T.C
MARMARA UNIVERSITY
INSTITUTE FOR GRADUATE STUDIES IN
PURE AND APPLIED SCIENCES

ANAEROBIC AND AEROBIC TREATMENT OF SANITARY LANDFILL LEACHATE

By
Barış ÇALLI

Thesis Supervisor
Assoc. Prof. Bülent İNANÇ

Submitted for Partial Fulfilment of the Requirements
for the Degree of Master In Science in
Environmental Engineering

İSTANBUL 1999

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Supervisor and Coordinator : Assoc. Prof. Bülent İNANÇ

Member : Assoc. Prof. Talha GÖNÜLLÜ

Member : Assoc. Prof. Mehmet Ali YÜKSELEN

İSTANBUL 1999

ACKNOWLEDGEMENT

I wish to express my sincere and profound gratitude to my supervisor Assoc. Prof. Dr. Bülent İNANÇ for his encouraging supports and valuable advises throughout the development of this study.

I'd like to thank also to Ela YILMAZ, Ayşegül KUŞAKSIZ, Melike HATİPOĞLU, Bülent MERTOĞLU, Özgür GÖLBAŞI, Hakan GÜL and all student assistants, for their helps in the analysis of the characterization and treatability samples.

Finally, assistance of Mustafa Atlan for providing leachate samples from Kömürcüoda Sanitary Landfill is also gratefully acknowledged.

ABSTRACT

In this study, characterization and biological treatability of leachate of Kömürcüoda Sanitary Landfill located on the Asian part of Istanbul were investigated. Biological treatability studies were composed of anaerobic treatability of raw leachate and aerobic treatability of anaerobic reactor effluents.

Characteristics of leachate were monitored from August 1998 until May 1999 for a 10 months period. Samples were taken from a 200 m^3 holding tank located at the lowest elevation of the landfill. COD concentrations have ranged between 18800 and 47800 mg/l while BOD_5 between 6820 and 38500 mg/L. COD and BOD_5 values were higher in summer and lower in winter due to dilution by precipitation. On the other hand, it was quite interesting that such dilution effect was not observed for ammonia. The highest ammonia concentration, 2690 mg/L was in November 1998. BOD_5/COD ratios around 0.7 for most samples has indicated high biodegradability, and acidic phase of decomposition in the landfill.

For anaerobic treatability, three different reactors, namely an upflow anaerobic sludge bed reactor, an anaerobic upflow filter and a hybrid bed reactor, were used. The anaerobic reactors were operated for more than 300 days and were continuing operation when this thesis was prepared. Organic loading was increased gradually from 1.3 kgCOD/m^3.day to 9.8 kg COD/m^3.day while hydraulic retention time was reduced from 2.4 days to 2.0 days after Day 180. All the reactors showed similar performances against organic loadings with efficiencies between 80% and 90%. However the reactors have experienced high ammonia concentrations several times throughout the experimental period, and showed different inhibition levels. Anaerobic filter was the least effected reactor while UASB was the most. Hybrid bed reactor has exhibited a similar performance to anaerobic filter although not at the same degree.

SBR's were used in the aerobic treatability of anaerobically treated leachate, same as in the leachate treatment plant currently under construction. Because of high ammonia concentrations and refractory organic content in the effluents of anaerobic reactors,

satisfactory COD removal efficiencies were not detected. Hence, to increase the biodegradability of anaerobic effluents by breaking down the refractory organics, ozonation and hydrogen peroxide oxidation were applied. However, no significant improvement was observed in aerobic treatability with application of these oxidants. Some other alternatives especially combinations should be investigated in further studies. Also, different seed sludges and long acclimation periods should be examined.

Although COD was not removed properly in SBR's, high nitrification efficiencies were observed under sufficient alkalinity concentrations. On the other hand, denitrification step of biological nitrogen removal was not carried out because of unsatisfactory conditions for denitrifiers. This subject should also be investigated in further studies, which is very important for the removal of high ammonia content in leachate.

ÖZET

Bu çalışmada, İstanbul'un Anadolu yakasında bulunan Kömürcüoda Sıhhi Katı Atık Depo Sahasının sızıntı sularının karakterizasyonu ve biyolojik arıtılabilirliği araştırılmıştır. Biyolojik arıtılabilirlik çalışmaları, ham çöp sızıntı suyunun anaerobik arıtılabilirliği ve anaerobik reaktör çıkışlarının aerobik arıtılabilirliği olarak gerçekleştirilmiştir.

Ağustos 1998 ve Mayıs 1999 arasındaki 10 aylık periyotta, çöp sızıntı suyunun karakterizasyonunu araştırılmıştır. Numuneler, depo sahasının en düşük kotunda bulunan 200 m^3'lük bir beton tanktan alınmıştır. Yapılan analizler sonucunda 18800 ile 47800 mg/L arasında değişen KOİ ve 6820 ile 38500 mg/L arasında değişen $BOİ_5$ değerleri elde edilmiştir. KOİ ve $BOİ_5$ değerleri yaz aylarında yüksek, kış aylarında ise yağmurdan kaynaklanan seyrelme sebebiyle düşük tespit edilmiştir. Diğer taraftan yağış ile seyrelmenin amonyak değerleri üzerinde görülmemesi ilginçtir. En yüksek amonyak konsantrasyonu 2690 mg/L olarak Kasım 1998'de tespit edilmiştir. Numunelerde tespit edilen 0.7 seviyelerindeki BOİ/KOİ değerleri, çöp sızıntı suyunun yüksek biyolojik ayrıştırılabilirliğini ve Kömürcüoda Çöp Depo Sahasının genç depo sahaların dan olduğunu göstermektedir.

Anaerobik arıtılabilirlik çalışmaları, anaerobik filtre, hibrid ve yukarı akışlı anaerobik çamur yatağı (UASB) reaktörü olamak üzere 3 ayrı reaktörde gerçekleştirilmiştir. Bu tezin hazırlandığı ana kadar 300 günden fazla çalıştırılmış olan reaktörler hala işletilmektedir. Reaktörlere uygulanan organik yükler basamak basamak 1.3 KOİ/m^3/gün'den 9.8 kg KOİ/m^3/gün'e kadar artırılmış, hidrolik bekletme süresi ise 180. günden sonra 2.4 günden 2 güne düşürülmüştür. 3 reaktöre de artan organik yükleri karışısında % 80 ile 90 arasında benzer KOİ giderim verimleri sergilemiştir. Aynı zamanda işletme esnasında reaktörler birkaç kez yüksek amonyak konsantrasyonlarına maruz kalmış ve farklı inhibisyon seviyeleri göstermiştir. Anaerobik Filtre, üç reaktör arasında, amonyak inhibisyonundan en az etkilenen olurken, UASB reaktörü en çok etkilenmiştir. Hibrid

reaktör ise inhibisyon karşısında Anaerobik Filtre'ye göre daha verimsiz olmak üzere benzer sonuçlar sergilemiştir.

Anaerobik olarak arıtılan çöp sızıntı suyunun aerobik arıtılabilirliği çalışmasında inşa edilmekte olan arıtma tesisi için de düşünülen ardışık kesikli reaktörler kullanılmıştır. Yüksek amonyak konsantrasyonu ve zor ayrıştırılabilir organik içerik sebebiyle bu reaktörlerde verimli KOİ giderimi tespit edilememiştir. Bu sebeple, anaerobik reaktör çıkışlarının biyolojik ayrıştırılabilirliğini arttırmak için, ozonlama ve hidrojen peroksit oksidasyonu denenmiştir. Ama bu kimyasal oksidasyon uygulamaları ile de olumlu gelişmeler gözlenmemiştir. Gelecek çalışmalarda, anaerobik reaktör çıkışlarının aerobik arıtılabilirliğini arttırabilmek için diğer alternatifler, özellikle kombinasyonlar, incelenmelidir. Ayrıca farklı aşı çamurları ve daha uzun aklimasyon periodları da denenebilir.

Ardışık kesikli reaktörlerde, KOİ tam olarak arıtılamasa da, yeterli alkalinite sağlanınca, yüksek nitrifikasyon verimleri tespit edilmiştir. Ama reaktörlerde denitrifikasyon bakterileri için uygun koşullar sağlanamadığı için, reaktörlerde denitrifikasyon gerçekleştirilememiştir. Çöp sızıntı suyundaki yüksek amonyak konsantrasyonun deşarj öncesi giderilmesi için çok önemli olan bu kademe muhakkak gelecek çalışmalarda incelenmelidir.

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CHAPTER 1

INTRODUCTION

Solid waste management may be defined as the discipline associated with the control of generation, storage, collection, transfer and transport, processing, and disposal of solid wastes in a manner that is in accord with the best principles of public health, economics, engineering, conservation, aesthetics, and other environmental considerations.

In recent years, especially in crowded cities, the municipalities are forced to manage more detailed programs for increasing solid wastes parallel to growing populations and developing human activities. The safe and long-term disposal of solid wastes is an important component of these management programs.

Landfilling or land disposal is today the most commonly used method for solid waste disposal by far. Compared to high-tech methods, sanitary landfilling is also the most economical one for industrializing countries as Turkiye. Besides economical advantages, landfilling also minimize adverse environmental effects and risks on human health comparing to wild dumping. But the principle outputs of landfills as landfill gases and leachate should be collected and managed to support these advantages.

Sanitary landfill leachate is composed of the liquid that has entered the landfill from external sources as rainfall, surface drainage and etc. and the liquid produced from the decomposition of the wastes. Leachates have variable characteristics according to amount of precipitation and waste composition and generally contain high concentrations of soluble organics and inorganic ions.

Istanbul, one of the most crowded metropolises in Europe with a population over 12 million in 1999, has to manage 8000 tons of solid waste every day. The population in Istanbul is expected to reach 17 million by the year 2020, so according to the forecasts, the solid waste generation will be 12000 tons/day only 20 years later. Amount of municipal solid waste has been increasing with increasing population and living standards. Currently, the waste generation rate is 0.65 kg/ca-day. Meanwhile, natural gas and low-ash coal consumption are increasing and expected to reach 40% for natural gas and 60% coal by the year 2000. As a result, it is expected that, ash content will decrease by about 50%, and waste generation rate will then be 0.6 kg/ca-day (Arikan et. al., 1997). More detailed studies about the characteristics of municipal solid waste in İstanbul is given in chapter 2.

At present, there are two sanitary landfills, one in European Side and the other in Asian Side, with total operational capacities of 20 years, have been under operation since 1995. On the other hand, construction of a compost plant with 1000tons/day capacity has been started in 1998. Some reduction is expected in organic content of the solid waste to be landfilled due to separation of organic contents for composting while sludges from newly constructed and future wastewater treatment plants, if landfilled, will result in increment. Hence, characteristics of solid wastes and leachates from landfills should be closely monitored. The municipality has started the construction of leachate treatment plants consisting of anaerobic and aerobic processes although the exact characteristics of the leachate and treatability results are not known.

Therefore, in this study three main investigation, which are not conducted before, were carried out on leachate of Kömürcüoda Sanitary Landfill,

- characterization
- anaerobic treatability
- and aerobic treatability of anaerobic effluents.

Kömürcüoda Sanitary Landfill is under operation since 1995 on Asian Side with a total area of 98 ha. Until the end of May 1999, about 3 million tons of solid waste has been landfilled to 25 ha area and it has been estimated that 5 ha will be used every year. While leachate recirculation has been practiced, excess amount of leachate produced is transported by tankers and disposed into municipal sewer system. Amount of leachate managed in this way was 110780 m^3 in 1997. This is equivalent of a flow rate of about 300 m^3/day. So, design capacity of the leachate treatment plant was selected as 500 m^3/d.

CHAPTER 2
SOLID WASTE MANAGEMENT

2.1 SOURCES, COMPOSITION AND PROPERTIES OF SOLID WASTE

Solid waste comprise all the wastes arising from human and animal activities that are normally solid and that are discussed as useless or unwanted.

Municipal solid waste, which is discussed in this chapter, is the smallest component of total waste. Currently, the total waste generated in industrialized countries yearly could be as high as 9 billion tones including;

- Residues from the production of energy, agricultural waste, mining spoil, demolition debris, dredge spoil, and sewage sludge
- Industrial waste and municipal solid waste.

2.1.1 Types and Sources

Municipal solid waste is the waste generated by households or by commercial and production facilities whose waste is similar to households. It is made up of packaging materials (i.e. plastics, metals, corrugated paperboard, etc.), paper, food, used objects and durable goods. Main sources of municipal solid waste are residential, commercial and institutional sites, municipal services, treatment plants and municipal incinerations.

2.1.2 Composition and Characteristics

The composition of solid waste is a crucial element for defining waste management strategies. Information about the nature of waste is critical for analyzing the single factors that determinate waste generation as well as for assessing the effects on the environment of specific components found in MSW. Moreover, knowledge on the composition of waste is essential for implementing the most appropriate waste reduction policies and for choosing the adequate waste treatment and disposal processes.

Differences may be detected in the compositions of solid wastes in different cities even in different parts of a city. There are many factors affecting the variability of MSW such as socio-economic conditions, weather, fuels combusted and etc. In Table 2.1, different MSW compositions in different countries, in Table 2.2 in different cities in Turkiye are presented. Table 2.3 represents the composition of MSW in Istanbul according to 4 different studies carried out between 1979 to 1996.

Table 2.1 Composition of MSW in Different Countries

Material Groups	**Germany** [1]	**Greece** [1]	**India** [2]	**Switzerland** [1]	**USA** [3]	**England** [1]
Organic Matter	44	48.5	75.0	30.0	27.5	19.8
Paper	17.9	22	2.0	31.0	41.1	34.8
Plastic	5.4	10.5	-	15.0	7.5	11.3
Glass	9.2	3.5	0.2	8.0	8.0	9.1
Textile	-	-	-	3.1	1.9	2.2
Metal	3.2	4.2	0.1	6.0	9.4	7.3
Others	20.3	11.3	22.7	6.9	4.6	10

All data are in % by weight

[1] Demir et. al.,1999,

[2] Baştürk, 1988

[3] Adapted from Tchobanoglous, 1993

Table 2.2 Composition of Municipal Solid Waste in Turkiye

Material Groups	**Tekirdağ** [1]	**Ş.Urfa** [2]	**Bolu** [1]	**İzmit** [2]
Organic Matter	24	44.7	0.5	64.5
Paper	1.8	1.4	2	6.6
Plastic	2.8	1.1	3	6.7
Glass	1.9	0.6	0.5	1.5
Textile	1.2	0.8	0.7	0.9
Metal	0.5	0.9	0.6	1.1
Ash	63.4	46.6	85	16.3
Others	4.4	3.9	7.7	2.4

All data are in % by weight

[1] Gören, 1997

[2] Baştürk, 1988

Table 2.3 Composition of Municipal Solid Waste in İstanbul

Material Groups	Baştürk (1979)	WHO (1981)	CH_2MHill-Antel (1992)	Arıkan (1996)
Ash	29.0	14.6	15.0	13.2
Organic Matter	46.5	60.6	45.0	48.0
Paper	12.0	10.0	14.5	8.4
Plastic	3.5	3.1	9.5	11.0
Glass	3.0	0.7	3.8	4.6
Textile	3.0	3.1	5.6	2.9
Metal	1.5	1.5	2.2	2.3
Others	1.5	6.1	4.4	6.3

All data are in % by weight

Istanbul Metropolitan City with a population over 12 million has to manage 8000 tons of solid waste every day, and the population is expected to reach 17 million by the year 2020. Amount of municipal solid waste has been increasing with increasing population and improved living standards. Currently, the waste generation rate is 0.65 kg/ca-day. Meanwhile, natural gas and low-ash coal use are increasing and expected to reach 40% for natural gas and 60% for coal by the year 2000. As a result, it is expected that, ash content will decrease by about 50%, and waste generation rate will then be 0.6 kg/ca-day (Arikan et.al., 1997) Some physical and chemical characteristics of the solid wastes in İstanbul are given in Table 2.4 as average values weighted by population.

Table 2.4 Some Physical and Chemical Characteristics of MSW in İstanbul.

Moisture Content (%)	Organic Matter (%)	C/N	Calorific Value (kcal/kg)	Unit Weight (kg/m^3)	Reference
46.3	50.1	30.5	811	416	Baştürk ,1979
-	51.7	-	-	410	WHO/UNDP,1981
54.5	-	-	1010	252	CH2M Hill-Antel , 1992
55.1	60	32	920	220	Arıkan, 1996

2.2 COLLECTION, TRANSFER AND TRANSPORTATION

2.2.1 Collection

Collection is the first fundamental function of solid waste management. It refers to the gathering of solid wastes from sources as residential, commercial, institutional and industrial areas, public parks and etc.

As the population and deviating urbanization increases, the collection of MSW is becoming harder and its cost is increasing. The amount of the fuel used for collection designate the cost of collection. The cost should be supplied from the citizens by MSW taxes, which will be indicated by municipalities to advance the collection system.

There are generally 2 methods of collection; *Hauled-container* and *Stationary container systems.* In hauled-container system, the container is hauled from the collection point to the final point of disposal. In stationary container system, the container is emptied into the vehicles at the point of collection.

In residential regions of İstanbul, generally stationary container systems are used in collection of MSWs. Hauled-container systems are used in some commercial and industrial zones.

2.2.2 Transfer Stations and Transportation

Transportation of MSW refers to the hauling of solid wastes to relatively far distances from the collection areas or transfer stations. The distance travelled may be to a final point of disposal or processing facility.

A transfer station is a facility where the wastes collected may be stored temporarily or transferred from smaller collection vehicles to bigger transport vehicles for transportation to the processing facility or disposal site.

In İstanbul, local municipalities are responsible from the transportation of MSWs, they collected, to the transfer stations. The operation of the transfer stations and transferring of MSW in the transfer stations to the disposal sites are the duties of metropolitan municipality.

There are 6 transfer stations in Istanbul. Halkalı, Yenibosna and Baruthane Transfer Stations are in European Side and Aydınlı, K.Bakkalköy and Hekimbaşı Transfer Stations are in Asian side (Figure2.1).

2.3 PROCESSING, RECOVERY & TRANSFORMATION

2.3.1 Processing and Recovery

Processing is the second fundamental function of solid waste management. It improves the efficiency of solid waste and prepares for subsequent recovery of materials and energy. Processing to segregate solid waste components may be done at the point of generation (on-site processing) or at a central processing facility. Operations in a central facility involve screening, shredding, air classifying and magnetic separations.

Recovery is a reusing of the recyclable solid wastes and gets back to the economy. Aims of recovery are to consume the materials in a small amount, prevention of decreasing of the air , water pollution and soil loss, decreasing consumption of energy resources by using energy content of the wastes and gaining disposal area.

2.3.2 Thermal Transformation Technologies

Incineration is the most common thermal conversion technique of MSW. It is the process of thermally reducing the volume of solid waste, while producing inoffensive gases and stabilized residue, by the application of the combustion process. It can also be defined as the conservation of solid wastes into gaseous, liquid and solid conversion products with the concurrent or subsequent release of heat energy.

Solid waste combustion system can be designed to operate with two types of solid waste fuel: commingled solid waste (mass-fired) and processed solid waste refuse-derived fuel (RDF-fired). Mass-fired combustion systems are the predominant type. Fluidized bed combustion (FBC) is an alternative design to conventional combustion systems. In its simplest form, an FBC system consists of a vertical steel cylinder, usually refractory-lined, with a sand bed, a supporting grid plate, and air injection nozzles.

Other thermal conversion techniques of MSWs are pyrolysis and gasification systems. Both pyrolysis and gasification systems are used to convert solid waste into gaseous, liquid, and soil fuels.

Incineration for Istanbul

The municipal solid waste in Istanbul contains 45% wet organic matter as vegetation and meal wastes with about 50% moisture content. 18 % of the remaining 55% is ash of coal. So, the calorific value of the solid waste is around 920 kcal/kg (Arikan, 1996), which is not suitable for incineration. With a pre-drying unit and additional fuel consumption for proper combustion, the cost of incineration is increased to 80-150 $/day.

2.3.3 Biological and Chemical Transformation Technologies

Composting is the most commonly used biological process for the conversion and stabilization of the organic portion of municipal solid waste and normally considered to be an aerobic process accomplished either by mixing or forced aeration. Applications of aerobic composting include yard waste, separated MSW, commingled MSW and co-composting with wastewater sludge. All aerobic composting processes are based on (1) preprocessing of the MSW, (2) aerobic decomposition of the organic fraction, and (3) product preparation and marketing. *Windrow*, *aerated static pile*, and *in-vessel* are the three principle methods used for aerobic composting (Tchobanoglous, 1993).

Anaerobic composting (digestion) techniques are also very promising since they not only produce a humus like residue comparable to the compost produced in aerobic conversion techniques, but also a form of energy, biogas, which can easily be upgraded to several forms of energy. There are two techniques in anaerobic composting as low and high solids digestion. In low-solids digestion, organic fraction of MSW is fermented at solids concentration equal to or less than 4 to 8 percent. To bring the solids content to this range, water must be added to the wastes and prior to disposal dewatering must be applied to remove this water. In high-solids digestion fermentation occurs at a total solids content of about 22 percent or higher. Lower water requirements and higher gas production per unit volume of the reactor are the two advantage of high-solids digestions.

Acid and alkaline hydrolysis are some of the chemical transformation processes for the recovery of conversion products as organic acids from MSWs. These chemical processes are not used routinely, because the conversion products can be easily manufactured from other cellulose containing wastes, such as wheat straw, sugar cane bagasse, and corncobs (Tchobanoglous, 1993).

Composting in İstanbul

A composting plant is under construction in European Side of Istanbul at Kemerburgaz with a capacity of 1000 tons/day of MSW. 500-600 tons/day of the waste will be used for composting after preprocessing and nearly 250 tons/day compost will be produced. The cost of the compost that will be produced in the plant is calculated as 20-22 $/day. If it is though that, the cost of landfilling is around 12 $/day in Istanbul, composting will increase the cost of ultimate disposal for 8-10 $/day (Demir, 1999).

2.4 DISPOSAL

2.4.1 Landfilling

Landfilling is one of the primary technologies used to dispose of solid waste and can be defined as a method of refuse disposal significantly limiting volume, where waste is systematically covered by layers of earth. Thus buried waste deteriorates as a result of natural oxidation and microbial degradation.

Sanitary landfills are those lined with materials preventing toxic substances from migrating towards surface or into ground water.

In sanitary landfill, all solid waste received at the site is spread and compacted to layers within a confined area. At the end of every working day, or more frequently, the compacted solid waste is completely covered with a relatively thin, continuous layer of soil which is then compacted solid waste varies depending on the type of waste landfilled and the techniques used for compacting.

Landfilling technologies

a) Trench Method; solid waste is spread and compacted in an excavated trench. Trench excavation tailings are used as cover material, which is readily available. The main advantage of this method is its adaptability to a wide variety of operations ad terrains.

b) Area Method; solid waste-cells are constructed in a large excavated area. Layers of cells are created until the permitted heights reached. Each cell represents the waste received and compacted in place and covered each day. This method of landfilling can accommodate very large-volume operations. Moreover, the installation of liners leachate collection systems is relatively easy.

c) Progressive Slope Method; is a combination of the previous ones, where solid waste is spread and compacted on a slope. Cover material is obtained from directly in front the working face and is compacted on the waste. The excavation of cover material provides a depression for the next day's waste.

Prior to landfilling, waste may be shredded or baled. The purpose of shredding is to increase the compaction rate and thus the capacity or life expectancy of the landfill. Indeed, shredded waste can be compacted to a density approximately 27% greater than unshredded waste. Moreover, gas production and land setting occur over a shorter period of time in shredded waste landfills, thus reducing site maintenance. On the other hand, the shredding process entails additional costs.

Historically, landfilling was thought to be the most economical waste disposal option. In the past, the cost of transporting the solid waste to the disposal site was not significant because there were many landfills located near the source of waste generation. Nowadays, the number of landfilling sites close to densely populated areas is declining. Accordingly to several estimates, the larger metropolitan areas will run out of landfill capacity in few years, and some of them are readily experiencing severe problems. As solid waste has to be hauled to more remote locations, transportation costs keep rising so landfilling become less attractive.

Recently, urban councils have been considering alternative methods of disposal based on recycling, material re-use and recovery of the energy content of waste, to the mere elimination of waste.

The energy crisis and problems of fuel availability have made waste-to-energy systems an interesting option for garbage disposal. Energy can be directly recovered from the waste simply by burning it or by turning it into a fuel for storage. On the other hand, re-use or recycling of waste as a substitute for raw materials represent an indirect way of recovering its energy content.

2.4.2 Composition, Characteristics And Management Of Landfill Gases

Concepts and Amounts of Landfill Gas

Landfill gas (LFG) is generated from the decomposition of municipal solid waste placed in sanitary landfills. Domestic solid wastes are usually formed by organic and inorganic substances that contain 20 –70% water. In organic substances, fats, proteins, carbonates are biodegradable substrates. The decomposition process that produces landfill gas is an anaerobic process, which means that free oxygen is not present. This decomposition differs from composting, which is an aerobic process that uses, and takes place in the presence of air. In the decomposition acids, alcohols, and CO_2 starts orderly.

Necessity of Landfill Gases

The need for gas movement control is primarily to prevent the gas from damaging plants and property or causing injury to people. Methane generated in landfills has killed vegetation; it displaces oxygen from the root zone. Most important thing is, If the gas that accumulated in the buildings exceed the lower limit of 5 %, there is danger of a methane explosion.

Gas Production

The sizing and implementation of gas handling equipment requires a prediction of gas production rates, yields, and gas composition from a particular landfill setting. Such a prediction may be based on theory or formulated from comparisons with empirical results from published laboratory and field experiences .In particular, the basic nature of landfill stabilization and the corresponding biophysical variations must be coupled to the refuse placement and leachate control technologies being utilized.

Table 2.5 Amount of Landfill Gas (EPA,1985)

Components	Typical Value %	Maximum %
CH_4	64	88
CO_2	34	89
O_2	0.16	21
N_2	2.4	87
H_2	0.05	21
CO	0.001	0.09
Ethane	0.005	0.01
Acetaldehyde	0.005	-
H_2S	0.00002	35
Alcohols	0.00001	0.127
Organosulfur compounds	0.00001	0.028
Unsaturated hydrocarbons	0.009	0.0048

Factors affecting Landfill Gas

1. Nature of refuse placed : The nature of the wastes influences the potential for gas production in terms of ; The relative availability of a usable substrate , The presence of potential inhibitors , The formation of localized " micro environments" which may be isolated from the overall liquid or gaseous transport phases.

2. Moisture content : Water or moisture provides the transport phase for organic substrates and nutrients and is also instrumental in establishing the anaerobic environment needed for methane production.

3. Particle size and degree of refuse compaction: Particle size reduction by refuse shredding may be expected to increase gas production rates by increasing the surface area available for leaching and/or biological activity and improving the ability to retain moisture.

4. Buffer Capacity : Buffer addition has been repeatedly demonstrated as beneficial to accelerating biological stabilization and increasing gas production rates.

5. Nutrients: Nutrient sufficiency may be best assured through initial addition or after leachate analysis by augmentation as needed again through leachate recycle or injection.

6. Temperature: Temperature affects microbial activity within landfill and vice versa. In the upper aerobic layers (1 to 2m), temperatures may range from 50 to 700°C; whereas, at lower aerobic levels (3m), temperatures generally range from 25 to 100° C.

7. Gas Extraction: The withdrawal of landfill gases at rates higher than their biological production the introduction of air into the landfill. This may not only inhibit the methanogens, but lead to excessive quantities of nitrogen and oxygen in the product gas.

Collection and Treatment of Landfill Gases

Gas collection systems employed in practice may consists of simple ventilation and/or flaring systems coupled with shallow trench induced exhaust networks intended primarily for migration control, and/or perforated pipe well matrices placed either vertically or horizontally.

Induced exhaust well systems are the most popular for energy recovery. These systems will generally encompass extraction equipment such as transport and well piping, backfill gravel systems generally incorporate perforated PVC pipe, although polyethylene or fiberglass pipes can be used

Landfill Gas Treatment

Raw landfill gases typically have allowed heating value due to the dilution of methane with CO_2, N2, and possibly O_2. They will likely contain troublesome constituents such as water and hydrogen sulfide (Table2.6).

Energy Recovery of Landfill Gas

The method of energy recovery will primarily depend open the available energy markets. The landfill gas is passed through filters to remove moisture and possible hydrogen sulfide and then injected into the furnace in combination with regular boiler fuel, which may be coal, oil, or natural gas. Boiler fuel is possibly the simplest approach for using landfill gas but availability of a boiler near a landfill is not common. Several different processes, including liquid solvent extraction and molecular sieves, are being employed to remove the carbon dioxide and other noncombustible constituents in landfill gas. The gas is upgraded to pipeline quality and injected into the natural gas distribution network. However, landfill gas can be directed to an engine – generator system for electricity production. Almost all landfills have electrical service and the generated power can be put back into the electric grid.

Table 2.6 Summary of gas treatment methods available for the removal of water, hydrocarbons, CO_2, and H_2S (EPA,1985)

Target Compound	Treatment process	Treatment process alternatives available
Water	Adsorption Absorption Refrigeration	Silica Del , Molecular sieves and Alumna Ethylene Glycol , Selector Chilling to 35°C
Hydrocarbons	Adsorption Absorption Combination	Activated carbon Lean oil absorption , Ethylene glacial and Selector Refrigeration
CO_2 and H_2S	Absorption Adsorption Membrane separation	Organic solvents, Alkaline salt solutions Molecular sieves , activated carbon Hollow fiber membrane

2.4.3. Composition, Characteristics and Management of Landfill Leachates

Formation of Leachate

Leachate is defined as any contaminated liquid that is generated from water percolating through a solid waste disposal site, accumulating contaminants, and moving into sub surface areas. A second source of leachate arises from the high moisture content of certain disposed wastes. As these wastes are compacted or chemically react, bound water is released as leachate. In the absence of a confining barrier beneath or surrounding the waste disposal site, this leachate can migrate and contaminate subsurface and surface waters.

Mechanisms of Leachate Formation

Leachate is produced when moisture enters the refuse in a landfill, extracts contaminants into the liquid phase, and produces a moisture content sufficiently high to initiate liquid flow. Sources of moisture entering the landfill include liquid present in the refuse at placement, precipitation falling on refuse at placement and infiltrating after cover application, and intrusion of groundwater from outside into the landfill.

Quantity of Leachate

The volume of leachate generated varies with the amount of precipitation and storm water run on and run off, the volume of ground water entering the waste-containing zone, and the moisture content and absorbent capacity of the waste material. When leachate is collected via perforated pipes, rainfall significantly affects the leachate volume and contaminant concentrations. Landfill age, ambient air temperature, precipitation and refuse permeability, depth, temperature and waste composition are among the factors that affect leachate quantity and composition. Leachate quantities are low in arid and hot regions, and high in rainy regions.

Characteristics of Leachate

Leachate has many components and is of changing character. Leachate character provides important clues on the landfill's age or status. Fundamental processes of waste degradation affects biogas characteristics and leachate character. These data are crucial for the design of the leachate treatment system and predicting possible changes.

Leachate characteristics vary depending on waste composition, pH, and redox potential, climatic conditions and landfill age. Solid waste composition affects leachate components and therefore, its treatability. Leachate contain several elements and compounds originating from solid waste's main components.

pH affects chemicals processes between waste and leachate such as dissolution, precipitation, redox and sorption. Redox potential affects the solubility of nutrients and metals in leachate.

Depony age, depending on the anaerobic treatment unit, is one of the most important factors affecting leachate characteristics. Readily degradable volatile fatty acids are high in percentage in young landfills. As the depony gets older, readily degradable organic matter drops in percentage due to the completion of biological degradation. Therefore, BOD/COD ratio is greater than 0.5 in young landfill leachates where it is lower than 0.2 in old landfill leachates. In 2-3 years old landfills, especially organic matter, microorganism species and inorganic pollution loads reach maximum levels. Leachate contains micro-pollutants as well as organic and inorganic ions and metals (Table 2.7).

Table 2.7 Change in leachate characteristics with landfill age (Timur, 1996)

Parameters	**< 1 year**	**5 year**	**10 year**
pH	4.8 – 5.2	5.0 – 6.6	5.6 – 6.1
COD	19700 – 45300	137 - 34900	293 – 10600
TOC	7300 – 16350	83 – 9150	108 – 3080
Total solids	10000 – 33000	718 – 18400	1920 – 5350
Total volatile solids	5350 – 20330	124 – 10300	770 - 3300
Alkalinity	4150 – 7700	184 - 7600	1240 – 2900
Chloride	620 – 1880	5.3 - 730	115 – 193
Cd	0.005 – 0.89	<0.001 – 0.162	<0.05 – 0.009
Cr	0.09 – 16.8	0.003 – 0.410	<0.025
Cu	0.03 – 0.12	0.009 – 0.09	<0.025
Fe	308 – 1.136	195 – 1820	98.7 – 855
Pb	0.077 – 3.15	0.003 – 0.082	<0.05 – 0.08
Ni	0.15 – 0.79	<0.005 – 0.342	<0.04 – 0.127
Zn	46 – 298	0.18 - 75	<0.025-0.167

All units in mg / L except pH

Chemical Composition of Leachate:

Organics: Organics are characterised by the carbon-carbon bond. One means of classifying the organics is into the three groupings indicated in the table below. There is increasing stability with respect to biodegradation from Group A through C (Table 2.8).

Table 2.8 Organic compound groupings

Group A Fatty Acids	**Group B Humic Acids**	**Group C Fulvic Acid-Like Substance**
Low molecular weight Examples: • Acetic • Propionic • Butyric	High molecular weight Carbohydrate-like substances • Carboxyl and aromatic hydroxyl groups	Intermediate molecular weight

Chian and deWalle (1975) conducted in-depth analyses to establish the composition of the organic fraction of the leachate. Their research revealed that "the volatile free fatty acids constituted from 20 to 70 percent of the total organic carbon (TOC) in leachate, depending on the age of the landfills." The percentages of these volatile fatty acids tend to decrease as the age of the landfill increases. The remainder of the organics was distributed approximately in equal amounts between the groups of compounds of fulvic acids, tannic acids, lignin, cellulose like material, and other organic matter.

Studies by Qasim and Burchinal (1970) showed that concentrations of organic substances (TOC, COD, and BOD) and the ratio of BOD/COD are generally highest during the active stages of decomposition and gradually decrease as the landfill stabilizes.

Analyses of leachate samples collected by Chian and de Walle (1975) from more stabilised landfills confirmed a general decrease in volatile fatty acids in leachate over time and showed that most of the organic material was present as fulvic-like refractory molecules. Only 0.5 percent of the TOC was present as humic material of high molecular weight. These authors suggested those physical and chemical processes as opposed to biological treatment processes might best treat such old or stabilised leachate.

Nitrogen: Ammonia and organic-N, collectively referred to as total Kjeldahl nitrogen (TKN), represent a high percentage of the total soluble nitrogen compounds in leachate. Combined, they are typically in the hundreds of milligrams per liter and may be considerably higher. Owing to the anaerobic conditions within landfills, concentrations of nitrite and nitrate are typically low.

If the resulting NH_3-N content is too high (e.g., greater than 1000 mg/l), nitrification may be inhibited. Nitrifying bacteria are also relatively sensitive to low temperatures. As a result, it may be necessary to partially reduce the level to a more acceptable concentration by physical-chemical methods prior to use of biological treatment methods.

Phosphorus: Landfill leachate is frequently deficient in phosphorus for effective biological treatment. Ratios of BOD: P of greater than 7000:1 have been found for leachate from recently emplaced wastes (Robinson and Maris, 1979). Since an optimum ratio of BOD: P of 100:1 is widely recommended for biological wastewater treatment processes (Metcalf and Eddy, 1991), biological treatment of the leachate will be inhibited owing to phosphorus deficiency unless additions of phosphorus are made.

pH: Particularly during the acid-phase stage of decomposition, pH levels are acidic, so they will require adjustments to allow biological treatment. In addition, if removal of metals is required, pH will have to be adjusted to form precipitates.

Heavy Metals: In general, removal of metals will occur if either aerobic or anaerobic biological treatment is used (Forgie, 1988) because the metals will precipitate out. If anaerobic treatment is used, the metals will tend to precipitate out as metal sulphides, whereas if aerobic treatment is selected, the metals will tend to oxidise and precipitate out as metal hydroxides. However, there may be specific instances when high concentrations of certain metals such as copper, zinc, and nickel may cause biological inhibition. In such cases, chemical precipitation may be needed.

On the negative side, however, removal of certain metallic ions such as Zn, Fe, Mn, and Pb in conjunction with sludge formation, either biologically or chemically, may lead to problems with sludge disposal. Because the sludges tend to concentrate the metals to two or three orders of magnitude greater than their influent concentrations, disposal problems arise.

Dissolved Solids: Leachate typically has high levels of total dissolved solids (e.g., chlorides, sulfates, and sodium). These constituents are not very reactive and therefore not easily removed. In the event of the need for removal, physical/chemical processes will be required.

Management of Landfill Leachate

The most widespread active leachate management strategy is "containment and collection" and the major components of the management systems at such sites are; leachate containment, leachate collection, recirculation and other landfill management techniques are treatment and discharge.

Containment can be achieved using mineral liners, synthetic liners, or composite liners, although synthetic liners are rarely used alone. Most common one for MSW is composite liner. Leakage rates are estimated to be from 0.04 to 0.2 mm/a, depending on the level of quality control.

Leachate collection typically consists of 0.3m inert coarse gravel, containing drainage pipes, leading to sumps for abstraction. The allowable head on the bottom liner is typically restricted to 0.3 to 1m. There is some evidence of clogging due to chemical precipitation and biological growth, but it appears to be dramatically reduced in methanogenic conditions.

Recirculation and the placement of a pre-composting layer of MSW, are both used to promote methanogenesis, the latter appearing to be universally effective.

The most common leachate disposal route is to sewer without any pre-treatment. This accounts for 60-80 % of MSW sites with active leachate removal.

Sources of Variability

Landfill leachates from municipal solid wastes generally contain high concentrations of organic and inorganic chemicals. The inorganic ions include chlorides and sulfates, and metals such as iron, sodium, potassium, calcium, manganese, and zinc. Evaluation of alternatives for treatment of leachate must consider the large temporal fluctuations in both the quantity and composition of leachate at a particular site. An important characteristic relevant to its treatability is then the change over time of certain components of leachate as the biological conditions change within the landfill.

Design of treatment systems for landfill leachate must consider the variability in the flow rates and the complex and temporally varying composition. To respond to the variability, the options for treatment of leachate may include (1) full treatment on site, (2) partial treatment on site and disposal to a publicly owned treatment works (POTW), or (3) transport offsite to a POTW directly. A part of the consideration of discharge possibilities is related to whether the leachate will have a detrimental impact on the POTW. The limited data that have been published on co-treatment indicate that generally 2 % leachate by volume will produce acceptable results. Due to the leachate volume contribution, the impacts on the POTW will probably include higher aeration requirements, a phosphorus deficiency for good biological treatment, higher sludge production due to the increased biomass and metal precipitates, elevated levels of metals in the sludge, increased foaming problems, and odors.

The option selected to treat leachate is a function of numerous factors, which are primarily a function of economics. The considerations include the relevant water quality standards or criteria that must be met prior to discharge, the extent of the variability in leachate flow and contaminant concentrations, the costs of exceeding the contaminant discharge criteria established by the POTW, and the physical proximity of the POTW. Thus, before decisions are made on whether to treat the leachate partially or fully on-site or off-site, it is first essential to understand the sources of both periodic and long-term variability.

Discharge Standards for Leachate

Two discharge standards are applicable for leachates as for other wastewater:

- Standard for discharge to the municipal sewer system after pretreatment,
- Receiving water bodies discharge standard.

Leachate discharged to the local sewer system should meet the discharge standards stated by the local municipality. If it is discharged to a receiving body, then it should meet the corresponding discharge standard. In İstanbul, the standard for discharges to local sewer system is "İSKİ Sewer Discharge Standards" and "Water Pollution Control Regulations" is valid when discharge is direct to a receiving water body. Both standards are given in Table 2.9.

Table 2.9 "Water Pollution Control Standards" and "ISKI Sewer Discharge Standards"

Parameters	**Water Pollution Control Regulations**		**İSKİ Sewer Discharge Standards**
	2 hr Composite Sample	24 hr composite Sample	2 hr composite sample
pH	6 - 9	6 - 9	
COD	160	100	800
BOD_5	100	50	-
Suspended Solids	200	100	350
Total Nitrogen	-	-	100
Total Phosphorus	2	1.0	10
Oil and Grease	20	10	100
Demir (Fe)	10	-	-
Cadmium (Cd)	0.1	-	2
Krom (Cr)	0.5	0.5	5
Copper (Cu)	3	-	5
Kurşun (Pb)	2	1.0	3
Nickel (Ni)	-	-	5
Zinc (Zn)	5	-	10
Silver (Ag)	-	-	5
Total Cyanide	1.0	0.5	10
Sulfate	-	-	1700

All units in mg / L except pH

2.4.4 Treatment Techniques for Leachate

Treatment Alternatives

The alternatives include aerobic or anaerobic biological processes, and physical and chemical treatment methods. In addition, it is possible to recycle the leachate through the landfill as a partial means of treatment and then spray the leachate onto land as a disposal method. The focus of anaerobic individual treatment alternative is typically toward anaerobic individual chemical, with the entire system being sequenced to collectively treat the array of contaminants within the leachate.

Table 2.10 Summary of treatment processes for the major constituents in leachate

Chemical	**Commentary**	**Likely Treatment Process**
Organic strength	*Young leachate* -BOD in 10,000s (mg/l) -in the form of volatile fatty acids -amenable to biological treatment processes, barring any toxic inhibition *Old leachate* -BOD in 100s and COD in 1000s -in the form of humic and fulvic acids	-Biological treatment -Carbon adsorption
Ammonia	-in the 1000s -some will be removed in biological-uptake	-Biological nitrification/ denitrification -Air stripping -Breakpoint chlorination -Ion exchange
Heavy metals	-in the 10s and 100s -iron (Fe) mainly, Zn, Pb, Cu also	-Chemical precipitation -Biological treatment
Phosphorus	-in the 10s (mg/l)	-Supplementary additions needed for biological activity
pH	-leachate usually acidic	-Neutralization using lime or caustic
Conservative ions	Cl^- and SO_4^{2-} in 1000s (mg/l), K^+ and Na^+ in 1000s (mg/l), leachate typically have high total dissolved solids	-Reverse osmosis and ultrafiltration

The treatment alternatives to be considered for application in a specific situation are a function of the necessary quality of the effluent at discharge. On-site treatment to allow discharge of the treated water to a nearby water body will necessarily involve much more extensive treatment than the alternative of partial treatment prior to discharge to a POTW.

Biological Treatment

The focus in biological treatment is to change the form of the organic constituents. The downside of biological treatment is that it may produce relatively large quantities of biomass sludge requiring subsequent disposal.

For a leachate with a high BOD/COD ratio (>0.4) the only major treatment options are aerobic or anaerobic biological treatment. In Table 2.11 some of the alternative approaches are listed. The first column differentiates between those processes in which the micro-organisms are (1) suspended or (2) attached to some type of fixed surface.

Table 2.11 Treatment summary of biological processes

Biomass	Treatment	Recycle	Comments
In suspension	Facultative ponds	No	Involves both aerobic and anaerobic decomposition. Odours will result during process. Care must be taken to avoid biomass washout.
	Aerated ponds	No	Effluent will be high in suspended solids Aeration requires substantial energy Care must be taken to avoid biomass washout
	Activated sludge	Yes	Less sensitive to shock loading Care must be taken to avoid biomass washout
Attached	Rotating biological contactors	Yes	Less sensitive to biomass washout
	Packed filter bed	Yes	Less sensitive to biomass washout
	Trickling filter bed	Yes	Less sensitive to biomass washout

Aerobic Treatment

Fatty acids are a product of the anaerobic decomposition of the organic materials within the landfill and thus are in the leachate. However, these fatty acids are easily biodegradable using aerobic processes, barring any toxic inhibition. They do, however, require very large quantities of oxygen, and the processes generate huge quantities of biomass that must be disposed of.

Aerobic biological treatment processes include activated sludge facilities, rotating biological conductors (RBCs), and trickling filters. All the aerobic processes work on the same principle-microorganisms acting on organic matter in the presence of oxygen. The processes are differentiated by whether the microorganisms are suspension organic fixed to a medium. Note that to keep the biological action in the aerobic phase, it is essential to supply large quantities of oxygen, particularly when treating young leachates with their associated high concentrations of organics.

Ammonia concentrations in the effluent from aerobic treatment facilities can be very low when nitrification has occurred; nitrate may be correspondingly high. If there is a nitrate concentration limitation for the effluent, biological denitrification may have to be included in the treatment process train.

Aerobic treatment systems are effective for young leaches when the BOD/COD ratio is greater than 5. Leachates from recently placed refuse may be treated satisfactorily by aerobic processes on a small scale. However, aerobic biological treatment of leachates will not be successful at high organic loading and low retention periods without the addition of nutrients and large aeration rates.

a) Activated Sludge

Activated sludge process successfully treated leachate in most cases. BOD and COD removed efficiency varied from 90 to 99 percent. Lu et al.(1984)analyzed the laboratory data of many researchers.

The operational parameters clearly indicate that large amounts of organic matter in leachate were not readily oxidized, and required extensive biological activity for stabilization.

b) Aerated Lagoon

Aerated lagoon organic ponds are aerated by a series of aerators (mechanical or diffuser). The diffused air can be used to create mixing as well as to supply the oxygen. Note that aerobic lagoons have a limited ability to effectively handle increasing leachate strength.

Chian and deWalle (1977b) showed that, in general most of organic matter in effluents from aerated lagoons consists of stable refractory materials, often with a high molecular weight. These effluents were similar to leachates from older, relatively stabilized landfills.

c) Rotating Biological Contactors (RBCs)

The RBC unit is a fixed-film biological processes in which the biomass accumulates on rotating disks. A fixed-film biomass system tolerates hydraulic organic chemical shock loading, which are common in landfill leachate, better than suspended growth processes. However, RBC units tend to clog with calcium deposits that prevent the growth of the biomass. As a result many treatment systems utilize NaOH preceding the RBC system to raise the pH and precipitate out the metals.

d) Trickling Filter

Trickling filters use a fixed surface, such as rocks, as the medium on which the biomass grows, and provide a discontinuous trickling of the water being treated over the biomass.

The air vents through a trickling filter from the bottom to the top. These methods consume low amounts of energy. However, there are limitations to the treatment capability for leachates high in organics, since the precipitates and / or biomass will clog the system and / or may not supply oxygen at the rate needed to keep the system aerobic.

Anaerobic Treatment

Anaerobic biological activity is the natural degradation process that occurs within a sanitary landfill. This can be followed by further anaerobic treatment in the leachate treatment facilities.

In an anaerobic wastewater treatment system, complex organic molecules in the influent wastewater are fermented by bacteria to volatile fatty acids, mainly acetic, propionic, butyric. These in turn are converted by methanogenic bacteria to methane and carbon dioxide, resulting in a low production of biological solids requiring disposal (A similar sequence of conversions occurs during the anaerobic decomposition of refuse within the landfill).

Anaerobic biological processes for the treatment of leachates have several potential advantages over aerobic biological processes. The advantages include the generation of methane gas as a byproduct and the much lower production of biological solids in the form of sludge organic suspensions. In addition, the systems have no need of aeration equipment and its considerable energy requirements.

Physical and Chemical Treatment of Leachate

As the landfill stabilizes, there is a decrease in the proportion of readily biodegradable organic compounds contained in the leachate. The effectiveness of biological leachate treatment processes therefore decreases as landfill waste stabilizes, and other forms of treatment such as physical and chemical techniques may become more appropriate.

Therefore, physical and chemical treatment alternatives represent an addition to, or replacement for, the aerobic and /or anaerobic biological treatment of landfill leachate.

A stand-alone physical/chemical system is applicable only for old leachates. Types of physical and chemical treatment processes;

a) Granular Filtration

Granular filtration removes suspended solids and is typical employed prior to the use of activated carbon to prevent clogging of the carbon by the suspended solids.

b) Carbon Adsorption

Treatment through sorption onto activated carbon generally occurs in a column configuration, although dispersed carbon treatment systems also exist. Columns may be downflow of upflow, with the latter having either packed or suspended carbon. Use of granular activated carbon or powdered activated carbon has been shown to be very effective for treatment of poorly biodegradable organics, solvents, pesticides, humic acids, and the like. Carbon adsorption can thus be used to reduce COD in old leachates or to remove color and refractory organics contributing to residual COD.

c) Chemical Precipitation

If the raw leachate is treated biologically, either anaerobically or aerobically, heavy metals will be removed either as metal sulfides in anaerobic treatment or as metal hydroxides in aerobic treatment. If no biological treatment is involved in the treatment process sequence, or if the metal content is still too high, the treatment options include chemical precipitation with lime or caustic, aeration, or chemical oxidants such as chlorine, hydrogen peroxide, or potassium permanganate.

d) Ultrafiltration

Ultrafiltration is an effective means of removing high molecular weight material from leachate. However, smaller molecular weight species frequently escape through the filter. Pretreatment with a biological process improves the effectiveness of ultrafiltration by removing particulate matter that would otherwise foul the filter.

e) Reverse Osmosis

Reverse osmosis is a separation process that may be appropriate in specific leachate treatment applications. With new developments in membrane material, it is possible to get very good quality effluent with the exception of acetic-phase leachate. The acetic phase leachate have very small organic molecules that tend to pass through ultrafiltration and are not retained in reverse osmosis (Forgie, 1988b).

f) Breakpoint Chlorination

The use of breakpoint chlorination provides a removal mechanism for nitrogen. Specifically, as chlorine is added, readily oxidizable substances and organic matter react with the chlorine. After meeting this immediate demand, the chlorine continues to react with ammonia to form chloroamines. With continued chlorine additions, some of the nitrogen is release as N_2 gas.

g) Air Stripping

Air stripping can be used in conjunction with anaerobic treatment to remove ammonia from the effluent. Air stripping can also be effected prior to biological treatment to remove ammonia nitrogen concentrations. An advantage of air stripping is that it can be adjusted relatively easily to changes in volume and strength of the leachate. A disadvantage of air stripping is that the efficiency drops off dramatically at low temperatures (e.g. 5- 10 °C). also, the air discharge often has to be treated with activated carbon.

h) Ion Exchange

The success of removal of organic matter by ion exchange is strongly dependent on organic matter type and the ion exchange resin being used. Most investigators have found relatively poor removal efficiencies, particularly with the low molecular weight organic compounds typical of raw young leachates. In contrast, ion exchange has achieved excellent COD removal from effluents from aerobic biological treatment systems.

CHAPTER 3
CHARACTERIZATION OF KÖMÜRCÜODA LANDFILL LEACHATE

3.1 DESCRIPTION OF KÖMÜRCÜODA LANDFILL

Kömürcüoda Sanitary Landfill is one of the two landfills in Istanbul. It is constructed in Şile for the Asian Side of the city in 1994 with a 20-years design capacity. It is under operation since January 1995. The total area of the landfill is 98 ha, and about 3 million tons of solid waste has been landfilled to 25 ha area at the end of May 1999 (Table 3.1).

Table 3.1. Solid Waste Landfilled in Kömürcüoda and Odayeri Landfills

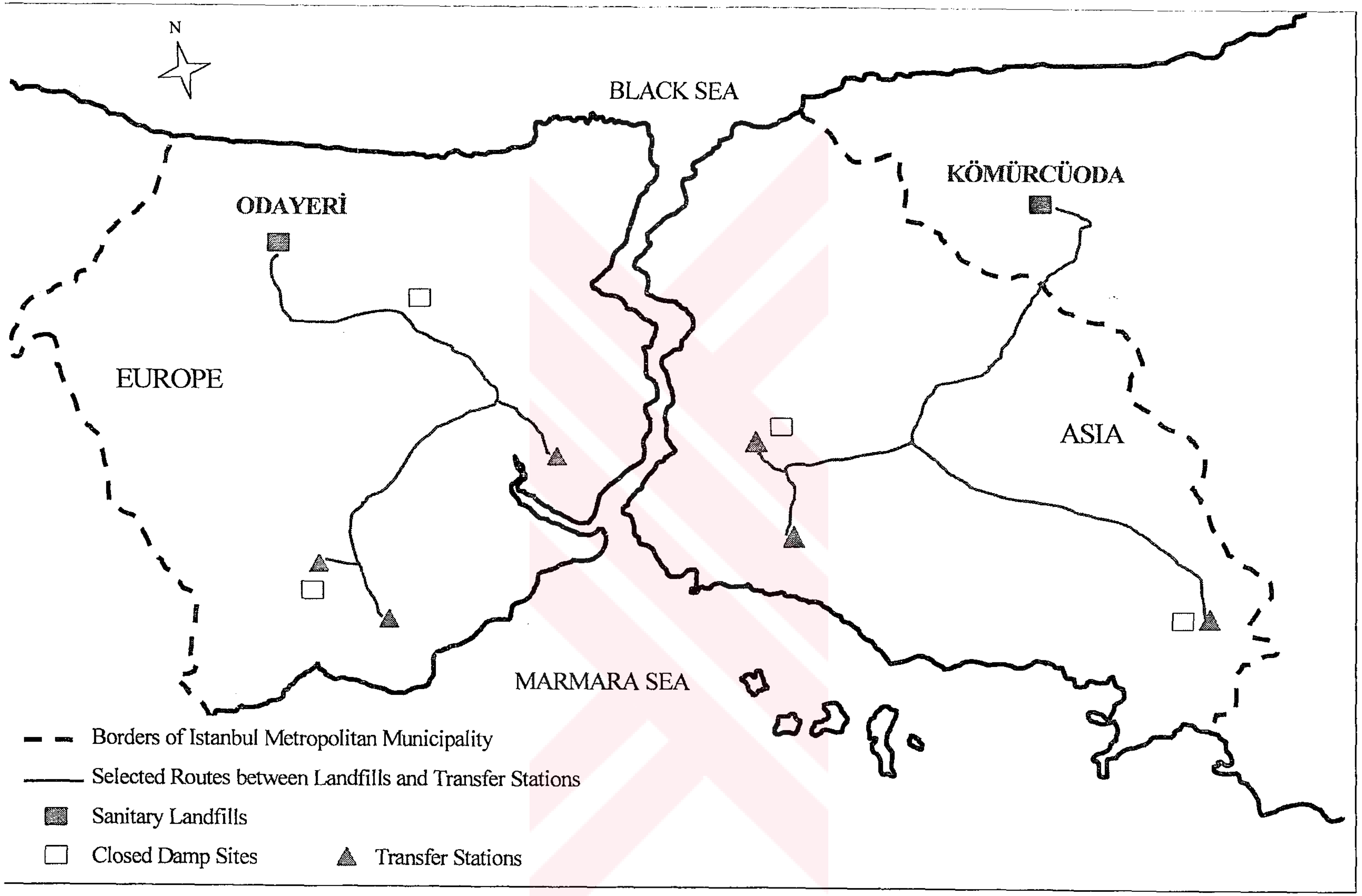
	1995	**1996**	**1997**	**1998**
Kömürcüoda	224235	631622	779412	879000
Odayeri	513017	1122443	1521396	1752000
Total	737252	1754065	2300808	2631000

All data are in tons/year

3.1.1 Location and Siting Considerations

After excluding the water catchment basins and valuable agricultural lands in Istanbul, there are not many alternatives for the landfill sites. On both sides, lands at the northern part are suitable for landfilling. These regions are covered with forests but there are lots of old mining sites left without closure and rehabilitation. Also the population density in these regions is very low and there are limited residential areas.

Kömürcüoda Sanitary Landfill is also located on a similar region, which was an old lignite and clay mining site. This site is very suitable for landfilling with a basement that is composed of clay and having a slight slope. The access to the landfill is carried out with a 5-km asphalt road that connects the landfill to the Şile Highway. By this highway Kömürcüoda Sanitary Landfill is 53km to Aydınlı, 44km to Hekimbaşı and 48km to K.Bakkalköy Transfer Stations (Figure 3.1).


N
BLACK SEA
KÖMÜRCÜODA
ODAYERİ
EUROPE
ASIA
MARMARA SEA
Borders of Istanbul Metropolitan Municipality
Selected Routes between Landfills and Transfer Stations
Sanitary Landfills
Closed Damp Sites
Transfer Stations

3.1.2 Operational Practices

The solid waste transferred to Kömürcüoda Sanitary Landfill from Aydınlı and Hekimbaşı transfer Stations is landfilled by cell method. The basement of the cells are covered with a 60 cm clay layer and on this layer 2 mm HDPE (high density polyethylene) geo-membranes are placed between two preserving sand layers. Then, the leachate collection pipes are laid in the drainage layer of gravel.

The solid waste is spread on the prepared cell with a 1/3 slope and 50-60 cm thickness and afterwards it is compacted. The thickness of compacted cells is approximately 5m. Everyday, the cells in the landfill site are covered with an approximately 15-cm soil layer. In filled lifts, an intermediate cover of 30 cm is used to control the storm water (Tüylüoğlu et. al., 1999).

For the collection of the landfill gases, stacks having effective collection diameters of 50-100m are placed vertically to the cells. Each stack is composed of 150 mm HDPE perforated pipes and drainage layer of gravel that placed in steel screens. A passive control of landfill gasses are carried out as releasing to the atmosphere, but some projects are on the way to produce energy from the collected landfill gases in Kömürcüoda Sanitary Landfill.

For the safety of the workers in the landfill, regular health controls are carried out periodically. Besides, all workers are forced to use protective equipment for providing occupational safety in the site.

After all landfilling operations have completed a final cover layer will be applied to the entire surface of Kömürcüoda Sanitary Landfill and the area will be rehabilitated.

3.1.3 Leachate Generation and Management

The leachate generated in Kömürcüoda Sanitary Landfill is collected in a 200 m^3 concrete tank that is located at the lowest elevation of the site. At the beginning years of operation in the landfill excess leachate was recirculated from the holding tank onto the cells to increase the biological degradation, but because of technical problems and the insufficiency of holding tank volume recirculation is left. Today, the excess leachate is transported by tankers and discharged to the nearest sewer line of ISKI. For a definite solution, a treatment plant is under construction that is mentioned in section 2.4.4.

The leachate generated in Kömürcüoda Sanitary Landfill is transferred to the nearest sewer line by tankers. In 1997 the volume of the transferred leachate was about 110780 m^3. In Table 3.2 leachate generation in 1998 and since June in 1999 is presented with respect to the volume transferred.

Table 3.2 Leachate Transferred from Kömürcüoda Sanitary Landfill

Year	**Month**	**Leachate Generation, m^3**
1998	January	7 500
	February	9 944
	March	13 001
	April	15 180
	May	16 362
	June	15 610
	July	16 050
	August	12 957
	September	11 541
	October	16 375
	November	13 343
	December	13 704
1999	January	14 936
	February	12 328
	March	14 490
	April	15 542
	May	15 661
	June	16 562

The leachate generation rate in Kömürcüoda Sanitary Landfill is based on meteorological conditions in the region and composition of the solid waste landfilled, The basic constituent of the solid waste coming to the landfill in summer is organic food waste and coal ash in winter. In summer, organic food wastes tend to increase the generation of leachate by rapid hydrolysis but generally because of low precipitation the increase in flow rate of the leachate is balanced in these months. On the other hand in winter, although the coal ash, having high absorption capacity, is dominant in the new cells of the landfill, the flowrate of the leachate does not decrease sharply because of high precipitation amount. The average precipitation amounts during the year in Asian Side of Istanbul are given in Figure3.2.

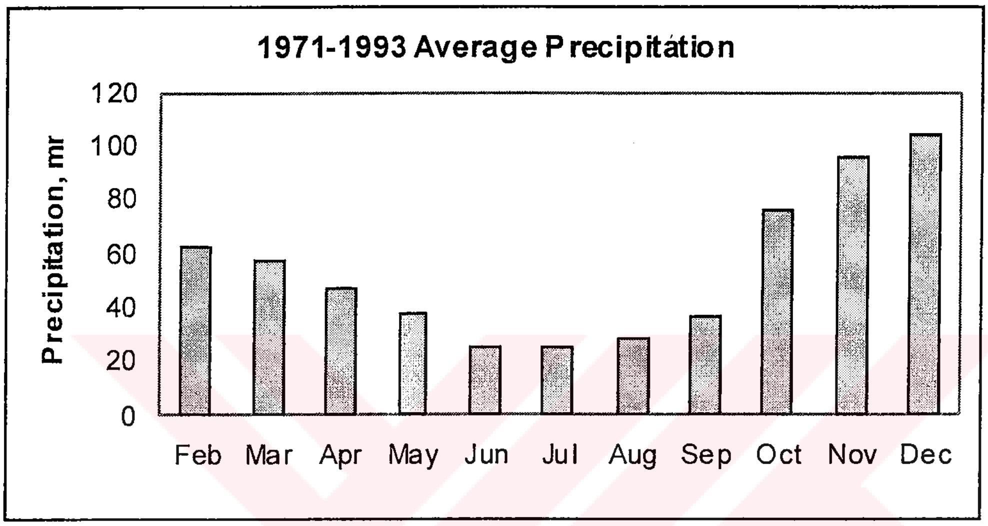


Figure 3.2 Monthly average precipitation in Asian Side of İstanbul (Meterological Data)

3.2 ANALYSES FOR CHARACTERIZATION OF LEACHATE

3.2.1 Collection and Storage of Leachate

For the characterization of Kömürcüoda Sanitary Landfill leachate, samples were taken, starting from August 1998, once every week from the 200-m^3 leachate holding tank located at the lowest elevation of the site. They are transported to the laboratory by İSTAÇ, the company operating the landfills in İstanbul, in 30lt containers. In the laboratory, these containers are stored in cool places away from direct sunlight.

3.2.2 Analyses

To monitor the time based fluctuations in the characteristics of leachate, several parameters are analysed in the samples. These parameters are; chemical oxygen demand (COD), 5-days biochemical oxygen demand (BOD_5), pH, alkalinity, chloride, ammonia-nitrogen (NH_3-N), total Kjeldahl nitrogen (TKN), color, total suspended solids (TSS), volatile suspended solids (VSS), total phosphorus, and heavy metals as iron (Fe), manganese (Mn), zinc (Zn), copper (Cu), lead (Pb), cadmium (Cd), chromium (Cr) and nickel (Ni). All analyses were carried out according to the Standard Methods (APHA , 1989). The analytical methods are given in Table 3.3.

Table 3.3 Analyses for Characterization of Leachate

Parameter	**Analytical Method**
pH	Electrode
Alkalinity	Titrimetric
COD	Closed Reflux
BOD_5	Winkler Method
NH_3-N	Nesslerization
TKN	Digestion + Nesslerization
SS & VSS	Gravimetric
Chloride(Cl^-)	Mercuric Thiocyanate
Color	Spectrophotometer(Pt. Co)
Total Phosphorous	Digestion + Aminoacid Method
Heavy Metals	Atomic absorption spectroscopy

Leachate samples brought from Kömürcüoda Sanitary Landfill were analyzed immediately after arriving at the laboratory. The results are presented in Table 3.4 and 3.5. The values in Table 3.4 are the averages of the samples taken in each month starting from August 1998 to May 1999. Samples were taken every week in August and September 1998, and once every week in the following months. Maximum, minimum and average values in Table 3.5 represent all samples these analyzed. pH has varied from 6.2 to 8.2 although Volatile fatty acids were high in the samples, which could be understood easily from smell. This was proved by qualitative analysis with gas chromatography. Relatively higher pH values were actually due to high alkalinity levels which was effected strongly by extraordinarily high ammonia concentrations. As it can be seen from Table 3.5, alkalinity has ranged between 6900 and 11750 mg$CaCO_3$/L while ammonia was between 1660 and 2690mg/L (Figure 3.3).

Table 3.4 Characteristics of leachate of Kömürcüoda MSW Landfill

	Monthly Average Values									
Parameters	**Aug 98**	**Sep 98**	**Oct 98**	**Nov 98**	**Dec 98**	**Jan 99**	**Feb 99**	**Mar 99**	**Apr 99**	**May 99**
pH	6.9	7.4	7.8	7.7	7.7	8.0	7.6	7.7	7.6	7.2
Alkalinity, mgCaCO3/L	10013	9888	9650	9000	7225	11100	10650	9800	9750	11175
COD, mg/L	34188	23840	19975	22440	20240	12810	13750	16650	15620	23475
BOD5, mg/L	24848	19700	13360	15828	9010	7650	8450	9500	10100	16190
NH3-N, mg/L	2218	2050	1975	2253	1890	2335	2380	2350	2430	2365
TKN, mg/L	2358	2416	2155	2518	2100	2510	2720	2640	2670	2610
SS, mg/L	1873	1896	2085	1837	1310	1400	1285	1470	1610	2020
TP, mg/L	3.1	3.1	4.4	1.9	3.7	1.6	0.8	2.3	3.5	3.3
Color, PtCo	14453	18728	12775	18190	11100	15900	17200	16500	22800	34850
Chloride, mg/L	3611	2429	3250	4097	1550	4125	4180	4900	5415	4200
Fe, mg/L	113.0	79.0	58.4	50.6	50.0	44.5	17.5	4.9	5.9	7.06
Mn, mg/L	8.48	5.12	2.26	4.48	1.23	0.92	0.82	0.84	0.86	0.97
Cu, mg/L	0.29	0.22	0.2	0.18	0.18	0.14	0.22	0.11	0.11	0.15
Zn, mg/L	1.67	0.84	1.30	0.89	0.89	0.68	0.79	0.85	0.84	0.94
Pb, mg/L	2.06	1.30	0.87	0.56	0.62	0.69	0.40	0.49	0.49	0.51
Cd, mg/L	0.10	0.12	0.12	0.11	0.08	0.05	0.07	0.08	0.06	0.09
Cr, mg/L	0.45	0.57	0.78	0.11	0.37	0.15	0.30	0.11	0.13	0.16
Ni, mg/L	1.12	1.04	1.03	0.50	0.64	0.56	0.39	0.57	0.62	0.72

Table 3.5. Characteristics of Leachate; Minimum, Maximum and Average Concentrations

Parameters	**Max**	**Min**	**Average**	**Std.Dev.**
pH	8,2	6,2	7,5	0,5
Alkalinity, mg$CaCO_3$/L	11750	6900	9682	1384
COD, mg/L	47800	18800	21635	10381
BOD_5, mg/L	38500	6820	15990	8377
NH_3-N, mg/L	2690	1660	2167	258
TKN, mg/L	2984	1825	2390	294
SS, mg/L	2915	940	1710	535
TP, mg/L	6,36	0,45	2,72	1,8
Color, Pt Co	30500	7400	15962	5577
Chloride, mg/L	5150	725	3378	1250
Fe, mg/L	245,5	4,91	63,2	51,6
Mn, mg/L	21,1	0,27	3,95	5,19
Cu, mg/L	0,42	0,11	0,21	0,07
Zn, mg/L	3,61	0,48	1,04	0,66
Pb, mg/L	3,57	0,31	1,04	0,77
Cd, mg/L	0,21	0,04	0,09	0,04
Cr, mg/L	0,91	0,07	0,38	0,28
Ni, mg/L	2,23	0,36	0,79	0,42

Similar characteristics and high ammonia levels around 2200 mg/L were also experienced in other young landfills in Harmandali-Izmir, Hamitler-Bursa and Odayeri - European side of Istanbul. The leachate characteristics of these landfills are given in Table 3.6 for comparison. These high concentrations of ammonia seem to be specific to Mediterranean countries, because similar concentrations represented in Table 3.7 are detected in Italy and Greece. Ammonia level in USA and France are comparatively lower. Ammonia levels are quite important for possibility of inhibition on anaerobic and nitrification processes. TKN/NH_3-N ratios around 1.15 indicate that most of the nitrogen is in ammonia form (Figure 3.3).

Table 3.6 Leachate Characteristics of Some Young Sanitary Landfills in Turkiye

Parameters	**Harmandali Izmir (1996)**	**Hamitler Bursa (1995-96)**	**Odayeri Istanbul (1995-98)**	**Kömürcüoda Istanbul (This study)**
PH	7.5-7.8	5.6-8.4	5.6-7.5	6.2-8.2
Alkalinity, mg/L	7040-13050	-	11500-13150	6900-11750
COD, mg/L	14900-19980	11760-32380	30100-70000	18800-47800
BOD_5, mg/L	6900-11000	6450-23000	21000-31000	6820-38500
NH_3-N, mg/L	1120-2780	1400	1345-2033	1660-2690
TKN, mg/L	1350-3280	-	1630-4490	1825-2984
TP, mg/L	-	8	1.0-6.0	0.45-6.4
Chloride, mg/L	5620-6330	1210-1706	-	725-8500
Fe, mg/L	14.2-44.0	1210-1706	60-130	4.91-245.5

COD concentrations have ranged between 18800 and 47800 mg/L while BOD_5 between 6820 and 38500 mg/L (Figure 3.2). COD and BOD_5 values were higher in summer and lower in winter due to dilution by precipitation in fall and winter. On the other hand, it was quite interesting that such dilution effect was not observed for ammonia. The highest ammonia concentration, 2690 mg/L was in November 1998. BOD_5/COD ratio was larger than 0.7 for most samples indicating high biodegradability, and acidic phase of decomposition in the landfill.

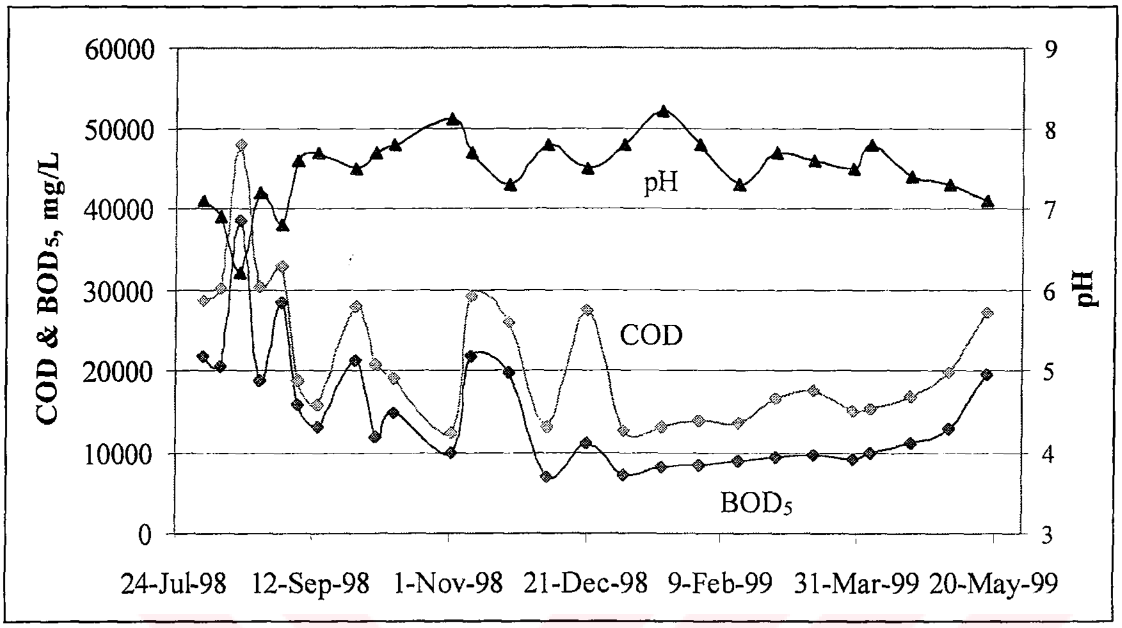


Figure 3.3. Time versus COD and BOD_5 data together with pH

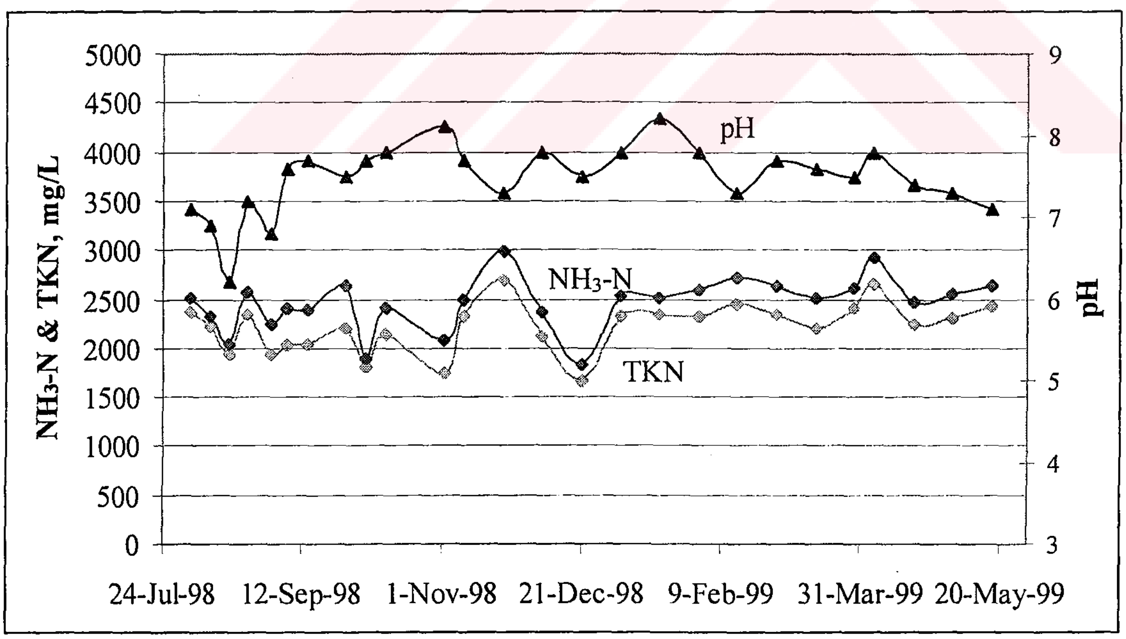


Figure 3.4. Time versus NH_3-N and TKN data together with pH

Table 3.7. Leachate Characteristics of Sanitary Landfills in Other Countries

Parameters	**Omega Hills USA (1984)**	**Serdiana Italy (1997)**	**Thessaloniki Greece (1993)**
pH	6.0-7.6	8.4	5.6-6.3
Alkalinity, mg/L	12260-15670	-	-
COD, mg/L	35800-60950	12950	60000-77500
BOD_5, mg/L	26120-45070	-	31500-41000
NH_3-N, mg/L	635-1020	2760	900-1510
TKN, mg/L	850-1410	2800	1560-2220
TP, mg/L	0.6-138	1.9	14.6-23.8
Chloride, mg/L	2990-3620	4715	3780-3820
Fe, mg/L	244-1710	-	8.7-43.0

Total phosphorus concentrations were very low indicating phosphorus deficiency of the leachate. Chloride concentrations have varied between 725 and 5150 mg/L (Table 3.5).

In general, heavy metal concentrations were low, except for iron which has varied between 4.91 and 245.5 mg/L. Compared with the Omega Hills Landfill (USA), these iron concentrations are quite low (Table3.7).

CHAPTER 4

ANAEROBIC TREATABILITY STUDIES ON KÖMÜRCÜODA LANDFILL LEACHATE

4.1 ANAEROBIC REACTORS

Set-Up Description

A leachate treatment plant is designed for Kömürcüoda Sanitary Landfill. In the process leachate is pumped to an UASB reactor from an eqaulization tank. Then an ammonia stripping is applied to anaerobically treated leachate. After the neutralization and intermediate ozonation, leachate is treated aerobically in SBR's. As tertiary treatment, stabilization, post-ozonation, sand and GAC filtration are carried out prior to discharge. The flow chart of this treatment plant is given in Figure 4.1.

Istanbul Metropolitan Municipality has started the construction of this treatment plant although the exact characteristics and performances of anaerobic reactors are not known. Therefore, in this study after the characterization of the leachate, anaerobic treatability using 3 different types of reactors namely anaerobic filter, UASB and hybrid bed reactors is investigated. Operational conditions and physical properties of the reactors are given in Table 4.1.

A constant temperature room was used to maintain the temperature of the reactors at 35 ± 2 °C. To provide this constant temperature in the room a heater controlled by a thermostat is utilised.

The reactors were seeded with sludge taken from an anaerobic treatment plant of a baker's yeast industry for the start-up. The set-up used in the anaerobic treatability study is presented in Figure 4.2.

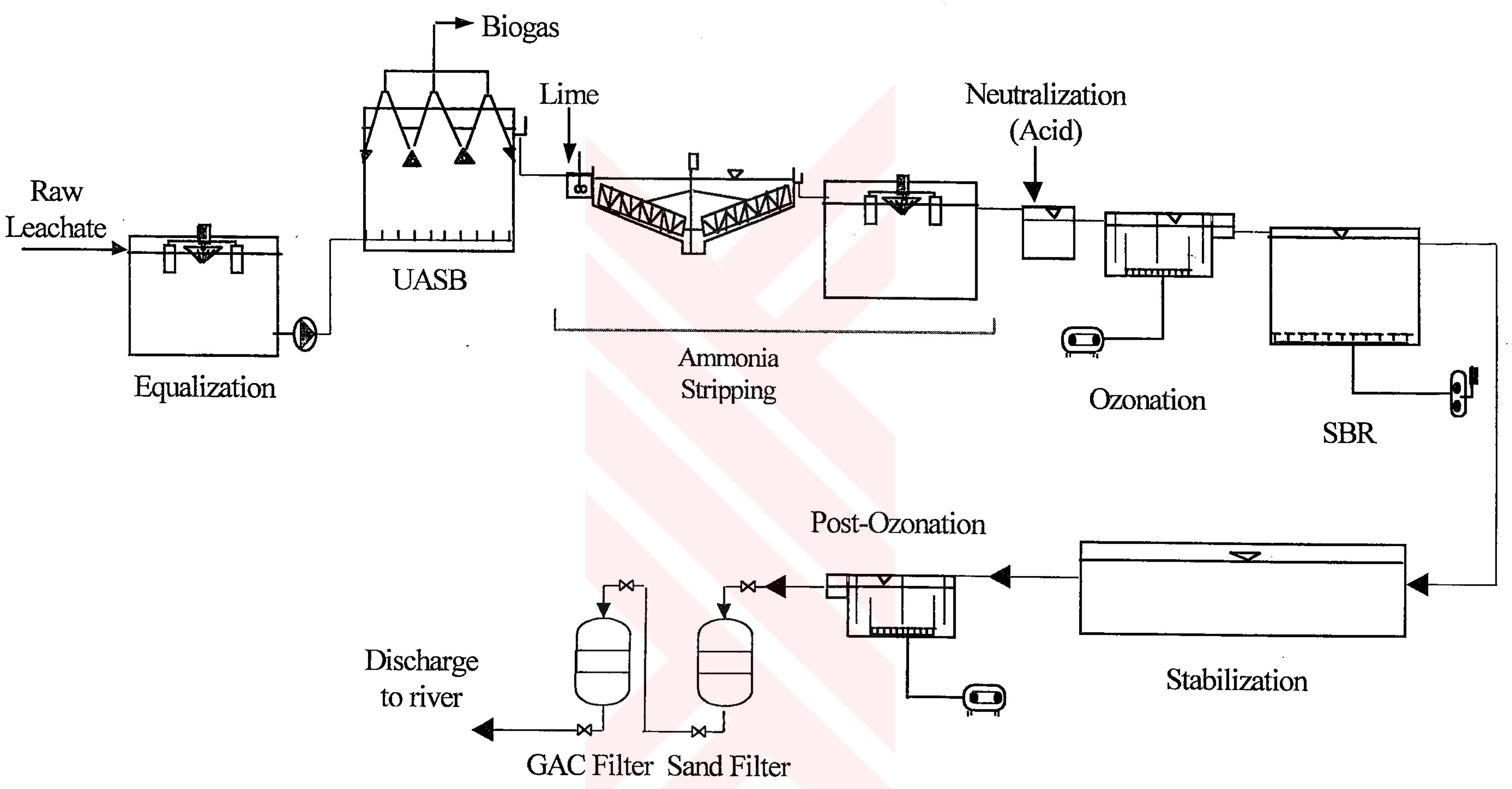


Figure 4.1. Flow Chart of the Leachate Treatment Plant in Kömürcüoda Sanitary Landfill.

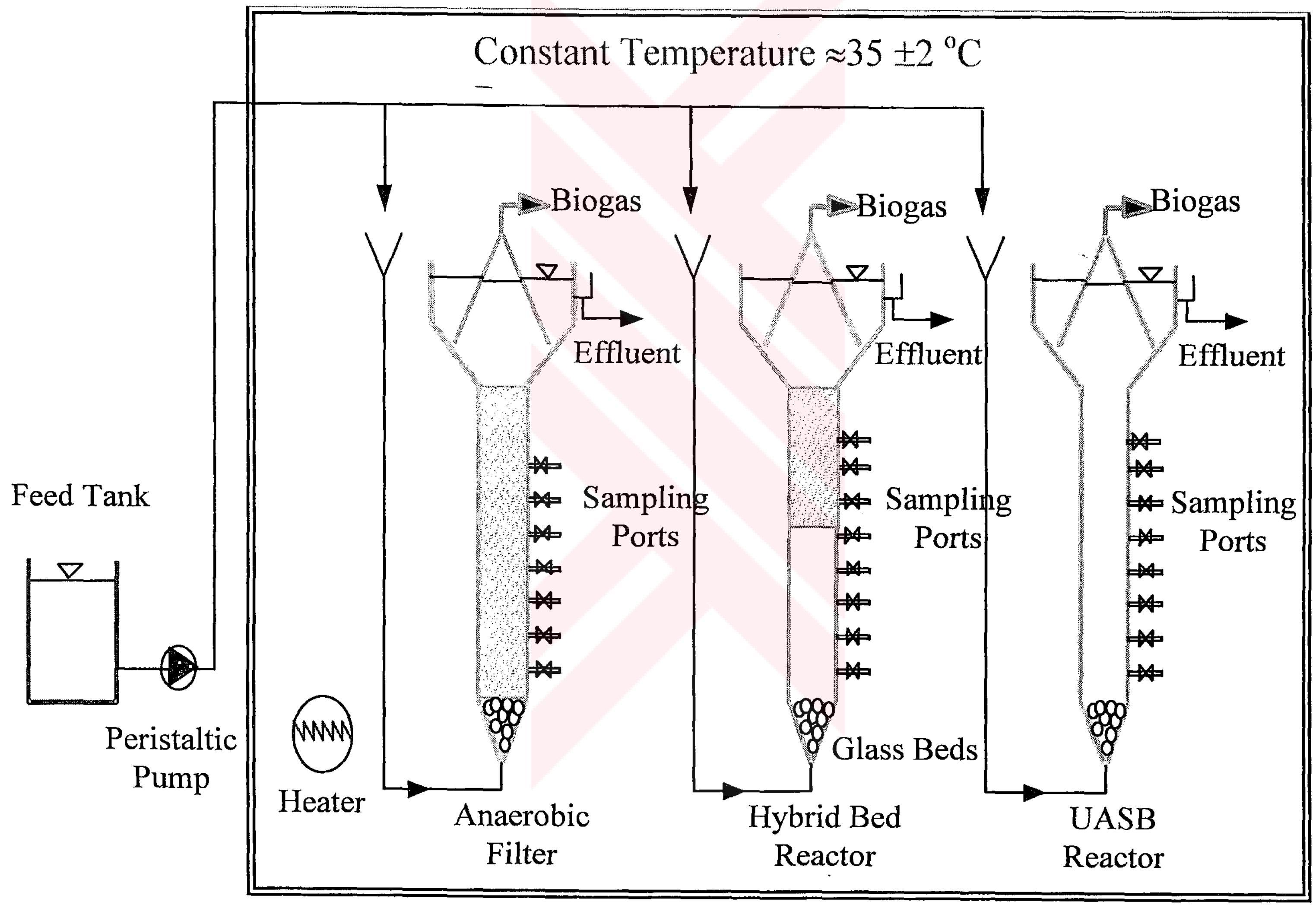


Figure 4.2. Anaerobic Treatability Set-up

Table 4.1 Operational Conditions and Physical Properties of the Reactors

	UASB	HYBRID	AF
Volume, L	7,8	7,8	7,8
HRT, days			
Day 1- Day 180	2.5	2.5	2.5
Day 180-Day300	2.0	2.0	2.0
Total Reactor Height, cm	115	115	115
Reaction Zone	100	40+60	100
Settling Zone	15	15	15
Temperature, °C	35±2	35±2	35±2
# of Sampling Ports	8	8	7

All reactors were made of PVC pipes with 10cm diameter and 100cm height. The influent was fed to the rectors by 2 peristaltic pumps from 9mm influent pipes and effluent was discharged from the same 9mm pipes. During the study, deposits of inorganic precipitates caused by high alkalinity in the reactors clogged these narrow pipes many times. So, it can be stated that, it would be better if wider influent and effluent structures and pipes were used. The bases of the reactors were filled with glass chips to support the biomass in the reactors. The effluent structures of the reactors were also made of PVC pipes but, to provide a settling in these zones, pipes with 15mm diameter were selected. The gas collection and outlet part in the reactors was consisted of a funnel placed about 5cm below the water levels in the settling zone. The collected gases were released out of the constant temperature room by pipes.

At the beginning of the study to provide the appropriate constant temperature in the reactors electrical blankets were used. But, because of the problems in the control of the temperature levels, the constant temperature with a thermostable heater was installed.

Filter media used in anaerobic filter and hybrid bed reactors were composed of helezonic rings cut from a plastic hose. The 40 cm filter media in the upper part of Hybrid Bed Reactor was supported by a screen located 60 cm above the bottom. The surface area of the media was 112 m^2/m^3 and void ratio was 87%. The density was determined as 0.1890 kg/L. All these properties of the filter media are given in Table 4.2.

Table 4.2. Physical Properties of Filter Media

	Anaerobic Filter	Hybrid
Surface Area (m^2/m^3)	112	112
Density (kg/L)	0.1890	0.1890
Void Ratio (%)	87	87
Height in reactor (cm)	90	40
Type	Helezonic Plastic Ring	Helezonic Plastic Ring

4.1.1 Start-Up and Operations of the Reactors

The anaerobic reactors were operated for more than 300 days in this study. At the beginning they were seeded with the sludge taken from an anaerobic treatment plant of a baker's yeast industry. All the reactors were started with an organic loading rate of 1.3 kg COD/m³.day. The raw leachate was diluted with dechlorinated tab water for adjusting the appropriate organic loading. Therefore, the parameters other than COD were also diluted during this start-up period. Influent COD concentrations were increased from about 3000 mg/L to about 25.000 mg/L with stepwise increase in organic loading up to 9.8 kg COD/m³.day. Higher loadings could not be applied due to dilution of COD in winter months. During this period of the study, leachate was fed to the reactors without dilution.

Because of phosphorus deficiency in raw leachate phosphorus in the form of diluted phosphoric acid was added to the influent to provide a 500/7/1 C/N/P ratio.

On the other hand, clogging was experienced in anaerobic filter in some duration of the experimental study due to deposition of inorganic precipitates and biomass growth. To overcome the clogging problem, the anaerobic filter was flushed with nitrogen gas from the bottom. No clogging was observed later on. In hybrid bed and UASB reactors no clogging problem was observed, except in feed and effluent pipes which are less than 1 cm in diameter.

4.1.2 Influent and Effluent Quality Monitoring

The performances of the reactors were routinely assessed by determination of pH, COD, alkalinity, NH_3-N and SS both in the influent and effluents. Analyses were carried out following the Standard Methods APHA, AWWA, 1989. The analyses, frequencies of them and analytical methods are listed in Table 4.3.

Table 4.3. Analyses carried out for the Assessment of the Reactor Performances

Parameter	Analysis Frequency	Method
pH	Every day	Electrode
Alkalinity	Every day	Titrimetric
COD	Every two days	Closed Reflux
NH_3-N	Every two days	Nesslerization
SS/VSS	Every week	Gravimetric

4.2 PERFORMANCES OF REACTORS

The evaluation of the reactor performances was based on COD removal efficiencies. pH and alkalinity analyses were used as emergency parameters for the performances of the reactors and ammonia was determined to state a relationship between the ammonia inhibitions and COD removal efficiencies. Suspended and volatile suspended solids in the effluents were observed to detect the type of the solids escaped from reactors.

4.2.1 Increasing Loading Rates and Changes in Removal Efficiencies

During the study, COD removal efficiency in the reactors has reached 90%'s. Especially, anaerobic filter reactor has achieved high COD removal efficiencies. But, five significant drops have occurred in COD removal, except the usual changes due to the increases in organic loading rate. In the first step start-up, at 1.3 kg COD/m^3.day organic loading rate, anaerobic filter reactor reached the highest COD removal efficiency of approximately 80%. The lowest removal efficiency has been determined in hybrid reactor. But the COD removal efficiency in this reactor reached to the efficiencies in the other reactors in the following days with the same organic loading rate. After 50 days, organic loading rate has been increased to 3.5 kg/m^3.day. Although anaerobic filter and UASB reactors have not given significant response to this increase, a reduction in the COD removal efficiency of hybrid reactor has been observed. But, it is not possible to conclude that this reduction is only because of increasing organic loading rate. So, the other concentration and temperature have been investigated in the mentioned days to better understand the reason. The discussions of these investigations will be presented later on the report. In the following months, organic loading rate has been risen to 9.8 kg COD/m^3.day step by step in all reactors. In Figure 4.3 COD concentrations in the influent and effluents of the reactors were given for more than 300 days. At the beginning of the start-up, the raw leachate was diluted with dechlorinated tap water for adjusting the appropriate organic loading. Influent COD concentrations were increased from about 3000 mg/L to about 25000 mg/L with a stepwise increase in organic loading. Higher loadings could not be applied due to the dilution of COD in winter although the leachate was fed to the reactors without any dilution.

In Figure 4.4 NH_3-N in the influent and effluents were presented. As in the COD graph the same stepwise increase in ammonia concentration of the influent and effluents of reactors can be seen.

In Figure 4.5 and 4.6, pH and alkalinity in the influent and effluents of reactors have been plotted. There were fluctuations in the influent pH and alkalinity. In the winter months a considerable increase in alkalinity, in the influent and as a result in the effluents has been monitored. This is caused by direct feed of raw leachate to the reactors without any dilution. By direct feed, the ammonia concentrations in the influent were also increased and resulted in higher pH values in the reactors. pH values in reactors decreased gradually below 8.0 following the pH control in the influent after day 175. For this decrease in the pH of effluents the influent pH was adjusted to 4.5 by using concentrated hydrochloric acid.

In Figure 4.7 the temperatures of the reactors were plotted. In decrease in temperature until day 120, beginning of winter, was not dealt with electrical blankets used for heating, so a constant temperature room mentioned before was installed. Sudden changes in temperature in this uncontrolled period have resulted in some decreases in COD removal efficiencies.

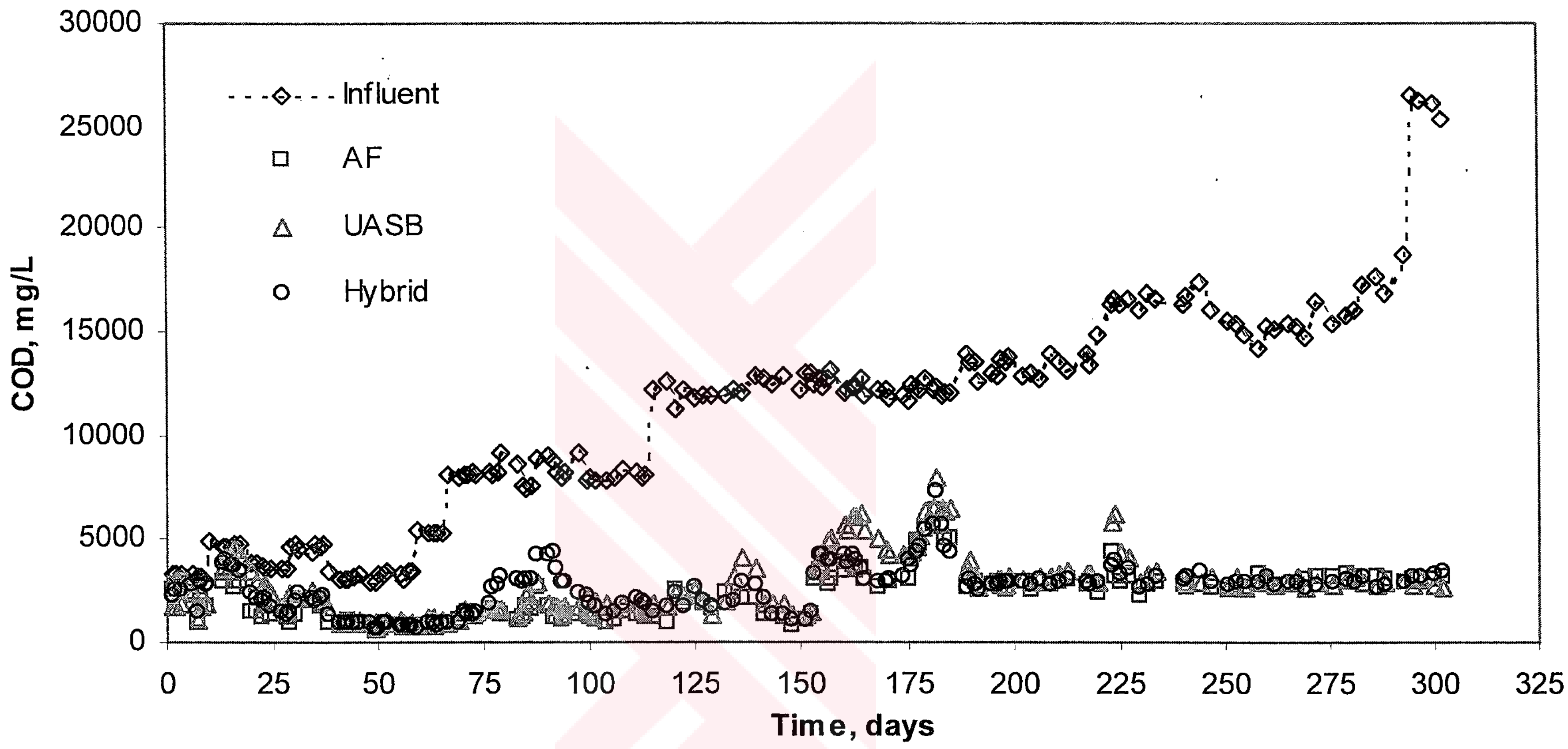


Figure 4.3. COD Concentrations in the Influent and Effluents of Reactors

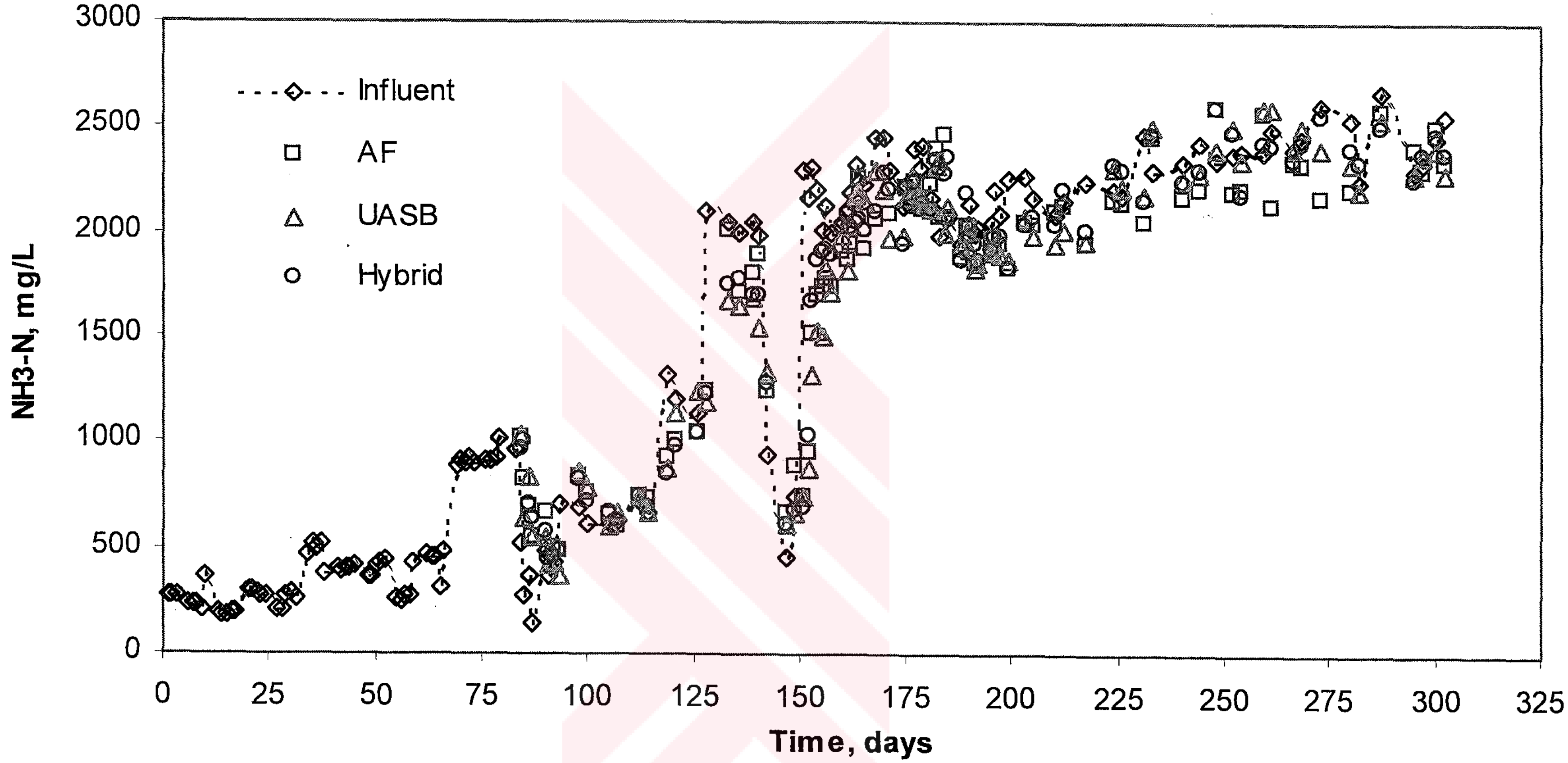


Figure 4.4. NH_3-N Concentrations in the Influent and Effluents of Reactors

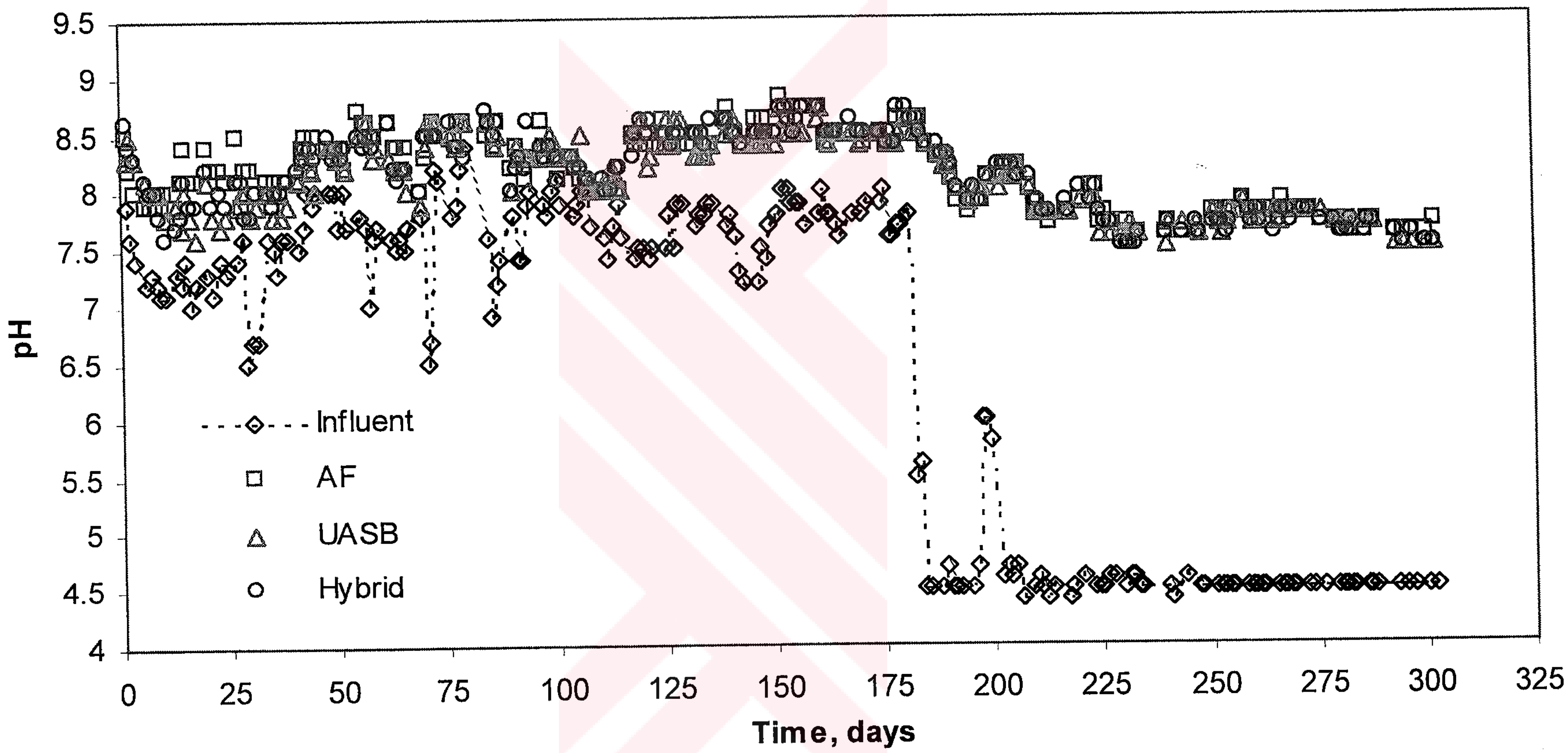


Figure 4.5. Change of pH throughout the experimental period

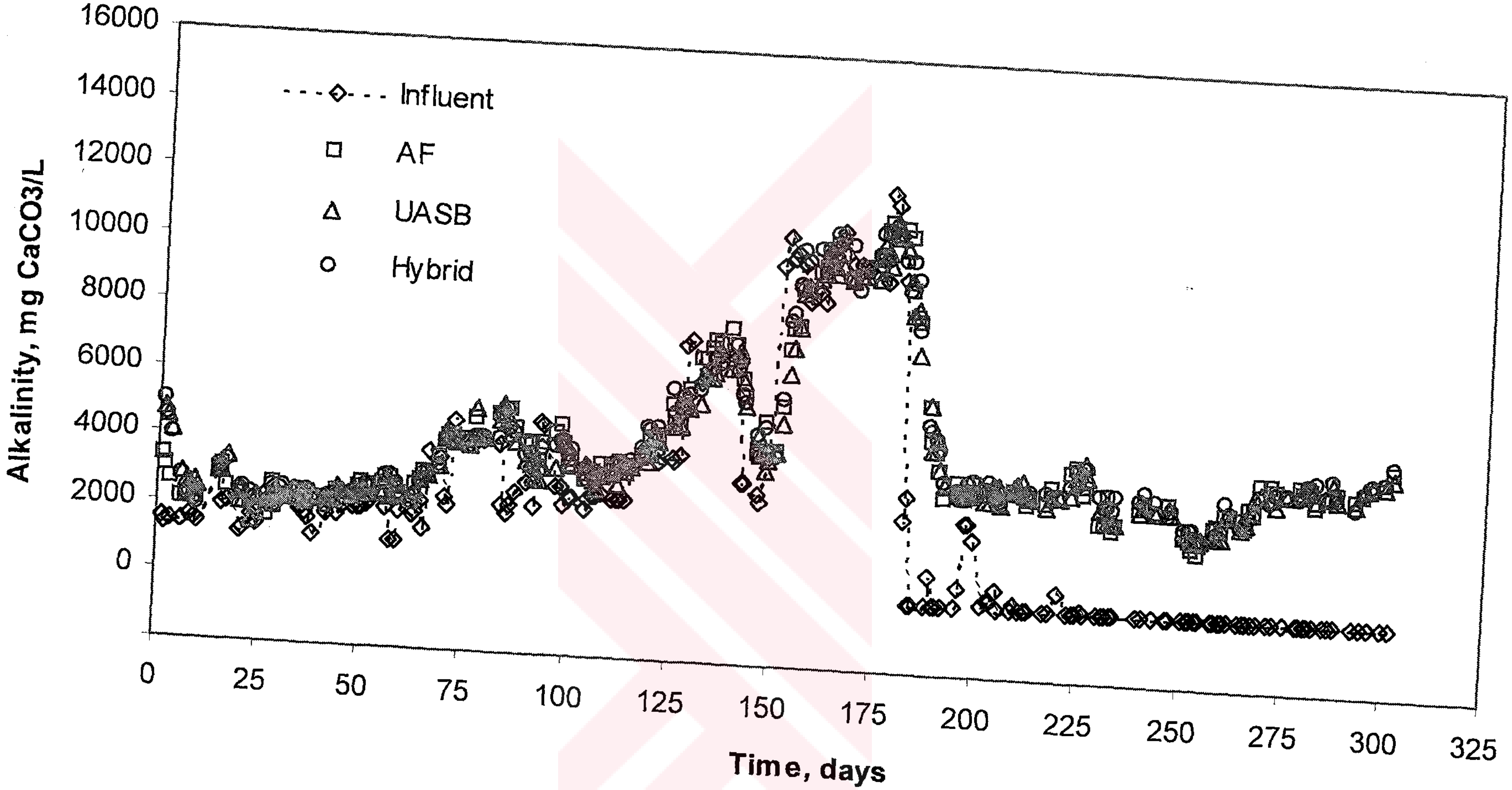


Figure 4.6. Change of Alkalinity throughout the experimental period

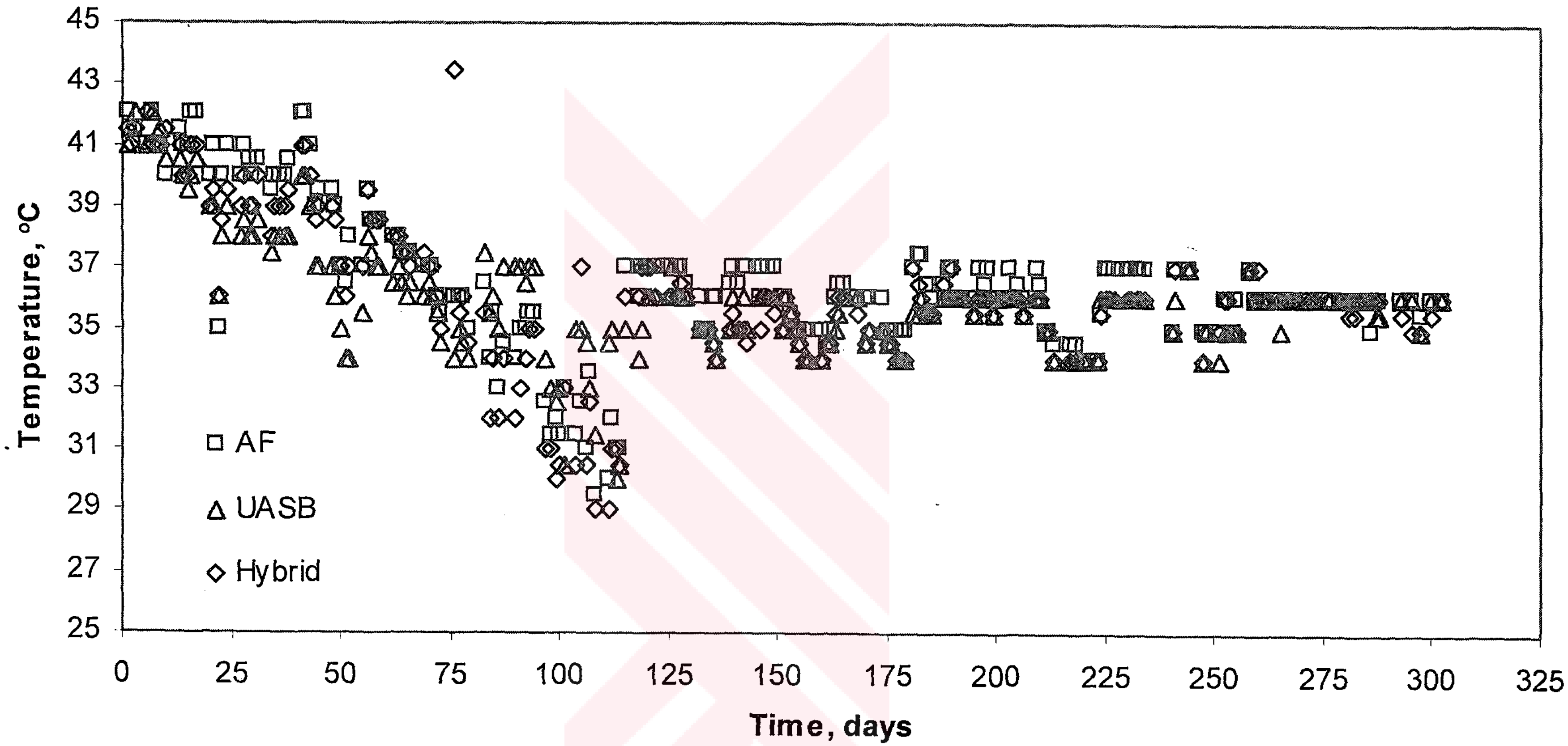


Figure 4.7. Change of Temperature throughout the experimental

4.2.2 Ammonia Inhibition

All the reactors showed similar performances against organic loadings with efficiencies between 80% and 90%. However the reactors have experienced high ammonia concentrations several times throughout the experimental period and showed different inhibition levels. The first ammonia inhibition has occurred around between Day 75 and Day 100. Other severe ammonia inhibitions were between Day 130 and Day 145, Day 150 and Day 170, Day 175 and Day 190 and Day 210 and day 240. No pH control was done until day 190 where ammonia inhibition became more severe. Since unionized ammonia is toxic to methane producing bacteria, total ammonia concentration is not a good parameter for understanding the ammonia inhibition. Therefore, unionized ammonia concentrations in the influent and effluents have been calculated according to the following formula and plotted as can be seen in Figure 4.9. pH values in the reactors were always above 8.0 and exceeded 8.5 in some duration of the experiments.

$$UAN = \frac{NH_3 - N}{1 + 10^{(pKa - pH)}}$$

UAN : unionized ammonia nitrogen, mg/L

pKa : dissociation constant for ammonium ion (8.95 at 35 °C)

The ammonia inhibition was reversible, as it can be seen from rapid increases in COD removal efficiencies of reactors, immediately after drop of unionized ammonia concentrations (Figure 4.8). Attached growth systems namely, anaerobic filter and hybrid bed reactor were more resistant to ammonia inhibition than UASB reactor. This was clearly seen in all cases of ammonia inhibition. Anaerobic filter was the least effected reactor from ammonia inhibition while UASB was the most. Hybrid bed reactor was exhibited a similar performance to anaerobic filter although not at the same degree. pH of the influent was decreased to 4.5 with concentrated HCl after day 175 to control ammonia inhibition. pH values of reactors decreased gradually below 8.0 following the pH control in the influent. Unionized ammonia nitrogen (UAN) values also decreased from about 800 mg/L to 200 mg/L in parallel to pH values. COD removal efficiency in all the reactors have restored back to 80% level soon after the drop in unionized ammonia levels.

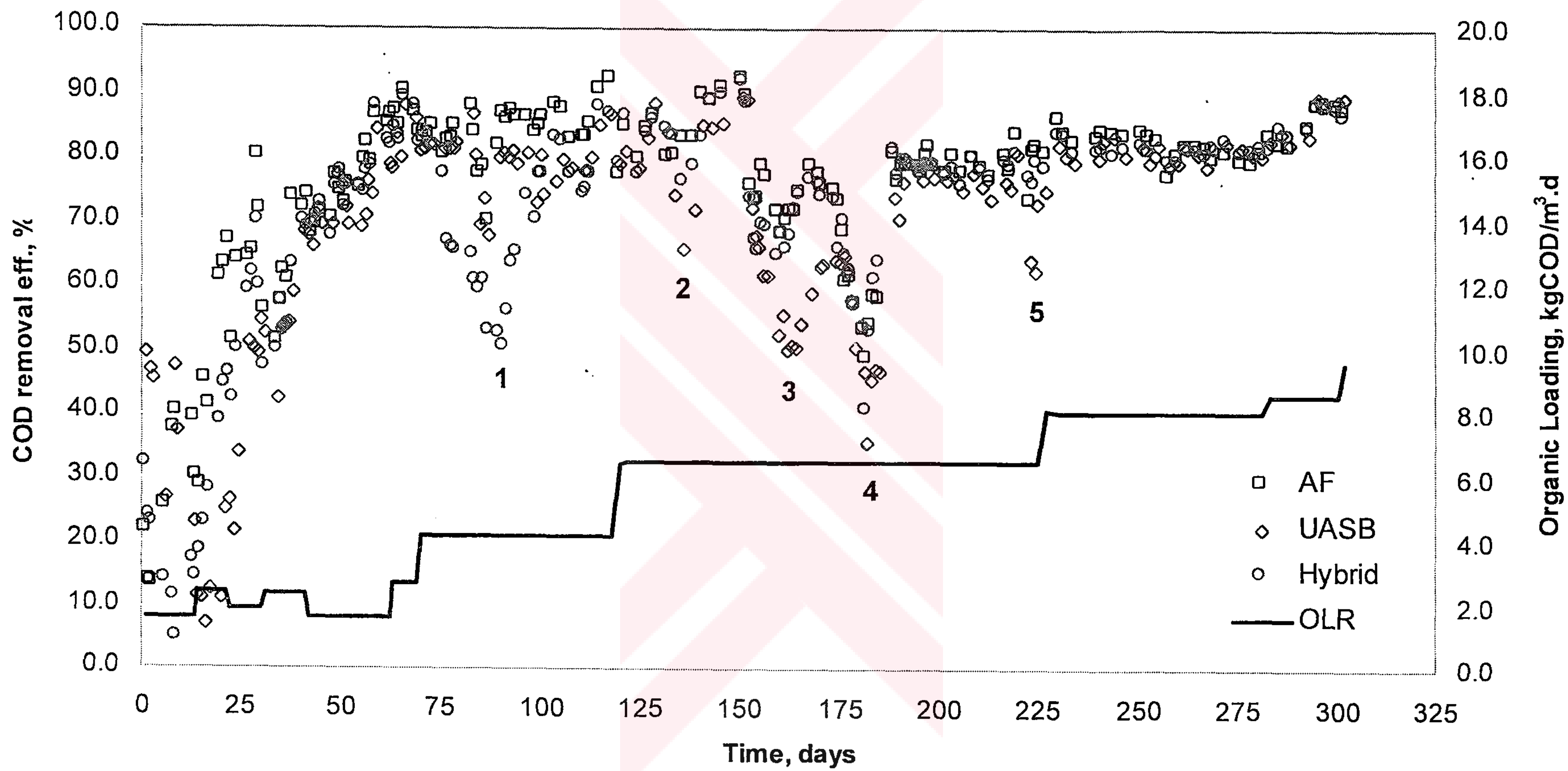


Figure 4.8. COD Removal Efficiencies versus Organic Loading Rate

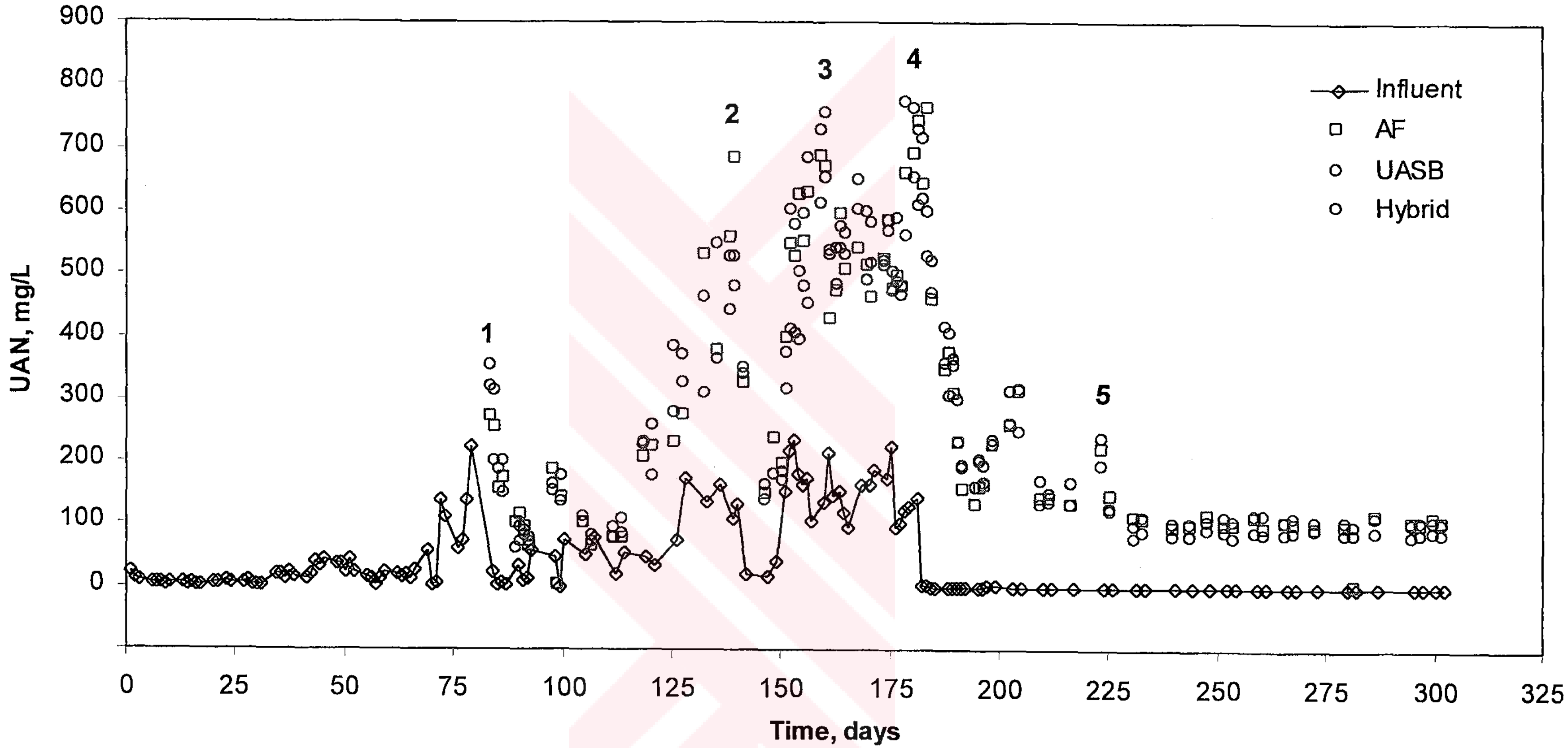
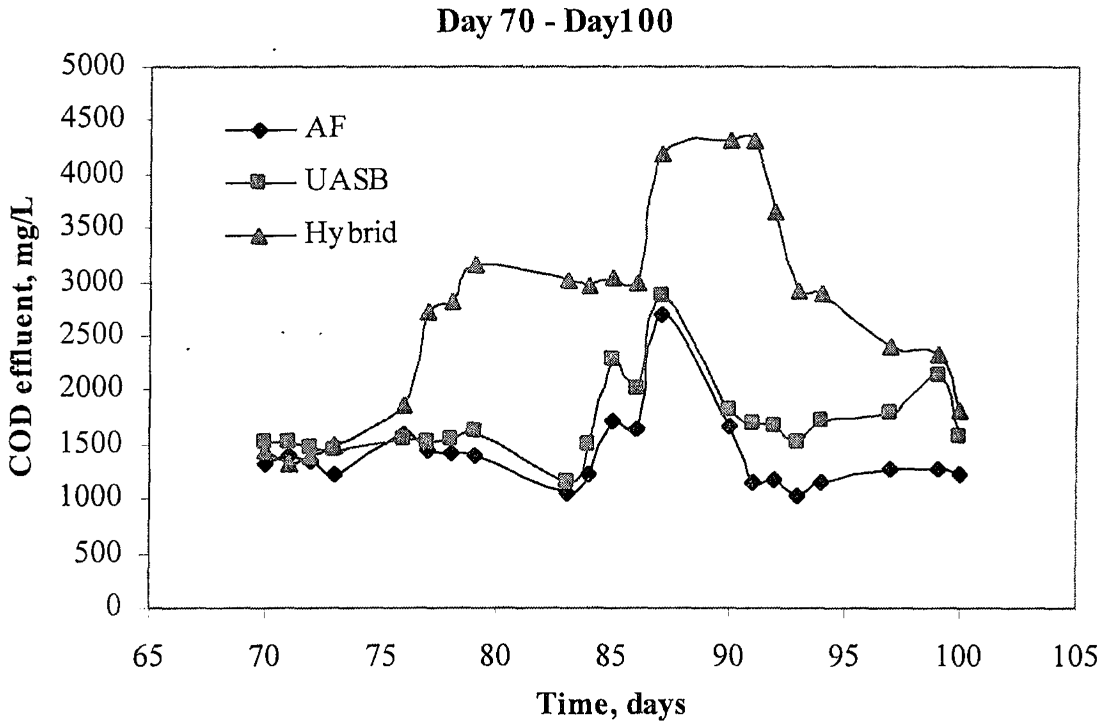


Figure 4.9. Unionized Ammonia Nitrogen Concentrations throughout the Experimental Period



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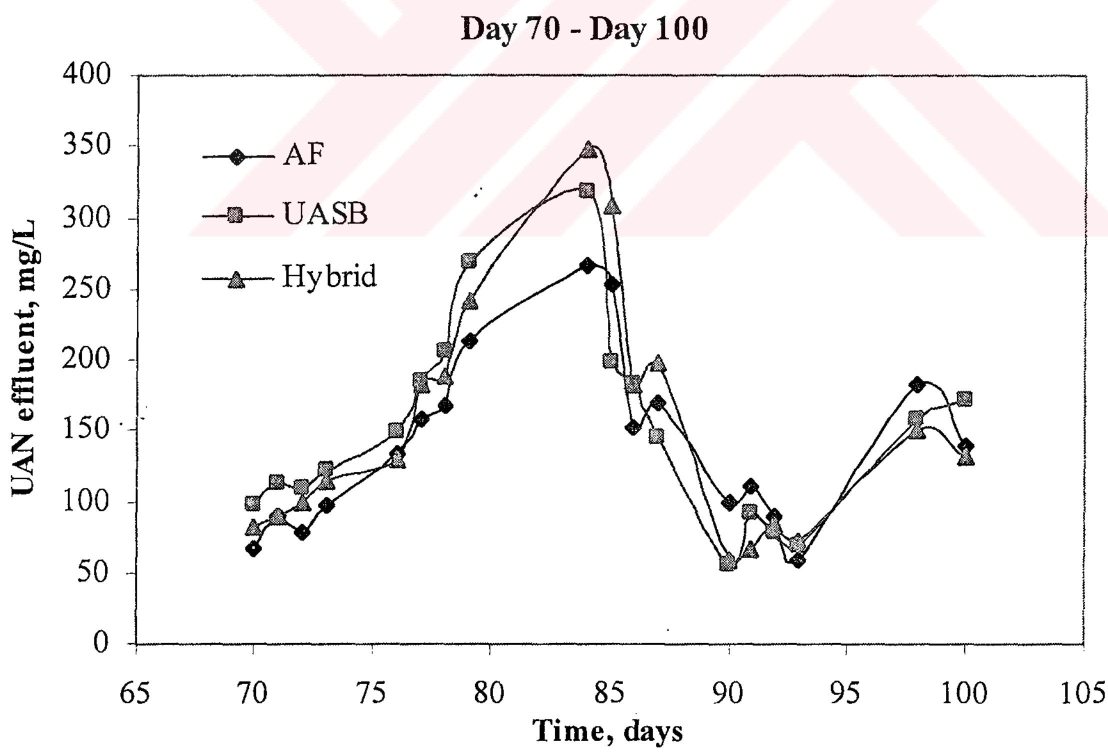

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Figure 4.10 Temperature and Ammonia Inhibition between Day 70 and Day 100

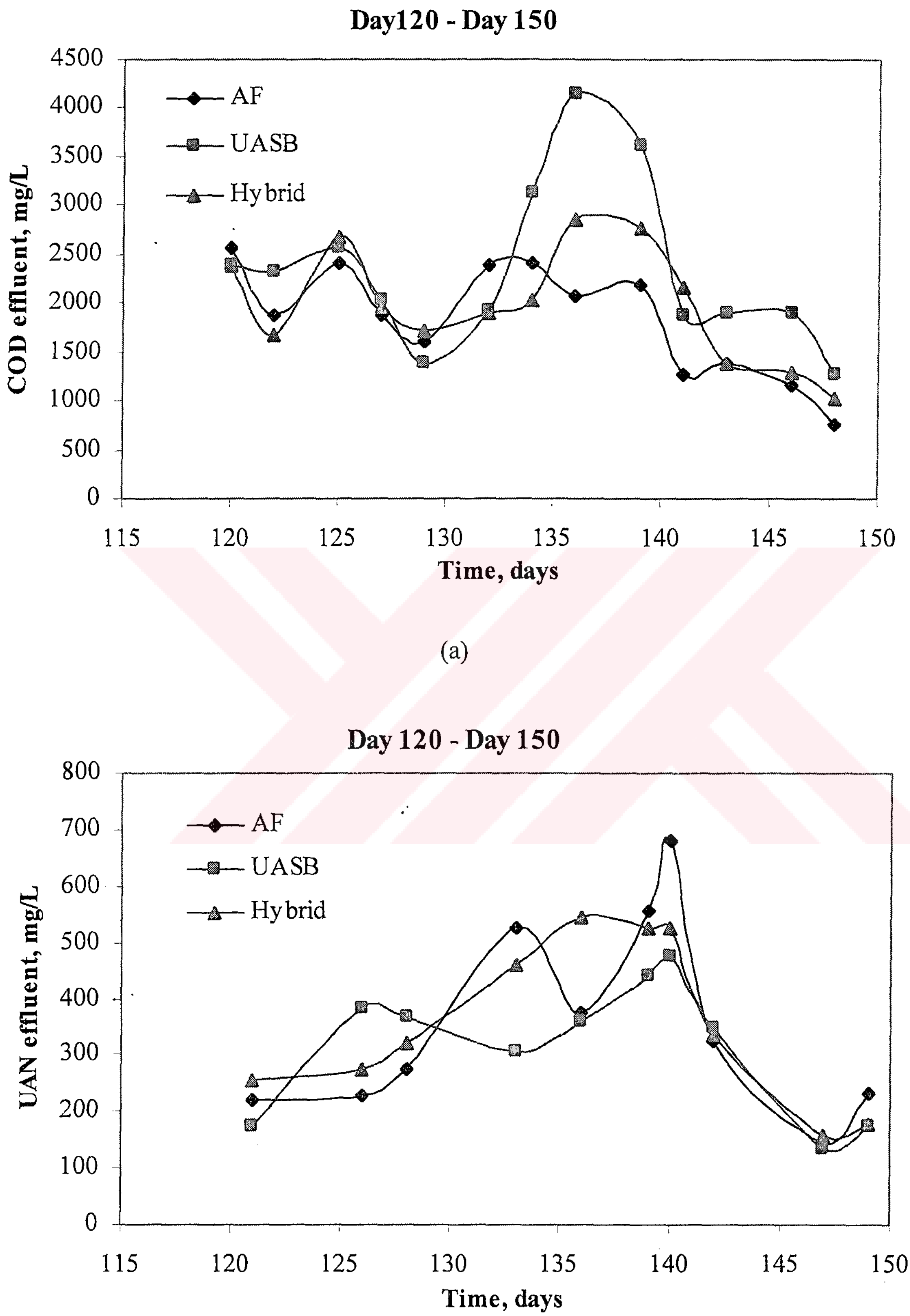

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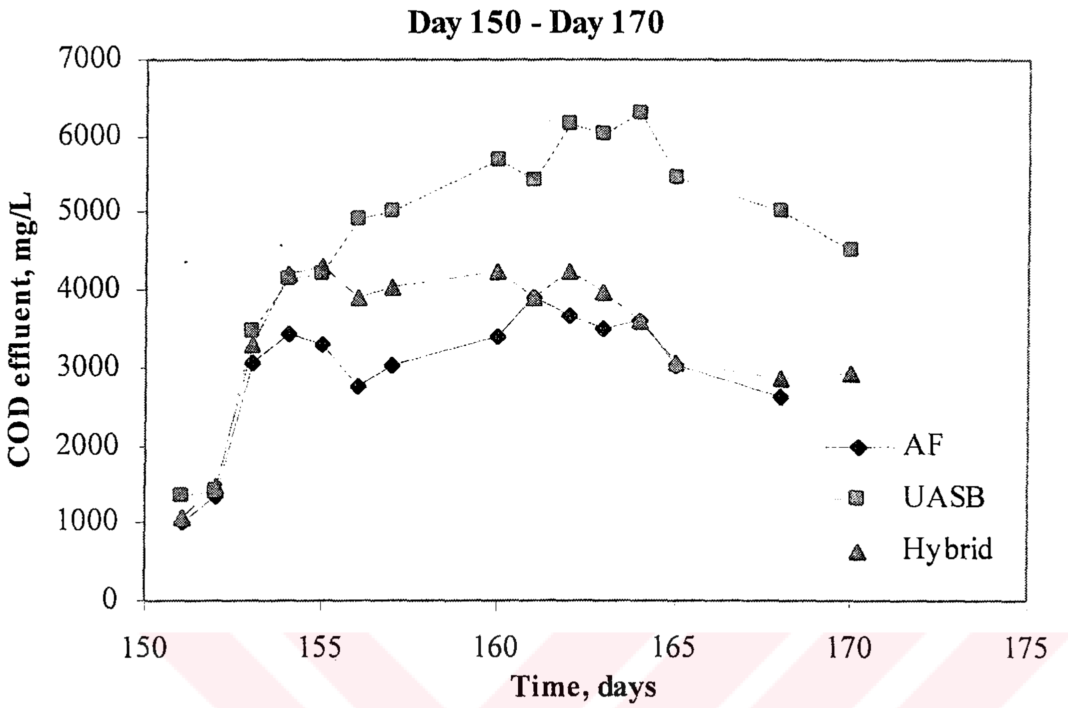
Figure 4.11 Ammonia Inhibition between Day 120 and Day 150



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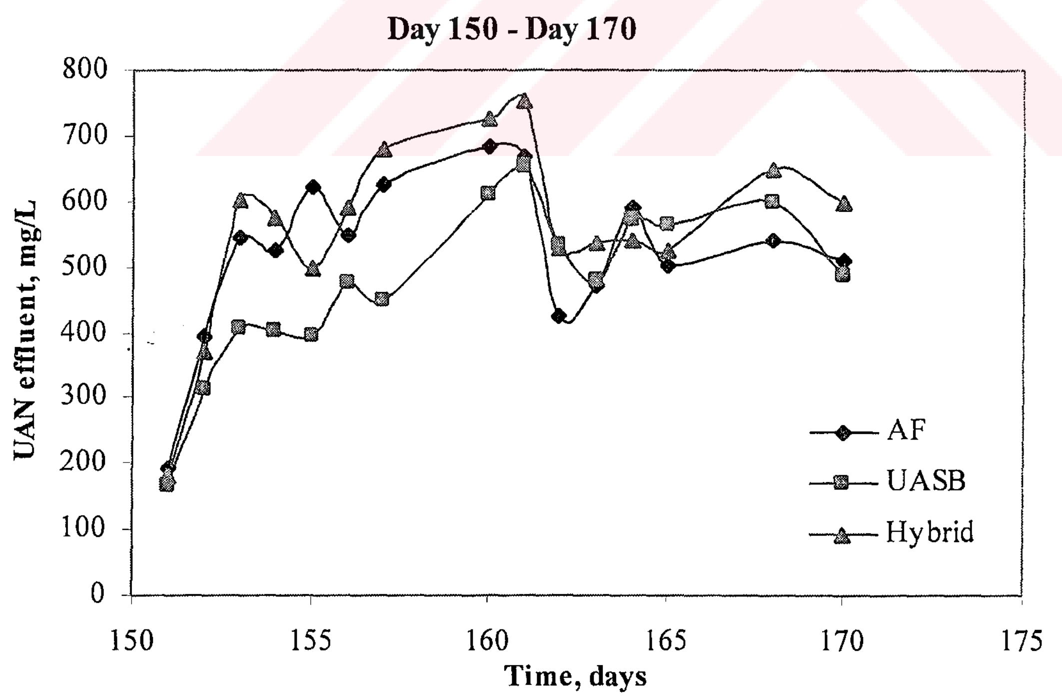

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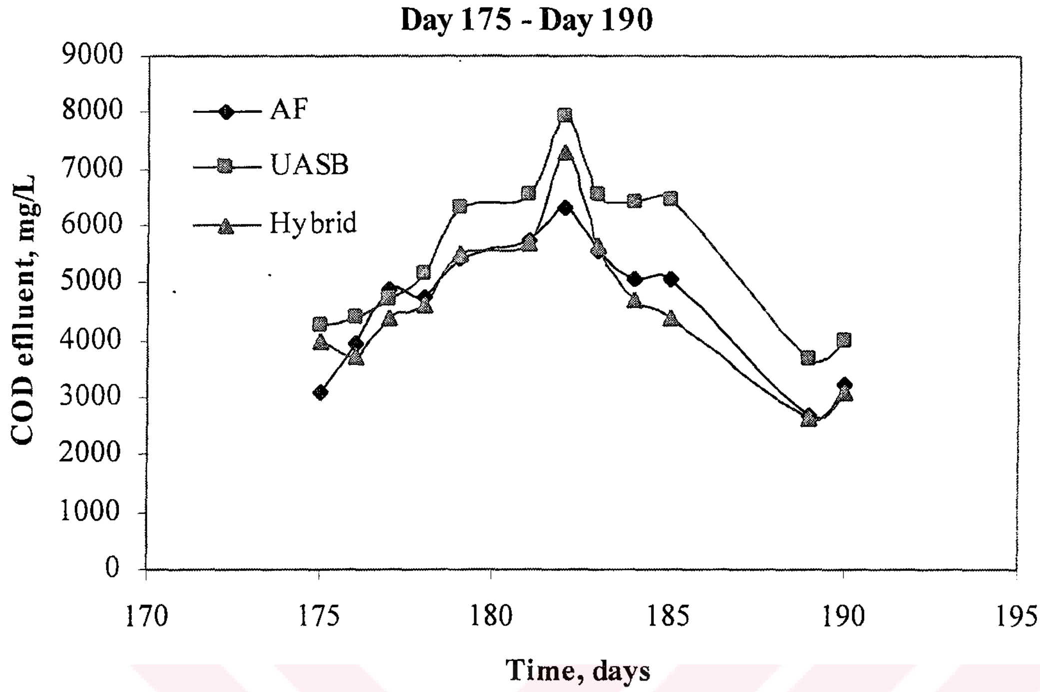
Figure 4.12 Ammonia Inhibition between Day 150 and Day 170



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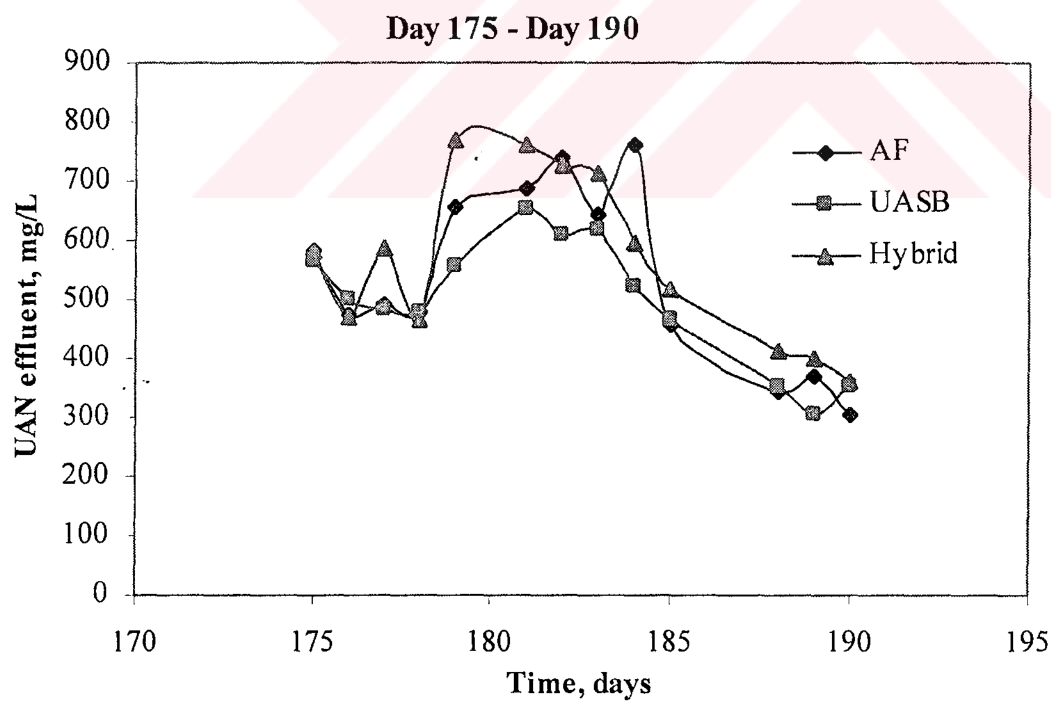

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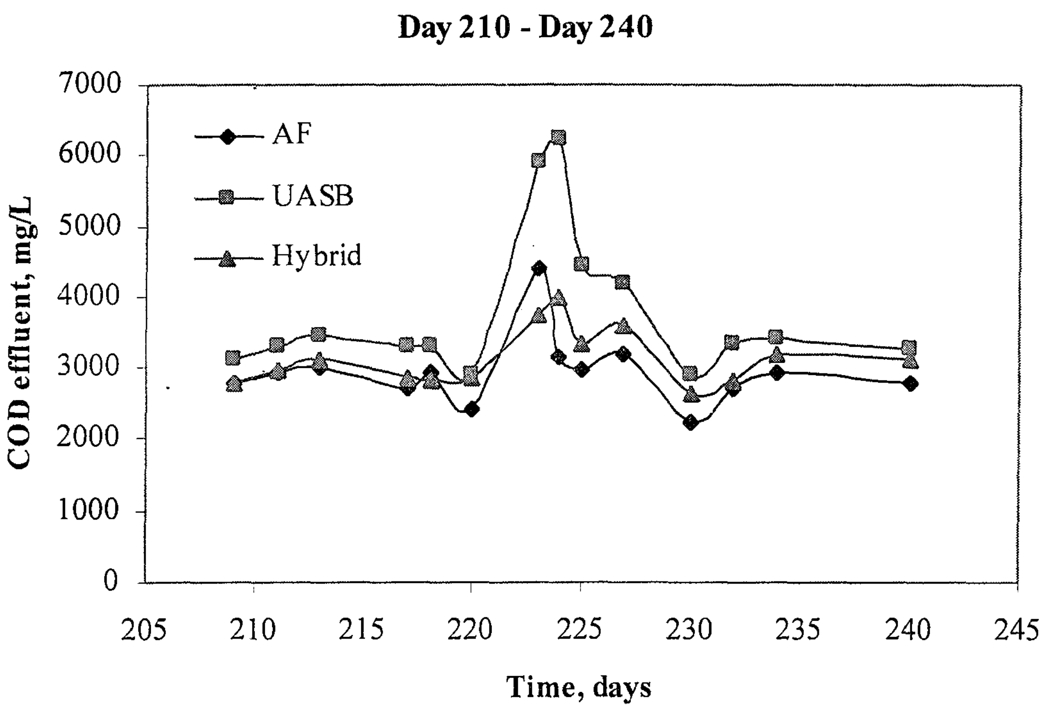
Figure 4.13 Ammonia Inhibition between Day 175 and Day 190



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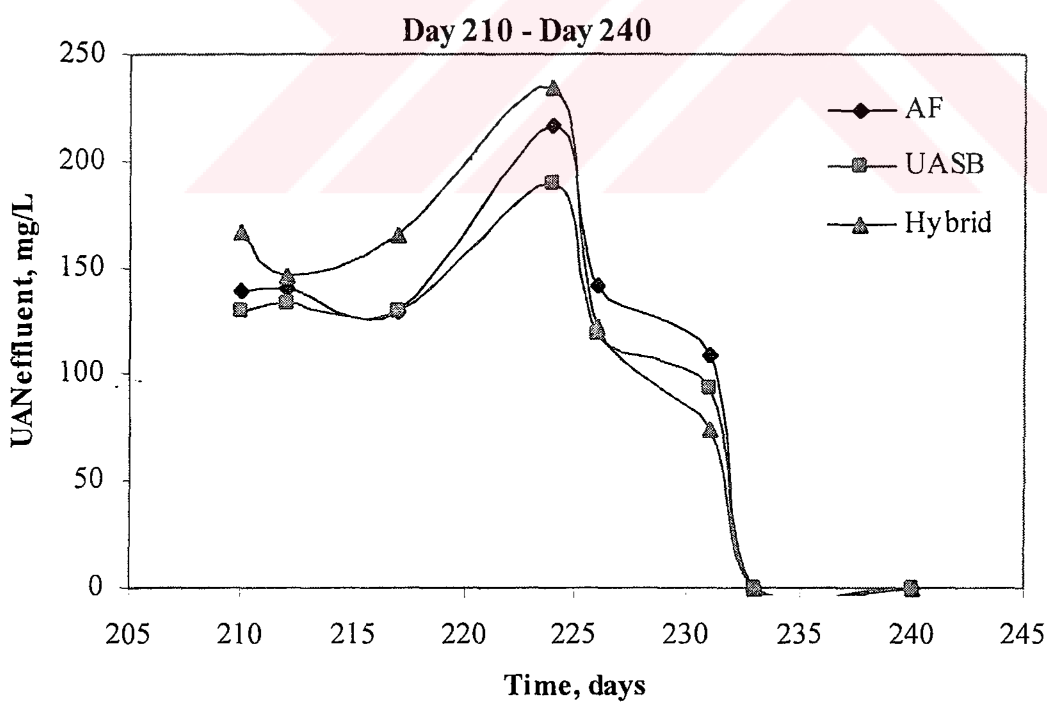

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Figure 4.14 Ammonia Inhibition between Day 210 and Day 240

To have more explanatory informations about the effects of ammonia inhibition on each reactor, more detailed graphs are plotted between days 70 and 100, 120 and 150, 150 and 170, 175 and 190, 210 and 240. These graphs are presented in Figure 4.10, 4.11, 4.12, 4.13 and Figure 4.14 respectively.

In Figure 4.10, the highest inhibition on COD removal is seen on Hybrid Bed reactor. Since on the other graphs, all the highest inhibitions are observed on UASB reactor, this high inhibition on Hybrid Bed reactor was because of the temperature problem at the electrical blanket of Hybrid Bed reactor at that period.

Especially, unionized ammonia nitrogen concentrations in the effluents of the reactors were at same levels as seen from the graphs. But, because of ammonia adsorption in the suspended growth biomass of UASB reactor, unionized ammonia nitrogen concentration in the effluent is a little bit lower in UASB reactor.

According to the detailed graphs of ammonia inhibition, it can be stated that, attached growth processes, as Anaerobic Filter and Hybrid Bed reactors are more resistant to ammonia inhibition.

4.3 DETERMINATION OF AMMONIA INHIBITION

4.3.1 Biogas Production Rate Test

Rates of biogas production, and more specifically the methane yield, can potentially be a good indicator of the metabolic status of anaerobic reactors. Lowering of methane production rates, when compared to the influent rate of organic matter gives warning of the accumulation of volatile fatty acids in the liquid phase. Unfortunately this is the result of an imbalance caused by the inhibition of the methanogenic bacteria.

By monitoring the biogas production, inhibitions in anaerobic reactors may be detected. A quantitative measurement is adequate to determine the levels of inhibition.

Monitoring and Measurements of Biogas

For biogas production rate tests, 100mL vial bottles are used as reactors. The biogas produced in these batch type reactors is transferred to the manometers by a fine transfer pipe connected to the tip of the bottle. An individual biogas measurement unit is given in Figure 4.15. In the study, 2 biogas measurement apparatuses composed of 33 interconnected units were used as shown in Figure 4.16. The manometer of the set-up is filled with salty water and this solution is supplied to the manometer tubes with a plastic hose connected to the holding containers. The pH of the salty water solution is reduced to 2 to prevent the dissolution of CO_2 present in the biogas. With this precaution, the biogas produced in the reactor is measured exactly. By reading the height of the displaced water, biogas is measured as height and multiplying it by cross sectional area of manometer tubes, volume is determined.

Preparation of Dilution Solution

The biogas production rate experiment begins with the preparation of dilution solution. Dilution solution is the mixture that represents content of the reactor in methanogenic phase.

Table 4.4. Chemical Content of Dilution Solution (Valcke and Verstraete, 1983)

Chemical	Amount
KH_2PO_4	2500 mg/L
$K_2HPO_4.3H_2O$	1000 mg/L
NH_4Cl	1000 mg/L
$CaCl_2.2H_2O$	150 mg/L
$Na_2S.7H_2O$	100 mg/L
Yeast Extract	200 mg/l

To calculate the volume of the dilution solution required for the test first of all the number of the cases that will be tried in the test is decided. Each case is studied triple for more accurate results and each vial bottle reactor is filled with 50mL dilution solution. So, the total volume is calculated as multiplying the number of cases, 50mL vial bottle content and number of reactor (3) per case. A spare volume should also be taken into account.

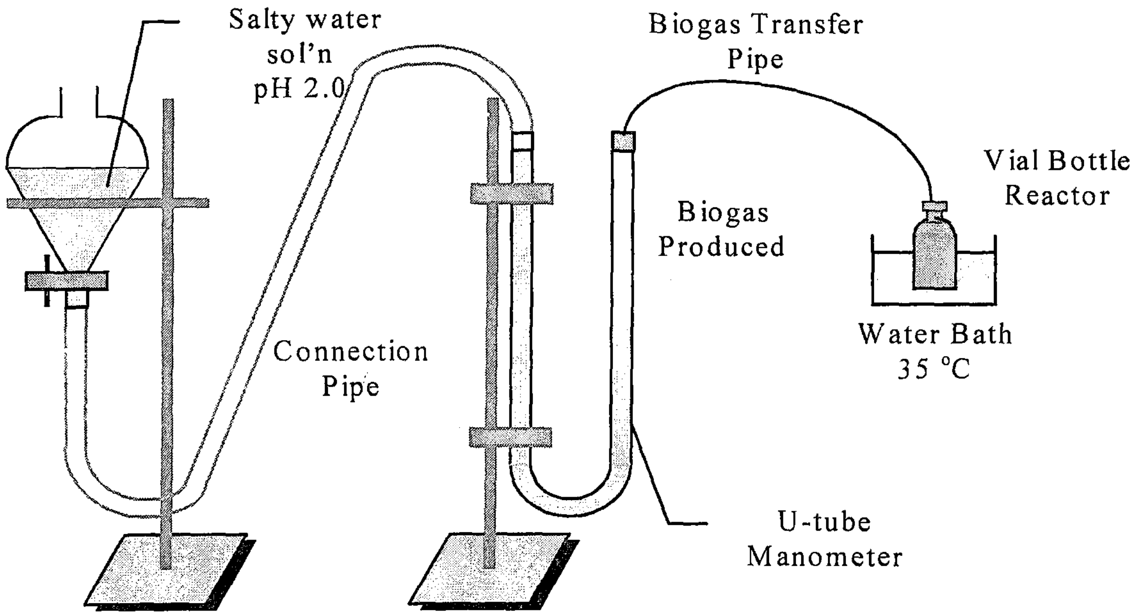


Figure 4.15. An individual Biogas Measurement Unit

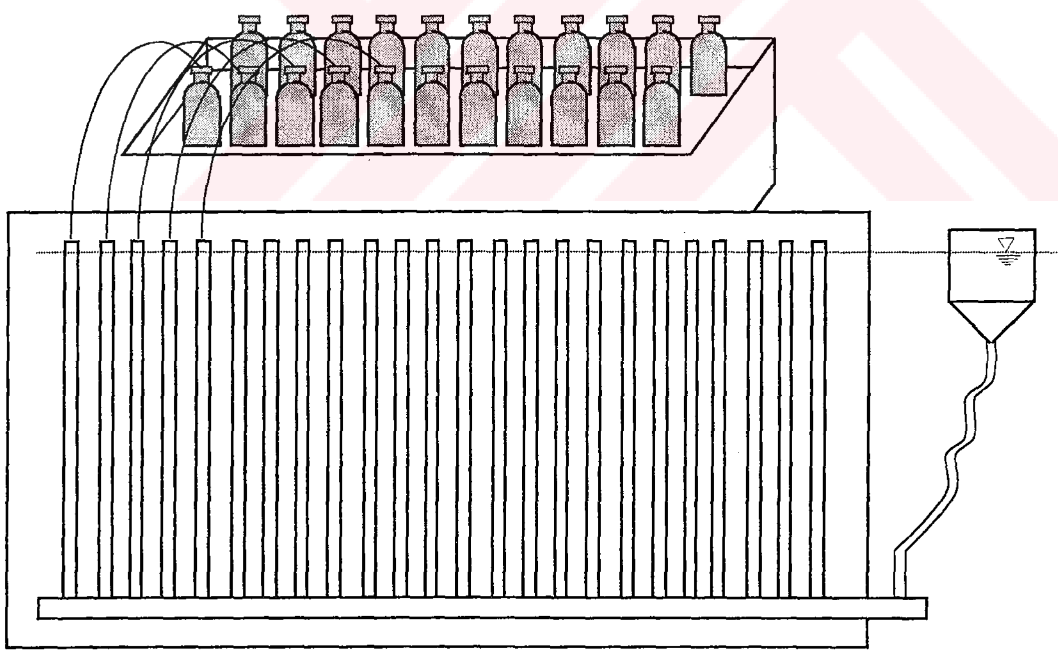


Figure 4.16 Multiple Manometer System

After determination of the volume of dilution solution, the chemicals, listed in Table 4.4, are dissolved in distilled water in a flask. Oxygen in this flask is removed under nitrogen purging. Then, the anaerobic feed sludge taken from the anaerobic reactors was added to the flask in adequate amounts to provide 2000 mg/L VSS in the dilution solution As in all transfers this transfer is also carried out under nitrogen gas. After these additions the dilution solution is incubated overnight in a 35 °C incubator. The flowchart of the dilution solution preparation is presented in Figure 4.17.

In biogas production test, for by-passing the acidogenic phase, acetic acid is used as substrate. After overnight incubation, stock acetic acid is added and 1000mg/L acetic acid concentration was provided in dilution solution. To satisfy a neutral pH in the solution acetic acid was added after being neutralized. This pH range should be satisfied for proper growth of methanogens.

Transfer of Dilution Solution to Vial Bottle Reactors

Before being filled, vial bottle reactors are washed with nitrogen gas to provide anaerobic conditions in. The transfer of dilution solution and sludge mixture to the vial bottle reactors is carried out with a partially automated transfer system, under nitrogen purging. The transfer system is composed of a programmable peristaltic pump, a magnetic stirrer and nitrogen supply. Peristaltic pump is used to deliver 50 mL dilution solution from the solution flask to each reactor. To distribute the dilution solution homogeneously to each reactor, the solution flask is agitated with a magnetic stirrer. The nitrogen supply is used to prevent the oxygen contamination during the transfer by purging. This transfer set-up is presented in Figure 4.1.

Transfer of Dilution Solution +Anaerobic Mixture to Vial Bottle Reactors

After the transfer of mixture, vial bottle reactors are plugged up with butyl rubber caps and aluminium seals are crimped onto them. Prior to labelling, for different ammonia concentrations and pH levels different amounts of ammonia and NaOH solutions are added into the vial bottles through the butyl rubber caps with a syringe. Then, the vial bottle reactors are placed into the 35 ± 2 °C water bath. To check the pH levels &NH_3-N concentrations, in each case 1 extra vial bottle reactor is opened.

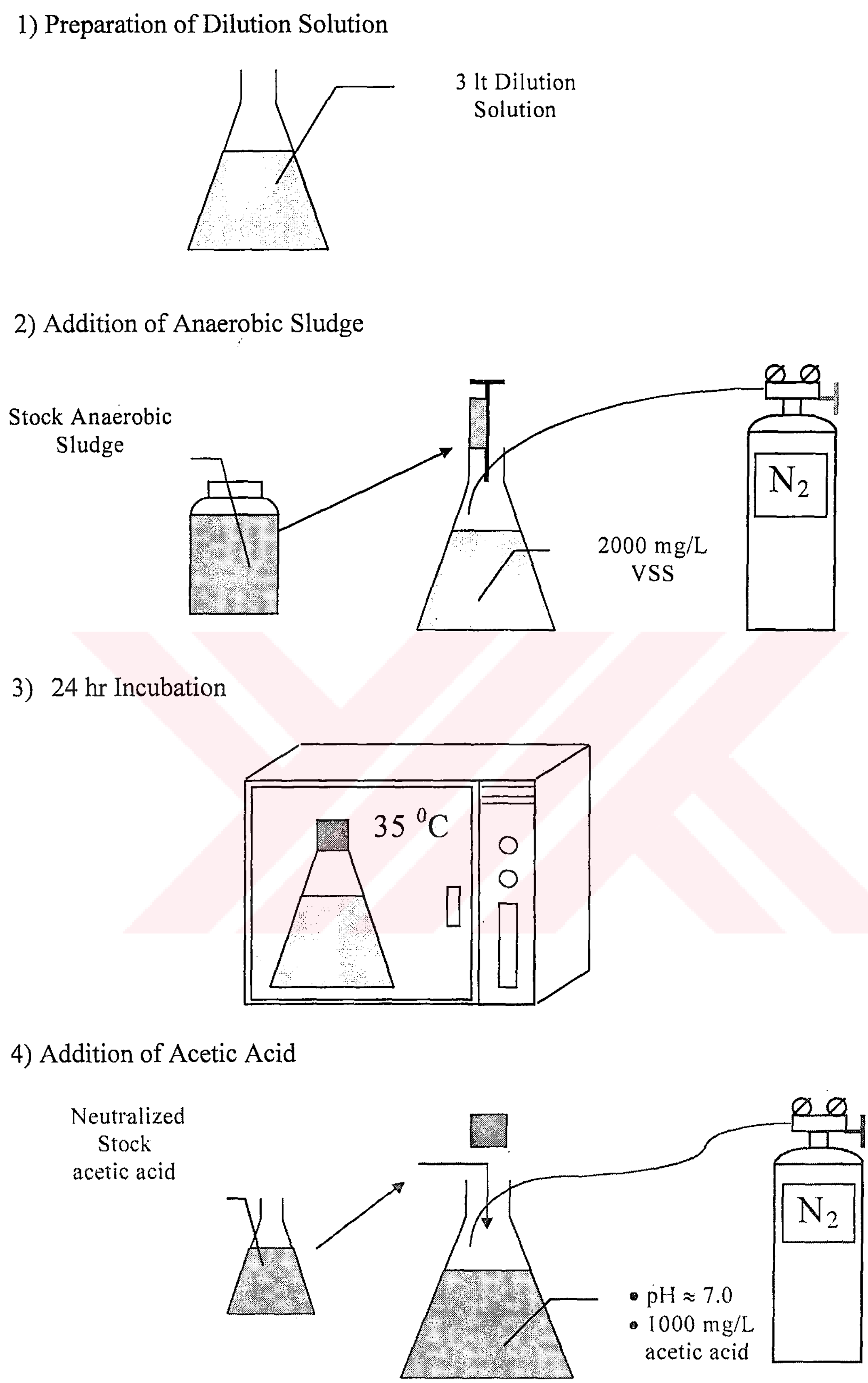


Figure 4.17 Preparation of Dilution Solution & Anaerobic Sludge Mixture

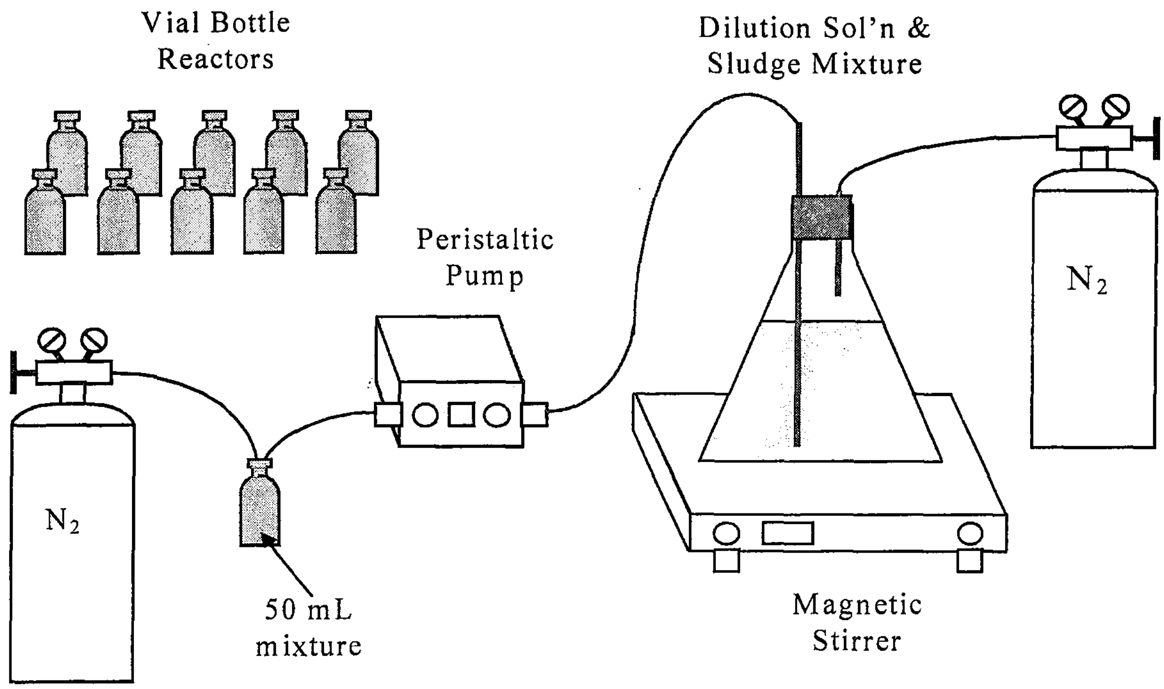


Figure 4.18 Automatic Transfer System

Collection of Data

The pressure in reactors is released after 30 min incubation in water bath to provide a better starting point with identical pressure in each of the reactor. After zeroing the pressure in the reactors, they are left to 35 ±2 °C water bath for incubation again.

Biogas production, in the vial bottle reactors is measured as the height of water displacement in the column of biogas measurement apparatus. The height of the water displacement may be converted to volume of the biogas by multiplying it with the cross sectional area of the manometer tube.

The measurements are carried twice a day, in the morning and in the afternoon. The data are noted on the data sheets. Generally, biogas production in the vial bottles end after 120-140 hours.

At the end of biogas measurement test, the inhibition percentages were calculated with the following formula:

Inhibition, % = (1-(Sample/Blank)) x 100

Sample = Biogas production in the reactors as water displacement (mL) that leachate influent is added

Blank = Biogas production in the blank reactors as water displacement (mL) the Blank reactors. (No influent is added)

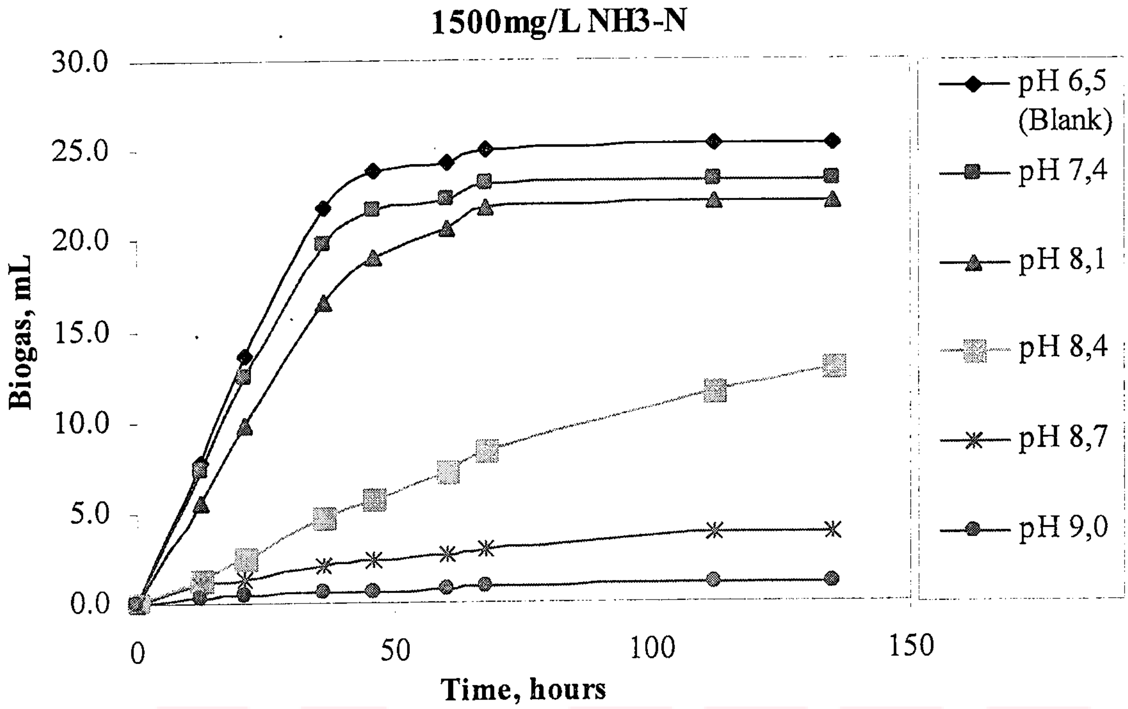
To determine the inhibitory pH levels and NH_3-N concentrations in the reactors, ammonia concentration of 1500, 2000, 2500 and 3000 mg/L and pH of 7.4, 8.1, 8.4 and 9.0 have been investigated in 16 different cases with the biogas production rate tests. Inhibition rates were determined by comparing the results of each case with the blank reactors. To control the pH levels and ammonia concentrations, before and after the test, one control reactor from each case was analysed.

The first ammonia concentration tested for inhibition was 1500mg/L. At this concentration, the inhibitory pH level was observed at 8.1. Above this pH, biogas production decreased sharply and inhibition reach to 75 %'s (Figure 4.19).

When NH_3-N concentration is increased to 2000mg/l, inhibition was determined as 35 % at pH 8.1 and 85 % at pH 8.4. Inhibitory pH level is just below 8.1in this ammonia concentration (Figure 4.20).

In 2500 mg/L NH_3-N concentration, a considerable decrease in biogas production and increase in inhibition were observed and at pH 7.4 40% and at pH 8.1 80% inhibition were recorded. Especially above pH 8.5, there existed almost no biogas production. Inhibitory pH level in this concentration is below 7.4 (Figures 4.21).

For NH_3-N concentration of 3000 mg/l, ammonia inhibition is very high for all pH levels. Inhibition as high as 80% was determined at pH 7.4, which means an inhibitory level around neutral pH. There were no significant differences between inhibition rates of different pH levels and biogas production due to the high NH_3-N concentration, and almost no biogas production existed (Figure 4.22).



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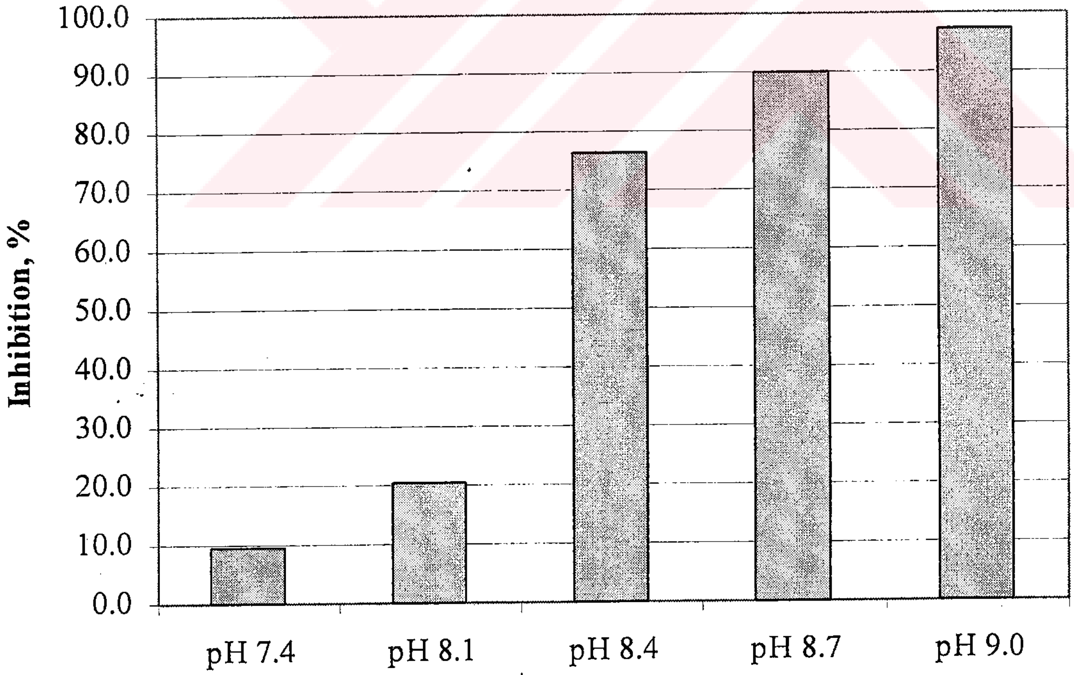

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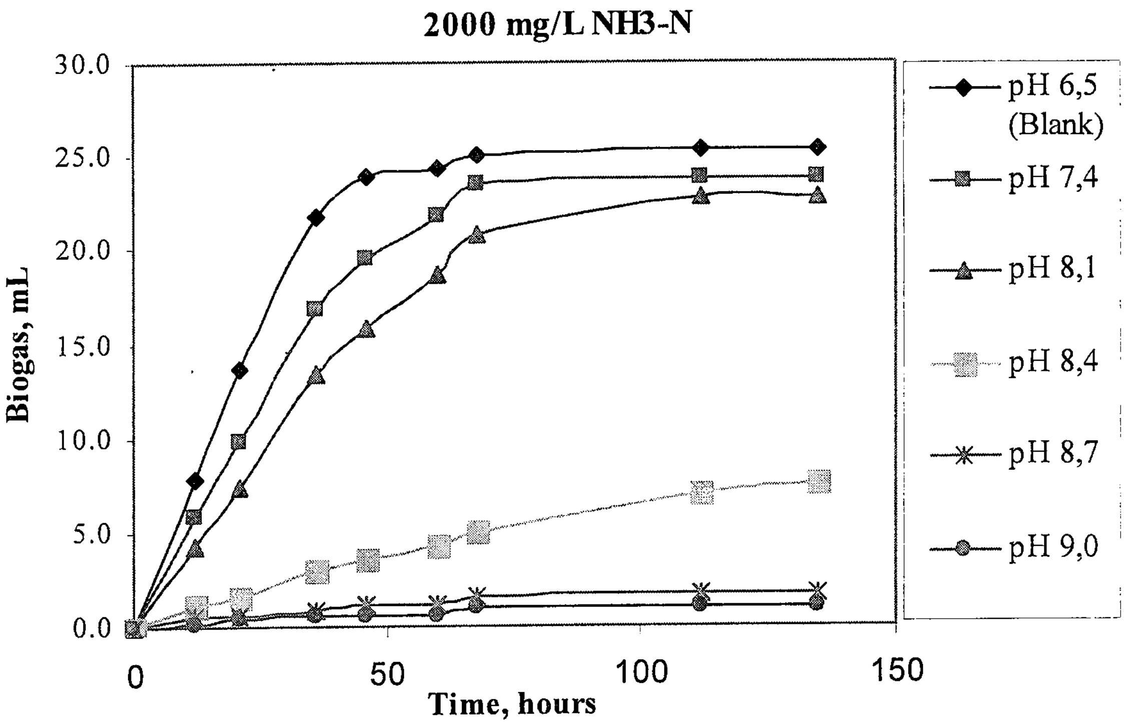
Figure 4.19 Ammonia Inhibition at 1500 $mgNH_3$-N/L (46th hour)



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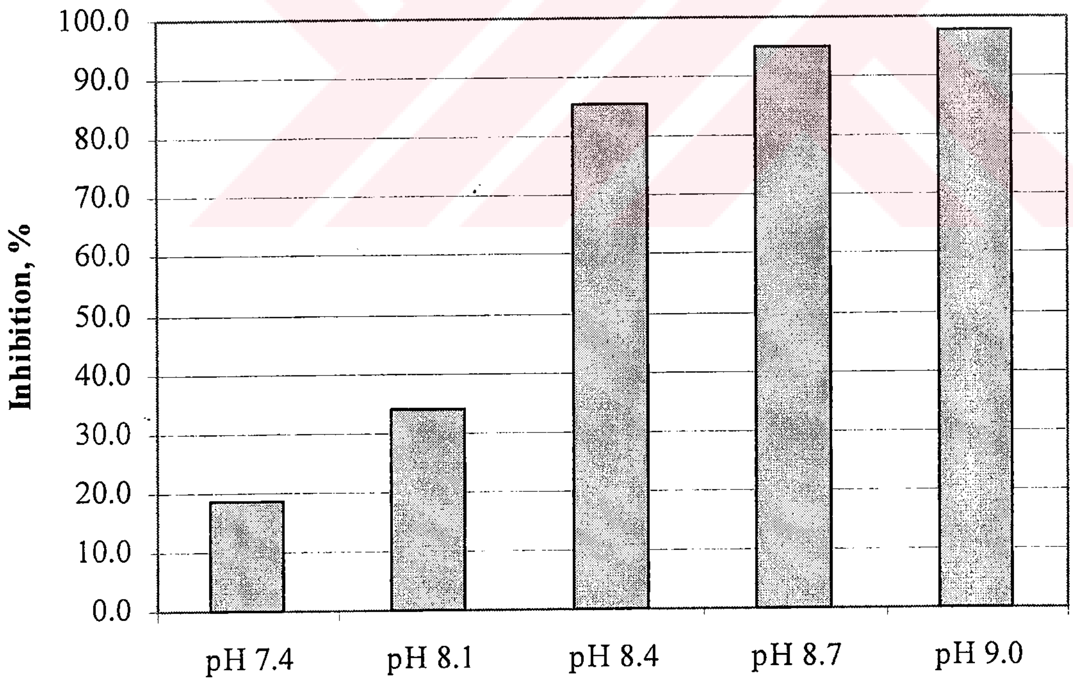

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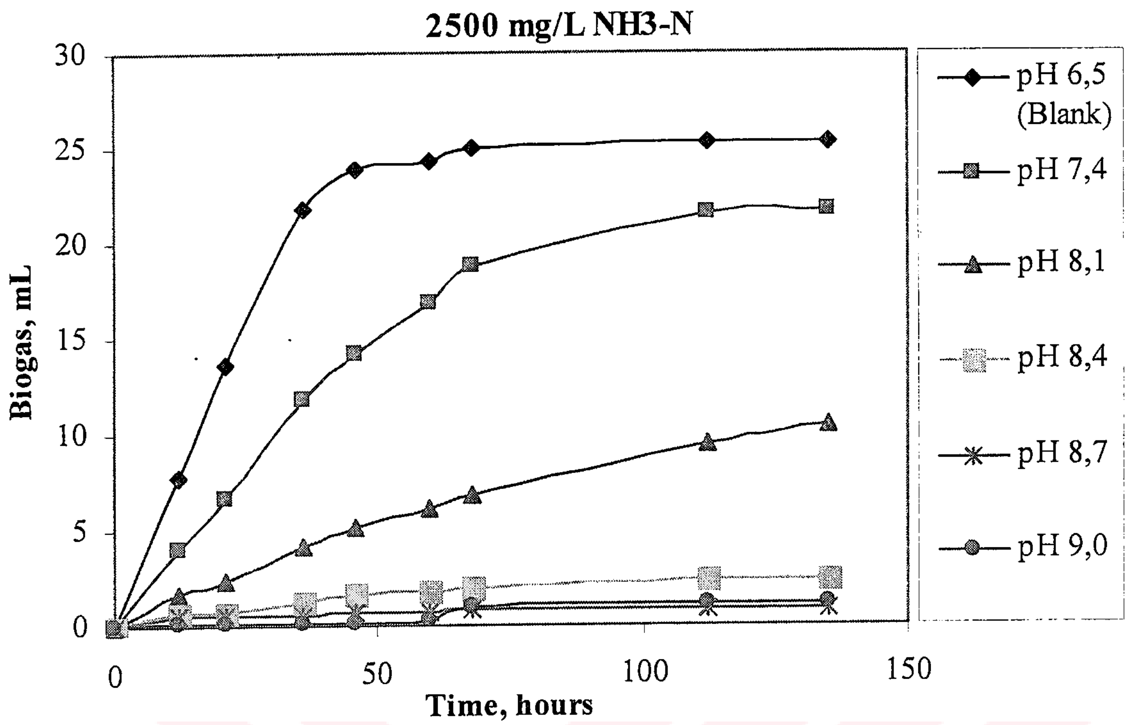
Figure 4.20 Ammonia Inhibition at 2000 mgNH_3-N/L (46th hour)



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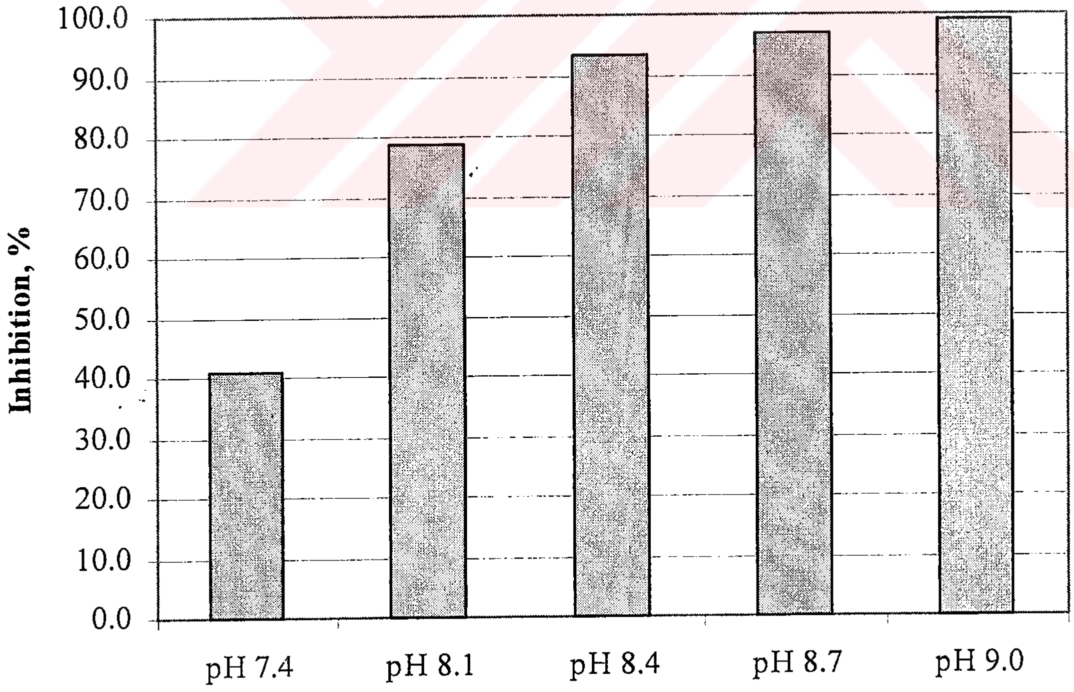

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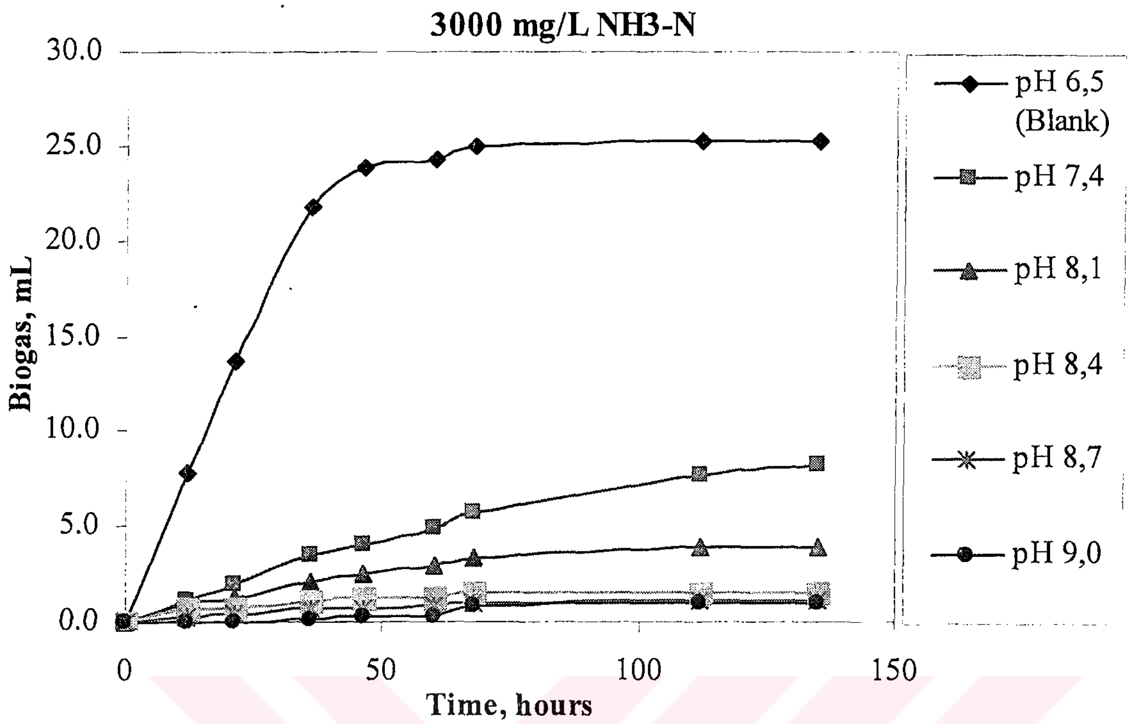
Figure 4.21 Ammonia Inhibition at 2500 mgNH_3-N/L (46th hour)



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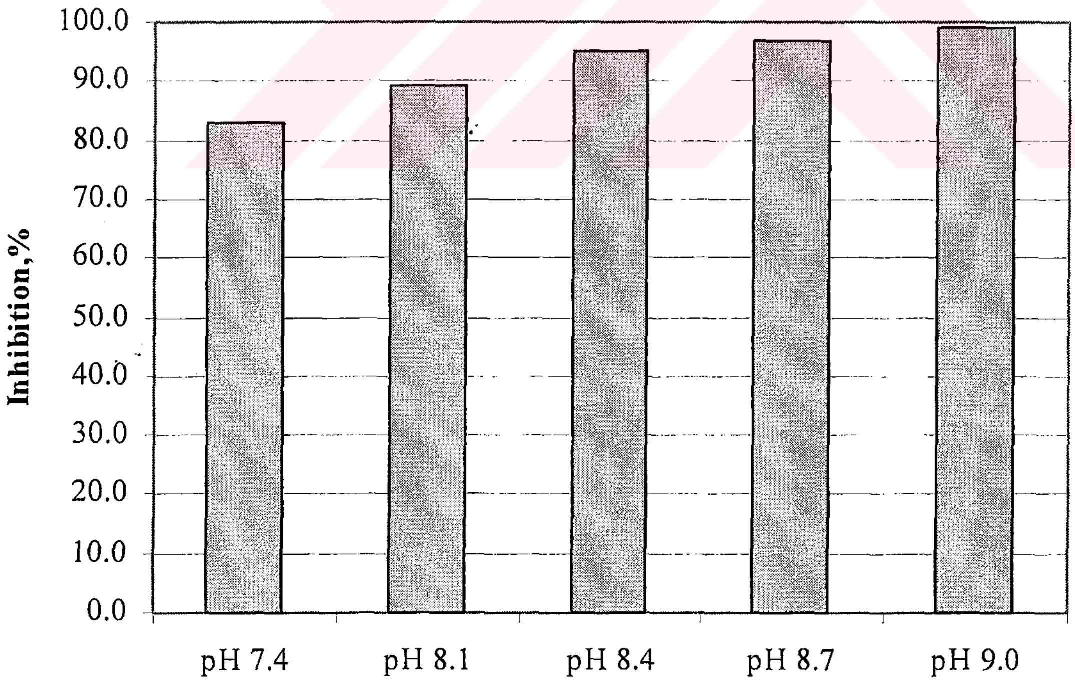

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Figure 4.22 Ammonia Inhibition at 3000 mgNH_3-N/L (46th hour)

CHAPTER 5

AEROBIC TREATABILITY STUDIES ON RAW AND ANAEROBICALLY TREATED LEACHATE

As it is mentioned in Chapter 2, landfill leachate initially is a high-strength wastewater, characterized by high organic constituents and by the presence of toxic chemicals. In addition, the leachate quality is variable from landfill to landfill, and over time as particular landfill ages. For a leachate with a high BOD/COD ratio (>0.4), the only major treatment option is biological treatment, but for a complete treatment according to the discharge standards in the region, a single biological treatment process is generally not enough. So, especially for young landfill leachates, treatment processes composed of anaerobic and aerobic steps are selected. Nowadays, advanced and tertiary treatment processes are usually forced for nutrient removal.

Only an anaerobic process by itself is not adequate to meet the discharge standards in the leachate treatment plant of Kömürcüoda Sanitary Landfill. To decrease the COD concentration in the anaerobic treatment effluents to the level in "İSKİ Discharge Standards" ($COD_{effluent}$<800 mg/L), an aerobic process is decided to be placed to the treatment plant.

In this part of the report, the studies on aerobic treatability of raw and anaerobically treated leachate is summarized.

5.1 AEROBIC REACTORS

As the aerobic treatment process, a squencing batch activated sludge system was selected for the leachate treatment plant in Kömürcüoda Sanitary Landfill. So, a set-up composed of two squencing batch reactors were used in aerobic treatability part of the study. Aerobic treatability is investigated on two step. First of all, in the acclimation phase of the reactors, raw leachate and then effluents of anaerobic reactors were fed.

5.1.1 Start-up and Operations of Sequencing Batch Reactors

A sequencing batch reactor (SBR) is a fill-and-draw activated sludge treatment system. The unit processes involved in the SBR and conventional activated sludge systems are identical. Aeration and sedimentation/clarification are carried out in both systems. However, there is one important difference. In conventional plants, the processes are carried out simultaneously in separate tanks, whereas in SBR operation the processes are carried out sequentially in the same tank.

Experimental Setup

The experimental study was carried out in a two-unit tank. Same conditions were conducted on both unit. The volume of each unit was 4L and they were diffused aerated. Diffusers were placed in the bases of the units and connected to an air pump. The air was controlled and equally distributed to the reactors by the valves on the connection pipes (Figure 5.1). In winter, to keep a constant temperature in the reactors a temperature controlled water bath was used. The temperatures in the reactors was kept between 20 to 25°C in winter, but in summer temperatures above 25°C were detected in some period.

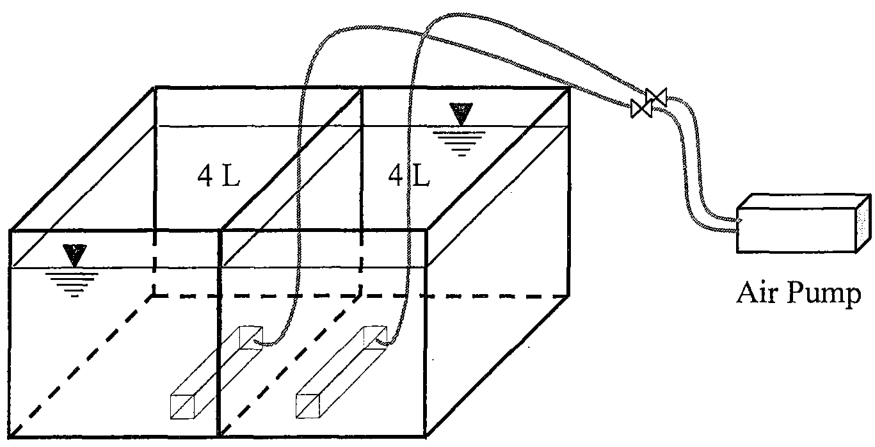


Figure 5.1 Experimental Set-up of Sequencing Batch Reactors

Seed Sludge

At the beginning of the experiments, SBR's were seeded with an aerobic sludge taken from a bakers' yeast industry. This sludge is selected because of the similar treatment applications in the treatment plant of this industry. The treatment, there, is composed of an anaerobic treatment (UASB reactor) followed by an activated sludge system as thought in Kömürcüoda leachate treatment plant.

5.1.2 Influent and Effluent Quality Monitoring

Determination of Total suspended solids (TSS), volatile suspended solids (VSS), sludge volume index (SVI), COD, pH and NH3-N routinely assessed the performances of the reactors. COD and NH_3-N were determined both in the influents and effluents, the others were monitored in mixed liquors of the reactors. All analyses were carried out following the Standard Methods APHA, AWWA, 1989. Analyses and the analytical methods are listed in Table 5.1.

Table 5.1 Routine Laboratory Analyses

Parameter	Analytical Method
pH	Electrode
SVI	Gravimetric
COD	Closed Reflux
NH_3-N	Nesslerization
TSS/ VSS	Gravimetric

5.1.3 Experiments

In the aerobic treatability part of this study, 4 different run were carried out using raw leachate and anaerobically treated leachate taken from the effluents of anaerobic reactors. At the beginning, in the first run, the reactors were fed with diluted raw leachate for the acclimation of the biomass. After the acclamation phase, anaerobically treated leachate was applied to the reactors in the second run and HRT was set as 1 day. This anaerobically treated leachate was obtained by mixed the effluents of anaerobic reactors equally. After a while the biomass concentration in the SBR's started to decrease. So, again, diluted raw leachate was fed to the reactors to increase the sludge concentration to 4000-5000 mgVSS/L levels.

At the forth run anaerobically treated leachate was fed to the reactors and the HRT was increased to 2 days. Again, unsatisfactory COD removal efficiencies were determined and decrease in sludge concentration has taken place. So, after this run, investigations were focused on the alternatives to increase the biodegradability of anaerobic effluent. Chemical oxidation alternatives as ozonation and hydrogen peroxide addition were applied to break down the non-biodegradable or slowly biodegradable contents of organics in the anaerobic effluent. The periods and types of feeds of these runs are presented in Table 5.2.

In all runs, to supply the adequate C/N/P ratio (100/5/1) necessary for bacterial growth, phosphoric acid was added to the reactors as phosphorus supply.

Table 5.2 Periods and Types of Feeds used in 4 Different Run

Run	**Period**	**Feeding**	**HRT**
1	Nov 07, 98 – Dec 06, 98	Diluted Raw Leachate	1 day
2	Dec 06, 98 – Dec 21, 98	Anaerobically Treated Leachate	1 day
3	Dec 27, 98 – Jan 08, 99	Diluted Raw Leachate	1 day
4	Feb 04, 99 – Mar 01, 99	Anaerobically Treated Leachate	2 days

RUN 1:

At the first run from 6^{th} of November to 2^{nd} of December 120 mL and from 2^{nd} of December to 5^{th} of December 200mL raw leachate was applied to the reactors and diluted to 4L in the reactors with dechlorinated tap water. During this period 3 raw leachate sample were taken from Kömürcüoda Sanitary Landfill. Characteristics of these leachate samples are given in Table 5.3.

Biomass concentration was kept around 5000 mgVSS/L in both reactors in this period. Influent COD concentration was fixed around 1000 mg/L in the first 25 days. After 2^{nd} of December it was increased to about 1500 mg/L to observe the performances of the reactors in higher COD levels. As seen from Figure 5.2, COD removal efficiencies as high as 85%'s were detected in this run. F/M ratio was between 0.20 to 0.25 in this period.

Table 5.3. Characteristics of Raw Leachate used in the First Run

Parameters	02/11/98 – 08/11/98	09/11/98 – 22/11/98	23/11/98 – 07/11/ 98
pH	8.1	7.7	7.3
Alkalinity, mg$CaCO_3$/L	6900	10400	9700
COD, mg/L	12350	29020	25950
BOD_5, mg/L	9890	21765	-
NH_3-N, mg/L	1750	2320	2690
TKN, mg/L	2070	2500	2984
SS, mg/L	940	2230	2340
TP, mg/L	2.11	-	1.74

RUN 2:

The second run was carried out with anaerobically treated leachate. Approximately 2L of anaerobic effluent was applied to both reactors everyday (HRT=1 day) from 6th of December to 21st of December. Rest of the reactors was completed with dechlorinated tab water. The characteristics of anaerobic effluents are given Table 5.4 in this period.

Table 5.4 Characteristics of Effluents of Anaerobic Reactors

Date	Influent (COD, mg/L)	Anaerobic Filter Effluent (COD, mg/L)	UASB Reactor Effluent (COD, mg/L)	Hybrid Bed Reactor Effluent (COD, mg/L)
Dec 7	11760	2400	2560	2670
Dec 8	12000	1880	2017	1957
Dec 9	11900	1595	1380	1705
Dec 11	11960	2380	1925	1885
Dec 14	12200	2395	3125	2025
Dec 16	12100	2065	4135	2855
Dec 18	12850	2185	3610	2765
Dec 21	12700	1275	1875	2155

In this run very low COD removal efficiencies and decrease in biomass concentration were determined. So, this run is stopped at 20th of December to prevent the wash out of microorganisms in the reactors. Results are presented in Figure 5.3.

RUN 3:

In the third run, 200mL of raw leachate were fed to the reactors to increase the biomass concentration, which was decreased in the second run carried out with anaerobic effluents. This time biomass concentration was kept around 4000mgVSS/L and again influent COD concentration was fixed around 1000mg/L to state the F/M ratio as 0.25. HRT's were 24 hr at this period. Characteristics of the raw leachate used in this period are given in Table 5.5.

As in run 1, high COD removal efficiencies around 80 to 85 % and nitrification efficiencies around 95 % were carried in this period. Biomass was started to growth and excess sludge was wasted every day (Figure 5.4).

Table 5.5. Characteristics of Raw Leachate used in the Third Run

Parameters	**21/12/98 – 04/01/99**	**04/01/99 – 10/01/99**
PH	7.5	7.8
Alkalinity, $mgCaCO_3/L$	6900	11200
COD, mg/L	27500	12600
BOD_5, mg/L	11200	7200
NH_3-N, mg/L	1660	2330
TKN, mg/L	1825	-
SS, mg/L	1560	1450
TP, mg/L	5.37	0.92

RUN 4:

In run 4, the effluents of anaerobic reactors were again fed to the SBR's from 4th of Feb to 1st of Mar. This time 1L was added to each tank. Rest is completed with dechlorinated tap water. Influent COD concentration was fluctuated between 600-1000 mg/L and biomass was kept around 4000 mgVSS/L. This time HRT was set as 2 days and F/M ratio between 0.2 to 0.3. The effluents of the anaerobic reactors used in feeding the SBR's are presented as COD data in Table 5.6.

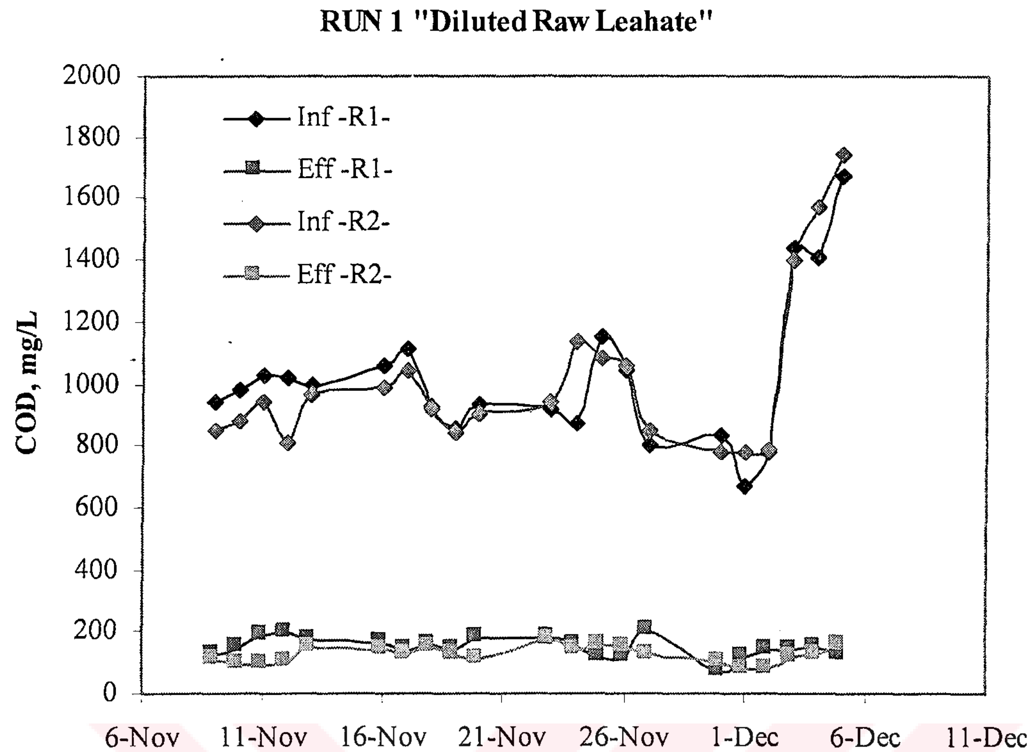
Table 5.6 Characteristics of Effluents of Anaerobic Reactors

Date	Influent (COD, mg/L)	Anaerobic Filter Effluent (COD, mg/L)	UASB Reactor Effluent (COD, mg/L)	Hybrid Bed Reactor Effluent (COD, mg/L)
Feb 04	12100	5050	6400	4700
Feb 05	12100	5070	6460	4400
Feb 06	14000	2690	3650	2630
Feb 07	13600	3240	4000	3100
Feb 08	13500	2870	3200	2750
Feb 11	12640	2640	2620	2570
Feb 13	13020	2760	2890	2820
Feb 15	12820	2880	3000	2790
Feb 18	13700	2700	3020	2890
Feb 20	13540	2500	2780	2900
Feb 22	13740	2925	3160	2910
Feb 24	12880	2885	3030	2930
Feb 27	13000	2575	2965	2825
Feb 28	12750	2855	3180	3110
Mar 01	13950	2795	3120	2775

Figure 5.5 represents the COD removal efficiencies in the reactors in RUN 4. Effluent COD concentrations were analysed between 700 - 1200mg/L therefore very low efficiencies were determined in this period. It is believed that, this was due to the high non-biodegradable or slowly biodegradable organic content in the anaerobically treated leachate.

As a result of these experiments, it can be indicated that satisfactory COD removal efficiencies with anaerobic effluents were not detected in SBR's because of high ammonia concentrations and high refractory or non-biodegradable organic content in the anaerobic effluents.

Although COD was not removed properly in SBR's, high nitrification efficiencies were observed under sufficient alkalinity concentrations. But, denitrification step of biological nitrogen removal was not carried out because of unsatisfactory conditions for denitrifiers. This subject should be investigated before the construction of SBR's in Kömürcüoda Landfill, which is very important for the removal of high ammonia content in leachate before the discharge.



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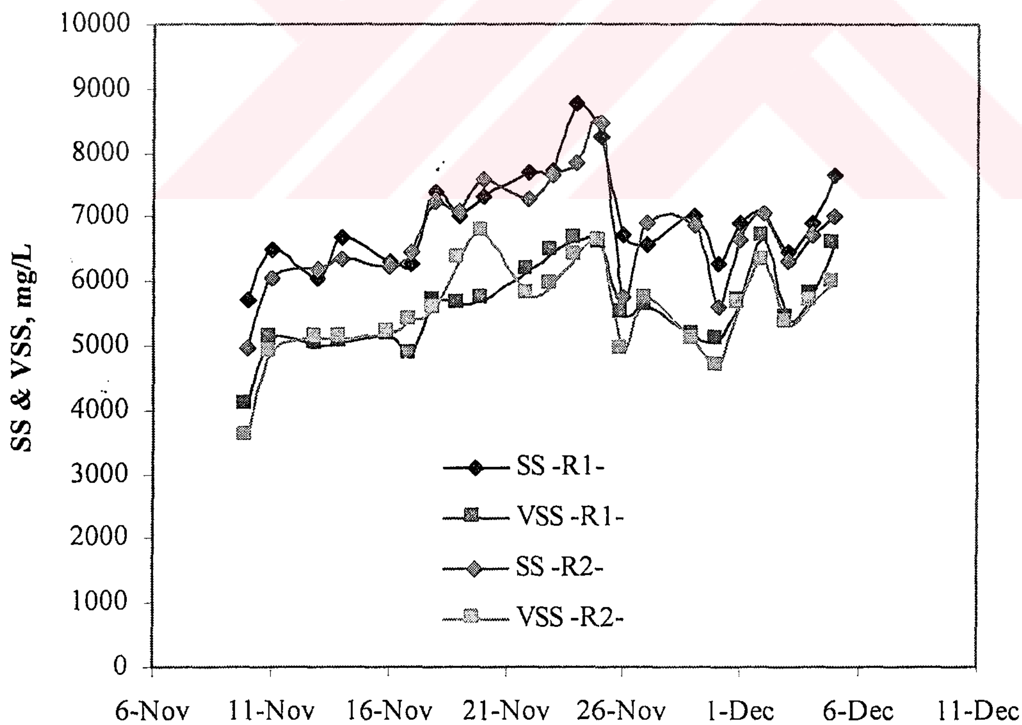
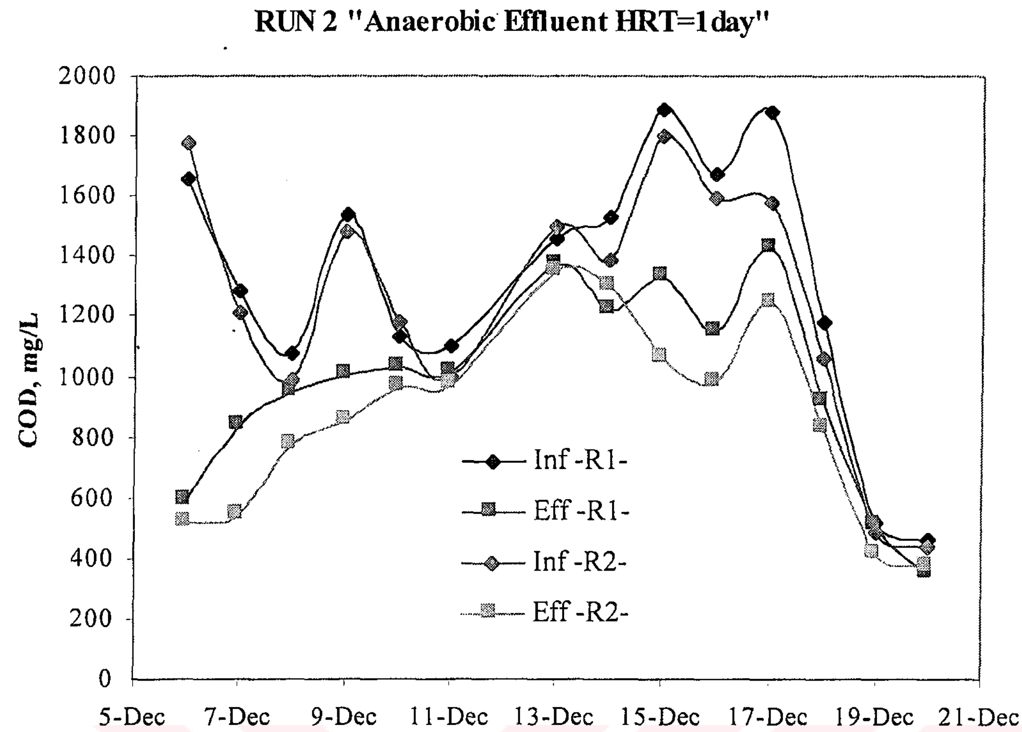


(b)

Figure 5.2 Aerobic Treatability of Raw Leachate; RUN 1



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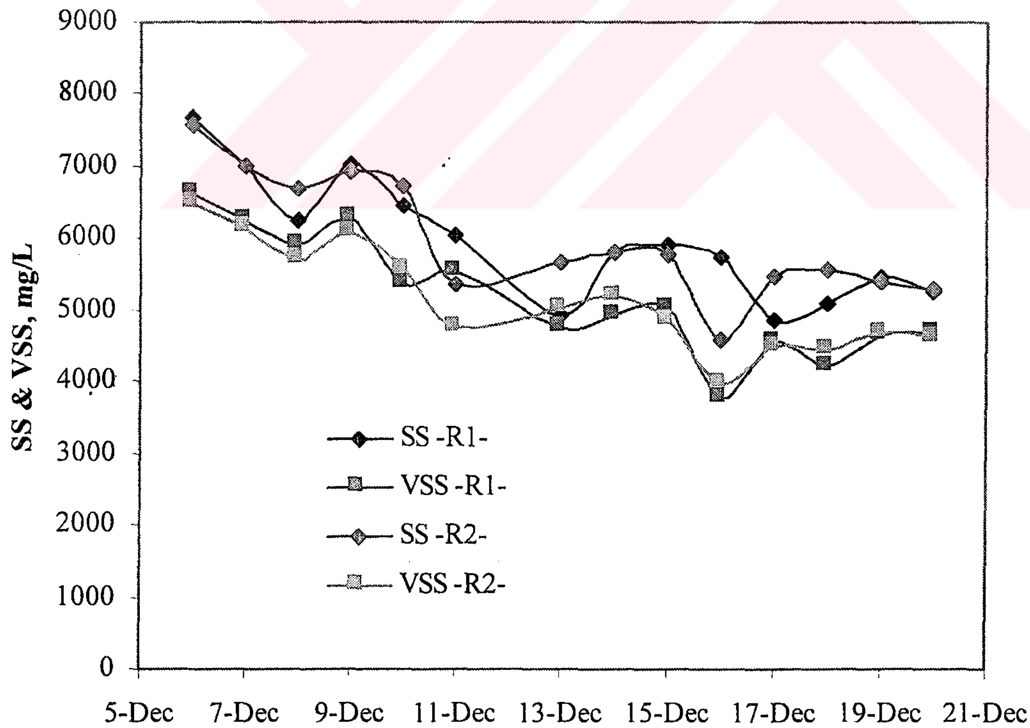

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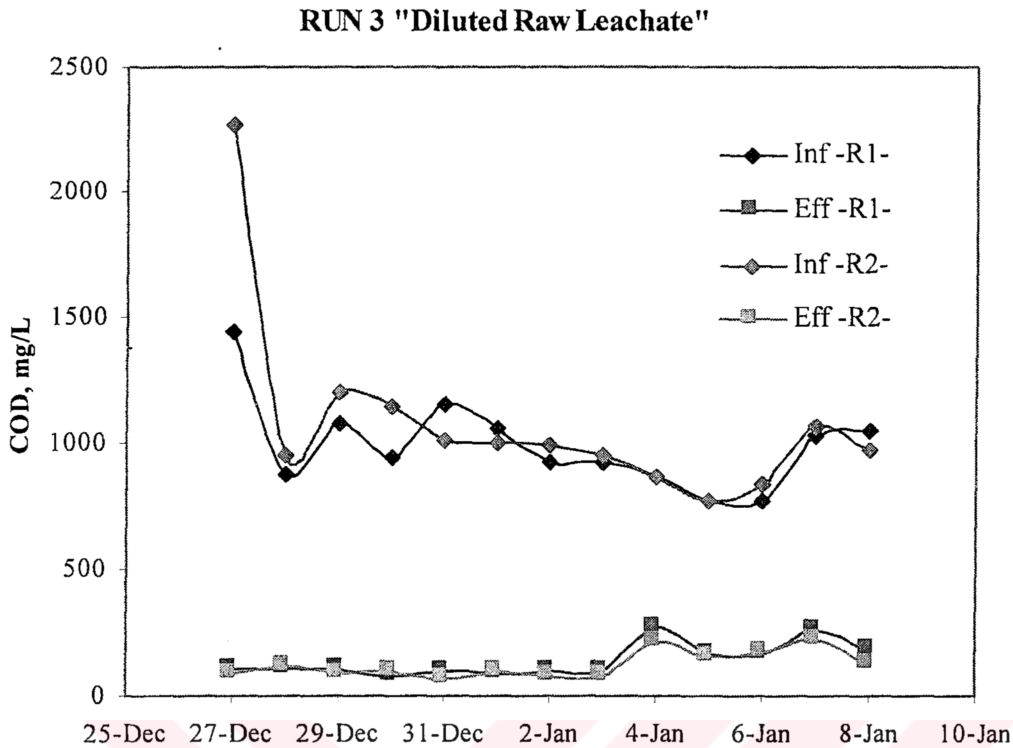
Figure 5.3 Aerobic Treatability of Anaerobic Effluents (HRT=1day); RUN 2



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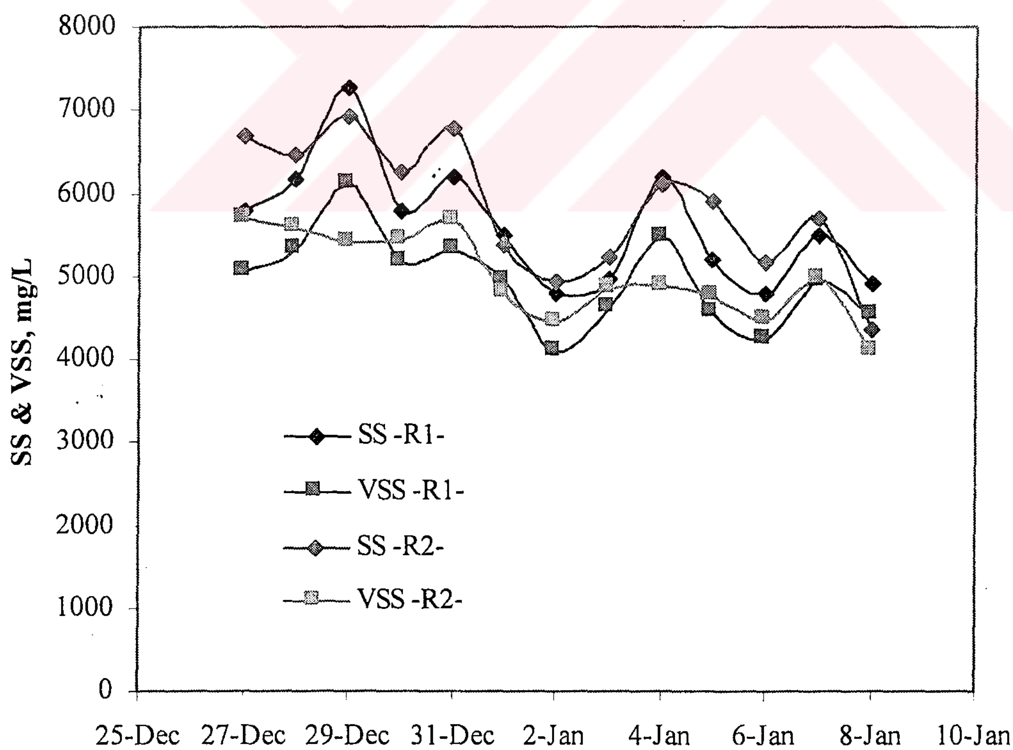

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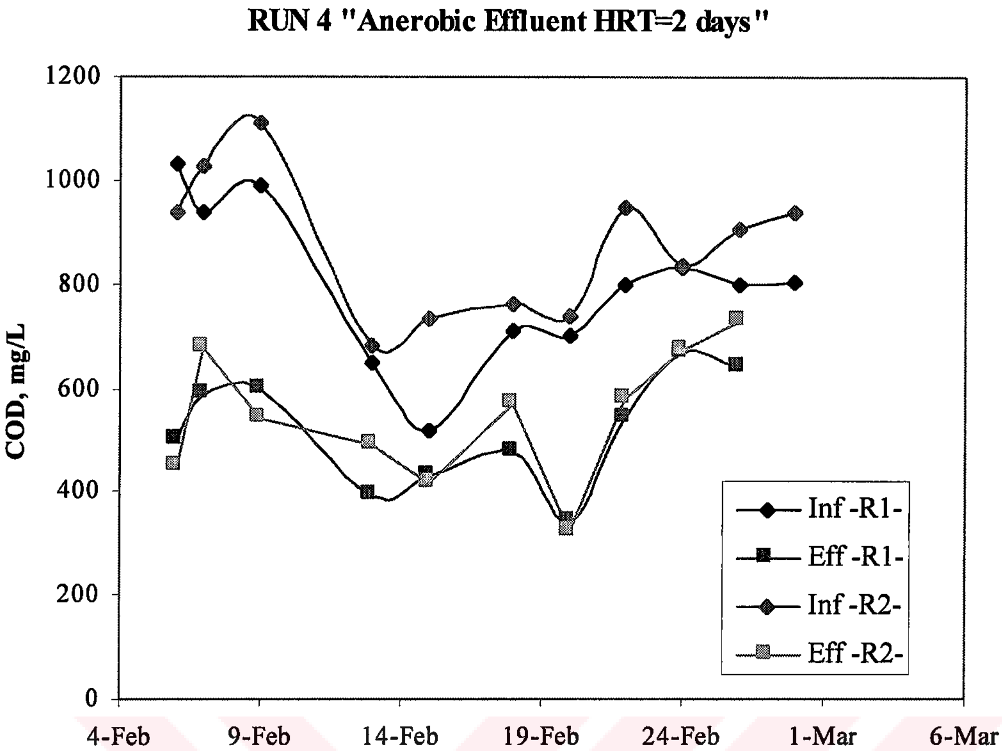
Figure 5.4 Aerobic Treatability of Raw Leachate; RUN 3



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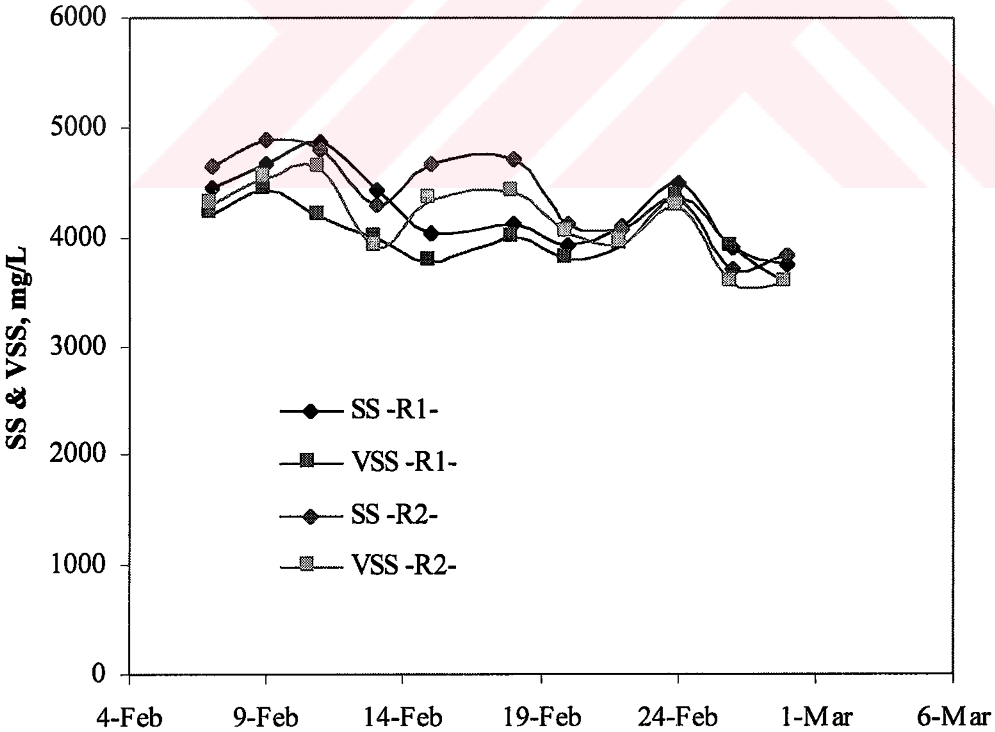

(b)

Figure 5.5 Aerobic Treatability of Anaerobic Effluents (HRT=2days); RUN 4

5.2 ALTERNATIVES TO INCREASE THE BIODEGRADABILITY OF ANAEROBIC EFFLUENTS

Because of low COD removal efficiencies in the second and forth run carried out with anaerobic effluent feeds, new investigations were focused on the alternatives to increase the biodegradability of anaerobically treated leachate.

Chemical oxidation is a potential treatment option for the removal or breaking down of certain organic pollutants in the landfill leachates. The amount of oxidant required in practice is generally greater than the theoretical mass calculated. The reasons for this are numerous and include incomplete oxidant consumption and oxidant demand caused by other species in solution. The process train for the chemical oxidation process includes adjustment of pH of the solution. The oxidizing agent is added gradually and mixed thoroughly. The oxidizing agent may be in the form of a gas (e.g., ozone, chlorine), a liquid (e.g., hydrogen peroxide), or a solid (e.g., potassium permanganate). Economics of treatment and treatability of a specific pollutant also govern the degree of oxidation. Partial oxidation followed by additional treatment options may be more efficient and cost effective than using a complete oxidation treatment scheme alone. An increase in the biodegradability of refractory organics due to chemical oxidation has been reported. Examples of common oxidants include ozone, chlorine, hydrogen peroxide, and UV radiation. So, to increase the COD removal efficiency in the SBR's chemical oxidants as ozone and hydrogen peroxide were applied to the effluents of anaerobic reactors in this part of the study.

5.2.1 Ozonation

Firstly, ozonation, the most common chemical oxidation technique, was applied to the anaerobically treated leachate. As investigated and observed in the previous treatability runs, high amounts of hardly biodegradable organic content in the anaerobic effluents were responsible from the unsatisfactory COD removal efficiencies.

Ozonation is carried out with the PCI Model GL-1 ozone generator. This ozone generator produces 1.134 kg/day ozone at three percent concentration by weight. It was set to 272 L/hr oxygen flow and 90 % ozone output. In the experiments, sample volume was selected as 16 lt. Ozone is applied from a porous diffuser and bubbled through the sample. Ozone transfer occured as fine bubbles containing ozone and oxygen rise slowly inside the column, contacting the leachate. The approximate ozone dose transfered to the sample was calculate as about 0.3kgO_3/min according to the previous ozonation studies.

The first ozonation experiment was performed with 30 minute contact time and samples were taken after 10 minute and at the end. COD about 2000 mg/L at the beginning was reduced to 1820mg/L after 10 minute and to 1470mg/L at the end parallel to the decrease in BOD_5. BOD_5 was decreased from 390 mg/L to 210 and 160 mg/L respectively, when pH increased from 8.46 to 8.76 and. The results of this ozonation experiment are presented in Table 5.7.

Table 5.7 Results of Ozonation Experiment 1

Parameter	Anaerobic eff. Before ozonation	Anaerobic eff. After 10 min ozonation	Anaerobic eff. After 30 min ozonation
pH	8.46	8.55	8.76
COD	1980	1820	1470
NH3-N	1855	1845	1755
NO3-N	7	8	14.5
BOD_5	390	210	160

The second ozonation experiment was performed with different pH levels between 7.6 to 9.5 and the contact time was set as 60 minute. Samples were taken at the 15th, 30th and 45th minutes and at the end. Color, pH, COD and NH_3-N analyses were carried out and the highest COD removal efficiency was detected as 35% at pH 8.5 and 60 minute contact time. The Color, pH, COD and NH_3-N concentrations were presented with respect to different pH levels and contact times in Table 5.8.

5.2.2 Hydrogen Peroxide

Hydrogen peroxide in the presence of a catalyst, e.g., iron, generates hydroxyl radicals which react with organics and reduced compounds in a similar manner to ozone. Cost-effective application of chemical oxidation suggests partial oxidation of refractory organics to render them more biodegradable. H_2O_2 oxidation is most commonly conducted at low pH levels. The optimal pH range for the reaction is 3 to 5.

Table 5.8 Results of Ozonation Experiment 2

Time, min	pH eff	Color eff PtCo	Removal Eff. %	COD eff Mg/L	Removal Eff. %	NH3-N eff mg/L	Removal Eff. %
pH 7.6							
0	7.65	6700	-	2880	-	2520	-
15	7.69	3100	0.54	2740	0.05	2520	0.00
30	7.72	790	0.88	2580	0.10	2360	0.06
45	7.77	580	0.91	2280	0.21	2420	0.04
60	7.81	420	0.94	2140	0.26	2360	0.06
pH 8.5							
0	8.53	4360	-	2800	-	2475	-
15	8.47	1320	0.70	2560	0.09	2475	0.00
30	8.43	680	0.84	2100	0.25	2425	0.02
45	8.38	560	0.87	1980	0.29	2150	0.13
60	8.36	520	0.88	1820	0.35	2175	0.12
pH 9.0							
0	9.00	6760	-	2480	-	2620	-
15	8.92	2320	0.66	1980	0.20	2560	0.02
30	8.86	960	0.86	1900	0.23	2540	0.03
45	8.83	680	0.90	1900	0.23	2500	0.05
60	8.00	600	0.91	1860	0.25	2420	0.08
pH 9.5							
0	9.46	4940	-	2260	-	2460	-
15	9.36	920	0.81	2060	0.09	2360	0.04
30	9.31	500	0.90	2040	0.10	2360	0.04
45	9.27	320	0.94	2020	0.11	2360	0.04
60	9.23	200	0.96	1920	0.15	2360	0.04

For hydrogen peroxide oxidation 30% by weight hydrogen peroxide was added to the anaerobically treated leachate with a concentration of 5 mL/L. No additional catalyst was added, because of high iron concentrations about 5 mg/L in the effluents of anaerobic reactors. Four different pH levels were set as 7.6, 6.5, 5.5 and 4.5 and a 10minute contact time was satisfied in a jar test apparatus with slow mixing. Color, COD, pH and NH_3-N analyses were carried out and the results presented in Table 5.9 were determined.

Table 5.9 Results of the Experiment carried out with H_2O_2

Sample	pH eff	Color eff PtCo	Removal Eff.,%	COD eff mg/L	Removal Eff.,%	NH3-N eff mg/L	Removal Eff.,%
Anaerobic Effluent	7.58	11750	-	3770	-	2460	-
pH 7.6	7.73	9520	0.19	3700	0.02	2360	0.04
pH 6.5	7.70	6400	0.45	2840	0.25	2360	0.04
pH 5.5	7.10	10260	0.13	3610	0.04	2440	0.01
pH 4.5	6.68	11280	0.04	3735	0.01	2460	0.00

Table 5.10 Results of the Experiment carried out with O_3 + H_2O_2

Time. Min	pH eff	Color eff, PtCo	Removal Eff.,%	COD eff mg/L	Removal Eff.,%	NH3-N eff mg/L	Removal Eff.,%
5 mL/L H_2O_2							
Raw	7.64	6260	-	2265	-	2375	-
0	7.77	5270	15.8	2095	7.5	-	-
15	7.79	2670	57.3	1885	16.8	-	-
30	7.89	2110	66.3	1670	26.3	-	-
45	7.93	1400	77.6	1670	26.3	-	-
60	7.99	1150	81.6	1600	29.4	2245	5.5
10 mL/L H_2O_2							
0	7.84	4230	32.4	2320	-	2315	-
15	7.81	1940	69.0	1805	22.2	-	-
30	7.92	950	84.8	1595	31.3	-	-
45	8.01	670	89.3	1535	33.8	-	-
60	8.08	490	92.2	1470	36.6	2185	5.6
15 mL H_2O_2							
0	7.95	4290	31.5	1870	-	2450	-
15	7.98	2120	66.1	1740	7.0	-	-
30	8	1690	73.0	1540	17.6	-	-
45	8.04	1300	79.2	1360	27.3	-	-
60	8.07	1100	82.4	1310	29.9	2345	4.3
20 mL H_2O_2							
0	7.96	5550	11.3	2050	-	2475	-
15	7.96	5170	17.4	2040	0.5	-	-
30	7.98	3740	40.3	2050	0.0	-	-
45	8.06	3240	48.2	1990	2.9	-	-
60	8.14	1940	69.0	1990	2.9	2450	1.0

5.2.3 Ozone + Hydrogen Peroxide

To increase the oxidation effect of ozone and improve the breaking down of refractory organics as an alternative ozonation was carried out together with hydrogen peroxide addition. 5, 10, 15 and 20 mL H_2O_2 were added to 1 L of leachate and 60minute ozonation was carried out. The highest COD removal efficiency was determined with 10mL/L H_2O_2 after 60 minute, but the additional removal effect of this case was detected as only 2 or 3 % with respect to 60min ozonation by itself. The results of this experiment are given in Table 5.10.

The aerobic treatability of anaerobic effluents after these applications given above is going on now under new studies. Some combinations of chemical oxidation techniques, different seed sludges and long acclimation periods are the subjects of this new study.

CHAPTER 6

CONCLUSIONS

Conclusions for characterization study;

- High COD and BOD_5 values in the leachate indicate high acidogenic activity in Kömürcüodă Sanitary Landfill. In addition to that BOD_5/COD ratios above 0.7 is an indication of high biodegradability.

- Heavy metal concentrations are below inhibition levels requiring no metal removal prior to biological treatment.

- Ammonia concentrations as high as 2700 mg/L with pH values around 8.0 in raw leachate increase the possibility of ammonia inhibition in anaerobic treatment.

Conclusions for anaerobic treatability study;

- Since unionized ammonia is toxic to methane producing bacteria, total ammonia concentration is not a good parameter for understanding the ammonia inhibition. Therefore, unionized ammonia concentrations should be used for evaluating the ammonia inhibition.

- COD removal efficiencies in anaerobic reactors can reach to approximately 90 % provided that unionized ammonia concentration is kept below inhibition levels with pH adjustment. This shows that , the leachate can be treated anaerobically.

- By controlling the pH in the influent of anaerobic reactors, ammonia inhibition can be eliminated. To investigate the ammonia inhibition, methanogenic activity test were carried out and critical levels of pH in the reactors were determined as 7.4 and 8.1 for ammonia nitrogen concentrations of 2000mg/L and 2500 mg/L, respectively.

- Anaerobic filter and hybrid bed reactor are more resistant to ammonia inhibition than UASB reactor according to the results of anaerobic treatability study. This indicates the resistance of attached growth processes to ammonia inhibition more than suspended growth systems.

- Although anaerobic filter is more resistant to inhibition, it has some operational difficulties. The main problem is the clogging of the system due to deposition of inorganic precipitate and biomass growth.

- For anaerobic treatment of Kömürcüoda Sanitary Landfill Leachate, UASB reactor is a right selection only if pH is controlled effectively or an ammonia stripping is located prior to anaerobic treatment. On the other hand, pH control will bring a rise in operational cost due to high amounts of HCl requirement and ammonia stripping will also increase the operational cost by lime consumption and sludge generation.

- This study does not involve any comparison between flocculated sludge and granular sludge for UASB reactor. Granular sludge might be more resistant to ammonia inhibition. But it should be investigated in details in further studies.

- High variability of quality and fluctuations in the quantity of leachate imposes longer experimental periods, in order to evaluate the behavior of anaerobic reactors. So, to evaluate the characteristics of the leachate and performances of the reactors exactly, this study should be continued at least one year.

Conclusions for aerobic treatability study;

- Because of high ammonia concentrations and high refractory or non-biodegradable organic content in the anaerobic effluents, satisfactory COD removal efficiencies were not detected in SBR's.

- To increase the biodegradability in the anaerobic effluents by breaking down the refractory organics, ozonation and hydrogen peroxide oxidation were applied. But the results obtained were not exactly satisfactory. So, some different alternatives especially combinations should be investigated in the further studies. Also, different seed sludges and long acclimation periods should be examined.

- Although COD was not removed properly, high nitrification efficiencies were observed if sufficient alkalinity was provided in SBR's. But, denitrification step of biological nitrogen removal was not carried out because of unfavourable conditions for denitrifiers as presence of high non-biodegradable organics. This part also should be investigated in further studies before the construction of SBR's in Kömürcüoda Landfill.

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APPENDIX

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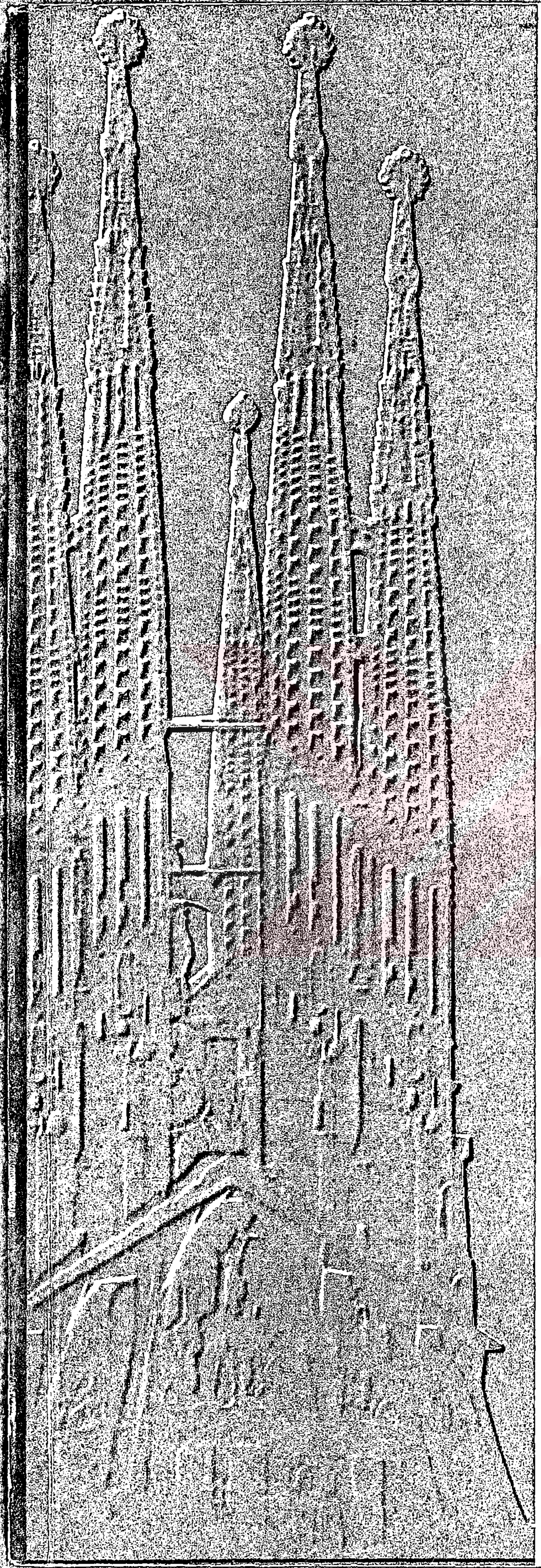

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CHARACTERIZATION AND ANAEROBIC TREATMENT OF THE SANITARY LANDFILL LEACHATE IN ISTANBUL

B. Inanc, B. Calli and A. Saatci

Marmara University, Faculty of Engineering, Dept. of Environmental Engineering, Goztepe Campus, Istanbul, Turkiye
e-mail: binanc@mutek.org.tr, bcalli@mutek.org.tr

ABSTRACT

In this study, characterization and anaerobic treatability of leachate from Komurcuoda Sanitary Landfill located on the Asian part of Istanbul were investigated. Time based fluctuations in characteristics of leachate were monitored for a 8 months period. Samples were taken from a 200 m^3 holding tank located at the lowest elevation of the landfill. COD concentrations have ranged between 18800 and 47800 mg/l while BOD_5 between 6820 and 38500 mg/L. COD and BOD_5 values were higher in summer and lower in winter due to dilution by precipitation. On the other hand, it was quite interesting that such dilution effect was not observed for ammonia. The highest ammonia concentration, 2690 mg/L was in November 1998. BOD_5/COD ratio was larger than 0.7 for most samples indicating high biodegradability, and acidic phase of decomposition in the landfill.

For anaerobic treatability, three different reactors, namely an upflow anaerobic sludge bed reactor, an anaerobic upflow filter and a hybrid bed reactor, were used. The anaerobic reactors were operated for more than 230 days and were continuing operation when this paper was prepared. Organic loading was increased gradually from 1.3 kgCOD/m^3.day to 8.2 kg COD/m^3.day while hydraulic retention time was reduced from 2.4 days to 2.0 days. All the reactors showed similar performances against organic loadings with efficiencies between 80% and 90%. However the reactors have experienced high ammonia concentrations several times throughout the experimental period, and showed different inhibition levels. Anaerobic filter was the least effected reactor while UASB was the most. Hybrid bed reactor has exhibited a similar performance to anaerobic filter although not at the same degree.

KEYWORDS

Solid wastes; leachate; sanitary landfills; anaerobic treatment; inhibition.

INTRODUCTION

Istanbul Metropolitan City with a population over 10 million has to manage 8000 tons of solid waste every day, and the population is expected to reach 17 million by the year 2020. Amount of municipal solid waste has been increasing with increasing population and improved living standards. Currently, the waste generation rate is 0.65 kg/ca-day. Meanwhile, natural gas and low-ash coal use are increasing and expected to reach 40% for natural gas and 60% for coal by the year 2000. As a result, it is expected that, ash content will decrease by about 50%, and waste generation rate will then be 0.6 kg/ca-day (Arikan et.al., 1997) Some physical and chemical characteristics of the solid wastes in the city are given in Table 1 as average values weighted by population.

TABLE 1 CHARACTERISTICS OF MUNICIPAL SOLID WASTE IN ISTANBUL (Arikan et al., 1997)

Moisture Content (%)	Organic Matter (%)	C/N	Calorific Value (kcal/kg)	Unit Weight (kg/m^3)	Reference
46.3	50.1	30.5	811	416	Basturk (1979)
-	51.7	-	-	410	WHO/UNDP (1981)
54.5	-	-	1010	252	CH2M Hill - Antel (1992)
55.1	60	32	920	220	Arikan (1996)

t present, there are two sanitary landfills, one at European part and the other at Asian part, with total ɔerational capacity of 20 years have been under operation since 1995. On the other hand, construction of a ɔmpost plant with 1000 tons/day capacity has been started in 1998. Some reduction is expected in organic ɔntent of the solid waste to be landfilled due to separation of organic contents for composting while sludges om newly constructed and future wastewater treatment plants, if landfilled, will result in increment. ence, characteristics of solid wastes and leachates from landfills should be closely monitored. The .unicipality has started the construction of leachate treatment plants consisting of anaerobic and aerobic :ocesses although the exact characteristics of the leachates are not known. Hence, in this study, ıaracterization and anaerobic treatability of leachate from Komurcuoda Sanitary Landfill, which is under ɔeration since 1995, were investigated.

:omurcuoda Landfill and The Leachate Treatment Plant

ntil 1997, 15 ha area has been utilized in Komurcuoda landfill, and it has been estimated that 5 ha will be sed every year. While leachate recirculation has been practiced, excess amount of leachate produced is ansported by tanker trucks, and disposed into municipal sewer system. Amount of leachate managed in ıis way was 110780 m^3 in 1997. This is equivalent of a flow rate of about 300 m^3/day. Design capacity of ıe leachate treatment plant was selected as 500 m^3/d. Flow diagram of the leachate treatment plant at :omurcuoda landfill (Asian part of Istanbul) which is under construction is shown in Figure 1.

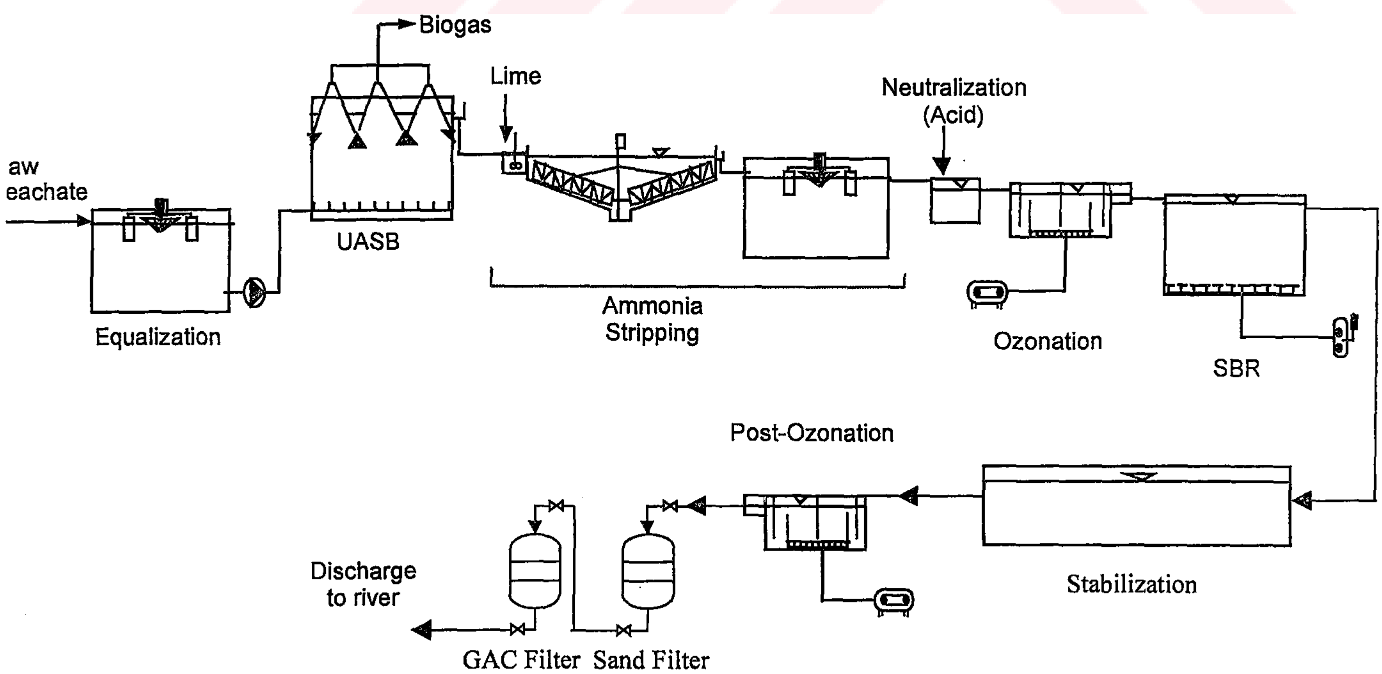

Fig. 1. Flow diagram of the leachate treatment plant at Komurcuoda landfill (under construction)

MATERIALS AND METHODS

haracterization of Leachate

ɔr characterization studies, leachate samples were taken from Komurcuoda Landfill once every week and ne based fluctuations in characteristics of leachate were monitored for a 8 months period. Samples were ken from a 200 m^3 holding tank located at the lowest elevation of the landfill. The parameters analyzed ere; pH, alkalinity, COD, BOD_5, NH_3-N, TKN, SS, TP, Color, Chloride, Fe, Mn, Cu, Zn, Pb, Cd, Cr and i. All the analyses were according to Standard Methods (AWWA, 1992). Heavy metals were analysed by ı atomic absorption spectrometer (UNICAM 919).

naerobic Treatability

ɔr anaerobic treatability, three different reactors, namely an upflow anaerobic sludge bed reactor, an ıaerobic upflow filter and a hybrid bed reactor, with an effective volume of 7.85 L, were used. The ameter of the reactors was 10 cm, and the effective height 100 cm. Anaerobic filter and hybrid bed reactors ere filled with packing material, 100 cm and 40 cm from the top, respectively. The packing material used as randomly packed roughened pipe pieces with helezonic support lines at the outer surface. The specific ea of the packing was 112 m^2/m^3, with a void ratio of 87%. A constant temperature room was used to ıaintain the temperature of the reactors at 35±2 °C. The reactors were seeded with sludges from an ıaerobic treatment plant of a beaker's yeast industry. The leachate samples used for feed in the experiments ere brought from the landfill once every week for characterization, and stored refrigerated. Dechlorinated ıp water was used for diluting the leachate for adjusting the organic loading. Phosphoric acid was added to ıe feed to provide a COD/N/P ratio of 500/7/1 due to phophorus deficiency of the leachate.

he reactors were operated for more than 8 months. Organic loading was increased gradually from 1.3 gCOD/m^3.day to 8.2 kg COD/m^3.day while hydraulic retention time, after Day 180, was reduced from 2.4 ays to 2.0 days due to low COD concentrations during this period caused by high dilution. In the effluents f the reactors, pH alkalinity and temperature were monitored on a daily basis, while COD and NH_3-N were ıonitored three times in a week. SS and VSS were monitored once a week. Sludge concentrations along ith the height of the UASB and Hybrid reactors were measured on Day 4, Day 55 and Day 140.

RESULTS AND DISCUSSION

haracterization of Leachate

eachate samples brought from Kömürcüoda Sanitary Landfill were analyzed immediately after arriving at ıe laboratory. The results are presented in Table 2.

he values in the table are the averages of the samples taken in each month starting from August 1998 to Iarch 1999. Samples were taken every week in August and September 1998, and once every week in the ollowing months. Maximum and minimum values represent these all samples analyzed. PH has varied from .2 to 8.2 although Volatile fatty acids were high in the samples, which could be understood easily from mell. This was proved by qualitative analysis with gas chromatography. Relatively higher pH values was ctually due to high alkalinity levels which was effected strongly by extraordinarily high ammonia oncentrations. As it can be seen from Table 2, alkalinity has ranged between 6900 and 11750 $mgCaCO_3/L$ vhile ammonia was between 1660 and 2690mg/L.

hese high concentrations of ammonia seems to be specific to Turkiye which can be understood from Table ı easily. Such high ammonia levels were also experienced in Harmandali Landfill of Izmir city. The next igh ammonia concentrations are reported from Thessaloniki of Greece with a maximum of 1510 mg/L. ımmonia level in USA and France are comparatively lower. Ammonia levels are quite important for

TABLE 2 CHARACTERISTICS OF LEACHATE OF KOMURCUODA MSW LANDFILL (ISTANBUL)

	Monthly Average Values										
Parameters	**Aug-98**	**Sep-98**	**Oct-98**	**Nov-98**	**Dec-98**	**Jan-99**	**Feb-99**	**Mar-99**	**Min**	**Max**	**Std. Dev.**
pH	6.9	7.4	7.8	7.7	7.7	8.0	7.6	7.7	6.2	8.2	0.5
Alkalinity, mg$CaCO_3$/L	10013	9888	9650	9000	7225	11100	10650	9800	6900	11750	1384
COD, mg/L	34188	23840	19975	22440	20240	12810	13750	16650	18800	47800	10381
BOD_5, mg/L	24848	19700	13360	15828	9010	7650	8450	9500	6820	38500	8377
NH_3-N, mg/L	2218	2050	1975	2253	1890	2335	2380	2350	1660	2690	258
TKN, mg/L	2358	2416	2155	2518	2100	2510	2720	2640	1825	2984	294
SS, mg/L	1873	1896	2085	1837	1310	1400	1285	1470	940	2915	535
TP, mg/L	3.1	3.1	4.4	1.9	3.7	1.6	0.8	2.34	0.45	6.36	1.8
Color, PtCo	14453	18728	12775	18190	11100	15900	17200	16500	7400	30500	5577
Chloride, mg/L	3611	2429	3250	4097	1550	4125	4180	4900	725	5150	1250
Fe, mg/L	113.1	79.0	58.4	50.6	50.0	44.5	17.5	4.9	4.91	245.5	51.6
Mn, mg/L	8.48	5.12	2.26	4.48	1.23	0.92	0.82	0.84	0.27	21.1	5.19
Cu, mg/L	0.29	0.22	0.20	0.18	0.18	0.14	0.22	0.11	0.11	0.42	0.07
Zn, mg/L	1.67	0.84	1.30	0.89	0.89	0.68	0.79	0.85	0.48	3.61	0.66
Pb, mg/L	2.06	1.30	0.87	0.56	0.62	0.69	0.40	0.49	0.31	3.57	0.77
Cd, mg/L	0.10	0.12	0.12	0.11	0.08	0.05	0.07	0.08	0.04	0.21	0.04
Cr, mg/L	0.45	0.57	0.78	0.11	0.37	0.15	0.30	0.11	0.07	0.91	0.28
Ni, mg/L	1.12	1.04	1.03	0.50	0.64	0.56	0.39	0.57	0.36	2.23	0.42

ıssibility of inhibition on anaerobic and nitrification processes. TKN/NH_3-N ratios around 1.1 indicate that ost of the nitrogen is in ammonia form.

ƆD concentrations have ranged between 18800 and 47800 mg/L while BOD_5 between 6820 and 38500 g/L. COD and BOD_5 values were higher in summer and lower in winter due to dilution by precipitation in ll and winter. On the other hand, it was quite interesting that such dilution effect was not observed for ımonia. The highest ammonia concentration, 2690 mg/L was in November 1998. BOD_5/COD ratio was ·ger than 0.7 for most samples indicating high biodegradability, and acidic phase of decomposition in the ıdfill.

ɔtal phosphorus concentrations were very low indicating phosphorus deficiency of the leachate. Chloride ncentrations have varied between 725 and 5150 mg/L.

general, heavy metal concentrations were low, except for iron which has varied between 4.91 and 245.5 g/L. Compared with the Omega Hills Landfill (USA), these iron concentrations are quite low (Table 3).

ıaerobic Treatability

ıe anaerobic reactors were operated for more than 230 days and were continuing operation when this paper ıs prepared. Figure 2 shows the changes in COD, pH, alkalinity and NH_3-N throughout the experimental riod of first 230 days. All the reactors (AF, Hybrid and UASB) were started with an organic loading rate 1.3 kg COD/m^3.day. The raw leachate was diluted with dechlorinated tap water for adjusting the propriate organic loading. Therefore, the parameters other than COD were also diluted during this start-up riod. Influent COD concentrations were increased from about 3000 mg/L to about 17000 mg/L with ɛpwise increase in organic loading up to 8.2 kg COD/m^3.day. Higher loadings could not be applied due to ution of COD in winter months. During the last period of experiments, the leachate was fed to the ıctors without and dilution. COD removal efficiencies in the reactors versus organic loading is presented Figure 3.

TABLE 3 CHARACTERISTICS OF LEACHATE IN TURKIYE AND SOME OTHER COUNTRIES

	TURKEY			USA	GREECE	FRANCE
Parameters	**Harmandali MSWL Izmir (1996)** Timur and Ozturk, 1997	**Odayeri MSWL İstanbul (1995-98)** Ozturk et al., 1999	**Hamitler MSWL Bursa (1995-96)**	**The Omega Hills MSWL (1984)** Carter et al. 1984	**Thessaloniki MSWL (1993)** Diamadopoulos, 1994	**Jeandelaincourt Town MSWL (1997)** Amokrane et al., 1997
pH	7.5-7.8	5.6-7.5	6.6-8.4	6.0-7.6	5.6-6.3	8.2
Alkalinity, $mgCaCO_3/L$	7040-13050	11500-13150	-	12260-15670	-	-
COD, mg/L	14900-19980	30100-70000	11760-32380	35800-60950	60000-77500	4100
BOD_5, mg/L	6900-11000	21000-31000	6450-23000	26120-45070	31500-41000	2000
NH_3-N, mg/L	1120-2780	1345-2033	1400	635-1020	900-1510	1040
TKN, mg/L	1350-3280	1630-4490	-	850-1410	1560-2020	-
SS, mg/L	-	1020-3930	1300	1065-2230	1120-7700	200
PO_4^{-2}-P, mg/L	48-79	-	-	-	6.5-22.8	2.4
TP, mg/L	-	1.0-6.0	8	0.6-138	14.6-23.8	-
Chloride, mg/L	5620-6330	-	1210-1706	2990-3620	3780-3820	5420
Fe, mg/L	14.2-44.0	60-130	22.2-95.9	244-1710	8.7-43.0	0.91
Mn, mg/L	0.11-5.3	1.3-2.1	-	-	3.4-12.0	-
Cu, mg/L	0.02-0.13	0.2-0.5	2.1-16.9	0.01-0.19	0.03-4.80	0.39
Zn, mg/L	0.38-1.06	0.4-0.8	8.3-31.8	18.6-82.8	0.50-8.56	0.73
Pb, mg/L	<0.04	0.08-1.4	-	0.01-0.34	<0.5	0.46
Cd, mg/L	<0.01	<0.2	-	0.03-0.17	-	0.10
Cr, mg/L	0.02-0.78	0.5-2.2	-	-	0.18-1.53	-
Ni, mg/L	0.32-0.45	0.65-1.3	-	0.53-2.16	0.23-0.82	0.81

ıll the reactors showed similar performances against organic loadings with efficiencies between 80% and 0%. However the reactors have experienced high ammonia concentrations several times throughout the xperimental period and showed different inhibition levels. The ammonia inhibition has occurred around etween Day 75 and Day 100. Other severe ammonia inhibitions were between Day 130 and Day 145, Day 50 and Day 170, Day 175 and Day 190. No pH control was done until day 190 where ammonia inhibition ecame more severe. Since unionized ammonia is toxic to methane producing bacteria, total ammonia oncentration is not a good parameter for understanding the ammonia inhibition. Therefore, unionized mmonia concentrations in the influent and effluents have been calculated and plotted as can be seen in 'igure 4. pH values in the reactors were always above 8.0 and exceeded 8.5 in some duration of the xperiments.

'he ammonia inhibition was reversible, as it can be seen from rapid increases in COD removal efficiencies ıf reactors, immediately after drop of unionized ammonia concentrations. Attached growth systems namely, naerobic filter and hybrid bed reactor were more resistant to ammonia inhibition than UASB reactor. This vas clearly seen in all cases of ammonia inhibition. Anaerobic filter was the least effected reactor from mmonia inhibition while UASB was the most. Hybrid bed reactor was exhibited a similar performance to naerobic filter although not at the same degree. pH of the influent was decreased to 4.5 with concentrated ICl after day 175 to control ammonia inhibition. pH values of reactors decreased gradually below 8.0 'ollowing the pH control in the influent. Unionized ammonia nitrogen (UAN) values also decreased from ıbout 800 mg/L to 200 mg/L in parallel to pH values. COD removal efficiency in all the reactors have estored back to 80% level soon after the drop in unionized ammonia levels. Details of the ammonia nhibition will be discussed in another paper, which is under preparation.

)n the other hand, clogging was experienced in anaerobic filter in some duration of the experimental study lue to deposition of inorganic precipitates and biomass growth. To overcome the clogging problem, the naerobic filter was flushed with nitrogen gas from the bottom. No clogging was observed later on. In hybrid ›ed and UASB reactors no clogging problem was observed, except in feed and effluent pipes which are less han 1 cm in diameter.

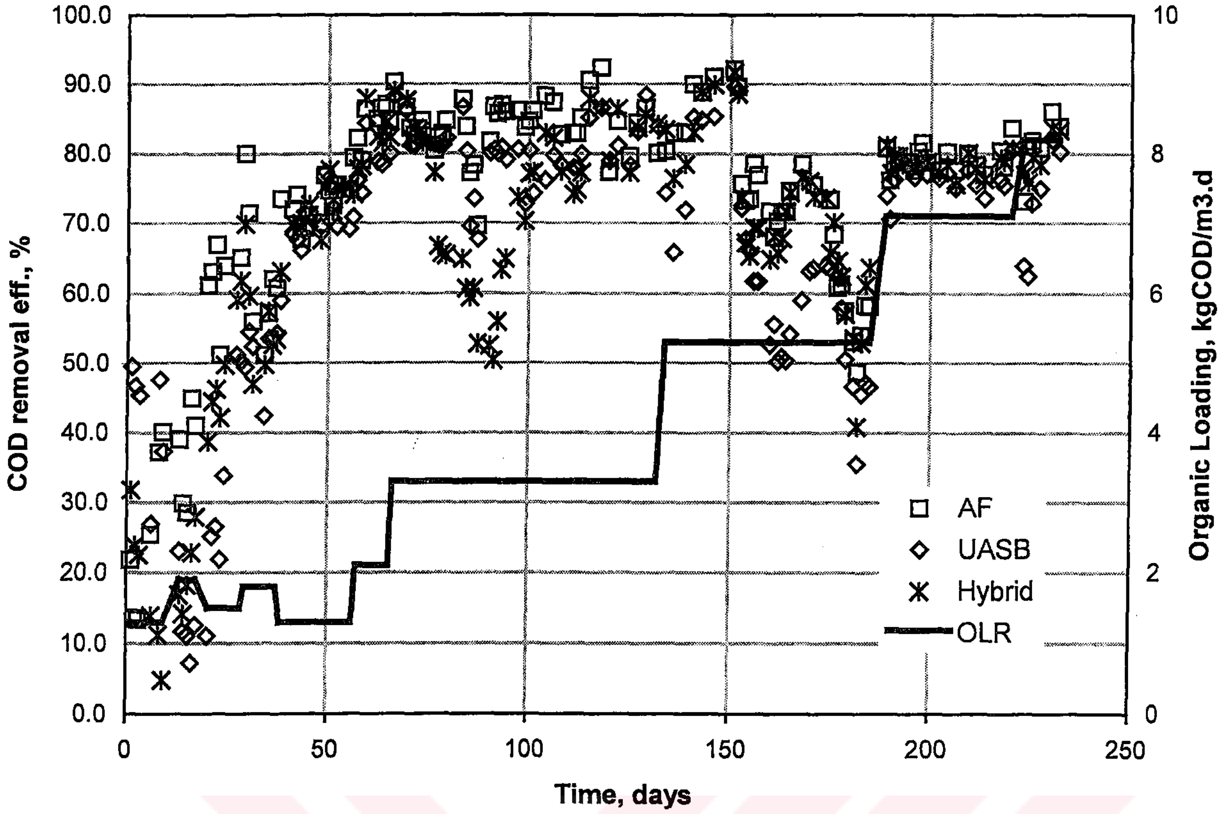

Fig. 3. Reactor efficiencies versus organic loading

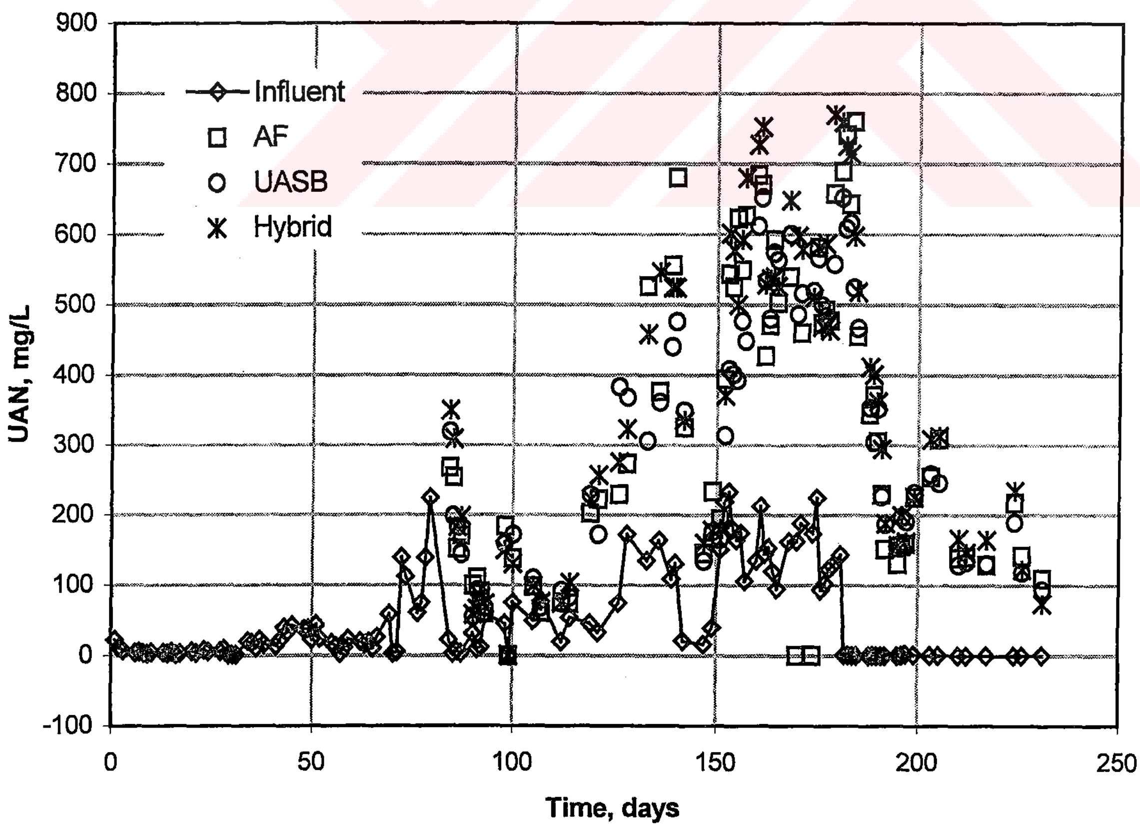

Fig. 4. Change of unionized ammonia concentrations throughout the experimental period

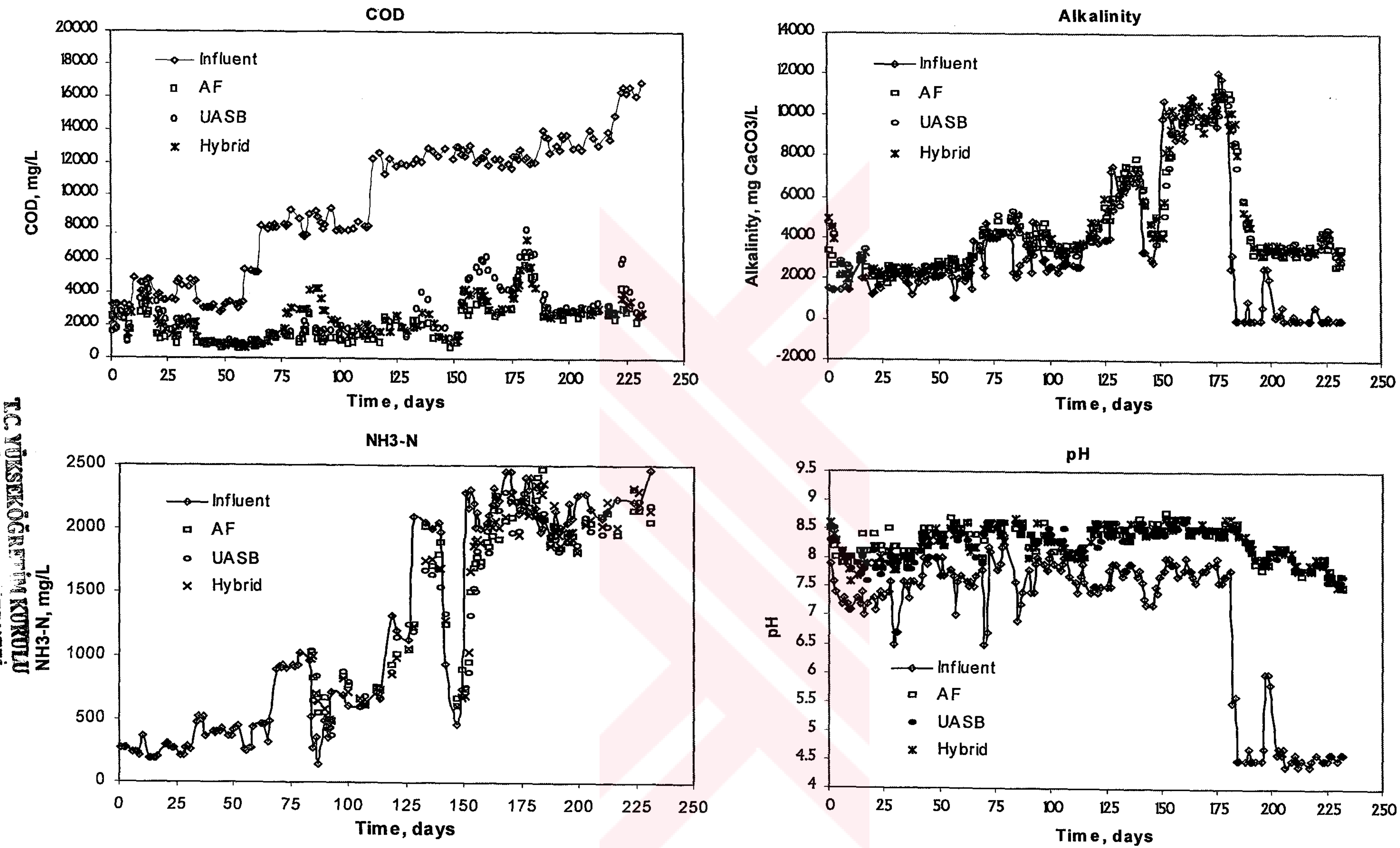

Fig. 2. Change of COD, pH, alkalinity and ammonia concentrations throughout the experimental period

CONCLUSIONS

he following conclusions were drawn from the characterization and anaerobic treatability of the leachate om Komurcuoda Landfill in Istanbul:

High COD and BOD_5 values in the leachate indicate high acidogenic activity in the young landfill.
High BOD_5/COD ratios above 0.7 is an indication of high biodegradability.
High ammonia concentrations as about 2700 mg/L with pH values around 8.0 in raw leachate increase the possibility of ammonia inhibition in anaerobic treatment.
Heavy metal concentrations are below inhibition levels requiring no metal removal prior to biological treatment.
The leachate can be treated anaerobically with efficiencies as high as 90% provided that unionized ammonia concentration is kept below inhibition levels with pH adjustment.
Attached growth processes, namely anaerobic filter and hybrid bed reactor are more resistant to ammonia inhibition than UASB reactor.

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