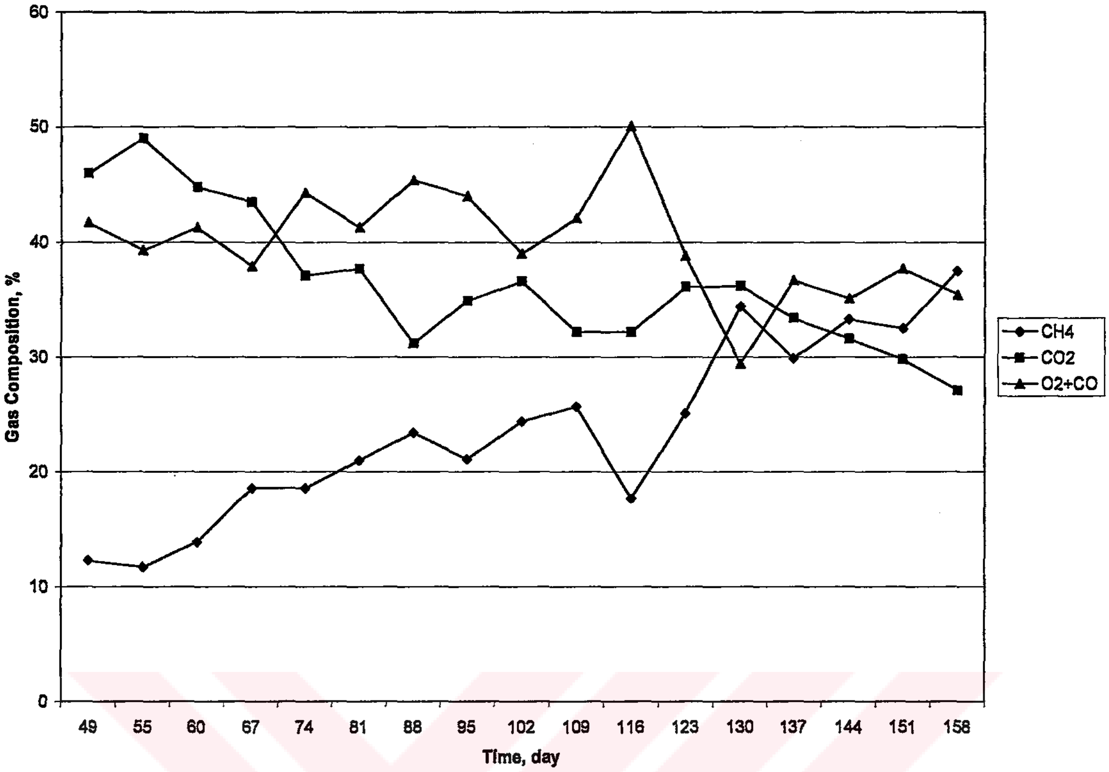


Figure 4.17. Gas composition of the anaerobic reactor 1.

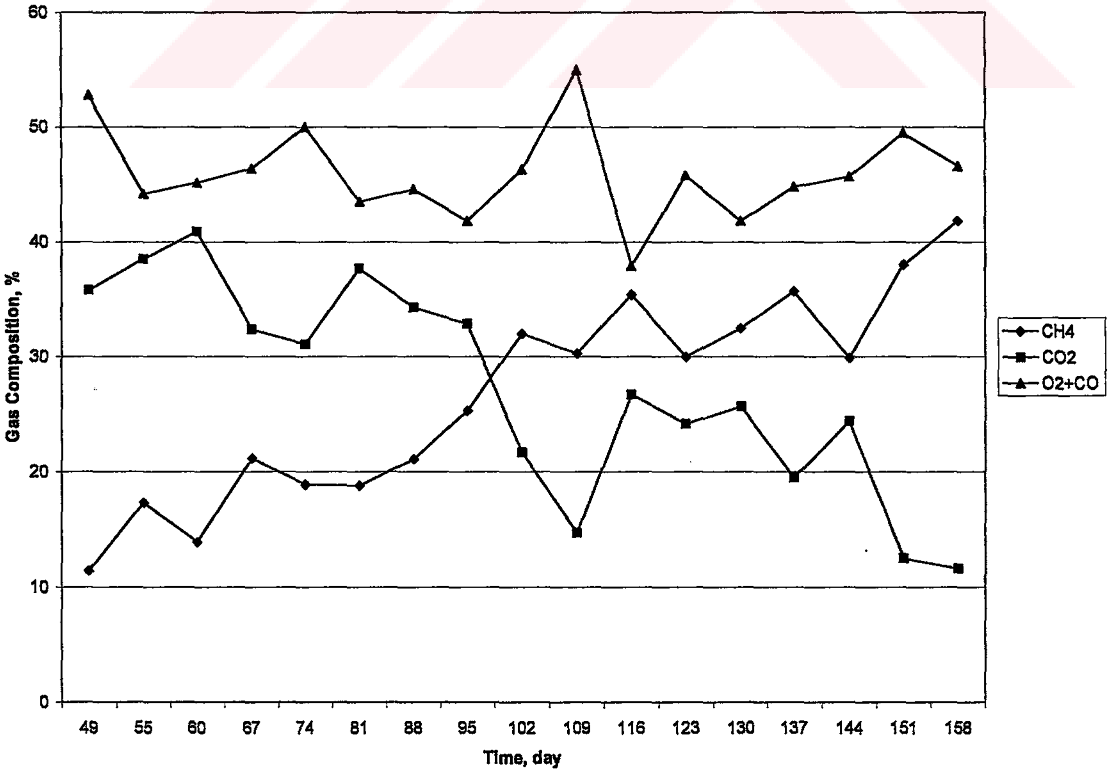


Figure 4.18. Gas composition of the anaerobic reactor 2.

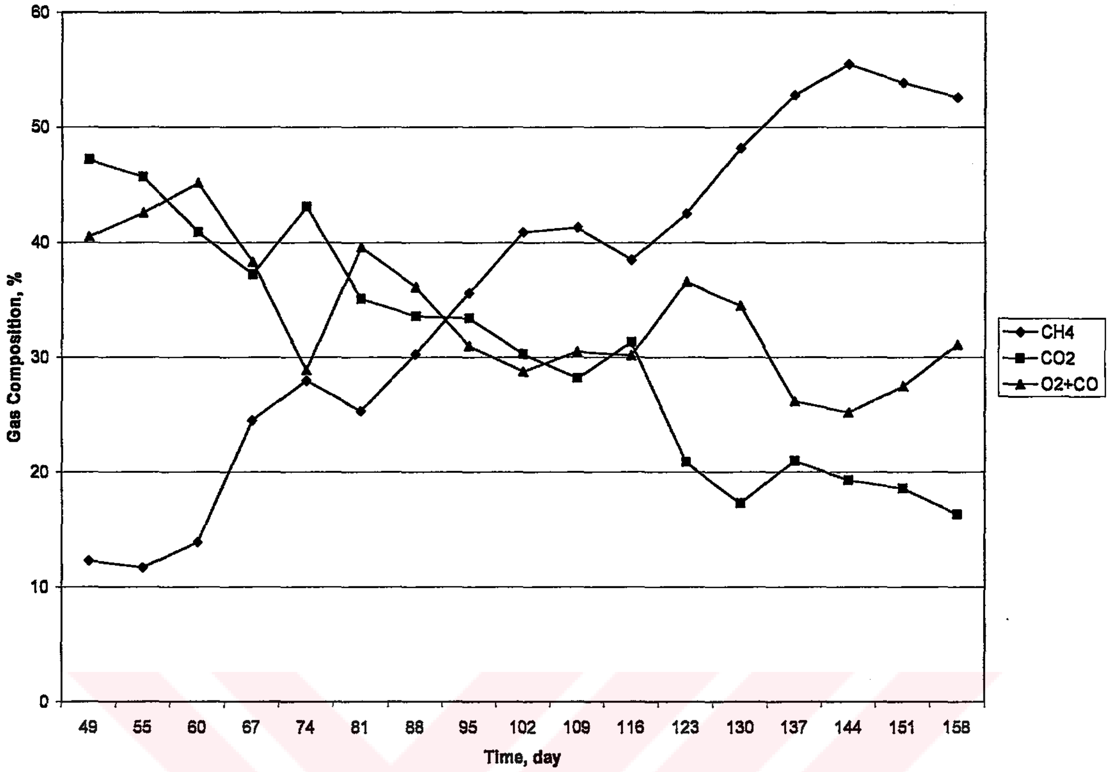


Figure 4.19. Gas composition of the anaerobic reactor 3.

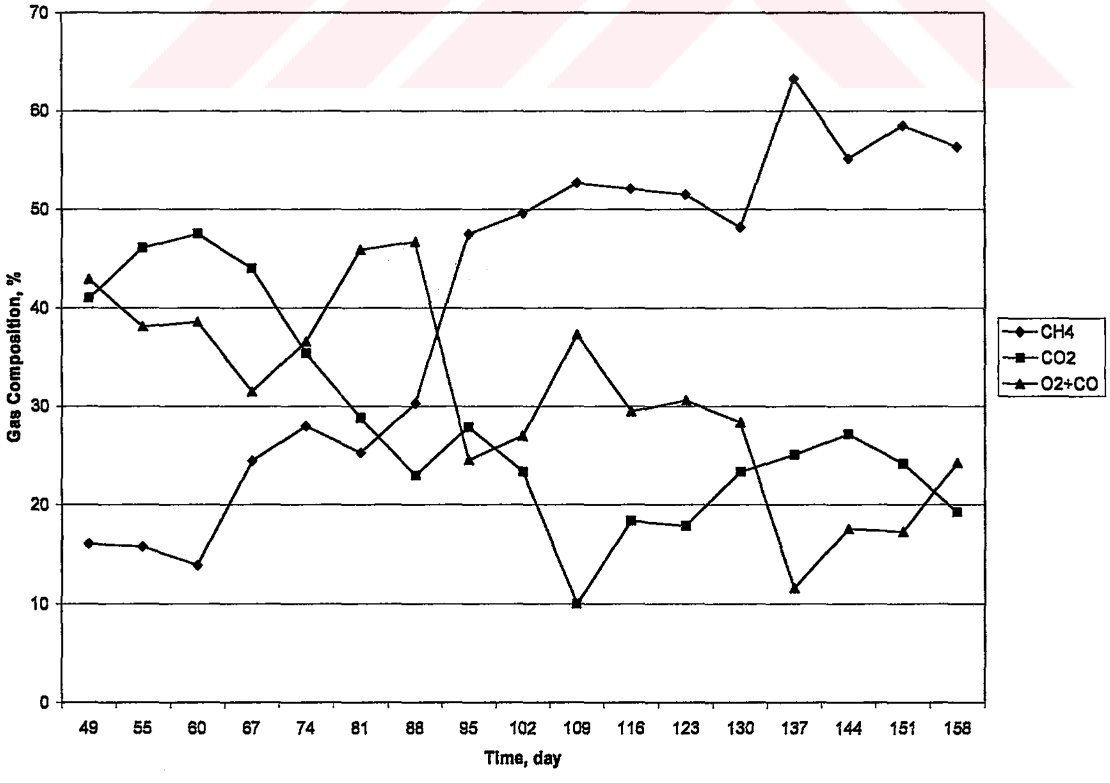


Figure 4.20. Gas composition of the anaerobic reactor 4.

4.3. Analyses of Leachate of the Aerobic Reactors

4.3.1. pH of Leachate of the Aerobic Reactors

pH is also an important parameter for the aerobic bacterial life. The aerobic bacteria are less sensitive to pH changes than the anaerobic ones. The optimal pH for the aerobic bacteria varies from 5 to 9. The pH of the each aerobic reactor is presented in Figure 4.21.

The initial pH values of leachate of the aerobic reactors (Reactors 1, 2, 3, and 4) are 6.35, 6.21, 6.18, and 6.03, respectively. As it can be seen from these values, the initial pH values of leachate of the reactors were very close to each other. At the early stages of the decomposition, the pH values of the reactors with sludge are lower than the Control Reactor without sludge because of the acid formation from the decomposition of organic substances. The pH values of leachate of the Reactors 2, 3 and 4 raised after the 31st day of the research. These increases might be resulted from the acid utilization, as the aerobic microorganisms used organics acids as substrate. By the utilization of organic acids, an increase in the pH of leachate of the reactors was observed and the pH of leachate of the Reactors 2, 3, and 4 reached neutral pH level at the end of the first stage. On the other hand, the pH of leachate of the Control Reactor reached neutral pH level at the 47th day of the research.

After the yeast addition, a stationary level in the pH was observed in the leachate of the Reactors 2, 3, and 4 until the 66th day of the research. The yeast addition might be responsible for these decreases. The COD values of leachate of these reactors also supported these stationary levels.

After the 66th day, a slight increase was observed for the three reactors until the end of the study. Conversion and utilization of acids by respiring bacteria caused this increase in pH and ORP values in the reactors. The pH of the Control Reactor also increased gradually during the same period. At the end of the research, the pH values of leachate of reactors were higher than the initial values of leachate of all aerobic reactors because of the consumption of the biodegradable organic materials.

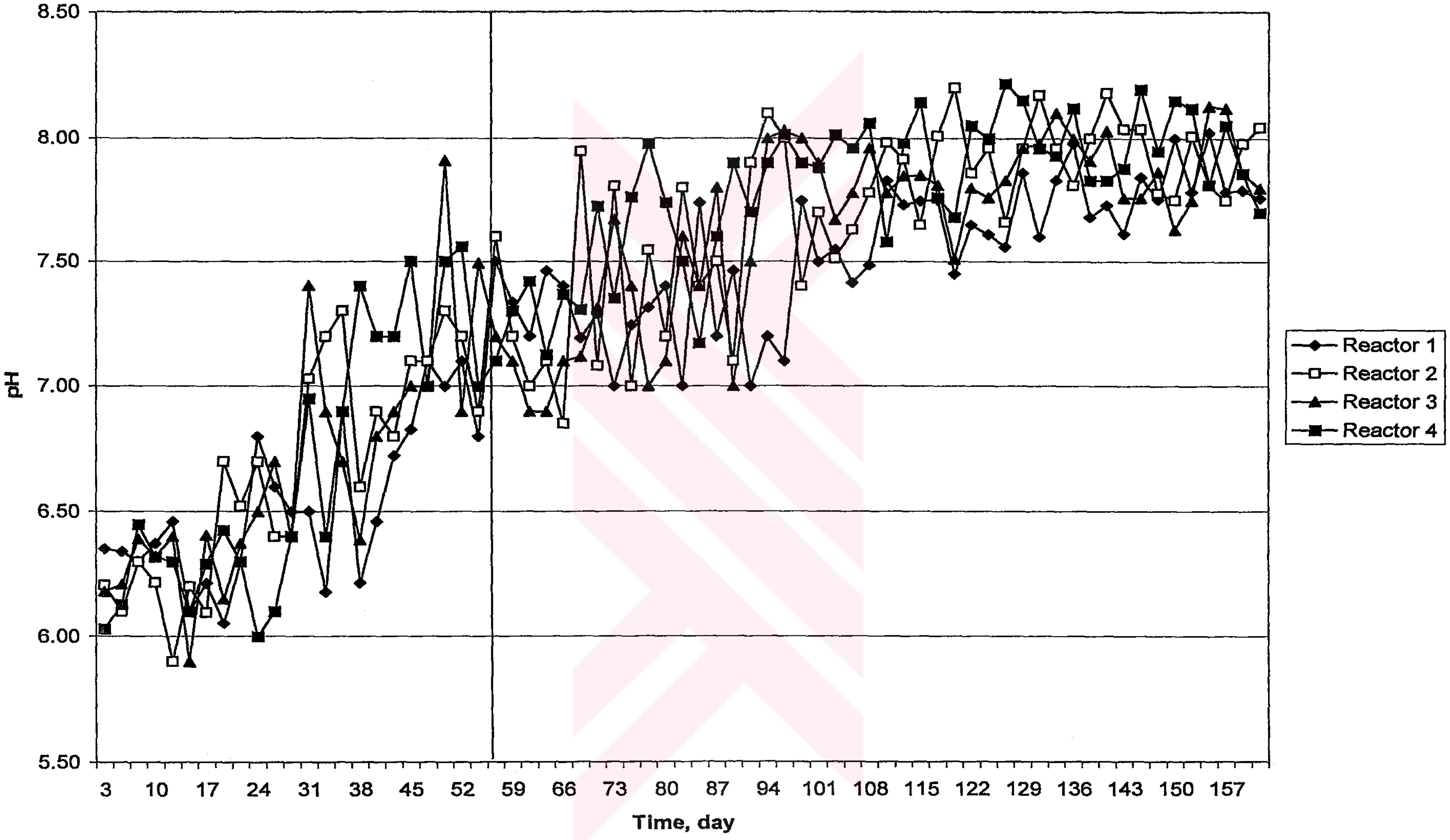


Figure 4.21. pH of the leachate generated from the aerobic reactors.

4.3.2. Oxygen Reduction Potential of Leachate of the Aerobic Reactors

As it is mentioned in the previous sections, a positive ORP value of the leachate indicates the aerobic conditions. In contrast to the positive value, the negative ORP value of the leachate shows the anaerobic conditions. The ORP data of the each aerobic reactor are presented in Figure 4.22.

The initial ORP values of leachate of the aerobic reactors (Reactors 1, 2, 3, and 4) were measured as 14.9 mV, -7.6 mV, 5 mV, and -17.4 mV, respectively. The negative ORP values observed in Reactor 2 and 4 at the begining of the the first stage. This was because of the fact that supply of the oxygen to the reactors was not enough for the aerobic microorganisms. Until the 17th day, the negative ORP values were observed in the leachate of all reactors. These negative results can be explained by the accumulation of the organic acids and insufficient aeration.

After the 17th day of the research, ORP values exhibited an increasing trend and at the end of the first stage, all the ORP values of leachate of the reactors were positive. The final ORP values of leachate of the Reactors 1, 2, 3, and 4 are 47.3 mV, 92.8 mV, 65.5 mV, and 56.7 mV, respectively.

During the second stage, the ORP values of leachate of the Reactors 1, 2, 3. and 4 ranged between 13-80 mV, 36-49.7 mV, 7-74 mV, and 53.4-92.8 mV respectively until the 61st day,. After the 61st day, a slight increasing trend was observed in the reactors with sludge till the end of the study.

At the end of the research, the ORP values of leachate of the Reactors 2, 3. and 4 were 128.6 mV, 123.1 mV, and 146.5 mV, respectively. On the other hand, the ORP of leachate of the Control Reactor were fluctuated until the 96th day. The ORP values of leachate of the Control Reactor ranged between 28.9-90 mV, respectively. Then, the ORP of the Control Reactor increased at the end of the research. At the end of the research, the ORP of the Control Reactor was 138.7 mV. This increasing trend might result from the conversion of acids and sufficient aeration rate.

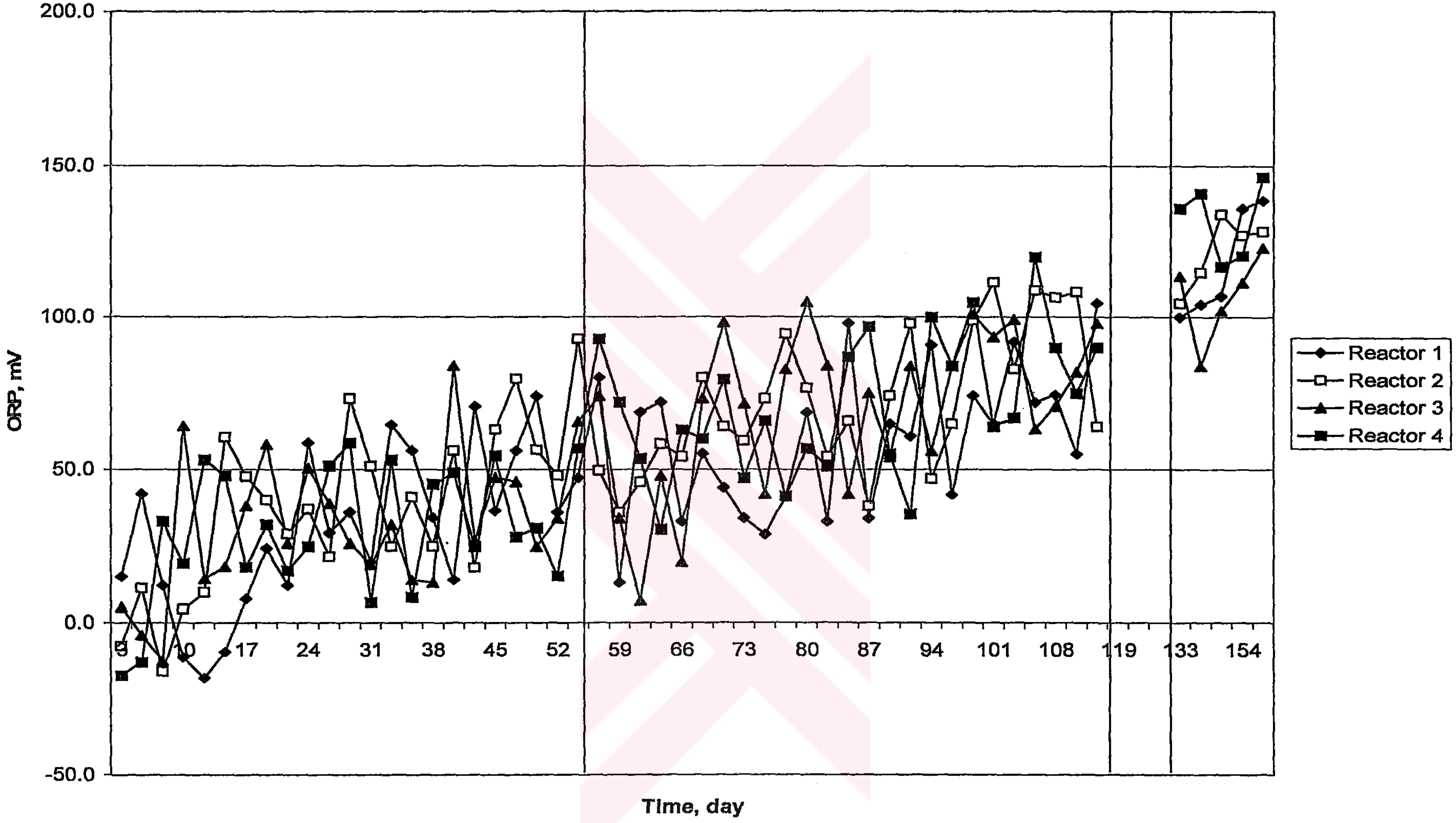


Figure 4.22. ORP of the leachate generated from the aerobic reactors.

4.3.3. Chemical Oxygen Demand of Leachate of the Aerobic Reactors

During the first stage, it was observed that the COD values of leachate of the all aerobic reactors showed a decreasing trend because of the decomposition of the organic materials under the aerobic conditions. The COD values of leachate generated from the aerobic reactors are presented in Figure 4.23.

At the 3rd day of the study, the initial COD values of leachate of the four reactors (Reactors 1, 2, 3 and 4) were found as 54322 mg/L, 50985 mg/L, 52744 mg/L, and 52600 mg/L, respectively. The initial COD differences in reactors were because of the amount and the type of the sludge mixed.

The reactors having sludge (Reactors 2, 3, and 4) reached to higher COD removal efficiencies, in comparison with the Control Reactor. In the first stage, the COD removal efficiencies of the Reactors 1, 2, 3, and 4 were 52, 80, 65, and 74 per cent, respectively. As it can be seen in Figure 4.23, the reactors with sludge represented a high COD removal efficiency. However, the Control Reactor exhibited a less COD removal efficiency.

During the second stage, the COD values of the reactors decreased significantly. The COD values of the reactors ranged between 2300 and 25300 mg/L. It could be determined that the reactors with yeast addition improved COD removal efficiency, since the Control Reactor without yeast had a low COD removal efficiency than the others. At end of the second stage, the COD removal efficiencies of the Reactors 1, 2, 3, and 4 were 67, 68, 76, and 82 per cent, respectively.

4.3.4. Alkalinity of Leachate of the Aerobic Reactors

Alkalinity results from the presence of the carbonates and bicarbonates, and especially hydroxides in the aerobic environments. The alkalinity data of leachate of the aerobic reactors are given in Figure 4.24.

The initial alkalinity values of leachate of the Reactors 1, 2, 3, and 4 are 4355 mg/L, 4784 mg/L, 3826 mg/L, and 2830 mg/L as $CaCO_3$, respectively. Then, the alkalinity values

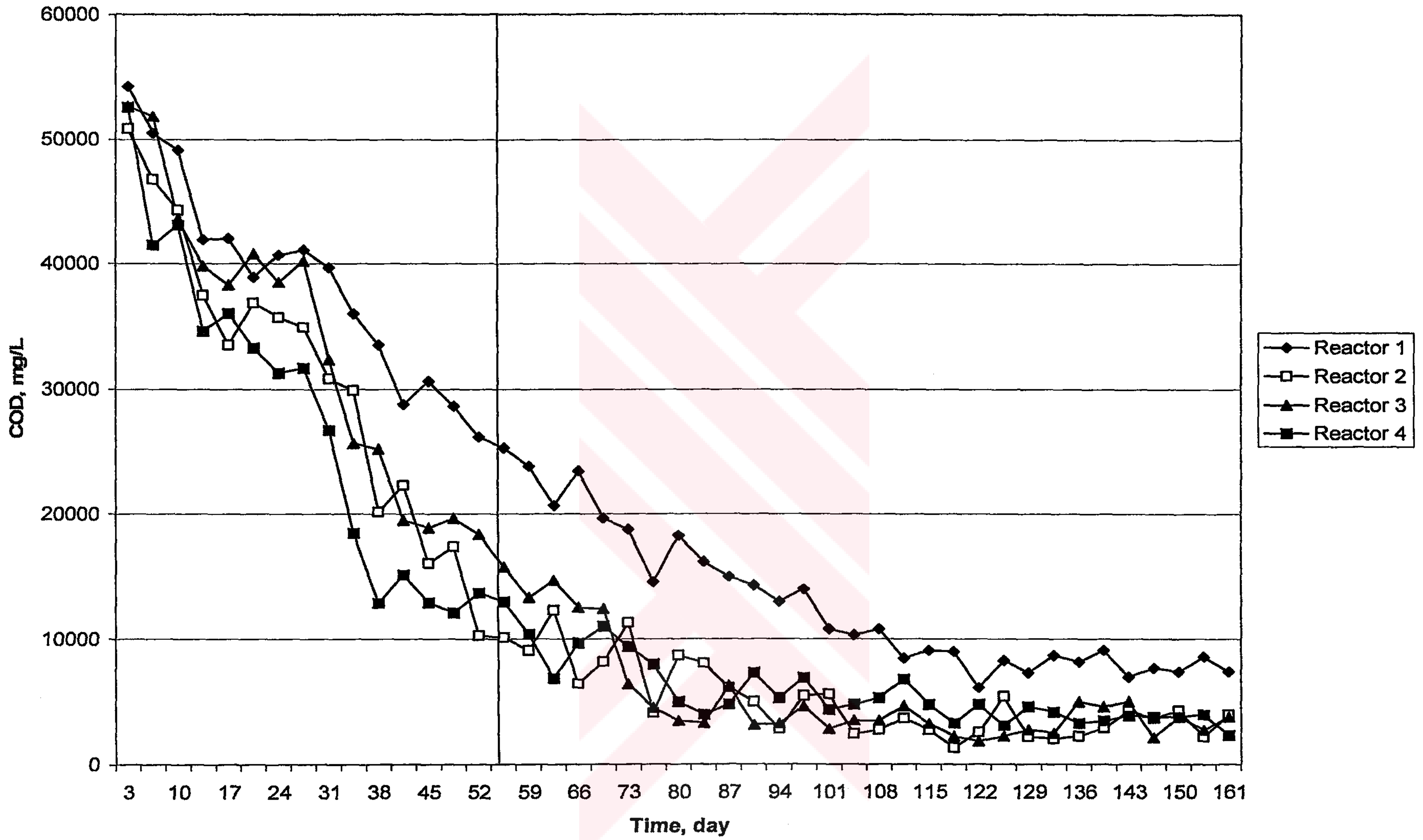


Figure 4.23. COD values of leachate of the aerobic reactors.

of leachate of aerobic reactors increased gradually at the end of the first stage. The alkalinity values of leachate of the aerobic reactors were sufficient to resist the sudden pH changes. The evaporation and utilization of volatile acids in the reactors quickly contributed to the presence of high alkalinity concentrations of the reactors.

After the addition of the yeast, the alkalinity values of leachate of the Reactors 2, 3, and 4 remained constant until the 62nd day of the research. It might be resulted from the yeast addition in the reactors. During the same period, the alkalinity of leachate of the Control Reactor was also close to the values of the others.

After the 62nd day, an increase in the alkalinity values of leachate of the all reactors was observed. This increase might be resulted from the consumption in the volatile fatty acid concentrations of the reactors as it was supported by the pH data and the COD removal data.

At the end of the study, the alkalinity values of the Reactors 1, 2, 3, and 4 were 9818 mg/L, 10170 mg/L, 10887 mg/L, and 11574 mg/L as $CaCO_3$, respectively. As it could be seen that the reactors having yeast had higher alkalinity contents. On the other hand, the Control Reactor without sludge and yeast had low alkalinity values. This might be explained by the rate of the aerobic decomposition in the reactors.

4.3.5. Orthophosphate Concentrations of Leachate of the Aerobic Reactors

The deficiency of the essential substances in the biological systems limits the growth of the microorganisms. In order to get a high biological efficiency, these substances must be monitored. The orthophosphate concentrations of leachate generated from the aerobic reactors are given in Figure 4.25.

At the 3rd day of the research, the initial orthophosphate concentrations of leachate of the Reactors 1, 2, 3, and 4 are 178 mg/L, 220 mg/L, 252 mg/L, and 208 mg/L, respectively. As it can be seen from these values, the initial orthophosphate concentrations differed from each other. Figure 4.5 shows that the loading of the sludge increased the concentration of orthophosphate in the Reactors 2, 3, and 4.

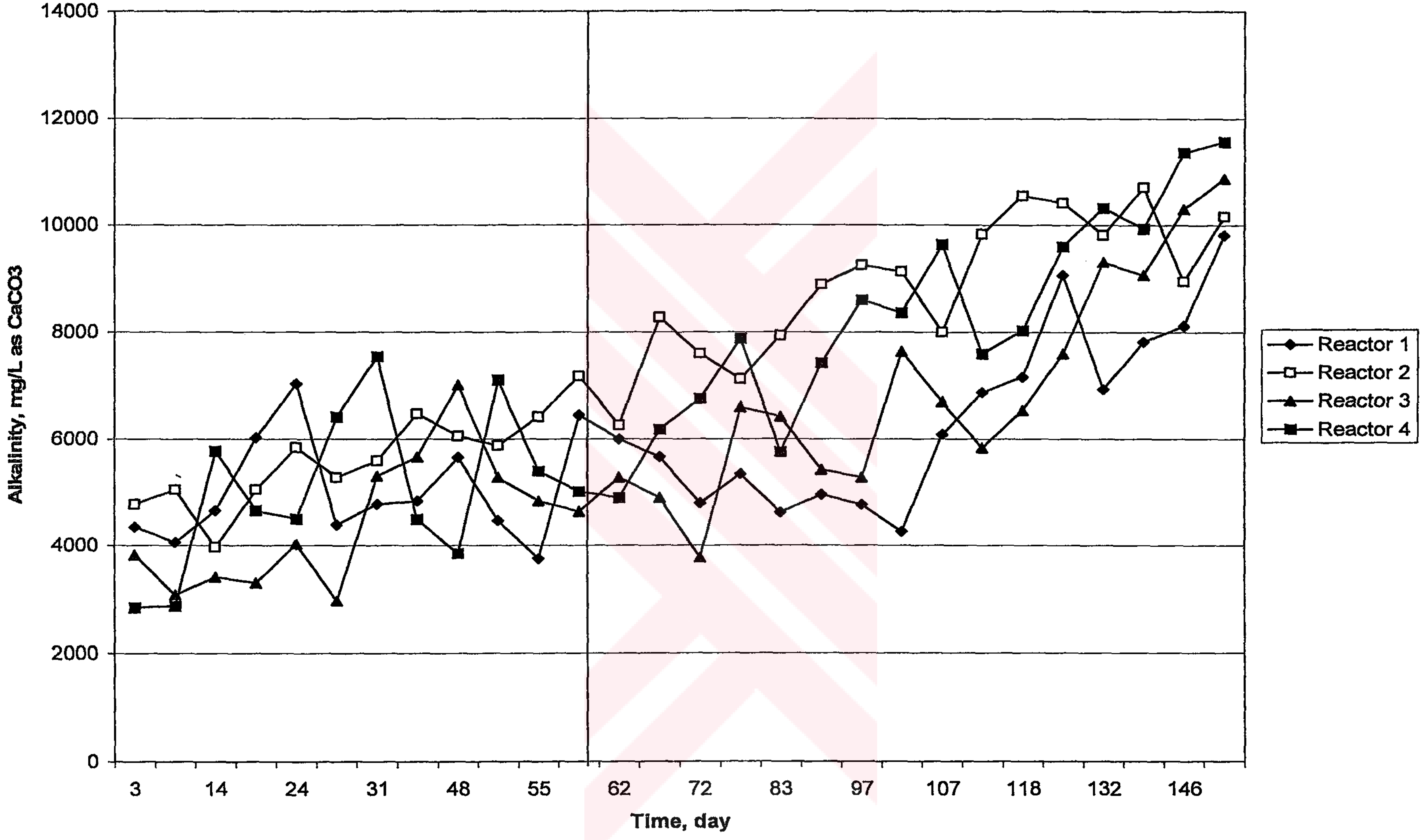


Figure 4.24. Alkalinity values of the leachate generated from the aerobic reactors.

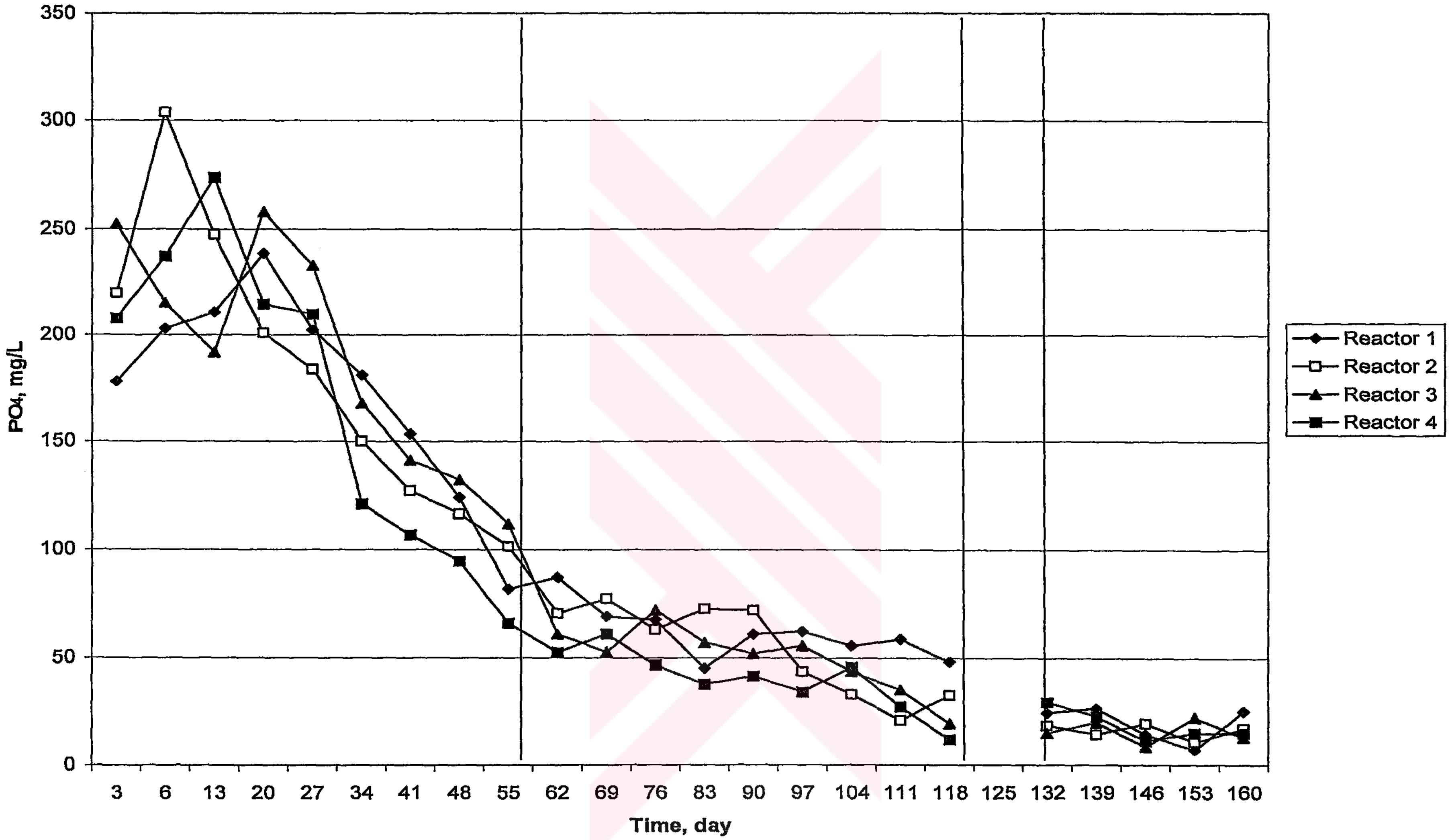


Figure 4.25. Orthophosphate concentrations of the leachate generated from the aerobic reactors.

Until the end of the first stage, the orthophosphate concentrations of the reactors decreased sharply. These sharp decreases were supported by the COD concentrations. The high speed of orthophosphates utilization by the aerobic microorganisms also caused these sharp decreases. At the end of the first stage, the final orthophosphate concentrations of leachate of the Reactors 1, 2, 3, and 4 are decreased to 82 mg/L, 74 mg/L, 71 mg/L, and 66 mg/L,respectively.

In contrast to the first stage, after the yeast addition, the orthophosphate concentrations of leachate of the Reactors 2, 3 and 4 followed a decreasing trend. In the same period, the orthophosphate concentrations of leachate of the Control Reactor also decreased. At the 62nd day of the research, the orthophosphate concentrations of leachate of the Reactors 1, 2, 3, and 4 were 76 mg/L, 71 mg/L, 61 mg/L, and 53 mg/L, respectively.

At the end of the research, the total phosphate removal efficiencies of the Reactors 1, 2, 3, and 4 are 90, 95, 96, and 95 per cent, respectively. The highest phosphate removal efficiency which was in Reactor 3, which was loaded with waste activated sludge, was related to the efficiency of the aerobic bacteria in the reactor.

4.3.6. Sulfate Concentrations of Leachate of the Aerobic Reactors

The sulfate measurement was started at the 3rd of the research. The initial sulfate values of leachate of the Reactors 1, 2, 3, and 4 are 279 mg/L, 345 mg/L, 383 mg/L, and 417 mg/L, respectively. When the sulfate concentrations of leachate of the Reactor 1 and Reactor 4 were compared in the first stage, it can be said that the value was the lowest (279 mg/L) in the Reactor 1 and the highest (827 mg/L) in the Reactor 4, respectively. This might be explained by the effects of the sludge. As it can be seen in Figure 4.26, the reactors with sludge had high sulfate concentrations than the values obtained for the Control Reactor.

By the yeast addition, sulfate concentrations of leachate of the reactors show an increased trend. After the 132th day, the sulfate concentrations of leachate of the reactors followed a stationary trend and this stationary trend was continued till the end of the

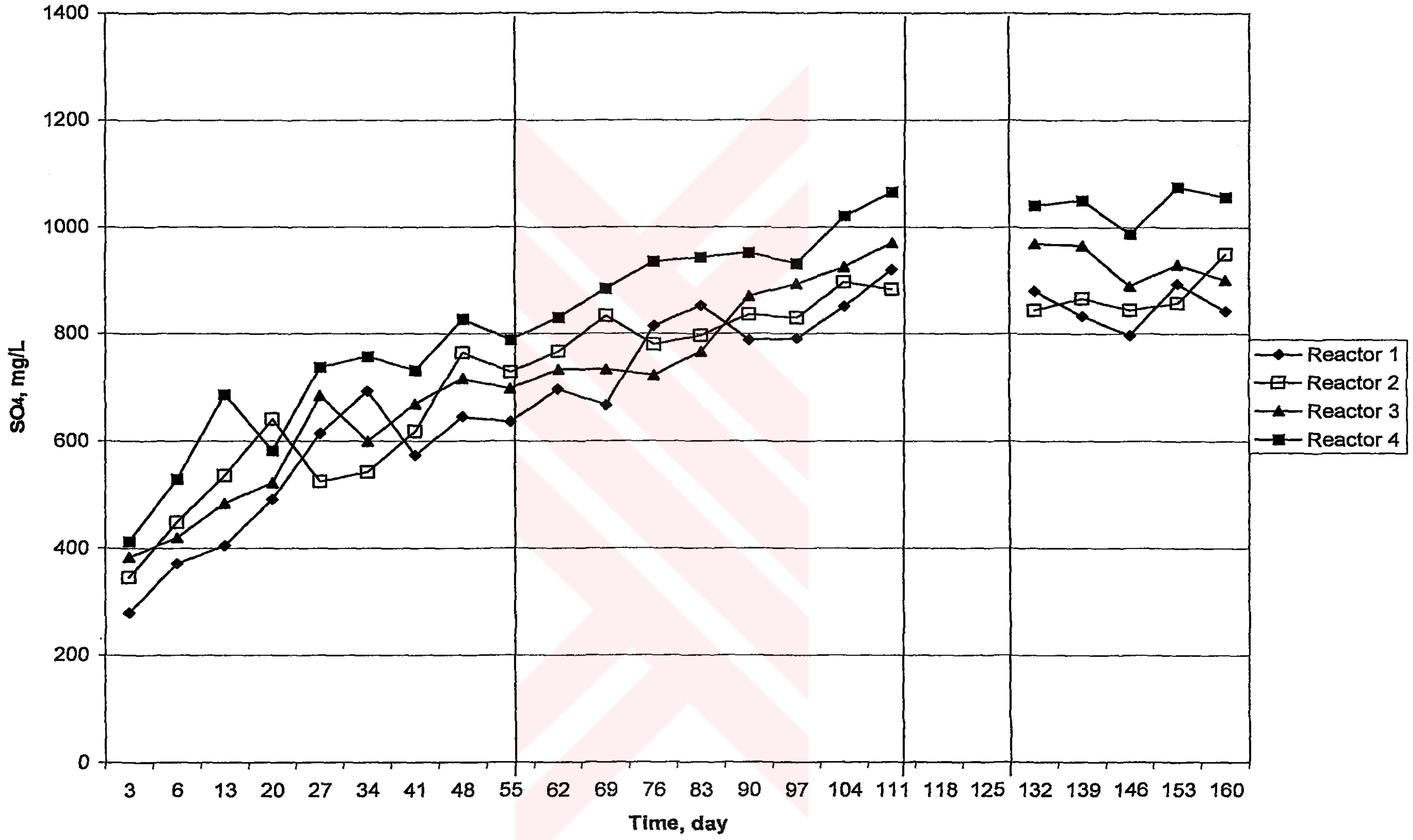


Figure 4.26. Sulfate concentrations of the leachate generated from the aerobic reactors.

research. At the beginning of the second stage, the sulfate concentrations of the Reactors 1, 2, 3, and 4 are 696 mg/L, 767 mg/L, 733 mg/L, and 830 mg/L, respectively. Whereas the final sulfate concentrations of the Reactors 1, 2, 3, and 4 reached to 842 mg/L, 945 mg/L, 901 mg/L, and 1057 mg/L, respectively.

4.3.7. Total Kjeldahl Nitrogen Concentrations of Leachate of the Aerobic Reactors

The TKN measurement of leachate started on the 3rd day of the research. The initial TKN values of leachate of the aerobic reactors (Reactors 1, 2, 3, and 4) were 4.69 mg/L, 6 mg/L, 4.87 mg/L, and 5.39 mg/L, respectively. At the end of the first stage, the maximum TKN concentration of leachate of the Reactors 1, 2, 3, and 4 were found as 6.48 mg/L, 7.30 mg/L, 7.21 mg/L, and 7.84 mg/L, respectively. In comparing the TKN concentrations, the lowest value (4.69 mg/L) was observed in Reactor 1, whereas the highest value (7.84 mg/L) was obtained in Reactor 4. This might be explained by the efficiency of the biologic decomposition. When the organic materials, which consist of simple nitrogen compunds, under the aerobic conditions was to be decomposed, the high TKN values decreased sharply until the end of the first stage.

After the yeast addition, the TKN values of leachate of the Reactors 2, 3, and 4 remained constant till the 69th day. They ranged from 4.70 to 5.44 mg/L. But After the 69th day, the TKN values of leachate of these reactors began to decrease again. In the same period, the TKN of leachate of the Control Reactor decreased gradually. This decreasing trend continued till the end of the research. The utilization of nitrogen for new cell constructions by microorganisms caused these decrease.

The final TKN values of leachate of the aerobic reactors (Reactors 1, 2, 3, and 4) are 1.97 mg/L, 1.64 mg/L, 1.35 mg/L, and 0.94 mg/L, respectively. As it can be seen in Figure 4.27, leachate of the reactors with sludge and yeast had low TKN concentrations than the ones obtained in the Control Reactor. This might be attributed to utilization rate of TKN by the microorganisms in the reactor. Because, the aerobic decomposition occurred in the Control Reactor was slower than in the other reactors.

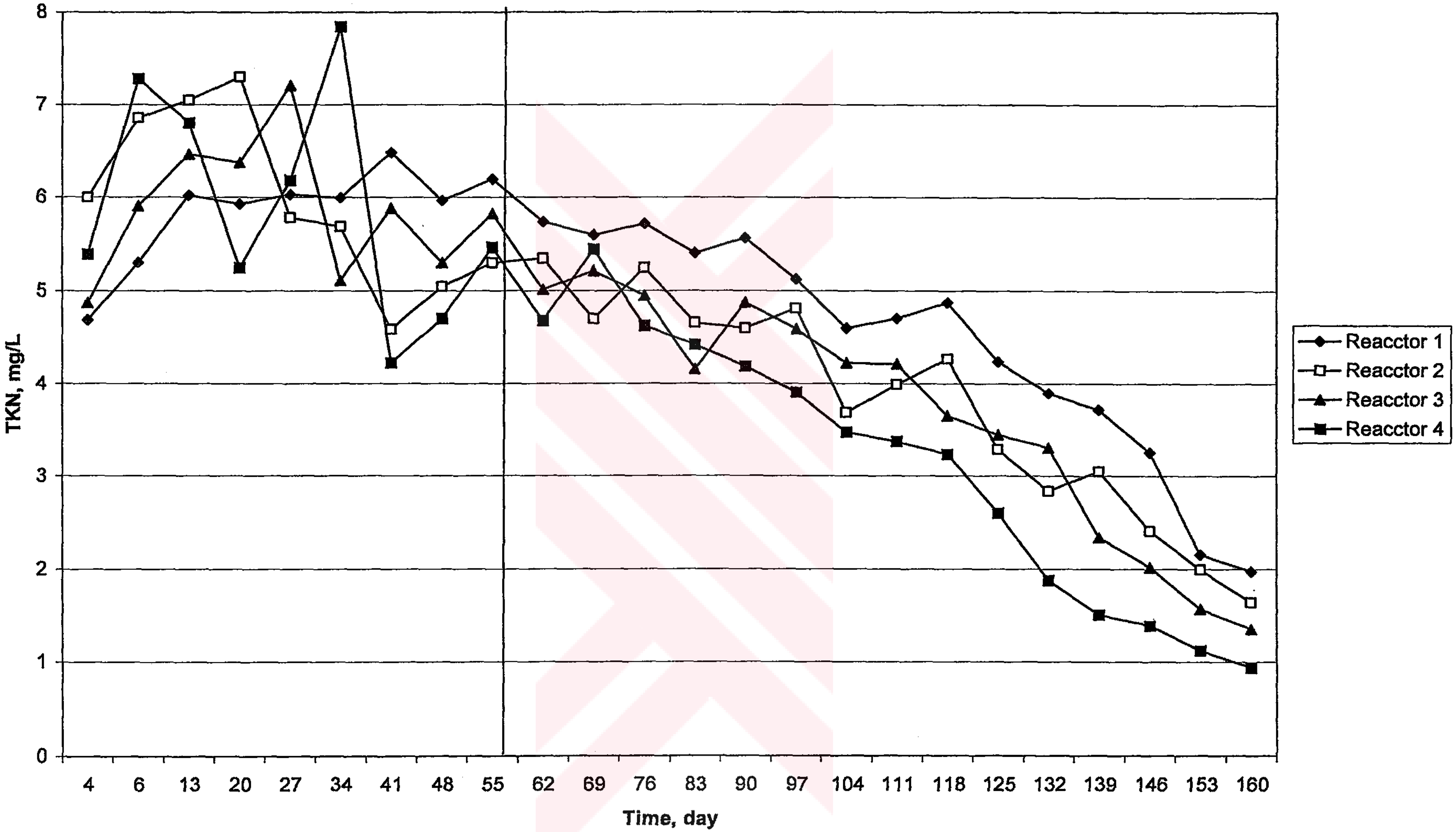


Figure 4.27. TKN values of the leachate generated from the aerobic reactors.

4.3.8. Chloride Concentrations of Leachate of the Aerobic Reactors

The chloride measurement of leachate of the aerobic reactors began at the 10th day of the study. The initial chloride concentrations of leachate of the aerobic reactors (Reactors 1, 2, 3, and 4) are 1620 mg/L, 1700 mg/L, 1540 mg/L, and 1830 mg/L, respectively. It was seen that the lowest chloride concentrations value (1620 mg/L) was observed in the Reactor 1, whereas the highest value (1830 mg/L) was obtained in Reactor 4. This might be attributed to the decomposition rates of these reactors. The Control Reactor did not have any sludge addition, on the other hand the Reactor 4 had 1/4 ratio of belt-press sludge consisted of available substrates and microorganisms. Therefore, water evaporated in the Reactor 4 faster than in the Control Reactor.

At the 45th day of the research, the chloride concentrations of leachate of all aerobic reactors decreased slightly. Addition of water was the reason of this decrease. Then, the chloride concentrations of leachate of the aerobic reactors increased until the end of the first stage. At the end of the first stage, the chloride concentrations of leachate of the aerobic reactors are 1690 mg/L, 1850 mg/L, 2120 mg/L, and 2200 mg/L, respectively.

After the yeast addition, chloride concentrations of leachate of the Reactors 2, 3, and 4 did not change significantly. On the other hand, chloride concentrations of leachate of the Control Reactor increased slightly. At the beginning of the second stage, the chloride concentrations of the Reactors 1, 2, 3, and 4 are 2030 mg/L, 1750 mg/L, 2140 mg/L, and 2330 mg/L, respectively. Then, the chloride concentrations of the reactors began to increase at the end of the research.

The final chloride concentrations of the Reactors 1, 2, 3, and 4 were increased to 2240 mg/L, 2630 mg/L, 2390 mg/L, and 2932 mg/L, respectively. The increase in the chloride concentrations of leachate of the aerobic reactors resulted from evaporation effects and aeration. The moisture contents of the aerobic reactors confirmed these results.

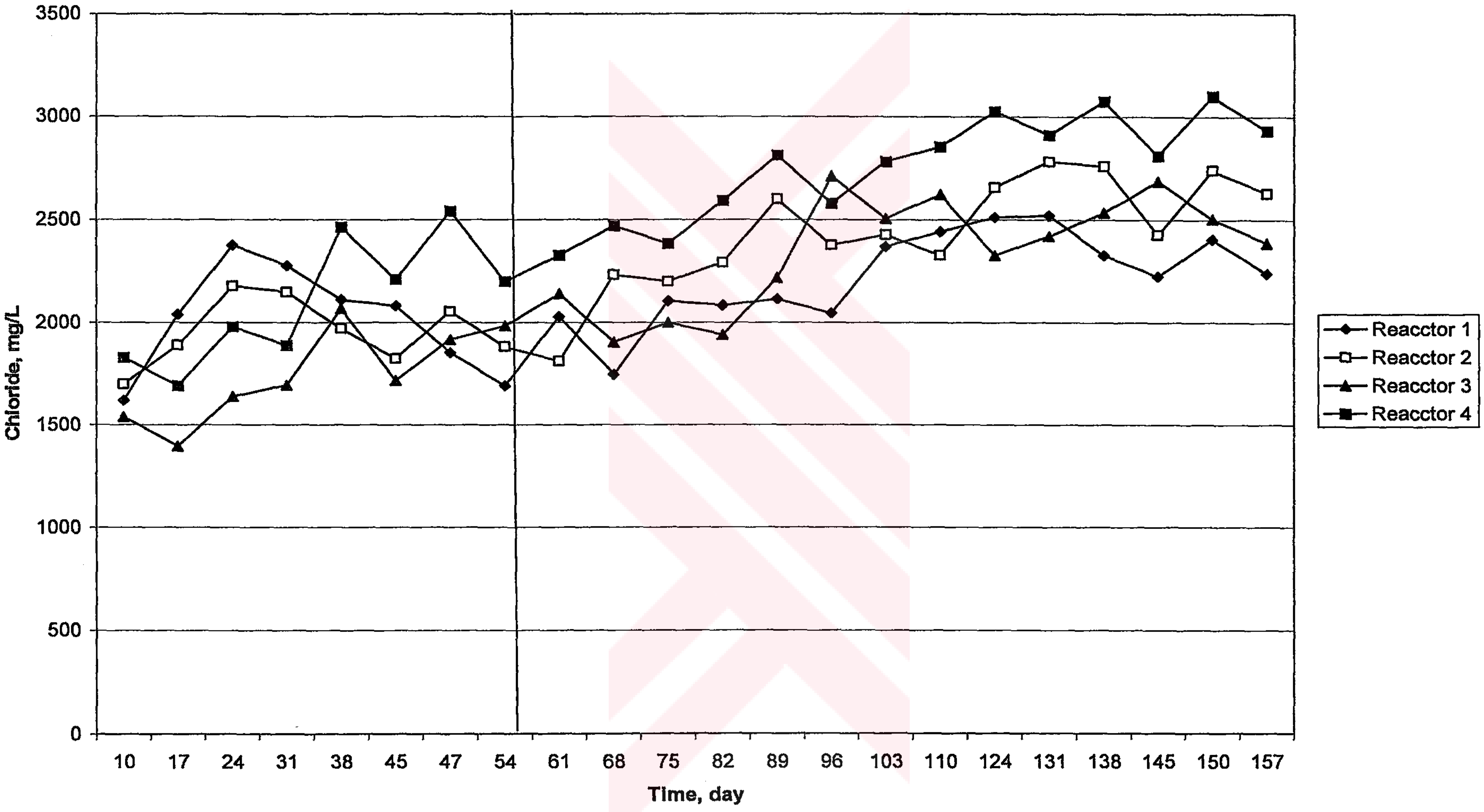


Figure 4.28. Chloride concentrations of the leachate generated from the aerobic reactors.

4.3.9. Heavy Metal Concentrations of Leachate of the Aerobic Reactors

The heavy metal concentrations of leachate generated from the aerobic reactors are given in Figures 4.29, 4.30, 4.31, 4.32, 4.33, and 4.34.

The initial heavy metal (copper, lead, nickel, zinc, cadmium and chromium) concentrations of leachate of the Reactors 1, 2, 3, and 4 were lower than the inhibition values of them in the environment. When the decomposition of the organic materials under the aerobic conditions continued, high heavy metal concentrations were observed in the aerobic reactors.

During the first stage, the Control Reactor without sludge had lower heavy metal concentrations than the reactors with sludge. The sludge increased the heavy metal concentrations of leachate of the aerobic reactors.

After the yeast addition, the heavy metal contents (copper, lead, nickel, zinc, and cadmium) of leachate of the Reactors 1, 2, 3, and 4 began to increase. The supply of aeration provided highly oxidizing conditions in the aerobic reactors. Under the highly oxidizing conditions in the aerobic reactors, sulfate did not reduce to sulfide. Therefore, sulfate did not precipitate with metals as sulfide compounds. The sulfate concentrations of leachate of the aerobic reactors might also confirm these heavy metal data.

On the other hand, chromium concentrations of leachate of the aerobic reactors decreased slightly at the end of the research. Chromium could be precipitated with hydroxide.

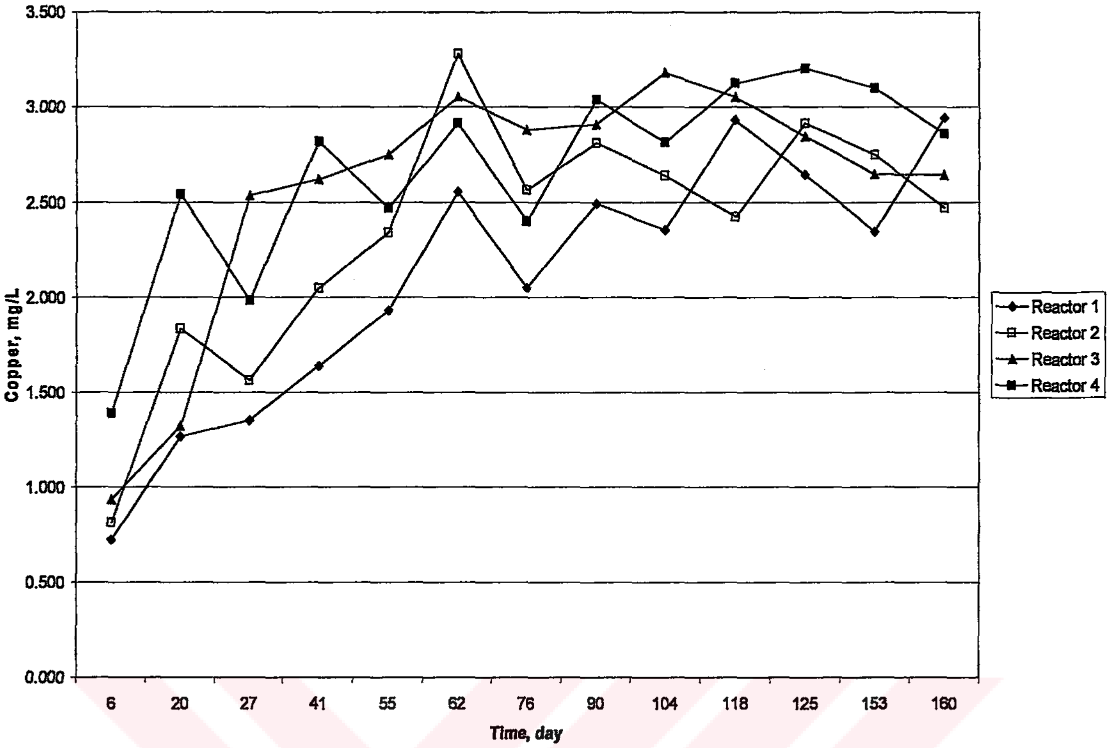


Figure 4.29. Copper concentrations of the leachate generated from the aerobic reactors.

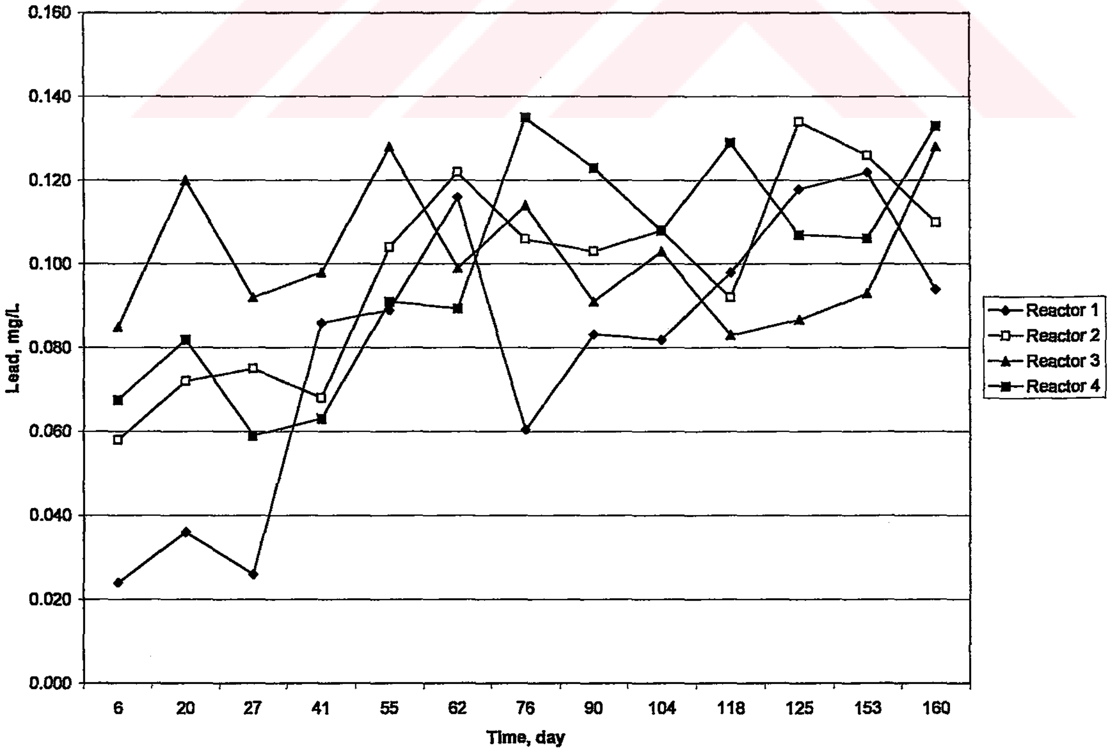


Figure 4.30. Lead concentrations of the leachate generated from the aerobic reactors.

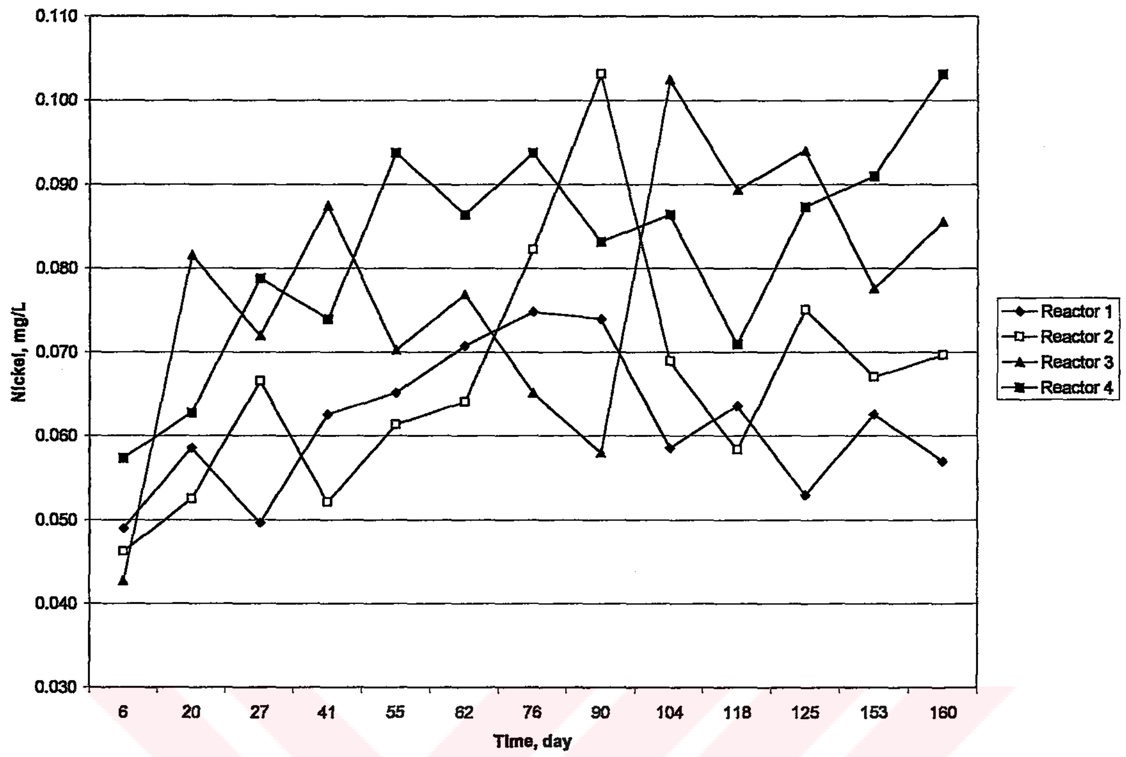


Figure 4.31. Nickel concentrations of the leachate generated from the aerobic reactors.

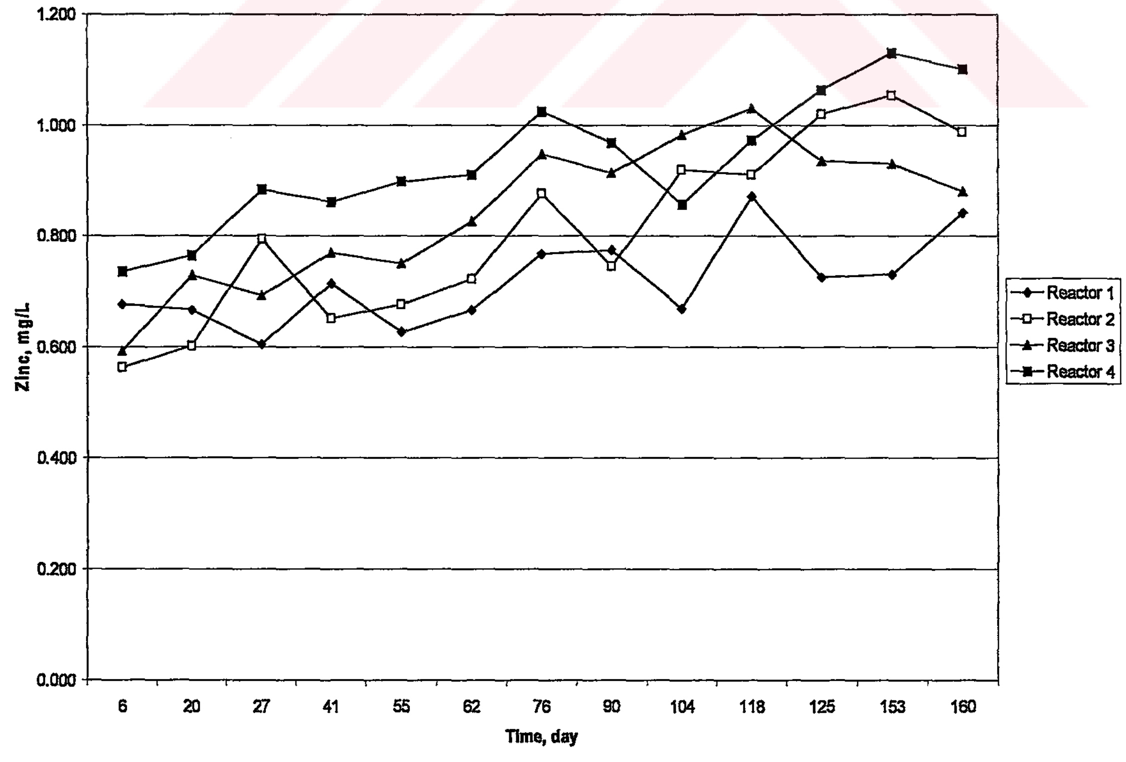


Figure 4.32. Zinc concentrations of the leachate generated from the aerobic reactors.

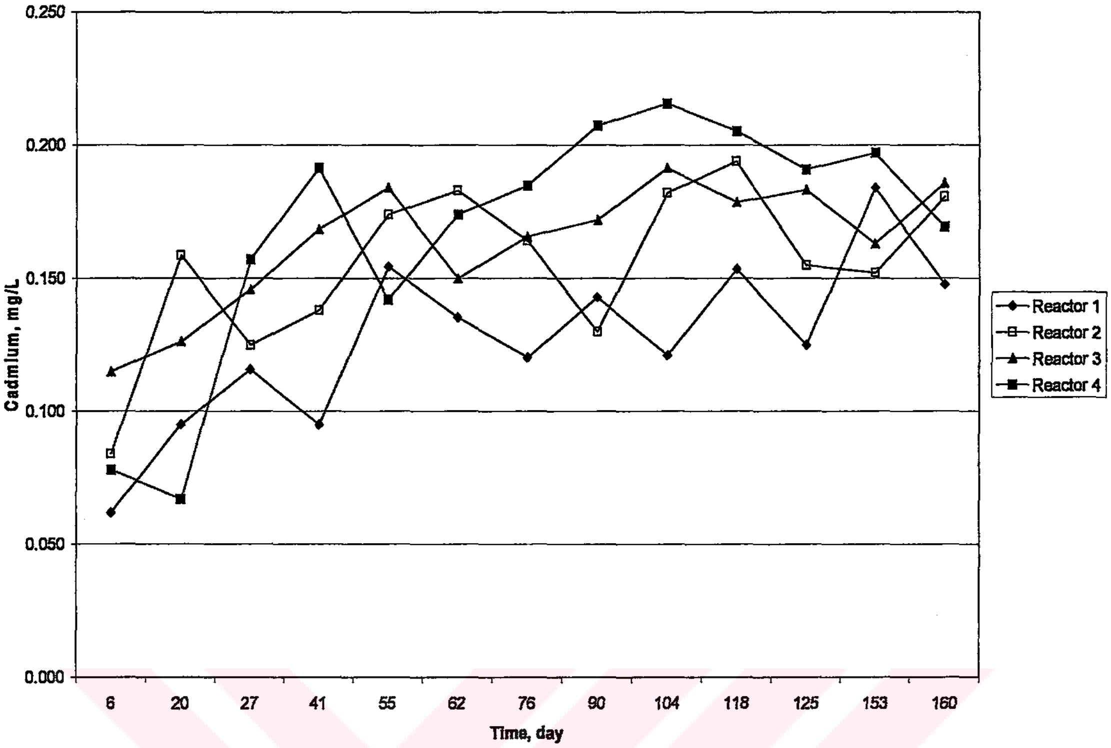


Figure 4.33. Cadmium concentrations of the leachate generated from the aerobic reactors.

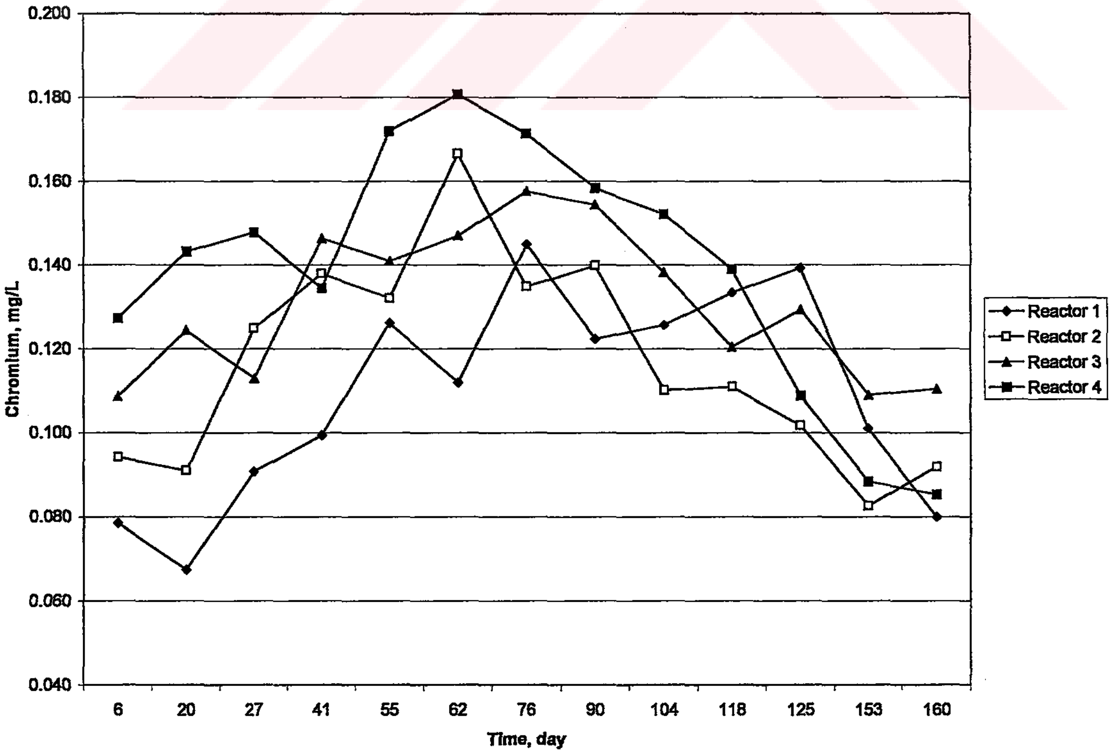


Figure 4.34. Chromium concentrations of the leachate generated from the aerobic reactors.

5. EVALUATION OF RESULTS

The results of the research are summarized in the following paragraphs:

1. During the first stage, the pH values of leachate of the anaerobic reactors were low because of the accumulation of the volatile acids in all anaerobic reactors. After the yeast addition, low pH values were observed in the Reactors 2, 3, and 4 until the 94th day. After then, this decrease was followed by a slight increase till the end of the study. Conversion of the volatile the fatty acids by the bacteria was the main reason for the increase of pH values in all anaerobic reactors.

   On the other hand, at the early stages of the decomposition, the pH values of leachate of the aerobic reactors with sludge are lower than the Control Reactor without sludge because of the acid formation from the decomposition of organic substances. By the utilization of organic acids, the pH of leachate of the aerobic reactors reached to neutral pH values during the first stage. After the yeast addition, the pH values of leachate of the aerobic reactors increased slightly at the end of the research.

2. The first ORP values of leachate of the anaerobic reactors were positive indicating the aerobic conditions in all the reactors. When the presence of oxygen was depleted, the negative ORP values were observed in all anaerobic reactors. After the yeast addition, the low negative ORP values indicated establishment of the anaerobic environment into the reactors.

   In contrast to the anaerobic reactors, the first ORP values of leachate of the aerobic reactors (Reactors 2 and 3) were negative showing the anaerobic conditions. Accumulation of the organic acids and insufficient aeration might be responsible for these negative results. By the time, ORP values of leachete in all aerobic reactors exhibited an increasing trend and positive values were obtained at the end of the first stage. After the yeast addition, the ORP values increased in reactors with yeast until the end of the study. The ORP of leachate of the Control Reactor also followed an increased trend until the end of the research. This increasing trend might be resulted from the conversion of acids and sufficient aeration rate.

3. Throughout the first stage of the research, it was observed that leachate generated from the anaerobic Reactors 2, 3, and 4 had higher COD concentrations than the Control Reactor. Mixing with sludge was responsible for these high COD concentrations. Any COD removal was not observed during this stage. After the yeast addition, the COD concentrations of the anaerobic reactors did not change significantly until the 77th day. Then, the COD concentrations began to decrease until the end of the study. The COD removal efficiencies of the Reactor 1, Reactor 2, Reactor 3, and Reactor 4 were found as 40, 60, 55, and 66 per cent, respectively. It can be seen from the values that the reactors with yeast addition had a higher COD removal efficiency than the Control Reactor.

   In contrast to the anaerobic reactors, a decreasing trend was observed in the COD concentrations of leachate of the aerobic reactors in the first stage. Mixing with sludge promoted the aerobic decomposition of the materials. Therefore, the higher COD removal efficiencies in each aerobic reactor were observed. During the first stage, the COD removal efficiencies of the Reactor 1, Reactor 2, Reactor 3, and Reactor 4 were found as 52, 80, 65, and 74 per cent, respectively. During the second stage, the decreasing trend was continued in the COD concentrations of the aerobic reactors. The addition of yeast increased the COD removal efficiency in each reactor. At the end of the second stage, the COD removal efficiencies of the Reactors 1, 2, 3, and 4 were measured as 67, 68, 76, and 82 per cent, respectively. These percentages indicated that the aerobic reactors reached higher COD removal efficiencies than the anaerobic reactors.

4. The alkalinity values of leachate of the all anaerobic reactors were sufficcent to buffer the reactors in the first stage. Presence of high volatile acids concentrations might

contribute to the presence of high alkalinity concentrations in the reactors. Afer the yeast addition, the alkalinity values of Reactors 2, 3, and 4 remained constant until the 79th day. Then, The alkalinity values of leachate of the all anaerobic reactors were flactuated.

Throughout the research, the alkalinity values of the aerobic reactors were higher than the ones obtained in the anaerobic reactors. Buffer addition was not made at the end of the research.

5. The orthophosphate concentrations of leachate of the anaerobic reactors decreased until the end of the first stage. Throughout the first stage of the research, it was observed that orthophosphate concentrations of leachate of the anaerobic Reactor 2, Reactor 3, and Reactor 4 were lower than the Control Reactor. After the yeast addition, orthophosphate concentrations of the Reactors 2, 3, and 4 followed a stationary trend until the 62nd day. Then, the orthophosphate concentrations of these reactors decreased gradually till the end of the research. The anaerobic Reactors 2, 3, and 4 with yeast addition had lower orthophosphate concentrations than the Control Reactor. At the end of the research, the orthophosphate removal efficiencies of the Reactors 1, 2, 3, and 4 were measured as 60, 85, 68, and 76 per cent, respectively.

   Similarly, the orthophosphate concentrations of leachate of the aerobic reactors also decreased. Orthophosphate was used by the aerobic microorganisms to construct the new cells. After the yeast addition, orthophosphate concentrations in all reactors followed a decreasing trend. At the end of the research, the aerobic reactors had lower orthophosphate concentrations than the anaerobic reactors. The orthophosphate removal efficiencies of the Reactor 1, Reactor 2, Reactor 3, and Reactor 4 were found as 90, 95., 96, and 95 per cent, respectively.

6. The initial sulfate values of leachate of the anaerobic reactors were low because of the insufficient biologic decomposition in the reactors. When the solid waste decomposed, the high sulfate concentrations were observed in the anaerobic reactors. The reactors

with sludge had higher sulfate concentrations than the Control Reactor. After the yeast addition, sulfate concentrations in all reactors followed a decreasing trend. This decreasing trend was continued till the end of the research. Conversion of sulfate to sulphide was the major reason for the decreasing of sulfate concentrations in the anaerobic reactors. At the end of the research, the sulfate removal efficiencies of the Reactor 1, Reactor 2, Reactor 3, and Reactor 4 were found as 51, 67, 67, and 68 per cent, respectively.

In contrast to the anaerobic reactors, during the first stage, the sulfate concentrations of the aerobic reactors were high because of the high decomposition of solid wastes under the aerobic conditions. After the yeast addition, the sulfate concentrations of the aerobic reactors continued to increase at the end of the research. . Any sulfate removal was not observed in the aerobic reactors.

7. The TKN values of leachate of the anaerobic reactors (Reactors 1, 2, 3, and 4) were low because of the insufficient decomposition of the organic materials in all reactors. When the decomposition of the organic materials under anaerobic conditions was completed, the higher TKN concentrations were observed. The yeast addition affect the TKN concentrations of the reactors after the 83rd day of the research significantly. Conversion and utilization of the nitrogen by the anaerobic microorganisms caused to a decrease in TKN concentrations at the end of the research. At the end of the research, the TKN removal efficiencies of the anaerobic Reactor 1, Reactor 2, Reactor 3, and Reactor 4 were found as 23, 34, 34, and 34 per cent, respectively.

In contrast to the anaerobic reactors, the TKN concentrations of leachate generated of the aerobic reactors were high because of the decomposition of the organic materials under aerobic conditions. The higher TKN concentrations were observed as the aerobic decomposition continued. High conversion and utilization of the nitrogen by the aerobic microorganisms and volatilization of ammonia caused sharp decreases in TKN concentrations of the reactors. At the end of the research, the TKN removal efficiencies of the Reactor 1, Reactor 2, Reactor 3, and Reactor 4 were found as 70, 78, 81, and 88 per cent, respectively.

8. During the first stage, it was observed that the chloride concentrations of leachate generated from the four anaerobic rectors showed a decreasing trend because of the dilution effect. After the yeast addition, the chloride concentrations did not change significantly.After a certain time, the chloride concentrations of leachate of the anaerobic reactors decreased gradually because of decomposition of solid wastes.

   In contrast to the anaerobic reactors, the chloride concentrations of leachte of the aerobic reactors were also high because of the evaporation in the reactors. During the second stage, the chloride concentrations of leachate of the aerobic reactors also increased until the end of the study. Aeration of the aerobic reactors and evaporation of leachate of the aerobic reactors were result in increasing of chloride concentrations.

9. Also, the high heavy metal concentrations were observed in the leachate of the aerobic reactors during the first stage. The leachate of the aerobic reactors had higher heavy metal concentrations than the leachate of the anaerobic reactors because of the highly oxidizing conditions. During the second stage, the metal contents of leachate of the anaerobic reactors fluctuated until the end of the research. On the other hand, the metal contents of leachate of the aerobic reactors increased except chromium until the end of the research

10. The cumulative gas productions of the anaerobic Reactor 1, Reactor 2, Reactor 3, and Reactor 4 were found as 26530 mL, 41170 mL, 38750 mL, and 46500 mL, respectively. It could be seen that Reactor 4 reached to a higher cumulative gas production than the others.

11. The initial methane percentage of the anaerobic Reactors 1, 2, 3, and 4 was low during the first stage. After the yeast addition, the methane contents of the reactors increased gradually. The maximum methane percentage of the Reactors 1, 2, 3, and 4 were

measured as 34, 58, 56, and 64 per cent, respectively. The yeast supplied available substrates for the methanogenic bacteria. When the reactors were compared to each other, the Reactor 4 was the most suitable reactor for obtaining a high methane content.

6. CONCLUSIONS

The objective of the experimental study is to compare the stabilization of the solid wastes by mixing different ratios and types of sludge under anaerobic and aerobic conditions. The other objective of this research is to detect the effects of the yeast addition on the decomposition rates of organic materials under aerobic and anaerobic conditions.

The following conclusions are provided from this research.

- It was observed that co-disposal of sludge and solid wastes was an effective technique for the stabilization of solid wastes and sludge in aerobic reactors. The aerobic reactors with sludge had the high COD, phosphate, and TKN removal efficiencies than the Control Reactor without sludge. On the other hand, the stabilization of sludge and solid wastes in the anaerobic reactors together were also effective in the COD, phosphate, sulfate, and TKN removals. The stabilization of solid wastes and sludge in the anaerobic reactors takes longer time than the aerobic reactors.
- Slowness and variability in the biologic decomposition rate of the Control Reactor was largely result in lack of sludge and yeast.
- Reactor 4, which was loaded with belt-press sludge and solid wastes at the ratio of 1/4, had the maximum COD, phosphate, sulfate, and TKN removal efficiencies under anaerobic conditions. Similarly, Reactor 4 showed the maximum COD and TKN removal efficiencies under aerobic conditions.
- The highest ratio 1:4 was found as the optimum sludge (belt-press) to solid wastes ratio for both the aerobic and the anaerobic reactors.
- The highest phosphate removal was observed at 1/7 secondary settling sludge to solid wastes ratio in the Reactor 3 under aerobic conditions.
- In addition to the presence of sludge, yeast solution addition also improved the rate of biologic decomposition. Yeast converted complex organic materials to their

simple forms. By this way, the available substrate concentrations increased in the reactors and the conversion of them by the microorganisms occurred easily.

- All aerobic reactors had higher COD, phosphate and TKN removal efficiencies than the anaerobic reactors.
- The methane contents of the anaerobic reactors with sludge and addition of yeast were higher than the Control Reactor without sludge and no addition of yeast.
- The highest methane content was observed at 1/4 belt-press sludge to solid wastes ratio.
- The heavy metal concentrations of leachate of the anaerobic reactors were lower than the heavy metals concentrations of leachate of the aerobic reactors.
- The waste settlement in the aerobic reactors occurred faster than the waste settlement in the anaerobic reactors.

7. RECOMMENDATION FOR FURTHER STUDIES

For further research, the following recommendations can be given.

In this study, different types of treatment plant sludges were mixed with municipal solid wastes in order to stabilize the municipal solid wastes under the aerobic and anaerobic conditions. The mixing ratios were based on EPA Sludge Manual. The different ratios of different types of sludge and solid wastes can be used for further studies.

Different yeast cultures can be tested in further studies. Moreover, yeast addition should be started at the beginning of the experiment in order to compare their complete effects on biological decomposition.

Aeration rate can affect the physical, chemical and biological reactions. The effect of different aeration rates should be investigated.

The results of this research should be applied at the landfill areas as a further studies.

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APPENDIX A

Table A.1. Daily gas production in the anaerobic reactors

Day	**Reactor 1**	**Reactor 2**	**Reactor 3**	**Reactor 4**
3	100	546	499	684
4	140	88	155	120
5	185	726	303	1080
6	386	797	1122	1057
7	255	551	314	764
8	177	785	869	940
9	284	126	416	160
10	205	1009	106	899
11	140	160	872	1160
12	875	476	403	704
13	204	238	513	322
14	132	0	235	162
15	184	90	123	299
16	0	520	137	460
17	183	141	378	304
18	106	315	135	224
19	475	151	435	266
20	179	274	0	0
21	137	0	0	331
22	105	120	173	242
23	0	112	346	457
24	0	280	107	370
25	113	166	104	406
26	0	196	110	287
27	219	188	391	160
28	0	284	0	307
29	107	451	207	250
30	108	214	242	328
31	340	401	140	331
32	104	280	138	203
33	120	250	270	499
34	371	240	197	151
35	112	367	212	182
36	125	282	87	320
37	146	421	168	266
38	0	179	0	450
39	100	436	99	290
40	105	155	148	212
41	281	275	250	376
43	114	211	183	331
44	115	230	155	370
45	182	445	205	165

Table A.1. *continued*

46	100	210	301	0
47	103	227	204	287
48	144	169	155	290
49	141	133	266	430
50	174	293	208	349
51	177	165	179	272
52	113	373	359	284
53	164	171	154	304
54	178	530	206	0
55	185	440	509	261
56	312	486	512	134
57	288	425	448	524
58	341	260	246	457
59	241	334	236	322
60	0	400	299	328
61	171	274	0	418
62	353	292	370	340
63	376	358	337	448
64	188	220	534	307
65	404	385	504	198
66	425	552	489	430
67	174	490	547	770
69	174	594	463	605
69	313	177	148	516
71	167	164	407	672
72	282	237	327	571
73	108	259	630	338
74	155	642	337	528
75	415	259	663	594
76	343	228	381	463
77	139	546	222	534
79	229	284	458	397
80	582	540	140	174
81	293	166	537	269
82	182	315	367	573
83	100	258	335	228
84	165	648	246	596
85	143	478	440	182
86	114	128	390	370
87	0	397	543	177
89	144	413	564	669
90	390	509	386	346
91	174	513	208	307
92	459	165	246	148
93	334	556	185	510
94	624	171	154	177

Table A.1. *continued*

95	528	556	592	190
96	457	570	384	274
97	376	470	316	385
98	238	425	423	582
99	341	391	308	543
100	167	403	431	397
101	183	141	160	254
102	171	388	404	436
104	309	171	357	188
105	140	511	257	256
106	412	368	257	449
107	312	155	202	370
108	175	0	291	133
109	0	201	235	404
110	89	275	309	227
111	93	227	412	0
112	124	180	148	0
113	0	164	315	358
114	109	237	268	209
115	108	0	0	406
116	352	305	128	0
118	337	259	0	0
119	143	173	155	210
120	139	324	0	421
121	135	223	195	226
122	102	156	140	346
123	117	166	245	257
124	97	129	113	367
126	105	0	195	349
129	0	343	242	120
130	144	169	155	310
131	141	316	371	0
132	203	258	336	343
133	177	403	179	244
136	178	382	206	263
137	185	200	148	80
138	0	94	200	235
139	91	106	214	272
140	0	195	129	248
143	171	134	280	0
144	148	138	179	217
145	135	106	275	212
146	123	263	0	322
147	97	212	275	0
150	0	0	109	0
151	93	169	69	272

Table A.1. *continued*

152	124	0	148	0
153	100	254	184	215
154	109	104	160	96
157	144	169	155	310
158	141	316	371	183
159	174	258	336	343