

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

**INVESTIGATION OF BURSA HAMİTLER LANDFILL**  
**LEACHATE TREATMENT PLANT**

**Korkut KAŞIKÇI**

**THESIS**  
**FOR THE DEGREE OF MASTER OF SCIENCE**  
**IN**  
**ENVIROMENTAL ENGINEERING**

**SUPERVISOR**  
**Assoc. Prof. Barış ÇALLI**

**İSTANBUL 2010**

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# **ABSTRACT**

## **INVESTIGATION OF BURSA HAMİTLER LANDFILL LEACHATE TREATMENT PLANT**

Hamitler Leachate Treatment Plant (HLTP) consisting of aerobic and facultative lagoons and sequencing batch reactors in two parallel lines was constructed in 2004 with 500 m<sup>3</sup>/day capacity to pre-treat the leachate generated in Bursa Hamitler Landfill. The experiences gained in operation of HLTP, which is the first full-scale leachate treatment plant constructed in Turkey, is very important for the design and more efficient operation of new leachate treatment plants. Therefore, in this study, the effects of influent leachate characteristics and different operation practices on COD, BOD<sub>5</sub> and NH<sub>4</sub>-N removal was investigated for 32 months in HLTP between January 2007 and August 2009.

Throughout the study, COD and BOD<sub>5</sub> concentrations ranged from 713 to 38400 mg/L and from 240 to 19000 mg/L, respectively. Meanwhile, incoming leachate flow rate fluctuated between 54 and 2447 m<sup>3</sup>/day. The flow exceeding 650 m<sup>3</sup>/day bypassed the HLTP. These fluctuations in influent leachate characteristics and flow rate due to precipitation deteriorated the treatment performance. In addition, after Line-1 has been closed in June 2008 as an operation practice, the hydraulic load on HLTP doubled and HRT decreased from 23 days to 11 days. As a result, the total BOD<sub>5</sub> removal efficiency decreased from 94% to 78% in average.

Although HLTP was not designed for nitrogen removal, significant NH<sub>4</sub>-N removal was achieved there in summer months. Removal efficiency reached up to 92% in July 2007. The theoretical calculations indicated that, maximum 32% of influent NH<sub>4</sub>-N could be removed via stripping. Besides, about 200 mg/L NO<sub>3</sub>-N measured in the effluent in same period showed that the main NH<sub>4</sub>-N removal process in HLTP is nitrification. Finally, it was estimated that in average while 9% of the influent NH<sub>4</sub>-N was stripped via aeration, 37% was removed via nitrification.

**September, 2010**

**Korkut KAŞIKÇI**

# ÖZET

## BURSA HAMİTLER KATI ATIK DEPO SAHASI SIZINTI SUYU ARITMA TESİSİNİN İNCELENMESİ

Aerobik ve fakültatif lagün ve ardışık kesikli reaktörleri içeren ve paralel 2 hattan oluşan Hamitler Sızıntı Suyu Arıtma Tesisi(HSAT), Bursa Hamitler Katı Atık Depolama Sahası'nda oluşan sızıntı suyunun ön arıtımı için, 2004 yılında 500 m<sup>3</sup>/gün kapasiteli olarak inşa edilmiştir. Ülkemizde kurulan ilk sızıntı suyu arıtma tesisi olan bu tesisin işletimi sırasında kazanılan tecrübeler yeni sızıntı suyu arıtma tesislerinin tasarımı ve daha verimli işletilmesi için oldukça önemlidir. Bu sebeple, Ocak 2007 ve Ağustos 2009 arasında 32 ay boyunca, giriş sızıntı suyu özellikleri ve farklı işletme uygulamalarının KOİ, BOİ<sub>5</sub> ve NH<sub>4</sub>-N giderimi üzerindeki etkileri bu çalışmada incelenmiştir.

Çalışma boyunca, KOİ ve BOİ<sub>5</sub> konsantrasyonları sırasıyla 713-38400 mg/L ve 240-19000 mg/L arasında değişmiştir. Aynı dönemde, sızıntı suyu debisi 54-2447 m<sup>3</sup>/gün aralığında olmuştur. 650 m<sup>3</sup>/gün' ü aşan debi arıtma tesisine alınmayıp by-pass edilmiştir. Giriş sızıntı suyu özelliklerindeki ve debideki yağışa bağlı bu dalgalanmalar arıtma performansına zarar vermiştir. Ayrıca, bir işletme uygulaması olarak Hat-1' in Haziran 2008' de kapatılmasıyla tesise gelen hidrolik yük iki katına çıkmış ve HBS 23 günden 11 güne düşmüştür. Bunun sonucu olarak BOİ<sub>5</sub> giderim verimi ortalama %94' ten %78'e gerilemiştir.

HSAT azot giderimi için inşa edilmemiş olsa bile, yaz aylarında tesiste ciddi NH<sub>4</sub>-N giderimi gerçekleşmiştir. Temmuz 2007' de giderim verimi %92' ye kadar yükselmiştir. Yapılan teorik hesaplamalar girişteki NH<sub>4</sub>-N' in en fazla %32' inin uçurularak giderilebileceğine işaret etmiştir. Bununla birlikte çıkışta ölçülen 200 mg/L NO<sub>3</sub>-N değeri HSAT' deki ana NH<sub>4</sub>-N giderim işleminin nitrifikasyon olduğunu göstermiştir. Son olarak, giriş NH<sub>4</sub>-N değerinin ortalama %9' unun havalandırma ile uçurulduğu, %37' sinin nitrifikasyon ile giderildiği hesaplanmıştır.

Eylül, 2010

Korkut KAŞIKÇI

## SYMBOLS

|                       |                      |
|-----------------------|----------------------|
| <b>E</b>              | : Leachate flow rate |
| <b>°C</b>             | : Degree Celcius     |
| <b>°F</b>             | : Degree Fahrenheit  |
| <b>I</b>              | : Infiltration       |
| <b>°K</b>             | : Degree Kelvin      |
| <b>P</b>              | : Precipitation      |
| <b>R<sub>in</sub></b> | : Surface run-off    |

## ABBREVIATIONS

|                        |                                              |
|------------------------|----------------------------------------------|
| <b>BOD<sub>5</sub></b> | : 5-day Biochemical Oxygen Demand (mg/L)     |
| <b>BNR</b>             | : Biological Nutrients Removal               |
| <b>BUSKİ</b>           | : Bursa Water and Sewerage Administration    |
| <b>COD</b>             | : Chemical Oxygen Demand (mg/L)              |
| <b>DO</b>              | : Dissolved Oxygen concentration (mg/L)      |
| <b>EB</b>              | : Electron Beam                              |
| <b>HLTP</b>            | : Hamitler Leachate Treatment Plant          |
| <b>HRT</b>             | : Hydraulic Retention Time (days)            |
| <b>MAP</b>             | : Magnesium Ammonium Phosphate               |
| <b>MBBR</b>            | : Moving-bed Biofilm Reactor                 |
| <b>MBR</b>             | : Membrane Bioreactors                       |
| <b>MF</b>              | : Microfiltration                            |
| <b>MSW</b>             | : Municipal Solid Waste                      |
| <b>NF</b>              | : Nanofiltration                             |
| <b>PAC</b>             | : Powdered Activated Carbon                  |
| <b>RO</b>              | : Reverse Osmosis                            |
| <b>SBR</b>             | : Sequencing Batch Reactor                   |
| <b>SS</b>              | : Suspended Solids                           |
| <b>TKN</b>             | : Total Kjeldahl Nitrogen                    |
| <b>TOC</b>             | : Total Organic Carbon                       |
| <b>TS</b>              | : Total Solids                               |
| <b>TSS</b>             | : Total Suspended Solids                     |
| <b>UF</b>              | : Ultrafiltration                            |
| <b>UF-BAC</b>          | : Ultrafiltration-biologically Active Carbon |
| <b>US</b>              | : Ultrasound                                 |
| <b>UV</b>              | : Ultraviolet                                |
| <b>WWTP</b>            | : Wastewater Treatment Plant                 |
| <b>VFA</b>             | : Volatile Fatty Acids                       |
| <b>VSS</b>             | : Volatile Suspended Solids                  |

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# **CHAPTER I**

## **INTRODUCTION and AIM**

Increasing wealth in many countries around the world in the past decade has been accompanied by rapid increases in municipal solid waste generation. Municipal Solid Waste (MSW) management constitutes today a major environmental, economical and social problem worldwide, mainly because the waste volume is growing faster than the world's population.

The sanitary landfill method for the final disposal of solid wastes continues to be widely accepted and used due to its economic advantages. Comparative studies of the various possible means of eliminating municipal solid waste (landfilling, incineration, composting and etc.) have shown that the cheapest, in term of exploitation and capital costs, is landfilling (Renou et al., 2008).

Leachates are defined as the aqueous effluent generated as a consequence of rainwater percolation through wastes, biochemical processes within the landfill and the inherent water content of landfilled wastes. As water percolates through the landfill, contaminants are leached from the solid waste. Effective pollution control through the proper design of landfill and leachate management facilities requires an understanding of leachate quality.

In spite of many advantages, generation of heavily polluted leachates, presenting significant variations in both volumetric flow and chemical composition, constitutes a major drawback. Year after year, the recognition of landfill leachate impact on environment has forced authorities to fix more stringent requirements for pollution control and treatment of landfill leachate have become a major environmental concern.

Leachates may contain large amounts of organic matter (biodegradable, but also refractory to biodegradation), where humic-type constituents consist an important group, as well as ammonia-nitrogen, heavy metals, chlorinated organic and inorganic salts. Chemical composition of leachate is directly related to the natural processes

occurring inside the landfill which is greatly depending on the age of landfill and the degree of stabilization (Kulikowska and Klimiuk, 2008).

The removal of organic material based on chemical oxygen demand (COD), biological oxygen demand (BOD<sub>5</sub>) and ammonium from leachate is the usual prerequisite before discharging the leachates into natural waters.

Selection and design of a leachate treatment process is not simple. Important factors that govern the selection and design of treatment facilities include leachate characteristics, effluent discharge alternatives, technological alternatives and costs (Qasim and Chiang, 1994).

In Bursa, MSW have been stored at Hamitler Landfill since May 1996. Hamitler Landfill which has 77 hectares of storage area is placed in the northwest of the city on Mudanya highway and operated by Bursa Metropolitan Municipality. Until 2025, 20 millions of MSW is expected to be landfilled here.

In 2004, a 500 m<sup>3</sup>/day capacity treatment plant was constructed in Hamitler Landfill to pre-treat the leachate before being discharged to sewer line. Hamitler Leachate Treatment Plant (HLTP) is a 3-stage plant consists of aerobic and facultative lagoons and sequencing batch reactors operated in two 2 parallel lines.

The aim of this study is to investigate the time based variation in the characteristics of leachate generated in Hamitler Landfill and COD, BOD<sub>5</sub> and NH<sub>4</sub>-N removal performances of HLTP for 32 months between January 2007 and August 2009. In order to evaluate the treatment performance of HLTP, organic and hydraulic loads applied, COD, BOD<sub>5</sub> and NH<sub>4</sub>-N removal efficiencies and kinetic coefficients of BOD<sub>5</sub> removal were calculated. In addition, the maximum amount of free ammonia that can be stripped was estimated and afterward compared with total NH<sub>4</sub>-N removal in order to explain the prevailing means in nitrogen removal.

## **CHAPTER II**

### **GENERAL BACKGROUND**

#### **II.1. LANDFILLING OF MUNICIPAL SOLID WASTE**

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

The term Municipal Solid Waste (MSW) normally is assumed to include all of the wastes generated in a community with the exception of industrial process wastes and agricultural solid wastes (Tchobanoglous et al., 1993).

Increasingly affluent lifestyles, continuing industrial and commercial growth in many countries around the world in the past decade has been accompanied by rapid increases in both the municipal and industrial solid waste production. MSW generation continues to grow both in per capita and overall terms (Renou et al., 2008). During the latter part of the 1990s, annual waste production ranged from 300 to 800 kg per person in the more developed countries to less than 200 kg in other countries (Warah, 2001).

The composition of solid waste is a crucial element for defining waste management strategies. 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. There are many factors affecting the variability of MSW such as socio-economic conditions, weather, fuels combusted etc. Typically, landfilled wastes are dominated by organics comprising of 50% cellulose, 15% lignin, 10 % hemicellulose, 5% protein as well as starch, pectin and other soluble sugars (Barlaz et al., 1992).

The sanitary landfill method for the ultimate disposal of solid waste material continues to be widely accepted and used due to its economic advantages. Comparative studies of the various possible means of eliminating solid urban waste (landfilling, incineration, composting, . . . , etc.) have shown that the cheapest, in term of exploitation and capital costs, is landfilling (Renou et al., 2008).

Landfilling is defined as “an engineered method of disposing of solid wastes on land in a manner that protects the environment, by spreading the wastes in thin layers, compacting it to smallest practical volume, and covering it with compacted soil by the end of each working day or at more frequent intervals if necessary” (Corbitt, 1989). Sanitary landfills are engineered land burial facilities for the disposal of solid waste which are located, designed, constructed, and operated to contain and isolate the waste so that it does not pose a substantial present or potential hazard to human health or the environment (Tchobanoglous *et al.*, 1993).

#### **II.1.1. Operation of MSW Landfill**

Construction and operation of landfills is generally straightforward, despite the complexity of the natural processes involved in waste decomposition. Before wastes are deposited in the ground, a site is often excavated to allow more waste to be deposited on a given plot of land. The lowest component of a landfill is the liner system, which includes drains and impermeable barriers designed to minimize the migration of the leachate to groundwater. Liners usually consist of layers of compacted clay and geomembrane material (Tchobanoglous *et al.*, 1993).

As solid waste is deposited in the landfills, waste is compacted and covered daily. The total amount of waste a site can receive is determined by the area of the plot and by the maximum slope of the sides of the landfill that still ensure slope stability. Once waste has reached the final design level of the landfill, a final cover is applied. The goals of the final cover are to minimize rainfall and snowfall infiltration, to limit the uncontrolled release of landfill gases, to suppress the proliferation of disease vectors, to limit the risk of fire, and to provide suitable conditions for use of the site after landfill closure.

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 settling 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.

After a load of solid waste is deposited in a landfill cell, the decay process that essentially constitutes the backbone of stabilization, involve both aerobic and anaerobic processes. The nature of microbial growth and the landfill environment

will give rise to a succession of major types of microorganisms depending on the stages of stabilization.

As waste degrades, it compacts as mass is converted to methane and escapes. The capacity of a landfill is defined by the available volume is a function of plot area and maximum slope. Faster decomposition provides faster settling in turn yields more waste capacity in a given landfill resulting in less need for more landfills. Understanding the decomposition processes is therefore important to reducing the need for more landfills.

Within the landfill biological, chemical, and physical processes occur that promote the degradation of wastes and result in the production of contaminated leachate and gas. Thus, the landfill design and construction must include elements that permit control of landfill leachate and gas.

#### **II.1.2. Landfill Gas**

Landfill gas is a product of anaerobic decomposition of biodegradable organic waste. For any MSW landfill site, the gas produced will consist primarily of methane ( $\text{CH}_4$ ) and carbon dioxide ( $\text{CO}_2$ ). In theory, the microbial process of waste degradation should yield equal amounts of methane and carbon dioxide. However, a small proportion of carbon dioxide dissolves in aqueous solution, resulting in typical concentrations of around 50-55% methane and 45% carbon dioxide. Trace gases may include nitrogen, oxygen, hydrogen sulfide, disulfides, and various volatile organic compounds (Tchobanoglous *et al.*, 1993). The rate of gas production commonly peaks at about 40-60 years.

Landfill gas must be controlled because of its potential explosiveness and controlled to reduce atmospheric emission, minimize release of odorous emissions, minimize subsurface gas migration, allow recovery if energy from methane (Tchobanoglous *et al.*, 1993). Also, methane and carbon dioxide are recognized as greenhouse gases that contribute to global warming. Some trace gases can be toxic and therefore pose risks to public health.

#### **II.1.3. Landfill Leachate**

Leachates are defined as the aqueous effluent generated as a consequence of rainwater percolation through wastes, biochemical processes in waste's cells and the inherent water content of wastes themselves.

As water percolates through the landfill, contaminants are leached from the solid waste. Effective pollution control through the proper design of landfill and leachate management facilities requires an understanding of leachate quality.

### II.1.3.1 Leachate Characteristics

Chemical composition of leachate varies greatly depending on the age of landfill and the degree of stabilization. As a result of the broad spectrum of solid waste characteristics, leachate composition will vary tremendously. Leachate may contain large amounts of organic matter (biodegradable, but also refractory to biodegradation), where humic-type constituents consist an important group, as well as ammonia-nitrogen, heavy metals, chlorinated organic and inorganic salts (Renou et al., 2008).

The comparison between the typical leachate characteristics of young and mature landfills are shown in Table II.1.

**Table II. 1** Typical Leachate Characteristics of New and Mature Landfills (Tchobanoglous et al., 1993)

| Constituent, mg/L <sup>a</sup> | New Landfill<br>(less than 2 years) | Mature Landfill<br>(greater than 10 years) |
|--------------------------------|-------------------------------------|--------------------------------------------|
| BOD <sub>5</sub>               | 2,000 – 30,000                      | 100 – 200                                  |
| COD                            | 3,000 – 60,000                      | 80 – 160                                   |
| Total suspended solids         | 200 – 2,000                         | 100 – 400                                  |
| Organic nitrogen               | 10 – 800                            | 80 – 120                                   |
| Ammonia nitrogen               | 10 – 800                            | 20 – 40                                    |
| Nitrate                        | 5 – 40                              | 5 – 10                                     |
| Total phosphorus               | 5 – 100                             | 5 – 10                                     |
| Ortho phosphorus               | 4 – 80                              | 4 – 8                                      |
| Alkalinity                     | 1,000 – 10,000                      | 200 – 1,000                                |
| pH                             | 4.5 – 7.5                           | 6.6 – 7.5                                  |
| Total hardness                 | 300 – 10,000                        | 200 – 500                                  |
| Calcium                        | 200 – 3,000                         | 100 – 400                                  |
| Magnesium                      | 50 – 1,500                          | 50 – 200                                   |
| Potassium                      | 200 – 1,000                         | 50 – 400                                   |
| Sodium                         | 200 – 2,500                         | 100 – 200                                  |
| Chloride                       | 200 – 3,000                         | 100 – 400                                  |
| Sulfate                        | 50 – 1,000                          | 20 – 50                                    |
| Total iron                     | 50 – 1,200                          | 20 – 200                                   |

<sup>a</sup> Except pH which has no unit

The removal of organic material based on chemical oxygen demand, biological oxygen demand and ammonium from leachate is the usual prerequisite before discharging the leachates into natural waters.

The biodegradability of the leachate varies with time. The BOD<sub>5</sub> to COD ratio is an indicator of the proportion of biologically degradable organic matter to total organic matter and it decreases as the landfill ages and more degradation products are leached from deposited residues (Copa *et al.*, 1995). Initially, the ratio will be in the range of 0.5 or greater. Ratio in the range of 0.4 to 0.6 are taken as an indication that the organic matter in the leachate is readily biodegradable. In mature landfills the BOD<sub>5</sub>/COD ratio is often in the range of 0.05-0.2. The ratio drops because of the humic and fulvic acids in the leachate from the mature landfills which are not readily biodegradable (Tchobanoglous *et al.*, 1993).

The class of organic compounds found at highest concentration in leachates is generally volatile fatty acids (e.g. acetic, propionic, and butyric) produced during the decomposition of lipids, proteins, and carbohydrates (Swarbrick, 2001). Aromatic hydrocarbons, including benzene, various xylenes, and toluene, are also frequently found at lower concentrations. According to Oman and Hynning (1993), a total of 150 different organic compounds have been identified in multiple studies, however only 29 were identified in more than one. And it was therefore concluded that leachate composition was quite site specific.

A variety of heavy metals are frequently found in landfill leachates including zinc, copper, cadmium, lead, nickel, chromium, and mercury (Lu, *et al.*, 1985). These metals are either soluble components of the refuse or are products of physical processes such as corrosion and complexation. Heavy metal release is a function of characteristics of the leachate such as pH, flow rate, and the concentration of complexing agents. Metal solubilities generally decrease with increasing pH.

#### **II.1.3.2 Factors Affecting Leachate Characteristics**

The two factors characterizing a liquid effluent are the volumetric flow rate and the composition which in the case of leachate are related. Figure II.1 illustrates water cycle in a landfill.

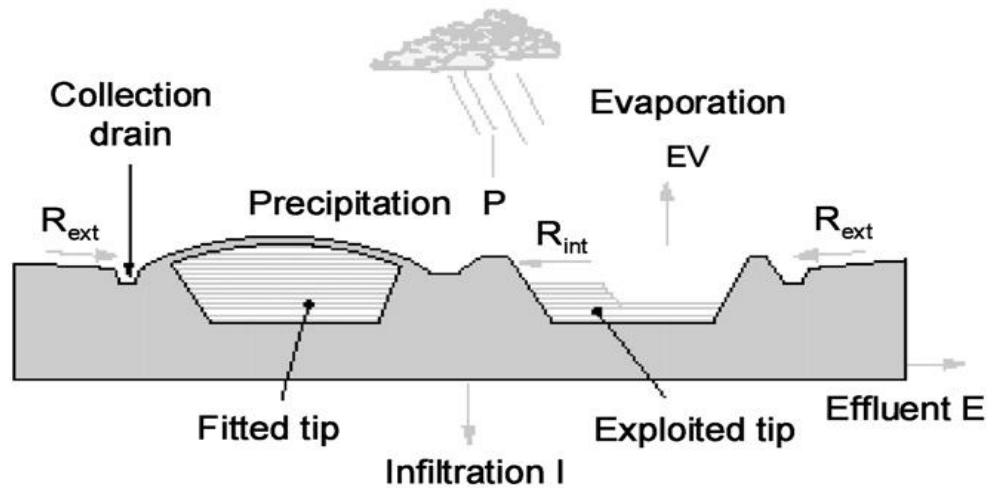


**Figure II. 1** Water cycle in a sanitary landfill (Billard, 2001)

Leachate flow rate ( $E$ ) is closely linked to precipitation ( $P$ ), surface run-off ( $R_{in}$ ,  $R_{ext}$ ), and infiltration ( $I$ ) or intrusion of groundwater percolating through the landfill. Landfilling technique (waterproof covers, liner requirements such as clay, geotextiles and/or plastics) remains primordial to control the quantity of water entering the tip and so, to reduce the threat pollution (Lema et al., 1988). The climate has also a great influence on leachate production because it affects the input of precipitation ( $P$ ) and losses through evaporation ( $EV$ ). Finally, leachate production depends on the nature of the waste itself, namely its water content and its degree of compaction into the tip. The production is generally greater whenever the waste is less compacted, since compaction reduces the filtration rate (Lema et al., 1988).

Chemical composition of the leachate is highly variable depending mainly on the age of landfill. The variation in leachate quality can be attributed to many interacting factors such as the composition and depth of waste, the availability of moisture and oxygen, landfill design and operation, and waste age.

**Waste Composition:** Municipal waste has great variation in composition and characteristics. The waste composition of refuse determines the extent of biological activity within the landfill. Rubbish, food and garden wastes, and crop and animal residues contribute to the organic material in leachate (Pohland and Harper, 1985). Inorganic constituents in leachate are often derived from ash wastes and construction and demolition debris (Pohland and Harper, 1985). Chen and Bowerman (1974) found that increased quantities of paper in solid waste resulted in a decreased rate of

waste decomposition. Lignin, the primary component of paper, is resistant to anaerobic decomposition which is the primary means of degradation in landfills.

Codisposal of municipal solid waste and sludge from municipal wastewater treatment plants (WWTP) can have a significant impact on leachate quality. Codisposal can accelerate leachate formation and the rate of biological stabilization through the addition of moisture, microbes, and nutrients provided by the sludge (Lu *et al.*, 1985). Codisposal of ash with municipal solid waste may provide a means to attenuate the toxic species within the ash (Westlake, 1995). Poon and Chu (1990) found that leachate from codisposal sites contained more chemical oxygen demand (COD), sulfide, and low molecular weight organic acids (acetate and formate). The greater amounts of acetate and formate generated were expected to depress pH and enhance metal mobility. However, sulfide would tend to immobilize metals due to the low solubility of most metal sulfides. Poon and Chu (1990) also determined that codisposal generated more lead and cadmium in the particulate form.

**Landfill Age:** Leachate quality is greatly influenced by the length of time which has elapsed since waste placement. The quantity of chemicals in the waste is finite and, therefore, leachate quality reaches a peak after approximately two to three years followed by a gradual decline in ensuing years (McBean *et al.*, 1995; Lu *et al.*, 1985). Generally, leachate from new landfills will be high in BOD and COD and will then steadily decline, leveling off after about 10 years (Akyurek, 1995). All contaminants do not peak at the same time. Due to their initially biodegradable nature, organic compounds decrease more rapidly than inorganics with increasing age of the landfill. Inorganics are only removed as a result of washout by infiltrating rainwater (Qasim and Chiang, 1994). Organic compounds, however, decrease in concentration through decomposition as well as washout.

**Temperature:** Landfill temperature is an uncontrollable factor influencing leachate quality. It has been shown to fluctuate with seasonal ambient temperature variations (Lu *et al.*, 1985). Temperature affects bacterial growth and chemical reactions within the landfill. Each microorganism possesses an optimum growth temperature, and any deviation from that temperature will decrease growth due to enzyme deactivation and cell wall rupture. Solubility of many salts (e.g.  $\text{Ca}_3(\text{PO}_4)_2$  and  $\text{NaCl}$ ) increases with temperature. However, a number of compounds in leachate, such as  $\text{CaCO}_3$  and  $\text{CaSO}_4$ , show a decrease in solubility with increasing temperature (Lu *et al.*, 1985).

**Moisture:** Water is the most significant factor influencing waste stabilization and leachate quality. Moisture within the landfill serves as a reactant in the hydrolysis reactions, transports nutrients and enzymes, dissolves metabolites, provides pH buffering, dilutes inhibitory compounds, exposes surface area to microbial attack, and controls microbial cell swelling (Noble and Arnold, 1991). Lu *et al.* (1985) stated that high moisture flow rates can flush soluble organics and microbial cells out of the landfill and in such cases microbial activity plays a lesser role in determining leachate quality. Also, high moisture application rates can remove the majority of waste contaminants early in the life of the fill. Under low flow rate conditions, anaerobic microbial activity is the significant factor governing leachate organic strength (McBean *et al.*, 1995). The quantity of moisture is important because it directly affects stabilization rates within the landfill. Sulfita *et al.* (1992) and Miller *et al.* (1994) both noted the important role of moisture in supporting methanogenic fermentation of solid waste when examining samples removed from operating landfills. Relatively dry landfills (i.e. 20 to 40 percent water) have very slow stabilization rates because there is only a small quantity of moisture to support biological degradation. Recommended moisture content reported in the literature ranges from a minimum of 25 percent (wet basis) to optimum levels of 40 to 70 percent (Barlaz *et al.*, 1990; Chen and Bowerman, 1974).

**Oxygen:** The quantity of free oxygen in a landfill indicates the type of decomposition (i.e. anaerobic or aerobic). Aerobic decomposition occurs during initial placement of waste, while oxygen is available. Aerobic degradation may continue to occur at, and just below, the surface of the fill (McBean *et al.*, 1995). Aerobic decomposition products differ greatly from those produced during anaerobic degradation (Bagchi, 1990). During aerobic decomposition, microorganisms degrade organic matter to CO<sub>2</sub>, H<sub>2</sub>O, and partially degraded residual organics, producing considerable heat. High concentrations of organic acids, ammonia, hydrogen, carbon dioxide, methane, and water are produced during anaerobic degradation (McBean *et al.*, 1995). Phase changes occur in a landfill as a result of reductions in the quantity of oxygen in the landfill. For example, a transitional change takes place when oxygen is depleted and an anaerobic environment develops.

## **II.2. TREATMENT OF LEACHATE**

Landfill leachate initially is a high strength wastewater, characterized by low pH, high BOD<sub>5</sub>, COD, and the presence of toxic chemicals (Qasim and Chiang, 1994).

Selection and design of a leachate treatment process is not simple. Important factors that govern the selection and design of treatment facilities include leachate characteristics, effluent discharge alternatives, technological alternatives, costs, and permit requirements (Qasim and Chiang, 1994).

### **II.2.1 Conventional Treatment**

During many years, conventional biological treatments and classical physico-chemical methods are being considered as the most appropriate technologies for manipulation and management of high strength effluents like landfill leachates. When, treating young leachate, biological techniques can yield a reasonable treatment performance with respect to COD, NH<sub>3</sub>-N and heavy metals. When treating stabilized (less biodegradable) leachate, physico-chemical treatments have been found to be suitable as a refining step for biologically treated leachate, in order to remove organic refractory substances. The integrated chemical–physical–biological processes (whatever the order) ameliorates the drawbacks of individual processes contributing to a higher efficacy of the overall treatment (Renou et al., 2008).

However, with the continuous hardening of the discharge standards in most countries and the ageing of landfill sites with more and more stabilized leachates, conventional treatments (biological or physico-chemical) are not sufficient anymore to reach the level of purification needed to fully reduce the negative impact of landfill leachates on the environment. It implies that new treatment alternatives species must be proposed. Therefore, in the last 20 years, more effective treatments based on membrane technology has emerged as a viable treatment alternative to comply and pending water quality regulations in most countries (Renou et al., 2008).

Conventional landfill leachate treatments can be classified into three major groups: (a) leachate transfer: recycling and combined treatment with domestic sewage, (b) biodegradation: aerobic and anaerobic processes and (c) chemical and physical methods: chemical oxidation, adsorption, chemical precipitation, coagulation/flocculation, sedimentation/flotation and air stripping.

### II.2.1.1 Leachate transfer

One of the methods of the leachate treatment is the circulation of high strength leachate back to landfill, so that it may again percolate through the refuse. Recycling leachate back through the tip has been largely used in the past decade because it was one of the least expensive options available (Lema et al., 1988). Bae et al. reported that leachate recirculation increased the moisture content in a controlled reactor system and provided the distribution of nutrients and enzymes between methanogens and solid/liquids. Significant lowering in methane production and COD was observed when the recirculated leachate volume was 30% of the initial waste bed volume (Chung et al., 1998). Also, Rodriguez et al. reported a 63–70% COD lowering in an anaerobic pilot plant with recirculation. The leachate recycle not only improves the leachate quality, but also shortens the time required for stabilization from several decades to 2–3 years (Reinhart and Al-Yousfi, 1996). Although positive effects have been reported on solid waste degradation, limited data are available concerning the recirculation rate impact on treatment efficiency in controlled anaerobic digesters (Rodriguez et al., 2004; Keefe and Chynoweth, 2000; Sponza and Agdag, 2004). High recirculation rates may adversely affect anaerobic degradation of solid wastes. For instance, Ledakowicz and Kaczorek observed that leachate recirculation can lead to the inhibition of methanogenesis as it may cause high concentrations of organic acids ( $\text{pH} < 5$ ) which are toxic for the methanogens. Furthermore, if the volume of leachate recirculated is very high, problems such as saturation, ponding and acidic conditions may occur (Chan et al., 2002; San and Onay, 2001).

Few years ago, a common solution was to treat the leachate together with municipal sewage in the municipal sewage treatment plant. It was preferred for its easy maintenance and low operating costs (Ahn et al., 2002). However, this option has been increasingly questioned due to the presence in the leachate of organic inhibitory compounds with low biodegradability and heavy metals that may reduce treatment efficiency and increase the effluent concentrations (Ceçen and Aktas, 2004). Combined treatment was investigated by Diamadopoulos et al. using a sequencing batch reactor (SBR) consisting of filling, anoxic, oxic and settling phases. When the ratio of sewage to leachate was 9/1, nearly 95%  $\text{BOD}_5$  and 50% nitrogen removals were obtained at the end of the daily cycles. COD and  $\text{NH}_4^+$ -N

reduction decreased with increasing landfill leachate/domestic wastewater ratio (Ceçen and Aktas, 2001). Moreover, the effluent quality may be improved with powdered activated carbon (PAC) addition, particularly if the leachate input exceeds 10%.

#### **II.2.1.2 Biological treatment**

Biological treatment processes involve placing a waste stream in contact with a mixed culture of microorganisms. The microorganisms consume the organic matter in the waste stream. The process must provide optimum environmental conditions for enhanced degradation include controlling the dissolved oxygen level, adding nutrients, increasing the concentration of microorganisms, and maintaining many environmental factors such as pH, temperature, and mixing.

Due to its reliability, simplicity and high cost-effectiveness, biological treatment (suspended/attached growth) is commonly used for the removal of the bulk of leachate containing high concentrations of BOD<sub>5</sub>. Biodegradation is carried out by microorganisms, which can degrade organics compounds to carbondioxide and sludge under aerobic conditions and to biogas (a mixture comprising chiefly CO<sub>2</sub> and CH<sub>4</sub>) under anaerobic conditions (Lema et al., 1988). Biological processes have been shown to be very effective in removing organic and nitrogenous matter from immature leachates when the BOD<sub>5</sub>/COD ratio has a high value (>0.5). With time, the major presence of refractory compounds (mainly humic and fulvic acids) tends to limit process's effectiveness (Renou et al., 2008).

**Aerobic treatment:** An aerobic treatment should allow a partial abatement of biodegradable organic pollutants and should also achieve the ammonium nitrogen nitrification. Aerobic biological processes based on suspended-growth biomass, such as aerated lagoons, conventional activated sludge processes and sequencing batch reactors (SBR), have been widely studied and adopted (Li and Zhao, 2001; Hosomi et al., 1989).

Due to main problems of sludge bulking or inadequate separability (Dollerer and Wilderer, 1996) in conventional aerobic systems, a number of innovative aerobic processes, called attached-growth biomass systems, using biofilm, have been recently developed. These systems present the advantage of not suffer from loss of active biomass. Also nitrification is less affected by low temperatures (Knox, 1985) than in suspended-growth systems, and by inhibition due to high nitrogen content.

Attached-growth systems have recently attracted major interest: the moving-bed biofilm reactor (MBBR) and biofilters. The combination of membrane separation technology and aerobic bioreactors, most commonly called membrane bioreactor, has also led to a new focus on leachate treatment.

*Lagooning:* The principal types of suspended growth lagoon processes, classified based on the manner in which the solids are handled, are; facultative lagoon, aerobic flow through lagoon and aerobic lagoon with solids recycle (Archevia, 1998). Differences in the manner in which the solids are handled affect the treatment efficiency, power requirements, hydraulic and solids retention time, sludge disposal, and environmental considerations (Tchobanoglous, Burton and Stensel, 2003). Aerated lagoons have generally been viewed as an effective and low-cost method for removing pathogens, organic and inorganic matters. Their low operation and maintenance costs have made them a popular choice for wastewater treatment, particularly in developing countries since there is a little need for specialised skills to run the system (Maynary et al., 1999). Maehlum used onsite anaerobic–aerobic lagoons and constructed wetlands for biological treatment of landfill leachate. Overall N, P and Fe removals obtained in this system were above 70% for diluted leachate. However, as stricter requirements are imposed, lagooning may not be a completely satisfactory treatment option for leachate in spite of its lower costs (Zaloum and Abbott, 1997). In particular, the temperature dependence of lagooning is a significant limitation because it mainly affects microbial activity (Renou et al., 2008).

*Activated sludge processes:* Active sludge processes are extensively applied for the treatment of domestic wastewater or for the co-treatment of leachate and sewage. However, this method has been shown in the more recent decades to be inadequate for handling landfill leachate treatment (Lin et al., 2000). Even if processes were proved to be effective for the removal of organic carbon, nutrients and ammonia content, too much disadvantages tend to focus on others technologies:

- inadequate sludge settleability and the need for longer aeration times (Loukidou and Zouboulis, 2001),
- high energy demand and excess sludge production (Hoilijoki et al., 2000),
- microbial inhibition due to high ammonium-nitrogen strength (Lema et al., 1988).

Hoilijoki et al. investigated nitrification of anaerobically pre-treated municipal landfill leachate in lab-scale activated sludge reactor, at different temperatures (5–10 °C) and with the addition of plastic carrier material. Aerobic post treatment produced effluent with 150–500 mg COD/L, less than 7 mg BOD<sub>5</sub>/L and on an average, less than 13 mg NH<sub>4</sub><sup>+</sup>-N/L.

*Sequencing batch reactor:* Sequencing batch reactors are ideally suited to nitrification–denitrification processes since it provides an operation regime compatible with concurrent organic carbon oxidation and nitrification (Diamadopoulos et al., 1997). Kargi and Pamukoglu observed 74 % COD removal with fed batch operation. Also 99% NH<sub>4</sub><sup>+</sup>-N removal has been observed by Lo during the aerobic treatment of domestic leachates in a SBR with a 20–40 days residence time. The greater process flexibility of SBR is particularly important when considering landfill leachate treatment, which have a high degree of variability in quality and quantity (Kennedy and Lentz, 2000).

*Trickling Filters:* Trickling Filter process has been investigated for the biological nitrogen lowering from municipal landfill leachate. Biofilters remain an interesting and attractive option for nitrification due to low-cost filter media (Jokela et al., 2002). In a recent work, above 90% nitrification of leachate was achieved in laboratory and on-site pilot aerobic crushed brick filters with loading rates between 100 and 130 mg NH<sub>4</sub><sup>+</sup>-N/L.day at 25 °C and 50 mg NH<sub>4</sub><sup>+</sup>-N/L.day even at temperatures as low as 5–10 °C, respectively (Jokela et al., 2002).

*Moving-bed biofilm reactor (MBBR):* MBBR process is based on the use of suspended porous polymeric carriers, kept in continuous movement in the aeration tank, while the active biomass grows as a biofilm on the surfaces of them. Main advantages of this method compared to conventional suspended-growth processes seems to be: higher biomass concentrations, no long sludge-settling periods, lower sensitivity to toxic compounds (Loukidou and Zouboulis, 2001) and both organic and high ammonia removals in a single process (Horan et al., 1997). In case of treating high strength ammonia leachate, no inhibition of nitrification is encountered (Welander et al., 1997).

**Anerobic treatment:** An anaerobic digestion treatment of leachates allows to end the process initiated in the tip, being thus particularly suitable for dealing with high strength organic effluents, such as leachate streams from young tips (Pokhrel and Viraraghavan, 2004). Contrary to aerobic processes, anaerobic digestion

conserves energy and produces very few solids, but suffers from low reaction rates (Berrueta and Castrillon, 1992). Moreover, it is possible to use the  $\text{CH}_4$  produced to warm the digester, that usually works at 35 °C and, under favorable conditions, for external purposes.

*Digester:* The high strength organic waste or sludge is processed in an anaerobic digester to stabilize organics and improve stability. The solids are digested by microorganisms, and gases rise bringing the scum to the surface. The gases are collected from the roof of the tank and used to maintain the temperature of the digester and sometimes to heat other parts of plant or produce energy (Qasim and Chiang, 1994).

Anaerobic digestion involves a complex biochemical process in which several groups of facultative and anaerobic organisms simultaneously assimilate and break down the organic material (Qasim and Chiang, 1994).

Typical values of 80–90% and nearly 55% COD removals were reached in anaerobic lab-scale tank at 35 °C and ambient temperature, respectively (Bull et al., 1983; Sung Sung et al., 1997).

*Sequencing batch reactor:* Recently, nutrient reduction from pre-treated leachate was carried out using a lab-scale SBR by Uygur and Kargi. Sequential anaerobic–aerobic operations resulted in COD,  $\text{NH}_4^+\text{-N}$  and  $\text{PO}_4\text{-P}$  removal of 62%, 31% and 19%, respectively, at the end of cycle time (21 h). Kettunen and Rintala showed that COD removal was 35% in the anaerobic stage while in the combined process the COD and  $\text{BOD}_7$  removals were up to 75% and 99%.

*Up-flow anaerobic sludge blanket (UASB) reactor:* UASB process is a modern anaerobic treatment that can have high treatment efficiency and a short hydraulic retention time (Lin et al., 2000). UASB reactors, when they are submitted to high volumetric organic loading rate values (Garcia et al., 1996), have exhibited higher performances compared to other kinds of anaerobic reactors. The process temperatures reported have generally been 20–35 °C for anaerobic treatment with UASB reactors. In these conditions, the average performance of COD decrease efficiency was always higher than 70% at ambient temperature (20–23 °C) and 80% at 35 °C. Up to 92% COD decreases were obtained by Kennedy and Lentz at low and intermediate organic loading rates (between 6 and 19.7 gCOD/L.day). These results show that high-rate treatment at low temperature may minimise the need for heating the leachate prior to treatment, which may thus provide an interesting cost-effective

option (Kettunen et al., 1996). The main disadvantages of such a treatment stay sensitivity to toxic substances (Sung Sung et al., 1997).

*Anaerobic filter:* The anaerobic filter is a high rate system that gathers the advantages of other anaerobic systems and that minimizes the disadvantages. In an up-flow anaerobic filter, biomass is retained as biofilms on support material, such as plastic rings (Nedwell and Reynolds, 1996). For instance, Henry et al. demonstrated that anaerobic filter could reduce the COD by 90%, at loading rates varying from 1.26 to 1.45 kg COD/m<sup>3</sup>.day, and this for different ages of landfill.

*Hybrid bed filter:* Hybrid bed filter consists on an up-flow sludge blanket at the bottom and an anaerobic filter on top. This device acts as a gas–solid separator and enhances solid's retention without causing channeling or short-circuiting (Timur and Ozturk, 1997). Enhanced performances of such a process results from maximization of the biomass concentration in the reactor. Nedwell and Reynolds reported steady-state COD removal efficiencies of 81–97% under methanogenic digestion, depending upon organic loading rate. One drawback of hybrid reactor, as well as anaerobic filter, is the added cost of the support media.

*Fluidized bed reactor:* The combined biodegradation and adsorption process provide a means for removing a variety of organic compounds (Suidan et al., 1993). Imai et al. found that the biological activated carbon fluidized bed process was much more effective for treating old landfill leachate than the conventional one such as activated sludge and fixed film processes.

### II.2.1.3 Physical/chemical treatment

Physical and chemical processes include reduction of suspended solids, colloidal particles, floating material, color, and toxic compounds by either flotation, coagulation/flocculation, adsorption, chemical oxidation and air stripping. Physical/chemical treatments for the landfill leachate are used in addition at the treatment line (pre-treatment or last purification) or to treat a specific pollutant (stripping for ammonia) (Renou et al., 2008).

**Flotation:** For many years, flotation has been extensively used and focused on the decrease of colloids, ions, macromolecules, microorganisms and fibers (Rubio et al., 2002). Recently, Zouboulis et al. investigated the use of flotation in column, as a post-treatment step for removing residual humic acids (nonbiodegradable

compounds) from simulated landfill leachates. Under optimized conditions, almost 60% humic acids removal has been reached.

**Coagulation-flocculation:** Coagulation and flocculation may be used successfully in treating stabilized and old landfill leachates (Silva et al., 2004). It is widely used as a pre-treatment (Tatsi et al., 2003), prior to biological or reverse osmosis step, or as a final polishing treatment step in order to remove non-biodegradable organic matter. Aluminum sulfate, ferrous sulfate, ferric chloride and ferric chloro-sulfate were commonly used as coagulants (Amokrane et al., 1997; Ehrig, 1984). However, this treatment presents some disadvantages: consistent sludge volume is produced and an increase on the concentration of aluminum or iron, in the liquid phase, may be observed (Silva et al., 2004).

**Chemical precipitation:** In the case of leachate treatment, chemical precipitation is widely used as pre-treatment in order to remove high strength of ammonium nitrogen ( $\text{NH}_4^+\text{-N}$ ) (Renou et al., 2008). Li et al. confirmed that the performance of a conventional activated sludge process could be significantly affected by a high concentration of  $\text{NH}_4^+\text{-N}$ . The COD removal declined from 95 to 79%, when the  $\text{NH}_4^+\text{-N}$  concentration in wastewater increased from 50 to 800 mg/l. Li et al. precipitated ammonium ions as magnesium ammonium phosphate (MAP) with the addition of  $\text{MgCl}_2 \cdot 6\text{H}_2\text{O}$  and  $\text{Na}_2\text{HPO}_4 \cdot 12\text{H}_2\text{O}$  with a  $\text{Mg}/\text{NH}_4/\text{PO}_4$  ratio of 1/1/1 at a pH of 8.5–9. Ammonium concentration was reduced from 5600 to 110 mg/l within 15 min by this method.

**Adsorption:** The adsorption of pollutants onto Activated Carbon in columns or in powder form provides better reduction in COD levels than the chemicals methods, whatever the initial organic matter concentration (Renou et al., 2008). The main drawback is the need for frequent regeneration of columns or an equivalently high consumption of powdered activated carbon (PAC). Adsorption by activated carbon has been used along with biological treatment for effective treatment of landfill leachate (Morawe et al., 1995). Non-biodegradable organics, inert COD and the color may be reduced to acceptable levels for biologically treated landfill leachate.

**Chemical oxidation:** Chemical oxidation is a widely studied method for the treatment of effluents containing refractory compounds such as landfill leachate. Growing interest has been recently focused on advanced oxidation processes (AOP). Most of them, except simple ozonation ( $\text{O}_3$ ), use a combination of strong oxidants,

e.g.  $O_3$  and  $H_2O_2$ , irradiation, e.g. ultraviolet (UV), ultrasound (US) or electron beam (EB), and catalysts, e.g. transition metal ions or photocatalyst.

Wable et al., Bigot et al. and Schulte et al. reported organic matter removal efficiency as high as 90% for the  $O_3/H_2O_2$  process. Concerning the  $H_2O_2/UV$  process, the  $BOD_5/COD$  ratio has been increased significantly from 0.1 to 0.45 by Qureshi et al. Also, Steensen reported 85–90% of COD reduction with a biologically pre-treated leachate. Fenton and photo-Fenton processes allow COD decrease efficiency of, respectively, 45–75% and 70–78%. In term of biodegradability improvement,  $BOD_5/COD$  ratios close to 0.5 after oxidation have been reported in recent works using Fenton process (Lopez et al., 2004; Kim et al., 2001).

Common drawbacks of AOP is the high demand of electrical energy for devices such as ozonizers, UV lamps, ultrasounds, which results in rather high treatment costs (Lopez et al., 2004). Also, for complete degradation (mineralization) of the pollutants to occur, high oxidant doses would be required, rendering the process economically expensive.

**Air stripping:** The most common method for eliminating a high concentration of  $NH_4^+$ -N involved in wastewater treatment technologies is air stripping (Renou et al., 2008). High levels of ammonium nitrogen are usually found in landfill leachates, and stripping can be successful for eliminating this pollutant, which can increase wastewater toxicity (Marttinen et al., 2002). If this method is to be efficient, high pH values must be used and the contaminated gas phase must be treated with either  $H_2SO_4$  or  $HCl$  (Renou et al., 2008).

Marttinen et al. reported a 89% ammonia reduction at  $pH = 11$  and  $20\text{ }^\circ\text{C}$  within 24 h retention time. High rates of ammonia removal have been achieved by Cheung et al. in spite of high initial ammonia concentration (0.5–0.7gN/l). Their results showed that 93% of 309–368 mg/l ammonia-nitrogen were removed in free stripping tanks with 1 day retention time. Major concern about ammonia air stripping is the release of  $NH_3$  into the atmosphere so as to cause severe air pollution if ammonia cannot be properly absorbed with either  $H_2SO_4$  or  $HCl$  (Renou et al., 2008). Other drawbacks are the calcium carbonate scaling of the stripping tower, when lime is used for pH adjustment, and the problem of foaming which imposes to use a large stripping tower (Li et al., 1999).

### **II.2.2 New Treatments: the use of membrane processes**

Microfiltration, ultrafiltration, nanofiltration and reverse osmosis are the main membrane processes applied in landfill leachates treatment.

#### **II.2.2.1 Microfiltration (MF)**

MF remains interesting each time that an effective method is required to eliminate colloids and the suspended matter like, for instance, in pre-treatment for another membrane process (UF, NF or RO) or in partnership with chemical treatments. But, it cannot be used alone (Renou et al., 2008). Only Piatkiewicz et al, reported the use of MF as prefiltration stage. No significant retention rate (COD reduction between 25 and 35%) was achieved.

#### **II.2.2.2 Ultrafiltration (UF)**

UF is effective to eliminate the macromolecules and the particles, but it is strongly dependant on the type of material constituting the membrane (Renou et al., 2008). UF may be used as a tool to fractionate organic matter and so to evaluate the preponderant molecular mass of organic pollutants in a given leachate. Also, tests with membrane permeates may give information about recalcitrance and toxicity of the permeated fractions. Syzdek and Ahlert suggested that UF might prove to be effective as a pre-treatment process for reverse osmosis (RO). UF can be used to remove the larger molecular weight components of leachate that tend to foul reverse osmosis membranes. The elimination of polluting substances is never complete (COD between 10 and 75%). More recently, UF has been applied to biological post-treatment of landfill leachate (Bohdziewicz et al., 2001).

#### **II.2.2.3 Membrane bioreactors (MBR)**

The combination of membrane separation technology and bioreactors has led to a new focus on wastewater treatment. It contributes to very compact systems working with a high biomass concentration and achieving a low sludge production with an excellent effluent quality. Membrane bioreactors have been widely applied at full scale on industrial wastewater treatment and some plants have been adapted to leachate treatment (Van Dijk, and Roncken, 1997). Pirbazari et al. used a hybrid technology known as the ultrafiltration-biologically active carbon (UF-BAC) process that amalgamates adsorption, biodegradation and membrane filtration. The process

efficiencies were in the range of 95–98% in terms of TOC reduction, and exceeded 97% for specific organic pollutants. Contrary to conventional systems, organisms such as nitrifiers or organisms which are able to degrade slowly biodegradable substances are not washed out of the system and no loss of process activity occurs.

#### **II.2.2.4 Nanofiltration (NF)**

NF technology offers a versatile approach to meet multiple water quality objectives, such as control of organic, inorganic, and microbial contaminants. NF studied membranes are usually made of polymeric films with a molecular cut-off between 200 and 2000 Da (Renou et al., 2008). The high rejection rate for sulfate ions and for dissolved organic matter together with very low rejection for chloride and sodium reduces the volume of concentrate (Peters, 1998). Nearly 60–70% COD and 50% ammonia were removed by NF, whatever membrane material and geometry (flat, tubular, or spiral wounded), with an average velocity of 3m/s and a transmembrane pressure between 6 and 30 bar. Physical methods were used in combination with nanofiltration and it was found satisfactory for removal of refractory COD from the leachate used. COD removal was 70–80% (Trebouet et al., 2001).

#### **II.2.2.5 Reverse osmosis (RO)**

RO seems to be one of the most promising and efficient methods among the new processes for landfill leachate treatment. In the past, several studies performed both at lab and industrial scale, have already demonstrated RO performances on the separation of pollutants from landfill leachate (Linde et al., 1995). Tubular and spiral wounded modules were the first medium used in the early RO systems for the purification of landfill leachate starting in 1984. An innovative technology was introduced to this market in 1988 with great success: the disc-tube-module (DT-module) developed by Pall-Exekia. Thanks to open channel module, systems can be cleaned with high efficiency with regard to scaling, fouling and especially biofouling (Peters, 1996).

Depending on the salt content of the feed water and the operation time between the cleaning cycles, the operating pressure ranges between 30 and 60 bar at ambient temperature and the specific permeate flux reach 15 L/h.m<sup>2</sup>. The average specific

energy demand is low with less than 5 kWh/m<sup>3</sup> of permeate for a recovery rate of 80% (Peters, 1996).

However, two issues have been identified, and remain today, as major drawbacks for the implementation of pressure-driven membrane processes, and particularly RO, to landfill leachate treatment: membrane fouling and the generation of large volume of concentrate (which is unusable and has to be discharged or further treated) (Renou et al., 2008). In the early 1990s, steady improvement of membrane technology and striving for high water recoveries in landfill leachate treatment resulted in development of a high pressure RO system based on the DT-module and operating at transmembrane pressures of 120 and 200 bar. An adapted process permits to reduce certain salt fractions by controlled precipitation. This means an increase of the permeate recovery from about 80% to 90% with a concentration factor of 10 and a reduction of concentrate volume (Rautenbach et al., 2000).

## **CHAPTER III**

### **THE STUDY**

#### **III.1. DESCRIPTION OF HAMİTLER LEACHATE TREATMENT PLANT**

In Bursa, municipal solid wastes have been stored at Hamitler Landfill which is placed northwest of the city on Mudanya highway with 77 hectares of storage area. Landfill is operated by Bursa Metropolitan Municipality and expected to accept 20 million tons MSW until 2025.

Hamitler Leachate Treatment Plant (HLTP), which consists of aerobic and facultative lagoons and sequencing batch activated sludge systems, with 2 parallel lines and 500 m<sup>3</sup>/day flow capacity, has been constructed as a 3 stage pre-treatment plant by EMIT-Sistemyapı-OTV Consortium in order to pre-treat the leachate generated in Bursa Hamitler Landfill.

Completely mixed aerobic and facultative lagoons followed by a sequencing batch reactor (SBR) were constructed to reduce the load of the BOD<sub>5</sub> and COD and consequently it was aimed to reduce the needs for the final active sludge facility by keeping the net sludge production at minor level which formed as a result of degradation and stabilization, with economic and low technology systems (lagoons).

The general layout of Hamitler Leachate Treatment Plant is given in Figure III.1.

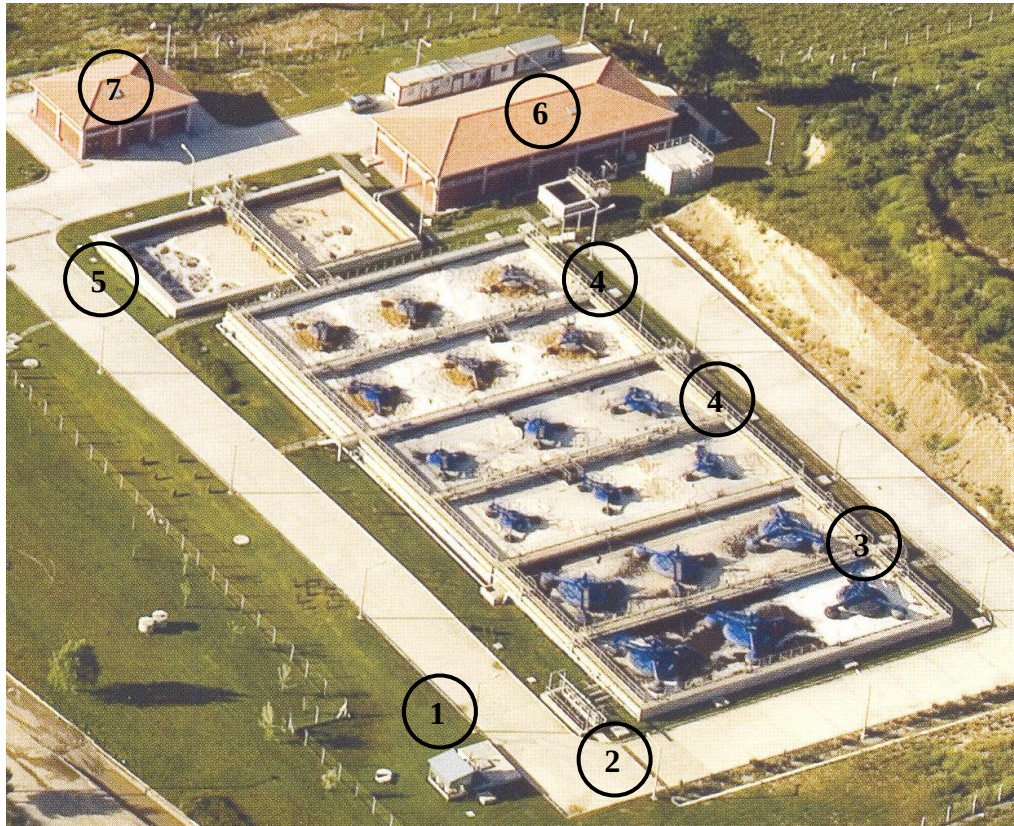


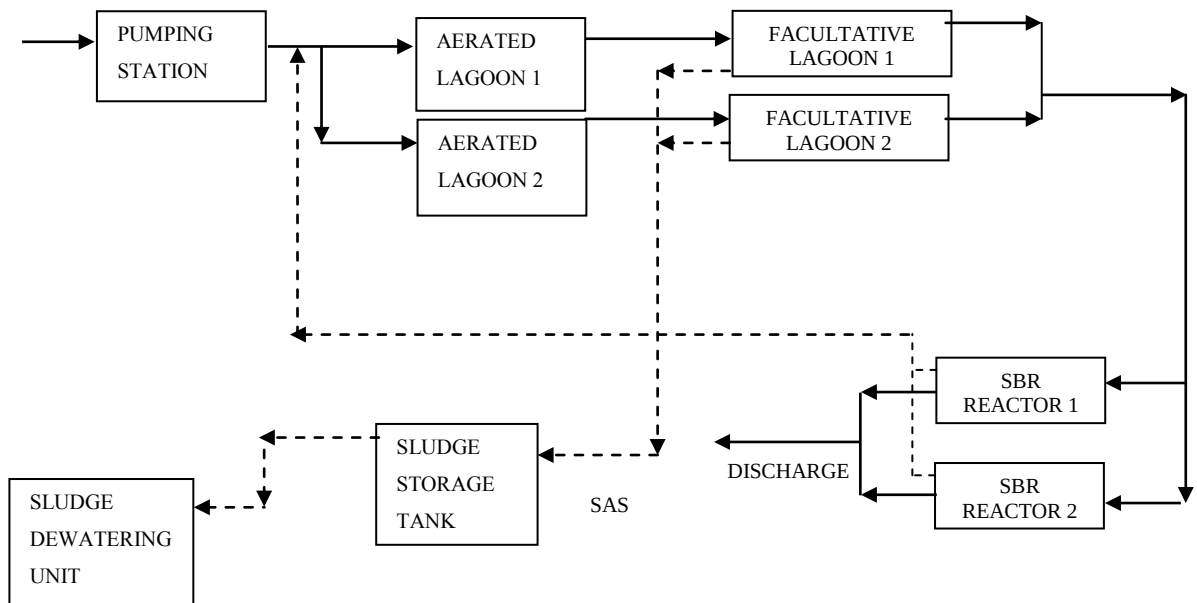
**Figure III. 1** General Layout of Hamitler Leachate Treatment Plant: 1.Acid Cleaning and Neutralization System, 2.Pumping Station, 3.Aerobic Lagoons, 4.Facultative Lagoons, 5.Sequencing Batch Reactors, 6.Operation Building, 7.Transformer Building

The main design parameters of Hamitler Leachate Treatment Plant are given in Table III.1.

**Table III. 1** Main design parameters of Hamitler Leachate Treatment Plant

| Parameter                    | Unit              | Value  |
|------------------------------|-------------------|--------|
| Daily flow rate              | m <sup>3</sup> /d | 500    |
| <u>BOD<sub>5</sub></u>       |                   |        |
| Daily load                   | kg/d              | 7,500  |
| Concentration                | mg/l              | 15,000 |
| <u>COD</u>                   |                   |        |
| Daily load                   | kg/d              | 15,000 |
| Concentration                | mg/l              | 30,000 |
| <u>SS</u>                    |                   |        |
| Daily load                   | kg/d              | 750    |
| Concentration                | mg/l              | 1,500  |
| <u>Temperature</u>           |                   |        |
| Average leachate temperature | °C                | 18     |

HLTP consists of the following units: Inlet Structure, Pumping Station, Acid Cleaning and Neutralization System, Aerobic Lagoons, Facultative Lagoons, Sequencing Batch Reactors, Sludge Storage Tank and Sludge Dewatering Unit. The flow diagram of HLTP is given in Figure III.2.

**Figure III.2** Flow Diagram of HLTP (straight arrows represent leachate lines, striped arrows represent sludge lines)

Discharge standard of HLTP based on COD was specified as below 400 mg/l in the contract if the amount of easily biodegradable COD is above 29,600 mg/l in the influent.

HLTP was designed and constructed in 2004 as a 3-step biological treatment plant targeting the discharge standard given above. However, when the treatment plant started to operate, between October 2004 and July 2005, the effluent COD concentration could not be reduced below 2000 mg/l. Treatability studies proved that raw leachate contains 2,000-2,500 mg/l inert COD on average. Therefore, on 12.07.2005 BUSKİ (Bursa Water and Sewerage Administration) increased the monthly discharge standard of leachate pretreatment to 3,000 mg/l COD and maximum 3-day consecutive discharge standard to 3,200 mg/l COD (Aydoğan et al. 2007). After this date, HLTP was operated according to discharge standard and treated leachate was pumped to Bursa-Western Wastewater Treatment Plant.

#### **III.1.1. Inlet Structure**

Raw leachate is collected with two parallel PVC pipes (DN 300), with gate valves for isolating the lines. One of the pipelines is stand-by. Under the normal operating conditions, leachate is sent to pumping station. In the case of emergency, for example when the treatment plant has to be closed, leachate can be by-passed via by pass pipe. By-pass flow rate is measured with an electromagnetic flow meter (Endress Hauser Promag50).

#### **III.1.2. Acid Cleaning and Neutralization System**

In order to remove the scale formed in leachate collectors, an acid cleaning is applied. The system is composed of:

- Two parallel (duty/stand-by) leachate collectors, with isolating valves;
- A recirculation pump with 50 m<sup>3</sup>/h of pumping capacity
- A hydrochloric acid dosing pump with 100 l/h of dosing capacity
- A neutralization tank that has a 15 m<sup>3</sup> volume with pH meter and a mixing pump with 15 m<sup>3</sup>/h capacity.

When necessary, one of the two collectors is closed and concentrated hydrochloric acid is dosed into this line. Then leachate is diverted to the stand-by line and the recirculation pump provides the necessary circulation of the acid solution, dissolving scales accumulated in the pipe.

After waiting for a while that is defined by operational experience, the acidic solution is taken to the neutralization tank, where caustic soda (commercial solution at  $\approx 30\%$ ) is used for neutralization. The pH of the neutralization tank is monitored by an online pH meter. The complete mixing of chemicals and leachate is done by means of a vertical mixing pump ( $Q=15 \text{ m}^3/\text{h}$ ). The neutralized leachate is then delivered to the pumping station.

The necessity of acid cleaning is determined by checking the pipe section and scale formed visually.

The photos of leachate collection pipe sections at different times during operation are given in Figure III.3.

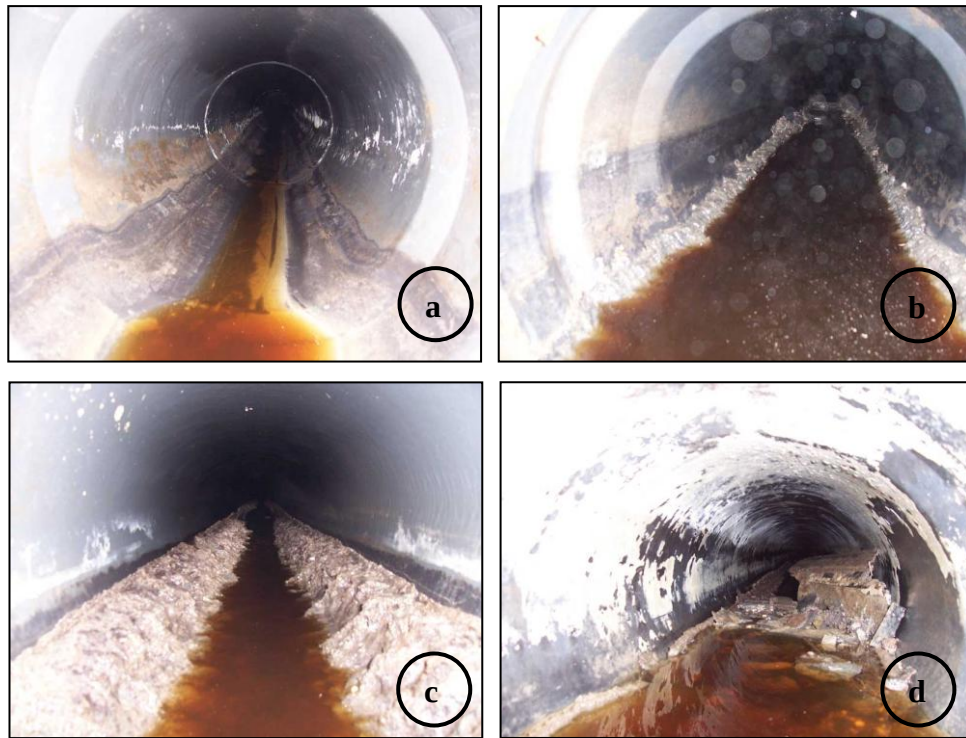


**Figure III.3** Pipe sections of leachate collectors at different times during operation

Figure 3.a was taken after just after the acid cleaning process. As it seen from the Figure 3.b, scale formation begins within the time at the inside sections of the collection pipe. Scale layer thickens gradually over the time (Figure 3.c) and after a certain period it causes partial blockages in the collector line (Figure 3.d).

### III.1.3. Pumping Station

The raw leachate comes by gravity to the sump of pumping station in which the leachate level is controlled automatically. It is then pumped to aerobic lagoons. There are 3 pumps at the pumping station; 2 of them are the main and one is the spare pump. The pH of leachate is measured by an online pH meter located at the sump of pumping station. There is a level sensor and transmitter in the sump to control the pumps by measuring the leachate level. Besides, pumps keep on pumping the leachate to the aerobic lagoons till the highest liquid level is reached. The valves on delivery line are kept open except for the maintenance periods. The flow rate of leachate pumped to Aerated Lagoons is measured by an electromagnetic flow meter (Promag50, Endress Hauser). There is also an automatic sampler (Liquiport2000-RPT20, Endress Hauser) located at the pumping station which takes daily composite raw leachate samples from the sump. A photograph of the pumping station is given in Figure III.4.



**Figure III. 4** The pumping station

### III.1.4. Aerobic Lagoons

In HLTP, the first stage stabilization of organic substances is performed in aerobic lagoons. In aerobic lagoons, organic substance removal occurs according to first degree reaction kinetics depending on leachate temperature and hydraulic retention time. The volume of each aerobic lagoon is  $1680 \text{ m}^3$  and design hydraulic retention time (HRT) is 6.72 days. In the design, it is expected to reduce the 15,000

mg/l influent BOD<sub>5</sub> concentration below 4.000 mg/l with 73% removal efficiency at an ambient temperature as low as 5.5 °C.

Each aerobic lagoon is aerated and mixed by three 37 kW floating mechanical aerators. The energy density is equal to 66 W/m<sup>3</sup> of lagoon and ideal for mixing and aeration (Tchobanoglous, Burton and Stensel, 2003). Dissolved oxygen (DO) concentration in the lagoons is monitored with oxygen meters (Endress Hauser, Liquisys M COM 253F). The speed of aerators is adjusted by a PLC according to minimum and maximum DO levels set at 1.0 and 2.0 mg/l, respectively. This system makes it possible to save energy. In order to provide aerobic conditions, it is necessary to keep DO concentration above 1,0 mg/l. In general, aerobic lagoons are operated at 1.5-2 mg/l DO concentration. When the leachate level is not high enough surface aerators are not operated.

Two aerobic lagoons can be operated in series or parallel depending on the flow rate and COD and BOD<sub>5</sub> concentration of leachate coming to the treatment plant. And sometimes, only one lagoon is used, if the treatment plant receives an average flow rate of lower than 400 m<sup>3</sup>/day. Leachate characteristics have to be taken into consideration when deciding on series or parallel operation and the number of lagoons used. Because of the low flow rate and organic load of the incoming leachate, in HLTP only one of the aerobic lagoons is under operation since January 2008.

A photograph of the Aerobic Lagoons is given in Figure III.5.



**Figure III. 5** Aerobic Lagoons

### III.1.5. Facultative Lagoons

Leachate partially stabilized in aerobic lagoons is taken to facultative lagoons through connection pipes. The facultative lagoons are deeper than aerobic lagoons, but aerated with smaller surface aerators. They are 6 meters deep. The upper part of the Facultative Lagoons (3 m depth) is aerated by 12 mechanical aerators each having 7.5 kW power. The energy density is equal to  $18 \text{ W/m}^3$ . COD removal takes place in the upper aerobic zone.

Facultative lagoons are designed to reduce  $\text{BOD}_5$  concentration from 4000 mg/l to 740 mg/l with 10 days HRT and at a temperature as low as  $5.5^\circ\text{C}$  level.  $\text{BOD}_5$  removal occurs according to 1<sup>st</sup> order reaction kinetics and it depends on ambient temperature and hydraulic retention time like in Aerobic Lagoons.

The bottom 3 m of facultative lagoons are not aerated and mixed. the excess sludge of the aerobic lagoons is taken to this un-mixed anaerobic zone and settles to the bottom. It is assumed that some portion of the sludge is digested here anaerobically. Odorous gases produced during the anaerobic decomposition of the sludge are eliminated at the aerated zone aerated and completely mixed by mechanical aerators. The sludge that accumulates at the bottom is taken with floated submersible pump periodically. During sludge withdrawal mechanical aerators are stopped and the sludge withdrawn is transferred to the sludge storage tank for dewatering.

In the upper zone of the facultative lagoons, DO concentration is kept at 1-2 mg/l (1 mg/l in average) by using DO meters connected to a PLC system.

Mechanical aerators are stopped and the sludge is settled for 15-20 minutes before the treated leachate in the supernatant is taken to the Sequencing Batch Reactors (SBR's). Special care is given not to transfer sludge to the SBR's from the facultative lagoons.

Like in aerobic lagoons, facultative lagoons can also be operated in series or parallel depending on the incoming leachate flow rate and  $\text{BOD}_5$  concentration. Since January 2008 only one of the facultative lagoons is under operation.

A photograph of the Facultative Lagoons is given in Figure III.6.



**Figure III. 6** Facultative Lagoons

### III.1.6. Sequencing Batch Reactor

Stabilization of the leachate in HLTP is completed in an SBR system. The SBR system had been subdivided in 2 reactors (with 1 cycle per day each one), with unitary volume of  $1,176 \text{ m}^3$  and net depth of 6 m. Reactors were designed with a sludge retention time (SRT) of 25.3 days. Because of its sophisticated structure, SBR system needs a close control on the contrary to aerobic and facultative lagoons. Therefore parameters such as time of each stage and oxygen concentration are monitored continuously.

Batch reactors are aerated with 3 blowers (2+1) each having a frequency converter and a capacity of  $512 - 1075 \text{ Nm}^3/\text{h}$ . The air is supplied by diffusers located at the bottom of the reactors. DO concentration is measured online with DO meters and kept at  $2.0 \text{ mg/l}$  by controlling the frequencies of the blowers.

The SBR's are operated as follows.

1-Fill: At this stage, leachate is filled to the batch reactors. During filling, blowers are kept open to maintain aerobic conditions. Filling lasts for approximately 3 hours and is completed when the leachate level reaches 6 meters in the reactor.

2-React: At this stage, reactor is aerated for 11 hours and the remaining BOD<sub>5</sub> in the leachate is removed. 3-Settle: At this stage, the blowers are shut down and the sludge is settled for 5 hours to separate the treated leachate from the sludge.

4-Draw: At this stage, the discharge valves are opened automatically and the treated leachate in the supernatant is transferred to the outlet structure. The drawl

takes approximately 3 hours and is completed when the leachate level decreases to 3.45 m in the reactor.

5-Idle: At this stage, blowers are started again. Meanwhile two submersible pumps run for 2 hours to withdraw the excess sludge and return back to the aerobic lagoons.

The operation of SBR's is summarized in Table III.2.

**Table III. 2** Operation sequences of SBR's in Hamitler Leachate Treatment Plant

| <b>Time, hours</b> | <b>Stage</b> | <b>Duration of Stage, hours</b> | <b>Position of the inlet valve</b> | <b>Position of the outlet valve</b> | <b>Blower</b> | <b>Sludge recirculation Pumps</b> |
|--------------------|--------------|---------------------------------|------------------------------------|-------------------------------------|---------------|-----------------------------------|
| 0 – 3              | Fill         | 3                               | Open                               | Closed                              | Open          | Closed                            |
| 3 – 14             | React        | 11                              | Closed                             | Closed                              | Open          | Closed                            |
| 14 – 19            | Settle       | 5                               | Closed                             | Closed                              | Closed        | Closed                            |
| 19 – 22            | Draw         | 3                               | Closed                             | Open                                | Closed        | Closed                            |
| 22 – 24            | Idle         | 2                               | Closed                             | Closed                              | Open          | Open                              |

A photograph of the SBRs is given in Figure III.7



**Figure III. 7** Sequencing Batch Reactors

### **III.1.7. Outlet Structure**

Treated leachate is sent to the collector with outlet structure. The flow rate of the treated leachate is measured with an electromagnetic flow meter (Promag50, Endress Hauser) located at outlet structure. Like in the inlet, there is an automatic sampler (Liquiport2000-RPT20, Endress Hauser) here to take daily composite samples from the discharge.

### **III.1.8. Sludge Storage Tank and Sludge Dewatering**

Sludge, which is periodically withdrawn from the bottom of the facultative lagoons is collected in the sludge storage tank, and aerated with a venturi aerator till being dewatered. The level in the storage tank is monitored by a level switch and the venturi aerator continues to run until the sludge level decreases below the set point.

Before dewatering the sludge by using a belt press a polyelectrolyte solution is first added to the storage tank. With belt pressing the solids content of the sludge is increased from 4% up to 20%. The dewatered sludge is collected in a container and then sent to the landfill for disposal. The filtrate of belt press is collected in filtrate reservoir and recirculated to the aerobic lagoons. The sludge dewatering facility is located in the management building.

A photograph of the Sludge Dewatering System is given in Figure III.8



**Figure III. 8** Sludge Dewatering System

The laboratory, workshop, blower room, and control room in which the PLC and SCADA systems are located are also in the operation building.

### **III.2. CHARACTERIZATION OF RAW LEACHATE**

In this study, which covers the 32 months between January 2007 and August 2009, the physical and chemical characteristics of Hamitler Landfill leachate such as flow rate, pH, temperature, COD, dissolved COD, BOD<sub>5</sub>, suspended solids (SS), chloride (Cl<sup>-</sup>), total nitrogen (TN), ammonium nitrogen (NH<sub>4</sub>-N) and nitrate nitrogen (NO<sub>3</sub>-N) was determined, and seasonal changes were monitored.

The pH, temperature, COD, dissolved COD and SS of raw leachate were analyzed 4-5 times a week. BOD<sub>5</sub>, Cl<sup>-</sup>, TN, NH<sub>4</sub>-N, and NO<sub>3</sub>-N were analyzed 1-2 times a week. In order to designate the biodegradability of leachate BOD<sub>5</sub>/COD ratio and to monitor the inorganic nitrogen content in total nitrogen NH<sub>4</sub>-N/TN ratio were calculated.

In addition to these parameters, incoming flow rate was measured continuously at the inlet by an electromagnetic flow meter (Promag50, Endress Hauser) to determine the daily produced leachate. In order to determine the effect of rainfall on the flow rate, the daily rainfall data (mm/day) was collected from the nearest meteorology station (Bursa No. 17116) of Turkish State Meteorological Service.

Composite raw leachate samples were taken daily from the inlet of the treatment plant using the automatic sampler (Liquiport 2000-RPT20, Endress Hauser). Daily composite samples were transferred to the laboratory on the site as soon as they were taken and the pH, temperature, COD, dissolved COD, BOD<sub>5</sub>, SS, Cl<sup>-</sup>, TN, NH<sub>4</sub>-N and NO<sub>3</sub>-N analyses were conducted. For dissolved COD analysis, leachate samples were filtered through 0,45 µm pore size filter paper after making the particles to settle down. For other analyses, no special pretreatment was done. In addition to routine analyses program, alkalinity concentrations have been also measured in the plant laboratory once a month between the period of February 2009 and August 2009.

#### **III.2.1. Analytical Methods**

Temperature and pH of the incoming leachate were measured from the composite sample taken from the inlet of treatment plant (Pumping Station) with WTW 330 pH meter. 60-70 ml of sample was taken into the beaker. Mixing of the sample was achieved by using magnetic stirrer and fish. Alkalinity analysis was made in accordance with Titration Method (APHA, AWWA, WEF, 1999).

COD and Soluble COD analyses were made in accordance with Closed Reflux Method (APHA, AWWA, WEF, 1999). For the Soluble COD analysis, 1 ml of zinc sulfate that has a concentration of 100 g/l was added to the 100 ml of sample and it was mixed with magnetic stirrer for 1 minute. pH of the sample was increased to 10.5 using 6M of NaOH . In order for the particulates in the sample to settle down, it was waited for 5 minutes. 20-30 ml of sample from the supernatant of the sample was taken and filtered through filter paper which has a pore size of 0.45  $\mu\text{m}$ . Remaining part of the soluble COD analysis was made in accordance with Closed Reflux Method (APHA, AWWA, WEF, 1999).

BOD<sub>5</sub> analysis was made in accordance with Manometric Method (APHA, AWWA, WEF, 1999). VELD Manometric BOD<sub>5</sub> Set was used for BOD<sub>5</sub> analysis. In order to prevent nitrification in BOD<sub>5</sub> bottles, nitrification inhibitor was added to BOD<sub>5</sub> bottles in accordance with the determined volume of sample for Veld Manometric BOD<sub>5</sub> Set. VELD Manometric BOD<sub>5</sub> Set with adjusted amount of leachate sample was put into incubator in which the temperature was adjusted to  $20 \pm 1$  °C. BOD<sub>5</sub> of the samples was measured for every 24 hours in 5 day period and at the end of the 5 days period BOD<sub>5</sub> was measured from the manometers.

Suspended Solids analysis was made in accordance with Gravimetric Method SM 2540 D (APHA, AWWA, WEF, 1999). Whatman 934-AH (47 mm diameter) glass fiber filter papers were used for SS analysis. SS concentration was calculated from the difference of weight for known amount of sample. Oven that has a temperature of 105 °C and desiccator was also used for SS analysis.

Cl<sup>-</sup>, TN, NH<sub>4</sub>-N and NO<sub>3</sub>-N concentration was measured with Lange LASA-100 Photometer. LCK 311 Chloride Kit was used for chloride analysis. LCK 338 A, LCK138/238/338 B, LCK 138/238/338 C and LCK 138/238/338 D kits were used for the TN analysis. DosiCap Zip kit was used for the NH<sub>4</sub>-N and LCK 339 A solution was used for the NO<sub>3</sub>-N analyses.

### **III.3. EVALUATION OF TREATMENT PERFORMANCE**

In the second part of this study, treatment performance of the HLTP was examined based on COD, BOD<sub>5</sub>, and NH<sub>4</sub>-N removal efficiencies, although there is not any discharge criteria for ammonia. Treatment performances were calculated in accordance to the difference between the inlet and outlet loads.

In this part of the study, BOD<sub>5</sub> removal of HLTP was modeled and kinetic coefficients of biological carbon removal were calculated for Aerobic and Facultative Lagoons in accordance with the real BOD<sub>5</sub> removal performances of each unit. BOD<sub>5</sub> removal efficiencies of SBR units were not modeled because of their low BOD<sub>5</sub> removal efficiency; i.e. high effluent BOD<sub>5</sub> values in accordance to the effluent design value of these units. In addition to this, NH<sub>4</sub>-N removal of HLTP was also modeled in accordance to ammonia stripping. MLVSS, pH and temperature parameters were also monitored in this study.

In addition to these parameters, inlet and outlet flow rates of HLTP were measured and monitored daily to determine the daily amount of leachate treated by two Endress Hauser, Promag50 electromagnetic flow meters located at the Pumping Station and Outlet Structure

Composite samples were taken from the inlet and outlet of the treatment plant with Endress Hauser Liquiport 2000-RPT20 automatic sampling machines every day. 0.5 liters of grab samples were taken from each unit of treatment plant. Daily composite and grab samples were taken to the plant laboratory without loss of time and analyses which are described above were made.

HLTP has operation flexibility and units of the plant can be operated either in serial or parallel mode. During this study period, it has been operated in parallel mode. Each line contains one Aerobic Lagoon, one Facultative Lagoon, and one SBR. In this study, parallel lines were called Line-1 and Line-2 and treatment performance of HLTP was evaluated separately for both of these parallel lines.

### **III.3.1. Analytical Methods**

BOD<sub>5</sub>, COD, NH<sub>4</sub>-N, TN, NO<sub>3</sub>-N, MLVSS, temperature, and pH were determined in accordance to the standard methods as described in Section III.2.1.

BOD<sub>5</sub> analysis was made twice a week for composite inlet and outlet samples and grab samples taken from Aerobic and Facultative Lagoons. COD was analyzed daily for composite inlet and outlet samples and grab samples taken from Aerobic and Facultative Lagoons. BOD<sub>5</sub> and COD analyses were not made for the SBR's and concentrations of these parameters measured at the outlet sample were assumed as the outlet of the SBR's in this study. In order to evaluate nitrogen removal in the HLTP, NH<sub>4</sub>-N, and TN concentrations were monitored at least once a week in the composite inlet and outlet samples. NO<sub>3</sub>-N analysis was also measured in the dates

between January 2007 and December 2008 once or twice a week for inlet and outlet. Bacterial mass concentrations in each unit were determined 6 times a week as MLVSS. Additionally, temperature values of grab samples taken from each unit and pH values of grab samples taken from Aerobic Lagoons and composite outlet samples were measured almost every day during this study period. pH values of Facultative Lagoons and SBR's were not measured, and pH values of Aerobic Lagoon samples were assumed as pH values of Facultative Lagoons and pH values of composite outlet samples were assumed as pH values of SBR units in this study. Monthly averages were used for evaluating treatment performance and modeling of HLTP.

Grab samples collected from Aerobic Lagoons were taken after the mechanical surface aerators have been switched off. Figure III.9 shows the sampling points in Aerobic Lagoon-1 and Aerobic Lagoon-2.



**Figure III. 9** Sampling Points of Aerobic Lagoon-1 (A) and Aerobic Lagoon-2 (B)

Grab samples collected from Facultative Lagoons were taken after the mechanical surface aerators have been switched off. Figure III.10 shows the sampling points of Facultative Lagoon-1 and Facultative Lagoon-2.



**Figure III. 10** Sampling Points of Facultative Lagoon-1 (C) and Facultative Lagoon-2 (D)

Grab supernatant samples collected from SBR's were taken just after the "React" period when the blowers have stopped and microbial mass has started to settle. Figure III.11 shows the sampling points of SBR-1 and SBR-2.



**Figure III. 11** Sampling Points of SBR-1 (E) and SBR-2 (F)

For the evaluation of  $BOD_5$  removal efficiency of each unit,  $BOD_5$  concentration in Aerobic Lagoons was taken as inlet  $BOD_5$  concentration of Facultative Lagoons

and BOD<sub>5</sub> concentration in the Facultative Lagoons was taken as inlet BOD<sub>5</sub> concentration of Sequencing Batch Reactors.

### III.3.2. Organic Matter Removal

Hamitler Leachate Treatment Plant is a 3 stage biological activated sludge system which contains aerobic and facultative lagoons and sequencing batch reactors. HLTP was designed according to aerobic and facultative flow-through lagoons without sludge recycle and activated sludge sequencing batch reactor system based on BOD<sub>5</sub> removal.

Organic substance removal occurs according to the first order reaction kinetics and it depends on leachate temperature and hydraulic retention time in lagoons. BOD<sub>5</sub> removal equation for flow-through lagoons, which is used in the design of aerobic and facultative lagoons, is described as follows (Tchobanoglous, Burton and Stensel, 2003);

$$S_e = \frac{S_0}{(1 + k \times HRT)} \quad (III.1)$$

where  $S_e$  = effluent BOD<sub>5</sub> concentration, mg/L

$S_0$  = influent BOD<sub>5</sub> concentration, mg/L

$k$  = overall first-order BOD<sub>5</sub> removal rate constant at 20 °C, d<sup>-1</sup>

HRT = hydraulic retention time, d

Influent BOD<sub>5</sub> concentration for HLTP was selected as 15,000 mg/L in the design. Overall first order BOD<sub>5</sub> removal-rate constant varies between 0.5 and 1.5 d<sup>-1</sup> for aerobic lagoons and 0.5 and 0.8 d<sup>-1</sup> for facultative lagoons (Tchobanoglous, Burton and Stensel, 2003). This parameter was taken as 1.3 d<sup>-1</sup> for aerobic lagoons, and 0.8 d<sup>-1</sup> for facultative lagoons in the design.

The effect of temperature on the reaction rate of a biological process is expressed as follows (Tchobanoglous, Burton and Stensel, 2003);

$$k_T = k_{20} \times \theta^{(T-20)} \quad (III.2)$$

where  $k_T$  = reaction rate coefficient at temperature T, °C

$k_{20}$  = reaction rate coefficient at 20 °C

$\theta$  = temperature activity coefficient

T = temperature, °C

Lowest temperature value was taken as 5.5 °C, and temperature activity coefficient was selected as standard value of 1.04 for first-order BOD<sub>5</sub> removal constant (Tchobanoglous, Burton and Stensel, 2003).

Hydraulic retention time of Aerobic and Facultative Lagoons were calculated according to the inlet flow rate as described as follows;

$$HRT = \frac{V_{lagoon}}{Q} \quad (III.3)$$

where  $V_{lagoon}$  = volume of each lagoon, m<sup>3</sup>

$Q$  = flow rate to the each lagoon, m<sup>3</sup>/day

The volume of each Aerobic Lagoon is 1680 m<sup>3</sup>, and volume of each Facultative Lagoon is 5040 m<sup>3</sup>. In facultative lagoons, the only 2520 m<sup>3</sup> upper aerobic zone volume was used in the evaluation of BOD<sub>5</sub> removal. In this manner, HRTs in aerobic and facultative lagoons were calculated as 6.72 days and 10.08 days with 500 m<sup>3</sup>/day design flow rate, respectively.

Aerobic Lagoons were designed to decrease BOD<sub>5</sub> concentration to 4,080 mg/L with 72.8% removal efficiency and Facultative Lagoons were designed to decrease this value down to 738 mg/L with 82% removal efficiency by using these parameters.

As the Sequencing Batch Reactors were designed as a polishing step after aerobic and facultative lagoons to remove the residual BOD<sub>5</sub>, it is difficult to determine the BOD<sub>5</sub> removal kinetics in SBRs based on highly fluctuating effluent quality of lagoons. Therefore, no study was performed here to evaluate the BOD<sub>5</sub> removal in SBR's.

#### **III.3.2.1. BOD<sub>5</sub> removal rate constants of Aerobic and Facultative Lagoons**

The kinetic coefficients for the biological carbon removal (first-order BOD<sub>5</sub> removal rate constant) were calculated for the Aerobic and Facultative Lagoons of the HLTP in accordance with the real BOD<sub>5</sub> removal performances of the each lagoon. Overall first-order BOD<sub>5</sub> removal rate constant was calculated for Line-1 and Line-2 lagoons separately. In order to find first order BOD<sub>5</sub> removal rate constant of lagoons at the measured lagoon temperature, equation described in III.1 was substituted into the following equation (III.4)

$$k = \frac{\left(\frac{S_0}{S_e}\right) - 1}{HRT} \quad (III.4)$$

Additionally, to find overall first-order BOD<sub>5</sub> removal rate constant, equation described in III.2 was substituted into equation (III.5)

$$k_{20} = \frac{k_T}{\theta^{(T-20)}} \quad (III.5)$$

Influent BOD<sub>5</sub> concentration was assumed to be same for Aerobic Lagoon-1 and Aerobic Lagoon-2 and the value determined in the composite sample taken from the inlet of the treatment plant was used. Because the aerobic lagoons are completely mixed condition in, BOD<sub>5</sub> concentration measured in the Aerobic Lagoon-1 and Aerobic Lagoon-2 was estimated as the effluent BOD<sub>5</sub> concentration.

Effluent BOD<sub>5</sub> concentration for Aerobic Lagoon-1 and Aerobic Lagoon-2 was taken as influent BOD<sub>5</sub> concentration for Facultative Lagoon-1 and Facultative Lagoon-2. The upper aerobic zone of the facultative lagoons are assumed to be completely mixed therefore, BOD<sub>5</sub> concentration measured in Facultative Lagoon-1 and Facultative Lagoon-2 was used as the effluent BOD<sub>5</sub> concentration.

### III.3.3. Ammonium Removal

The nitrification/denitrification is the most common process used for removal of ammonium from municipal wastewater, but this is not the only common way of ammonium removal for high-strength ammonium containing wastewater like leachate, where physical–chemical systems such as stripping are also frequently used. In HLTP, ammonium removal is considered to occur in the above mentioned two different ways: biologically via nitrification/denitrification and physically via stripping.

In this study, the amount of NH<sub>4</sub>-N removed via stripping in HLTP was calculated based on free ammonia concentration and the amount of air used in aeration. Subsequently, the calculated results were compared with the total amount of NH<sub>4</sub>-N removed. In addition, the effect of free ammonia on biological nitrogen removal was assessed by evaluating the overall ammonium performance of HLTP.

### III.3.3.1. Ammonia Stripping

Gas stripping involves the mass transfer of a gas from the liquid phase to the gas phase. The transfer is accomplished by contacting the liquid containing the gas that is to be stripped with a gas (usually air) that does not contain the gas initially (Tchobanoglous, Burton and Stensel, 2003).

The air stripping of ammonia from leachate requires that the ammonia present as gas. Ammonium ion in leachate exists in equilibrium with gaseous ammonia, shown as follows (Eldem et al, 2004);



Relative amounts of ammonium ion ( $\text{NH}_4$ ) and free ammonia ( $\text{NH}_3$ ) is depended on pH and temperature, and free ammonia concentration was calculated by using the equation given as follows (Eldem et al, 2004);

$$\text{NH}_3 - \text{N} = \frac{\text{NH}_4 - \text{N}}{1 + 10^{(pK_a - pH)}} \quad (\text{III.7})$$

where  $pK_a$  = ionization constant of ammonium

Ionization constant of the ammonium was calculated by using equation given as follows (Emerson et al, 1975);

$$pK_a = 0.09018 + \frac{2729.92}{T} \quad (\text{III.8})$$

where  $T$  = temperature, ° K

Ammonium ion concentration was only measured at the inlet and outlet of the treatment plant. With the lack of ammonium concentration at each unit, the amount of ammonia stripped was calculated according to the effluent  $\text{NH}_4\text{-N}$  concentrations measured in the composite sample taken from the outlet structure of the HLTP. Averages of temperature and pH values, measured from grab samples taken from each unit, were used for calculating free ammonia concentration in these units.

Ammonia stripping modeling is based on standard equation given as follows (Tchobanoglous, Burton and Stensel, 2003);

$$\frac{G}{L} = \frac{C_o - C_e}{y_e} \quad (\text{III.9})$$

where  $G/L$  = Gas-to-liquid ratio, mole air / mole  $\text{H}_2\text{O}$  (leachate)

$C_o$  = influent mole fraction of free ammonia, mole  $\text{NH}_3$  / mole leachate

$C_e$  = effluent mole fraction of free ammonia, mole  $\text{NH}_3$  / mole leachate

$y_e$  = mole fraction of ammonia in the air leaving the surface of unit, mole  $\text{NH}_3$ / mole air

In this equation, influent mole fraction of free ammonia ( $C_0$ ) in SBR units was calculated in accordance with effluent mole fraction of free ammonia ( $C_e$ ) and difference is estimated as stripped in this unit. The amount of ammonia stripped in SBR units was calculated by using the  $\text{NH}_4\text{-N}$  concentration in the effluent of HLTP, and temperature measured in SBR units and pH measured at the outlet of plant (considered as pH in SBR units).

The value of G/L represents the theoretical minimum amount of air that can be used for stripping for the given conditions. Oxygen introduced to each unit is important for calculating G/L and it was calculated for each unit in this study.

Aeration of Sequencing Batch Reactor is supplied with three blowers (2+1) equipped with frequency converter and each blower has a capacity of 512  $\text{Nm}^3/\text{h}$ -1075  $\text{Nm}^3/\text{h}$ . According to the operational conditions, SBR-1 and SBR-2 are aerated for eight hours with a capacity of 1075  $\text{Nm}^3/\text{h}$  and these operating parameters were accepted for the Ammonia Stripping Modeling for SBR's.

Afterwards, specific air input was calculated as follows;

$$\text{Specific air flow rate} = \frac{Q_{\text{air}}}{Q} \quad (\text{III.10})$$

where Specific air flowrate =  $\text{m}^3$  of air/ $\text{m}^3$  of leachate (or L of air/L of leachate)

$Q$  = leachate flowrate,  $\text{m}^3/\text{day}$

Leachate flow rate, measured with electromagnetic flow meter at the inlet pumping station, was used in the calculation of specific air flow rate. The molecular weight and density of water was taken as 18 g/mole and 1000 g/L, respectively. As a result volume of one mol of water was calculated as 0.018 L. The volume of one mol of air is assumed to be 24.1 L/mole at 20 °C (Tchobanoglous, Burton and Stensel, 2003). Using these parameters, G/L was calculated according to following formula

$$\frac{G}{L} = \text{Specific air flowrate} \times \frac{0.018}{24.1} \quad (\text{III.11})$$

where G/L = gas-to-liquid ratio, mole air / mole leachate

The mole fraction of free ammonia was calculated using the following equation;

$$C_o = \frac{n_{NH_3}}{n_{leachate} + n_{NH_3}} \quad (III.12)$$

where  $n_{NH_3}$  = number of moles of  $NH_3$   
 $n_{leachate}$  = number of moles of leachate, 55.5 moles

As it is formerly stated, one mol of leachate which is assumed to be water is equal to 0.018 moles. In this condition one liter of water equals to 55.5 moles as it is seen with equation given as

$$n_{leachate} = 1 \text{ liter of leachate} \times \frac{\text{mole}}{0.018 \text{ liter}} = 55.5 \text{ moles leachate} \quad (III.13)$$

Mole fraction of ammonia in the air leaving the surface of unit ( $y_e$ ) was calculated with equation given as

$$y_e = \frac{H}{P_T} \times C_o \quad (III.14)$$

where  $H$  = Henry's law constant,  $\frac{\text{atm}(\text{mole gas} / \text{mole air})}{(\text{mole gas} / \text{mole water})}$

$P_T$  = total pressure, 1 atm

In the literature, Henry's law constant is often reported as atm, with the mole fraction being implied. Henry's law constant is a function of type of gas, temperature and nature of the liquid (Tchobanoglous, Burton and Stensel, 2003). The change in the Henry's law constant with temperature can be estimated using the following modified form of the van't Hoff-Arrhenius relationship given as follows (Tchobanoglous, Burton and Stensel, 2003);

$$\log_{10} H = \frac{-A}{T} \times B \quad (III.15)$$

where  $A$  = empirical constant at temperature  $T$ , atm due to the dissolution of a component in water and universal gas law constant

$T$  = temperature, K = 273.15 + °C

$B$  = empirical constant

Values of Henry's law constants for various gases in the water at 20 °C and temperature dependent coefficients are given in Table III.3.

**Table III. 3** Henry's law constants at 20 °C and temperature dependent coefficients (Tchobanoglous, Burton and Stensel, 2003)

| Parameter      | Henry's constant,<br>atm | Temperature coefficients |       |
|----------------|--------------------------|--------------------------|-------|
|                |                          | A                        | B     |
| Air            | 66,400                   | 557.60                   | 6.724 |
| Ammonia        | 0.75                     | 1887.12                  | 6.315 |
| Carbon dioxide | 1420                     | 1012.40                  | 6.606 |
| Nitrogen       | 80,400                   | 537.62                   | 6.739 |
| Oxygen         | 41,100                   | 595.27                   | 6.644 |

Henry's law constant and temperature coefficients for ammonia was assumed as 0,75 atm from Table III.3 and the Henry's constant that represents the temperature recorded in each unit was calculated using the following equation;

$$H = 10^{\left(\frac{-188712}{T+27315} + 6.315\right)} \quad (\text{III.16})$$

where  $H$  = Henry's constant at temperature  $T$ , atm  $\left(\frac{\text{mole}.H_2O}{\text{mole}.air}\right)$

The mole fraction of free ammonia in the influent of SBR units ( $C_0$ ) was calculated by using equation III.9. Then,  $C_0$  was converted into mg  $\text{NH}_3\text{-N/L}$  using equation given in III.12. In order to convert free ammonia concentration to  $\text{NH}_4\text{-N}$  concentration, equations given in III.7 and III.8 were used again.

The amount of ammonia that can be stripped from SBR's, facultative lagoons and aerobic lagoons were estimated separately by assuming that the calculated influent ammonium concentration of a unit is the effluent concentration of the previous one.

The amount of ammonia stripped in facultative lagoons was calculated by using the  $\text{NH}_4\text{-N}$  concentration in the influent of SBR units, and temperature measured in facultative lagoons and pH measured in aerobic lagoons (considered as pH in facultative lagoons). Facultative Lagoon-1 and Facultative Lagoon-2 each have six 7.5 kW floating mechanical aerators. Specific oxygen input capacity of each of these mechanical aerators is 2.4 kg  $\text{O}_2/\text{kW.h}$ . These mechanical surface aerators are assumed to be operated for 18 hours in average every day.

Oxygen input to the facultative lagoons as kg O<sub>2</sub>/day was calculated in accordance with the following equation.

$$O_2 \text{ Input, } \frac{kg.O_2}{day} = Sp.O_2 Input \times T \times P \quad (III.17)$$

where Sp. O<sub>2</sub> Input = specific O<sub>2</sub> input, kg O<sub>2</sub>/kW.h

T = operation hour in a day, h/d

T = total power input, kW

The amount of air introduced to the facultative lagoons was calculated as daily air flow rate using equation III.18.

$$Q_{air} = \frac{O_2 Input}{\eta \times \delta} \quad (III.18)$$

where Q<sub>air</sub> = Air flowrate, Nm<sup>3</sup>/d

η = fraction of O<sub>2</sub> in air, 21%

δ = density of air, 1.225 kg/m<sup>3</sup>

The amount of ammonia stripped in aerobic lagoons was calculated by using the NH<sub>4</sub>-N concentration in the influent of facultative lagoons, and temperature and pH measured in aerobic lagoons. Aerobic Lagoon-1 and Aerobic Lagoon-2 are each equipped with three 37 kW floating mechanical aerators. The specific oxygen input capacity of each mechanical aerator was taken as 2.28 kg O<sub>2</sub>/kW.h according to the technical specifications. These mechanical surface aerators in aerobic lagoons are operated for 22 hours in average.

In free ammonia stripping, it was estimated that nitrification activity only took place in aerobic lagoons due to low oxygen concentrations in facultative lagoons and short retention time in SBR units. So, nitrification activity in SBR units and facultative lagoons neglected.

## CHAPTER IV

### RESULTS and DISCUSSION

#### IV.1. CHARACTERIZATION OF RAW LEACHATE

The first part of the study includes chemical and physical characterization of Hamitler Landfill leachate for 32 months between January 2007 and August 2009. There is only limited long-term characterization study performed in our country about landfill leachate comprising seasonal changes (Inanc et al., 2000; Calli et al., 2005). Designing and operation of the new leachate treatment plants will be easier with interpretation of realistic leachate characteristics.

The general characteristics of leachate generated in Hamitler Landfill between January 2007 and August 2009 are given in the Table IV.1. The monthly averages of characterization results are given in the Table IV.2 below.

**Table IV. 1** General characteristics of Hamitler Landfill leachate

| Parameter                     | Number of Measurement | Minimum | Maximum | Average | Standard Deviation |
|-------------------------------|-----------------------|---------|---------|---------|--------------------|
| Flowrate, m <sup>3</sup> /day | 974                   | 54      | 2447    | 455     | 280                |
| pH                            | 973                   | 6.63    | 8.79    | 7.68    | 0.27               |
| Temperature, °C               | 942                   | 3.4     | 29.4    | 17.6    | 5.99               |
| COD, mg/l                     | 973                   | 713     | 38400   | 8154    | 6792               |
| Dissolved COD, mg/l           | 567                   | 690     | 35072   | 7847    | 6476               |
| BOD <sub>5</sub> , mg/l       | 236                   | 240     | 19000   | 4088    | 3655               |
| BOD <sub>5</sub> /COD         | 236                   | 0,19    | 0,88    | 0,49    | 0,11               |
| SS, mg/l                      | 677                   | 38      | 3850    | 584     | 684                |
| Chloride, mg/l                | 119                   | 93      | 7740    | 2982    | 2406               |
| TN, mg/l                      | 138                   | 560     | 3916    | 2363    | 657                |
| NH <sub>4</sub> -N, mg/l      | 135                   | 100     | 3392    | 1723    | 546                |
| NH <sub>4</sub> -N/TN         | 135                   | 0.14    | 0.98    | 0.72    | 0.18               |
| NO <sub>3</sub> -N, mg/l      | 82                    | 5       | 23      | 14      | 4.22               |

**Table IV. 2** Monthly averages of characterization results

| DATE         | Flowrate,<br>m <sup>3</sup> /day | Precipitation,<br>mm | pH   | °C   | COD,<br>mg/l | DCOD,<br>mg/l | SS,<br>mg/l | BOD <sub>5</sub> ,<br>mg/l | Cl <sup>-</sup> ,<br>mg/l | TN,<br>mg/l | NH <sub>4</sub> -N,<br>mg/l | NO <sub>3</sub> -N,<br>mg/l | BOD <sub>5</sub> /<br>COD | NH <sub>4</sub> -N/<br>TN |
|--------------|----------------------------------|----------------------|------|------|--------------|---------------|-------------|----------------------------|---------------------------|-------------|-----------------------------|-----------------------------|---------------------------|---------------------------|
| January-07   | 401                              | 2.8                  | 7.80 | 18.6 | 5727         | 4933          | 166         | 2256                       | 527                       | 2491        | 570                         | 13                          | 0.39                      | 0.23                      |
| February-07  | 356                              | 0.8                  | 7.92 | 10.8 | 4580         | 3893          | 211         | 1694                       | 481                       | 2488        | 1693                        | 11                          | 0.37                      | 0.68                      |
| March-07     | 293                              | 1.9                  | 8.05 | 10.9 | 4124         | 3259          | 276         | 1506                       | 797                       | 2260        | 1998                        | 11                          | 0.37                      | 0.88                      |
| April-07     | 229                              | 1.1                  | 8.09 | 11.8 | 4483         | 3968          | 258         | 1425                       | 875                       | 3118        | 2172                        | 15                          | 0.32                      | 0.70                      |
| May-07       | 196                              | 0.4                  | 8.01 | 19.4 | 5071         | 4495          | 361         | 1481                       | 781                       | 2770        | 2217                        | 18                          | 0.29                      | 0.80                      |
| June-07      | 282                              | 1.6                  | 7.78 | 23.6 | 10684        | 9494          | 874         | 4736                       | 698                       | 2595        | 1996                        | 19                          | 0.44                      | 0.77                      |
| July-07      | 348                              | 0.4                  | 7.54 | 25.9 | 19252        | 16163         | 2,070       | 9814                       | 641                       | 2496        | 1929                        | 21                          | 0.51                      | 0.77                      |
| August-07    | 346                              | 0.0                  | 7.41 | 25.6 | 22947        | 20268         | 2,005       | 11675                      | 614                       | 2619        | 2101                        | 19                          | 0.51                      | 0.80                      |
| September-07 | 255                              | 0.1                  | 7.42 | 21.1 | 23747        | 21463         | 1,351       | 11456                      | 585                       | 3038        | 2551                        | 19                          | 0.48                      | 0.84                      |
| October-07   | 307                              | 3.1                  | 7.47 | 18.3 | 17200        | 14902         | 1,331       | 7650                       | 613                       | 3117        | 2202                        | 18                          | 0.44                      | 0.71                      |
| November-07  | 588                              | 4.7                  | 7.51 | 15.9 | 10640        | 9129          | 621         | 5044                       | 536                       | 1433        | 1019                        | 10                          | 0.47                      | 0.71                      |
| December-07  | 718                              | 5.8                  | 7.51 | 11.2 | 6851         | 6316          | 277         | 3775                       | 324                       | 1358        | 917                         | 8                           | 0.55                      | 0.68                      |
| January-08   | 404                              | 1.8                  | 7.77 | 10.8 | 11105        | 9924          | 347         | 6200                       | 315                       | 2158        | 1881                        | 13                          | 0.56                      | 0.87                      |
| February-08  | 427                              | 1.6                  | 7.82 | 11.5 | 6783         | 5983          | 189         | 3221                       | 3140                      | 2276        | 1653                        | 12                          | 0.47                      | 0.73                      |
| March-08     | 746                              | 3.8                  | 7.74 | 13.1 | 4514         | 3819          | 151         | 2308                       | 1732                      | 1779        | 1367                        | 8                           | 0.51                      | 0.77                      |

**Table IV. 2 (Continued)**

| DATE         | Flowrate,<br>m <sup>3</sup> /day | Precipitation,<br>mm | pH   | °C   | COD,<br>mg/l | DCOD,<br>mg/l | SS,<br>mg/l | BOD <sub>5</sub> ,<br>mg/l | Cl <sup>-</sup> ,<br>mg/l | TN,<br>mg/l | NH <sub>4</sub> -N,<br>mg/l | NO <sub>3</sub> -N,<br>mg/l | BOD <sub>5</sub> /<br>COD | NH <sub>4</sub> -N/<br>TN |
|--------------|----------------------------------|----------------------|------|------|--------------|---------------|-------------|----------------------------|---------------------------|-------------|-----------------------------|-----------------------------|---------------------------|---------------------------|
| April-08     | 457                              | 1.3                  | 7.79 | 16.5 | 4321         | 3922          | 135         | 2113                       | 1978                      | 2148        | 1662                        | 11                          | 0.49                      | 0.77                      |
| May-08       | 490                              | 0.7                  | 7.74 | 18.5 | 4788         | 3523          | 398         | 1575                       | 2989                      | 2476        | 1800                        | 13                          | 0.33                      | 0.73                      |
| June-08      | 291                              | 1.0                  | 7.68 | 18.9 | 5320         | 3732          | 391         | 2017                       | 4342                      | 2405        | 1859                        | 14                          | 0.38                      | 0.77                      |
| July-08      | 319                              | 0.0                  | 7.70 | 23.5 | 5262         | 4467          | 353         | 2910                       | 3515                      | 2435        | 1974                        | 14                          | 0.55                      | 0.81                      |
| August-08    | 369                              | 0.0                  | 7.49 | 26.8 | 9405         | 8354          | 665         | 5338                       | 6857                      | 2733        | 2128                        | 20                          | 0.57                      | 0.78                      |
| September-08 | 596                              | 4.4                  | 7.44 | 23.9 | 10713        | 8334          | 759         | 6300                       | 6623                      | 2499        | 1904                        | 28                          | 0.59                      | 0.76                      |
| October-08   | 522                              | 1.2                  | 7.77 | 20.3 | 3700         | 3124          | 163         | 2261                       | 4302                      | 2523        | 1719                        | 21                          | 0.61                      | 0.68                      |
| November-08  | 423                              | 2.2                  | 7.82 | 16.4 | 3524         | 3188          | 113         | 1571                       | 4927                      | 3155        | 1873                        | 21                          | 0.45                      | 0.59                      |
| December-08  | 535                              | 3.0                  | 7.84 | 14.6 | 3400         | 3061          | 178         | 1404                       | 4993                      | 2286        | 1350                        | 23                          | 0.41                      | 0.59                      |
| January-09   | 721                              | 3.8                  | 7.70 | 11.4 | 3232         | 2766          | 108         | 1672                       | 5435                      | 1688        | 957                         | -                           | 0.52                      | 0.57                      |
| February-09  | 837                              | 5.6                  | 7.70 | 9.9  | 3025         | 2471          | 158         | 1550                       | 2942                      | 1942        | 865                         | -                           | 0.51                      | 0.45                      |
| March-09     | 880                              | 3.9                  | 7.54 | 11.5 | 2514         | 2101          | 173         | 1433                       | 2354                      | 1541        | 1067                        | -                           | 0.57                      | 0.69                      |
| April-09     | 534                              | 0.9                  | 7.41 | 13.3 | 3268         | 3236          | 210         | 1606                       | 3439                      | 2075        | 1530                        | -                           | 0.49                      | 0.74                      |
| May-09       | 415                              | 0.6                  | 7.70 | 17.6 | 3288         | 2516          | 135         | 1621                       | 4657                      | 2028        | 1774                        | -                           | 0.49                      | 0.87                      |
| June-09      | 290                              | 0.3                  | 7.75 | 21.5 | 4469         | 3977          | 497         | 2756                       | 5623                      | 2529        | 2014                        | -                           | 0.62                      | 0.80                      |
| July-09      | 385                              | 0.1                  | 7.58 | 25.9 | 10769        | 10163         | 564         | 6446                       | 5495                      | 2635        | 2016                        | -                           | 0.60                      | 0.76                      |
| August-09    | 541                              | 0.0                  | 7.19 | 25.0 | 21830        | 23083         | 943         | 13829                      | 5886                      | 2624        | 2037                        | -                           | 0.63                      | 0.78                      |

#### **IV.1.1. Flow rate and Daily Precipitation**

Many factors interact to produce variable quantity and quality of leachate from landfills. Leckie et al. (1979) reported some of the most relevant factors that influence leachate generation as follows: annual precipitation, runoff, infiltration, evaporation, transpiration, freezing, mean ambient temperature, waste composition, waste density, initial moisture content and depth of the landfill. They also reported that the stabilization of solid waste placed in sanitary landfill, and quality of leachate are principally the result of physical, chemical, and biological processes; thus, additional variables are water movement, macronutrients, micronutrients, and presence or absence of toxic or inhibitory elements and compounds.

The potential for the formation of leachate can be assessed by preparing water balance on the landfill. The water balance involves summing the amounts of water entering the landfill and subtracting the amounts of water consumed in chemical reactions and the quantity leaving as water vapor (Tchobanoglous et al., 1993). In this study, we evaluated the effect of daily precipitation on the flow rate of landfill leachate generated. Leachate flow rate produced in Hamitler Landfill and daily precipitation amount during this study period are given in Figure IV.1.

In this period, leachate flow rate has changed between 54-2447 m<sup>3</sup>/day. Significant increases have been observed in the flow coming to the plant, especially during periods of increased rainfall as winter and spring. But the 454 m<sup>3</sup>/day of daily average flow rate during this study period is very close to the design flow rate of 500 m<sup>3</sup>/day. The leachate flow rate was always lower during summer months than in winter season throughout the three years study period. Additionally, depending on increasing amount of municipal solid waste landfilled in Hamitler Landfill a gradual increase in leachate flow rate was observed from 2007 to 2009. Yearly average flow rate have been recorded as 363 m<sup>3</sup>/day in 2007, 461 m<sup>3</sup>/day in 2008 and 581 m<sup>3</sup>/day in 2009.

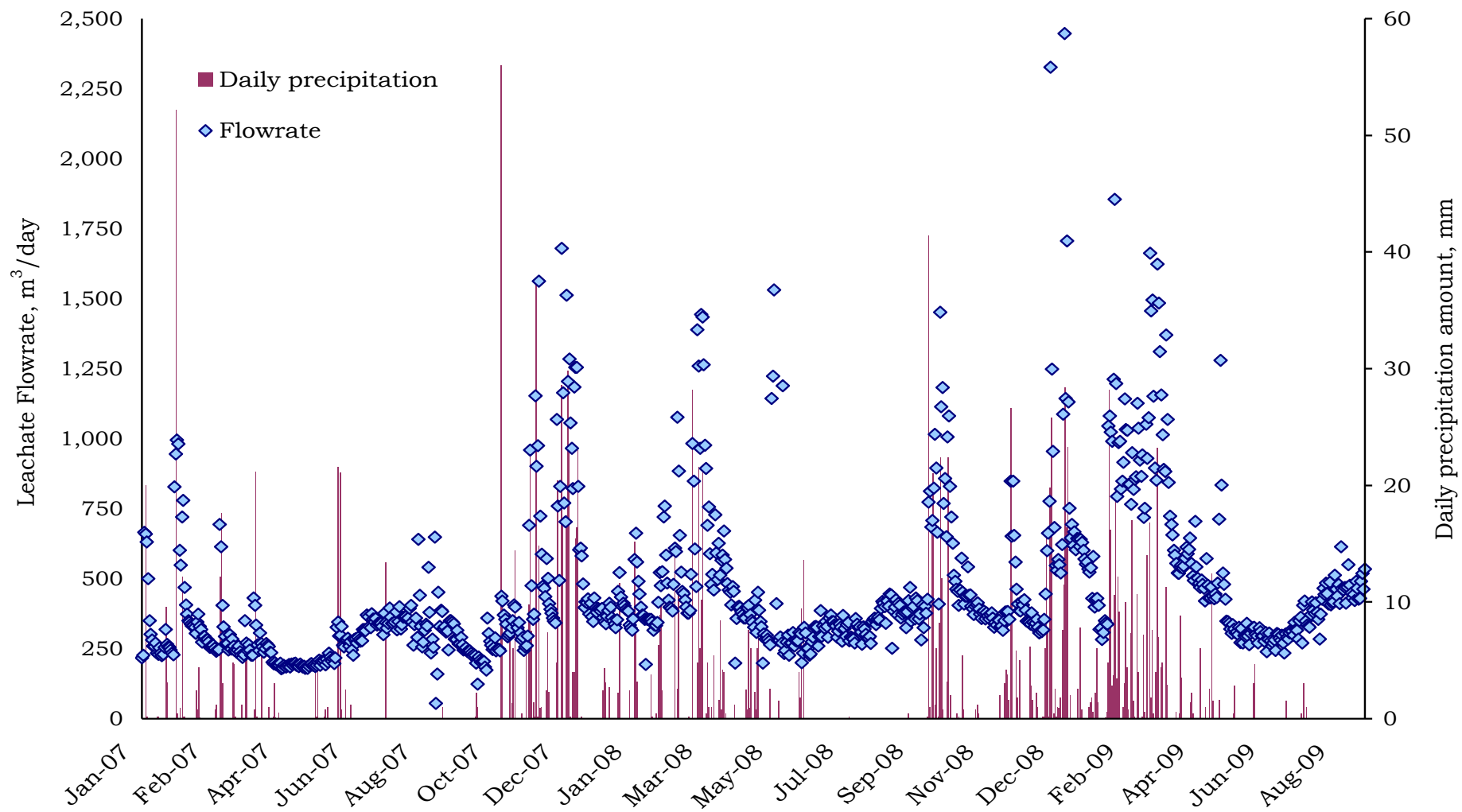


**Figure IV. 1** Change in leachate flow rate and daily precipitation amount between January 2007 and August 2009

#### IV.1.2. COD and BOD<sub>5</sub>

COD represents the total organics and BOD<sub>5</sub> represents the biodegradable portion of organics present in the leachate. Quality of the leachate produced from different landfills in the world changes considerably. Authors reported that there is a wide reduction with time in concentrations of organic matter in leachate (Chian and DeWalle, 1977; Chian, 1977). According to Alvarez-Vazquez et al., (2004) average COD concentration is higher than 15,000 mg/L for young landfill leachate, where it changes between 3000 mg/L and 15000 mg/L for medium age landfill leachate, and it is lower than 3000 mg/L for old landfill leachate.

BOD<sub>5</sub>/COD ratio of leachate is a parameter that can be used to evaluate the stability of the waste in the landfill. The BOD<sub>5</sub>/COD ratio decreases rapidly from 0.70 to 0.04, with the aging of the landfills. This is due to the release of the large recalcitrant organic molecules from the solid wastes (Chian and DeWalle, 1976). According to Heyer and Stegmann (1998), BOD<sub>5</sub>/COD ratio of 0.4 and above indicate acid formation phase and below 0.2 shows methanogenic phase. According to Abbas et al., (2009) average BOD<sub>5</sub>/COD ratio changes between 0.5 and 1 for young landfill leachate, where it changes between 0.1 and 0.5 for medium age landfill leachate, and it is lower than 0.1 for old landfill leachate. Consequently, old landfill leachate is characterized by its low ratio of BOD<sub>5</sub>/COD. Changes in the COD and BOD<sub>5</sub> concentrations and BOD<sub>5</sub>/COD ratio of the Hamitler Landfill leachate are given in Figure IV.2.

High concentrations of COD and BOD<sub>5</sub> ranging from 713 to 38400 mg/l and from 240 to 19000 mg/l respectively, shows that Hamitler Landfill is at acid formation phase. COD and BOD<sub>5</sub> values were higher in summer and lower in winter during 32-month study period due to dilution by precipitation in fall and winter.

BOD<sub>5</sub>/COD ratio of Hamitler Landfill leachate has changed between 0.19 and 0.88 during 32-month study period. Average BOD<sub>5</sub>/COD ratio of Hamitler Landfill leachate has been calculated as 0.49 indicates that it contains high amount of biodegradable organic matter. The ratio higher than 0.4 points out acidic phase of decomposition in the Hamitler Landfill, and leachate produced here can be considered as young to medium age leachate considering the average ratio.

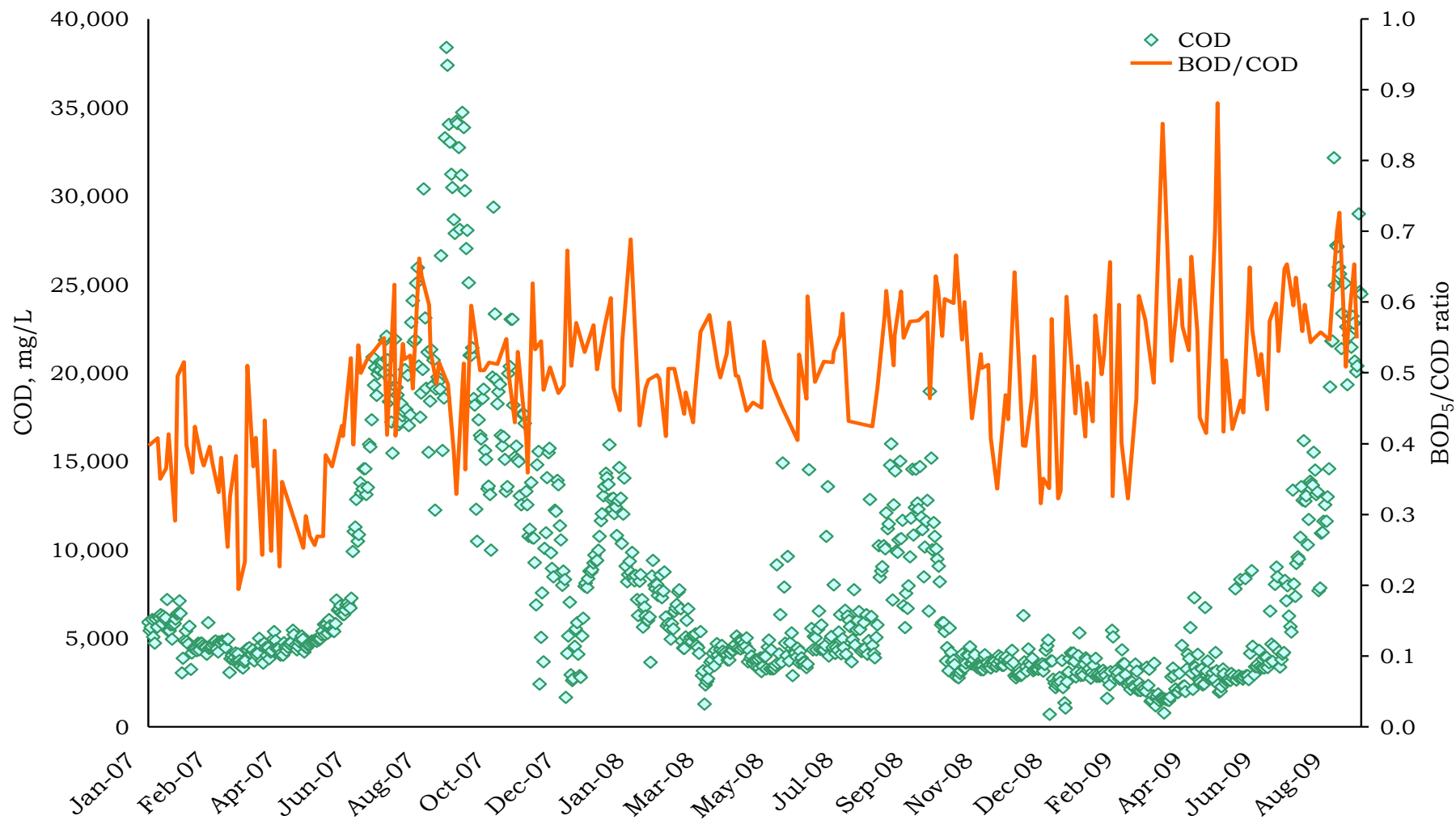


**Figure IV. 2.A** Change in COD concentrations and BOD<sub>5</sub>/COD ratio of Hamitler Landfill leachate between January 2007 and August 2009

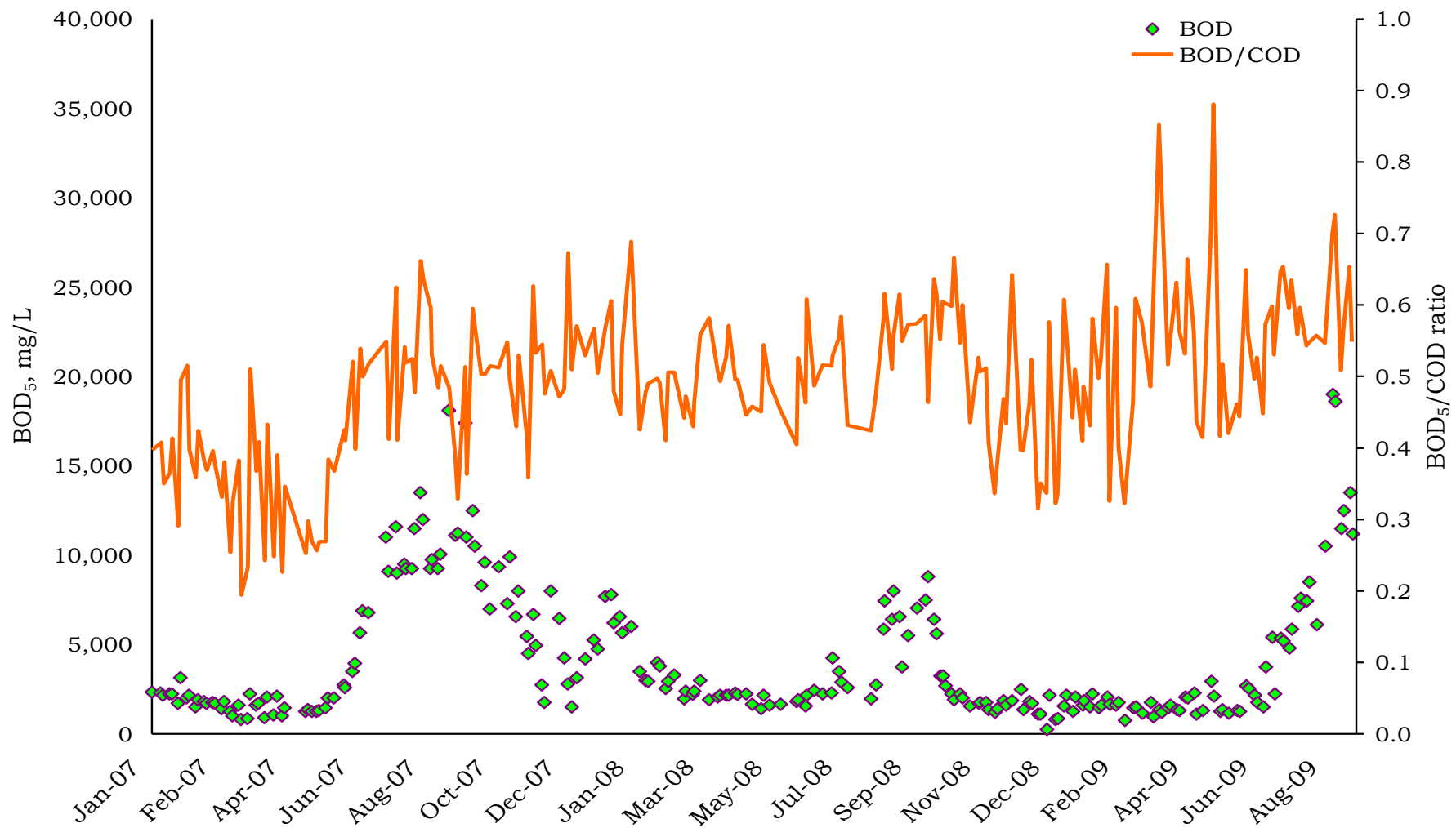


**Figure IV. 2.B** Change in BOD<sub>5</sub> concentrations and BOD<sub>5</sub>/COD ratio of Hamitler Landfill leachate between January 2007 and August 2009

#### **IV.1.3. Total Nitrogen and Ammonium Nitrogen**

Ammonia in leachate is an end product of anaerobic degradation of proteins in the landfilled solid waste. It represents the larger fraction of the total amount of nitrogen in older landfills due to complete hydrolysis and fermentation of organic nitrogen in the landfill (Carley and Mavinic, 1991). According to Chu et al. (1994), ammonia concentration of landfill leachate varies between 500 and 1500 mg/L after 3-8 years period and it is expected to remain at this level for at least 50 years generally.  $\text{NH}_4\text{-N}/\text{TKN}$  ratio is another parameter, which indicates the stabilization of waste within landfill. A ratio close to one demonstrates that hydrolysis of organic nitrogen occurs effectively in the landfill, in other words acidogenesis phase prevails (Inanc et al., 2000). In this study TKN was not measured, but due to low influent  $\text{NO}_3\text{-N}$  concentrations (5-23 mg/L), TN which represents the sum of organic-N,  $\text{NH}_4\text{-N}$  and  $\text{NO}_3\text{-N}$  was considered as TKN by neglecting the minor  $\text{NO}_3\text{-N}$  concentration. TN concentration in Hamitler Landfill leachate has fluctuated between 560 and 3916 mg/l during 32-month study period. The TN concentrations measured and  $\text{TN}/\text{NH}_4\text{-N}$  ratio calculated are shown in Figure IV.3.

High concentrations of  $\text{NH}_4\text{-N}$  that have been observed in Hamitler Landfill leachate, is a typical characteristic of old landfill leachates (Renou et al., 2008). According to previous studies, it can be concluded that high ammonia concentrations are observed in landfill leachates in Turkey (Inanc et al., 2000; Calli et al., 2005). Ammonia levels in USA and France are comparatively lower than Turkey. Whereas, such elevated ammonia concentration was also reported from the neighbor country, Greece, which may be related to similar living habits and solid waste compositions (Tasti et al., 2003).

$\text{NH}_4\text{-N}/\text{TN}$  ratio of Hamitler Landfill leachate has varied between 0.14 and 0.98 during the 32-month study period. The  $\text{NH}_4\text{-N}/\text{TN}$  ratio of Hamitler Landfill leachate calculated as 0.72 in average shows the acid formation phase in Hamitler Landfill.

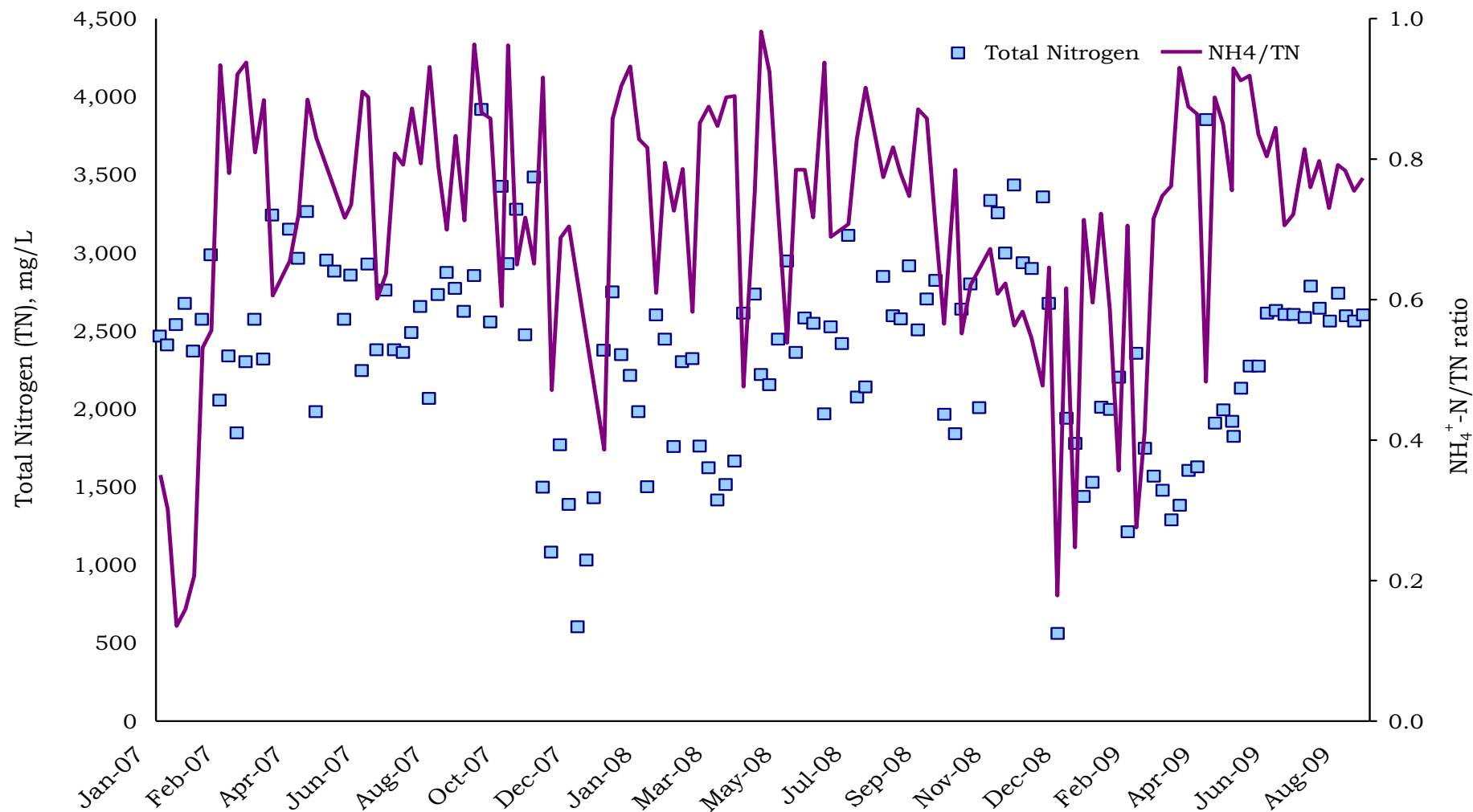


**Figure IV. 3** Change in TN concentration and TN/NH<sub>4</sub>-N ratio of Hamitler Landfill leachate between January 2007 and August 2009

#### IV.1.4. pH, Alkalinity and Temperature

Alkalinity and pH are important parameters for estimating the stabilization phase of the landfills. In young acidogenic landfills pH is low because of VFA production. As stabilization continues in landfill VFAs are consumed by methanogens and so pH and alkalinity increases (Qasim and Chiang, 1994).

pH has varied from 6.63 to 8.79 although Hamitler landfill is in acidic phase of decomposition. Relatively higher pH values were actually due to high alkalinity which was strongly affected by the extraordinarily high ammonia concentrations. Additionally, pH has always dropped in the summer months contrary to the increase in the temperature and VFA concentrations in leachate. Changes in the pH and temperature of the Hamitler Landfill leachate are given in Figure IV.4.

Differing than other characterization parameters, alkalinity was only measured between February and August 2009. The results are given in Table IV.3. Parallel to the pH and ammonia concentrations, alkalinity has increased from 5524 mg/L in February to 11174 mg/L in July 2009. These alkalinity results showed that ammonia and VFAs are the two significant parameters controlling the alkalinity in Hamitler Landfill leachate. Although the alkalinity was not monitored in 2007 and 2008 it can be concluded that alkalinity was followed a similar profile in that period.

**Table IV. 3** Alkalinity concentration of Hamitler Landfill leachate in 2009

| Month         | Alkalinity, mg CaCO <sub>3</sub> /l |
|---------------|-------------------------------------|
| February 2009 | 5,524                               |
| March 2009    | 5,395                               |
| April 2009    | 7,389                               |
| May 2009      | 8,232                               |
| June 2009     | 10,980                              |
| July 2009     | 11,174                              |
| August 2009   | 10,554                              |

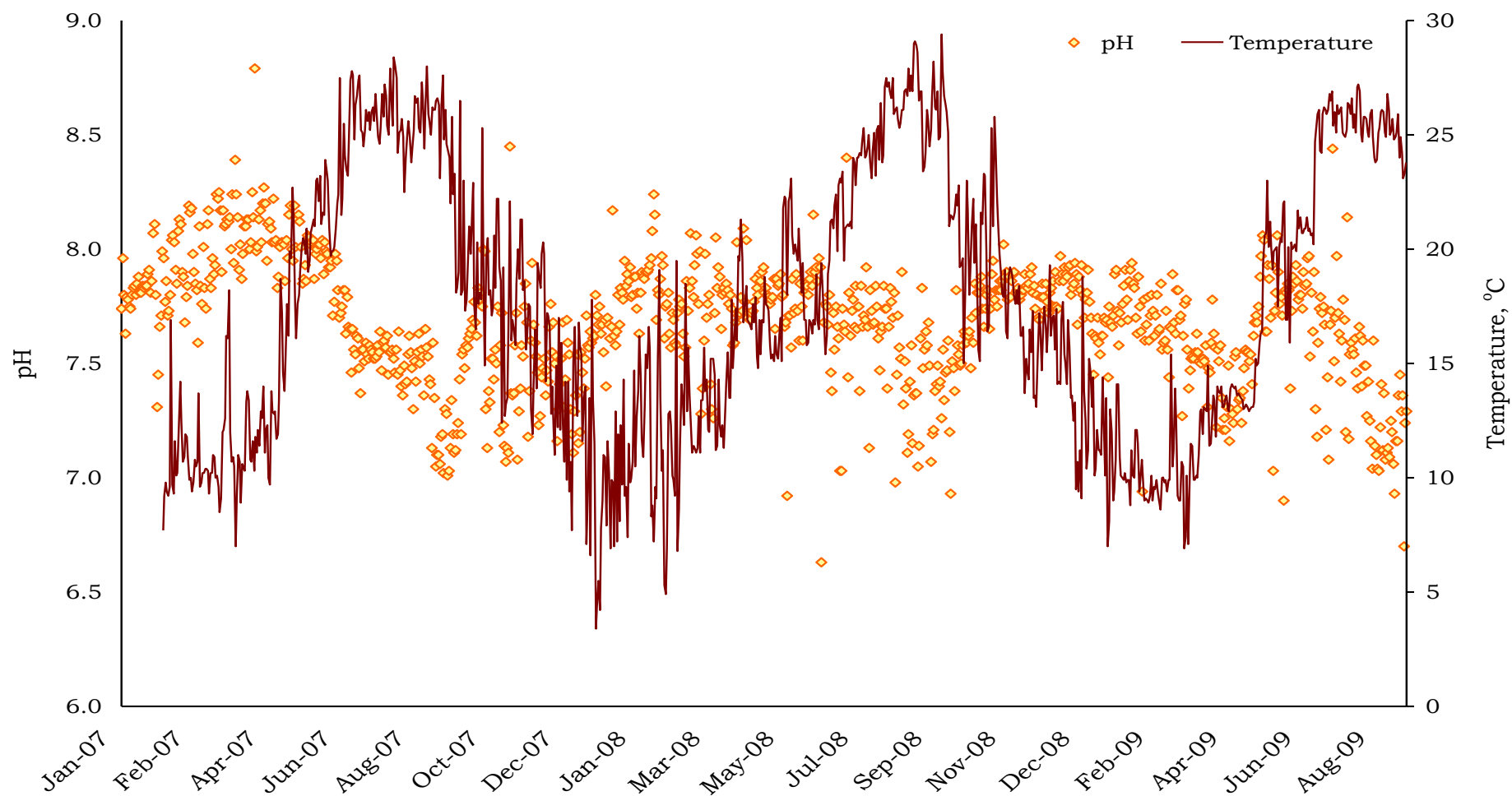


**Figure IV. 4** Change in pH and temperature of raw Hamitler Landfill leachate in the inlet of treatment plant between January 2007 and August 2009

Landfill temperature is an uncontrollable factor influencing leachate quality. It has been shown to fluctuate with seasonal ambient temperature variations (Lu et al., 1985). During decomposition of MSW a large amount of heat is produced that rises the landfill temperature above the ambient temperature. Peak temperature of 71 °C (160 °F) can occur a few day to a few weeks after the application of cover (Oweis and Khera, 1998). Temperature of raw leachate was measured in the open top collector at the inlet of HLTP and therefore considerably affected from the ambient temperature. It has decreased below 4 °C in winters, and elevated up to 29.4 °C in summer months depending on the seasonal temperature change.

#### **IV.1.5. Suspended Solids**

Suspended solids (SS) in leachate can be affected by different storage types of solid wastes in landfills. If the solid wastes are compacted sufficiently the filtration ability of landfill would increase so SS in leachate decreases. In landfills where acidogenic phase prevails, organic suspended solids (VSS) in leachate increase due to the breakdown of biomass to small organic particles which can be swept by the leachate percolating through the waste matrix. Thus, another indication of the acidogenic activity in the landfills may be high SS and VSS concentrations in leachate (Qasim and Chiang, 1994). The variation in suspended solids concentration of Hamitler Landfill leachate is given in Figure IV.5.

SS concentration of Hamitler Landfill leachate was fluctuated between 38 mg/l and 3850 mg/L during 32-month study period. Similar SS concentrations were observed in some other studies in which raw leachates from different landfills from Turkey was used (Ceçen and Aktaş, 2004; Uygur and Kargi, 2004). It is thought that increase in SS concentrations during summer periods was due to high ambient temperatures and the consequent increase biological activity. Increases in COD and BOD<sub>5</sub> concentrations in these periods support this result.

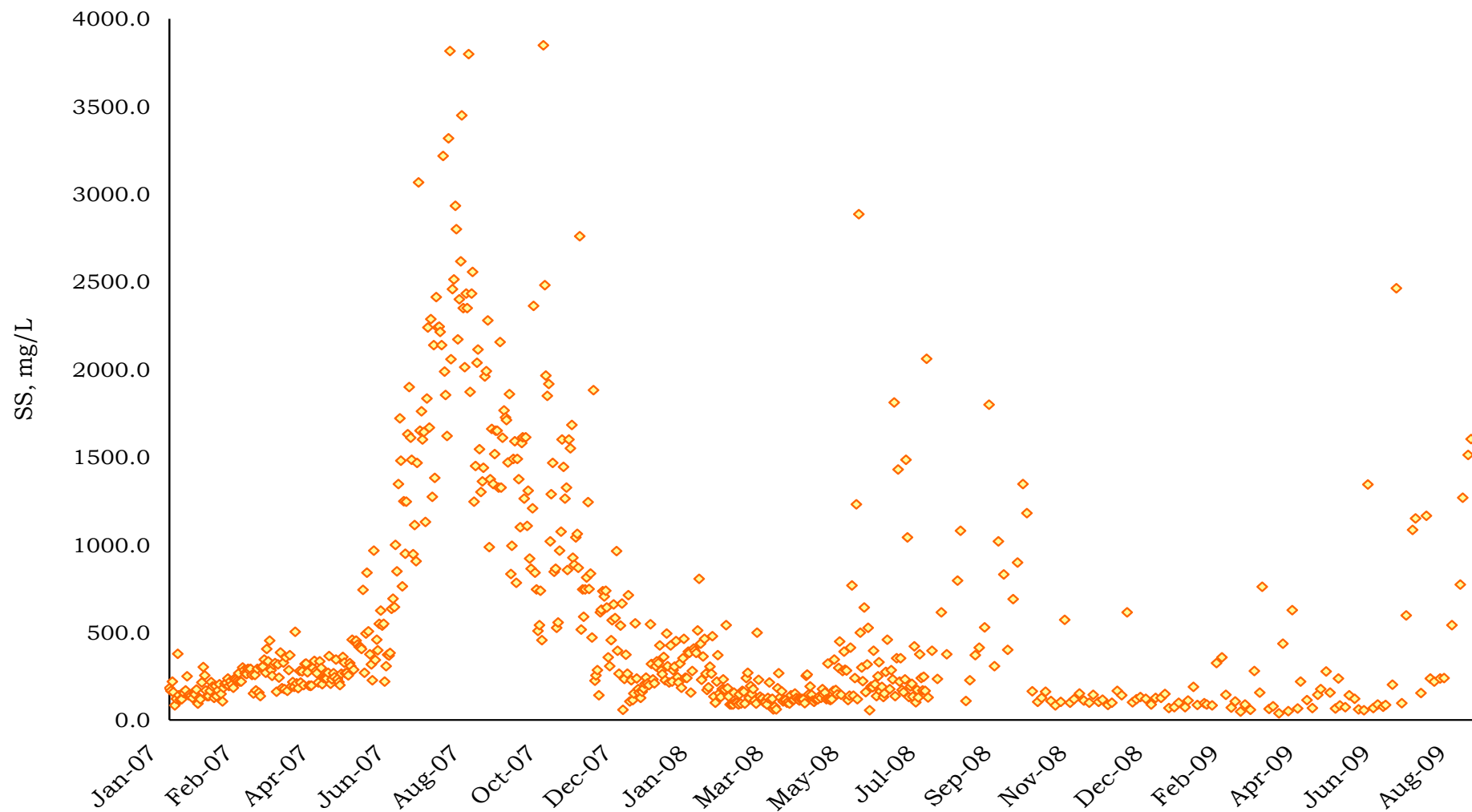


**Figure IV. 5** Change in SS concentration of Hamitler Landfill leachate between January 2007 and August 2009

## **IV.2. EVALUATION OF TREATMENT PERFORMANCE**

In the second part of this thesis study, treatment performance of HLTP was evaluated based on COD, BOD<sub>5</sub>, and NH<sub>4</sub>-N removal efficiencies during 32 months between January 2007 and August 2009. In evaluating the treatment performance of HLTP the followings were calculated; organic and hydraulic loads applied, COD, BOD<sub>5</sub> and NH<sub>4</sub>-N removal efficiencies, kinetic coefficients of BOD<sub>5</sub> removal. In addition, the maximum amount of free ammonia that can be stripped was estimated and afterward compared with total NH<sub>4</sub>-N removal in order to explain the prevailing means in nitrogen removal.

### **IV.2.1. Organic Matter Removal**

COD load (kg/day) and hydraulic load (m<sup>3</sup>/day) applied to each line (Line-1 and Line 2) of HLTP between January 2007 and August 2009 are shown in Figure IV.6. Because the COD load applied was lower than the design load of 15,000 kg/day (7500 kg/day per each parallel line), Line-1 was closed and aerobic and facultative lagoons and SBRs were emptied in June 2008. Then, HLTP was operated with single line (Line-2). As a result, the hydraulic load applied to each line ranging between 98 and 285 m<sup>3</sup>/day until June 2008, gradually rose, and reached up to 640 m<sup>3</sup>/day in March 2009 (Figure IV.6b). Line-2 has been operated with 184 m<sup>3</sup>/day average hydraulic loading and 23 days of hydraulic retention time (HRT) until June 2008. After this date, due to operation of the plant with single line and increase in leachate flow rate because of heavy rainfall and increasing hydraulic loading the HRT was reduced to 11 days in Line-2. Due to increasing COD concentration in summer months and increasing hydraulic load to Line-2, COD load elevated up to 9425 kg/day in August 2009. However, because of the dilution effect of heavy rainfall it remained below 1900 kg/day in between fall 2008 and spring 2009 (Figure IV.6b). In contrast with the considerable increase in hydraulic loading, the COD load of about 1750 kg/day in average when two lines were under operation did not change noticeably and excluding August 2009 remained about 2000 kg/day in average for Line-2 alone.

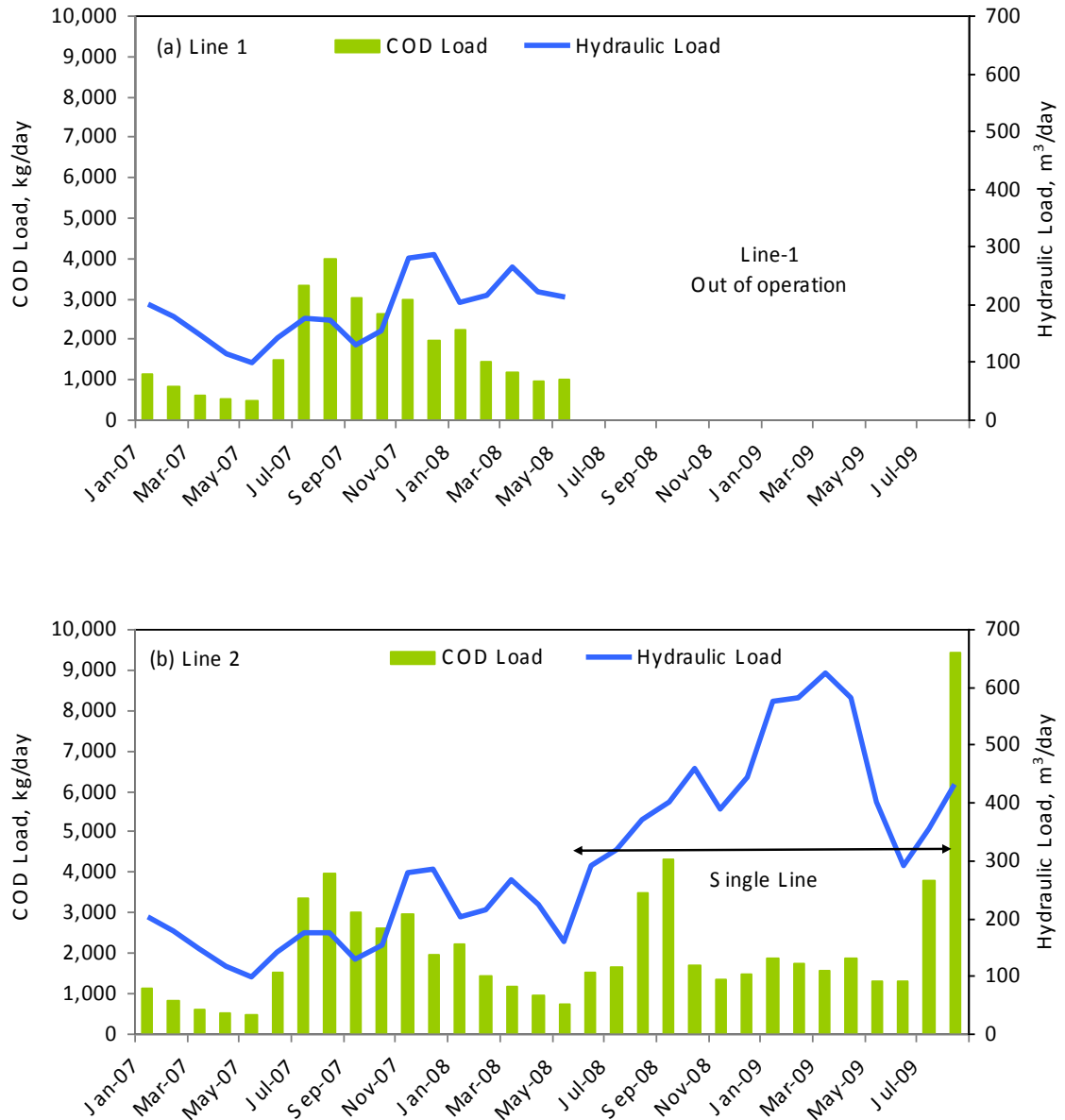


**Figure IV. 6** COD and hydraulic load applied to the HLTP between January 2007 and August 2009; (a) Line-1 (b) Line-2

The monthly average of COD concentrations measured in the influent and aerobic lagoons, facultative lagoons and SBR's (effluent) are shown in Figure IV.7.

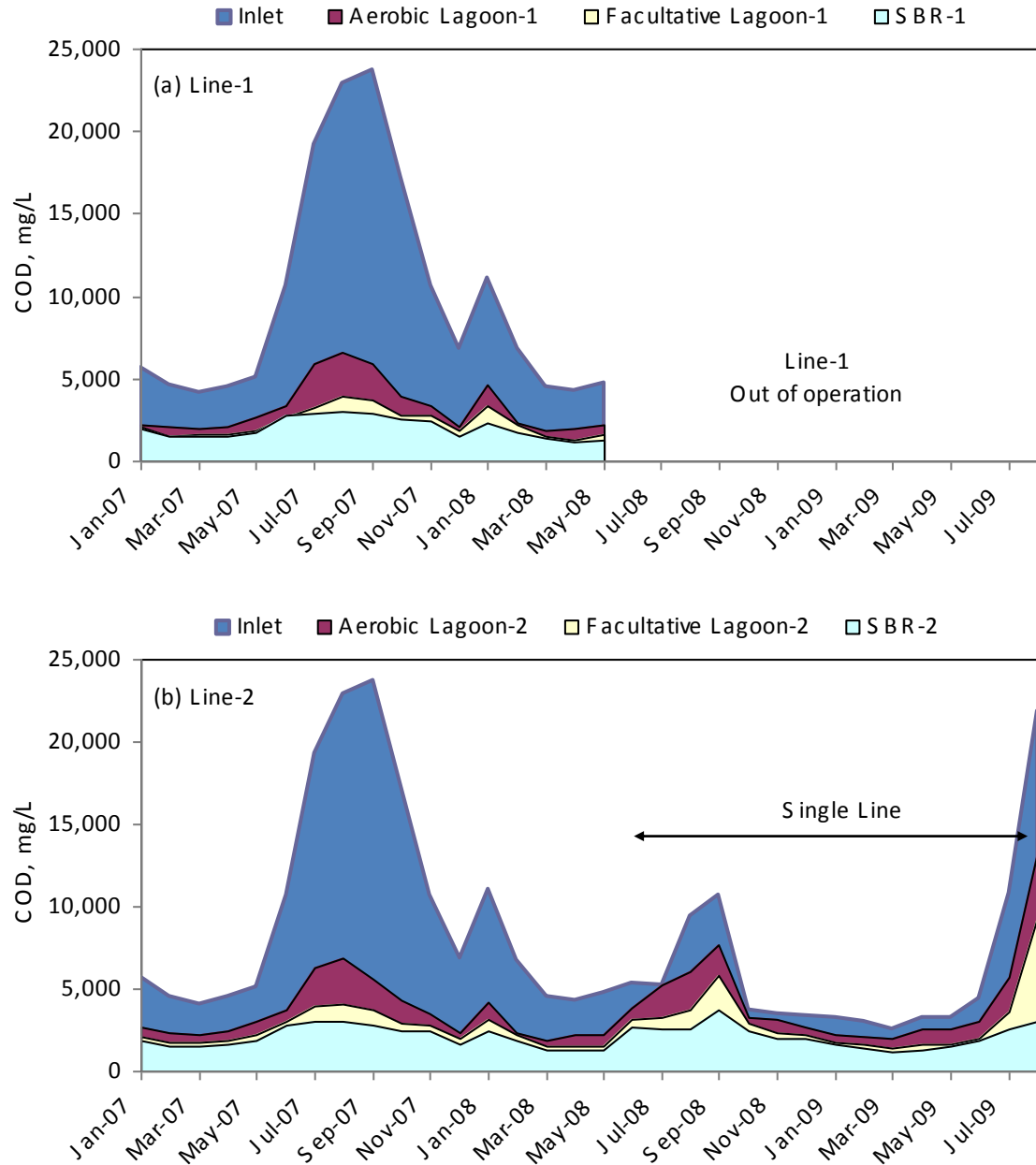


**Figure IV. 7** COD concentrations measured in the inlet and aerobic lagoons, facultative lagoons and SBR's between January 2007 and August 2009; (a) Line-1 (b) Line-2

In the winter and spring months of 2007 (January-May), average influent COD concentration was about 4797 mg/L, and dropped below 2360 mg/L in Aerobic Lagoons with 51% removal efficiency. In this period, COD concentration dropped to 1806 mg/L in Facultative Lagoons with 23% average removal efficiency and thereafter dropped to 1643 mg/L in SBR's with 9% removal efficiency.

After May 2007, influent COD concentration started to increase and reached up to 23747 mg/L in September 2007. Between June 2007 and September 2007, average influent COD concentration was 19157 mg/L. In this period, COD removal efficiencies increased to 71%, 34%, and 17% in Aerobic Lagoons, Facultative Lagoons and SBR's, respectively.

Starting from September 2007 except January 2008, influent COD concentrations decreased back to 4400 mg/L. Meanwhile, COD removal efficiencies decreased to 63%, 22%, and 16% in Aerobic Lagoons, Facultative Lagoons, and SBR's, respectively. This deterioration in COD removal is supposed to be related to the increasing inert COD in influent.

After Line-1 has been closed and Line-2 started to operate alone, the COD removal efficiency decreased considerably in Aerobic Lagoon-2 while remained almost same in Facultative Lagoon-2. On the other hand, because more soluble COD came into the SBRs, its COD removal performance increased noticeably although higher COD concentrations were measured in the effluent of HLTP after June 2008.

While Line-2 was operating alone, 6248 mg/L average influent COD concentration reduced to 4357 mg/L in Aerobic Lagoon-2 with 30% removal efficiency. In Facultative Lagoon-2, COD concentration decreased to 3075 mg/L with 29% removal efficiency. The average COD removal efficiency in SBR-2 was about 30% and effluent COD was 2165 mg/L. In the overall, the COD removal efficiency of HLTP decreased from 79% to 65% after Line-1 has been closed in June 2008.

COD and BOD<sub>5</sub> removal efficiencies followed a similar profile during 32-month study period. The average BOD<sub>5</sub> removal in both aerobic and facultative lagoons and SBR's were 75%, 44% and 55%, respectively which makes 94% in total. After Line-1 has been closed, BOD<sub>5</sub> removal decreased to 45%, 42%, and 30% in Aerobic Lagoon-2, Facultative Lagoon-2 and SBR-2, respectively. As a result, the total BOD<sub>5</sub> removal efficiency decreased to 78% when HLTP was operated with single line. The considerable decreases in BOD<sub>5</sub> and COD removal efficiencies in HLTP after Line-1 has been closed may be explained as a result of increase in hydraulic load and the consequent decrease in HRT.

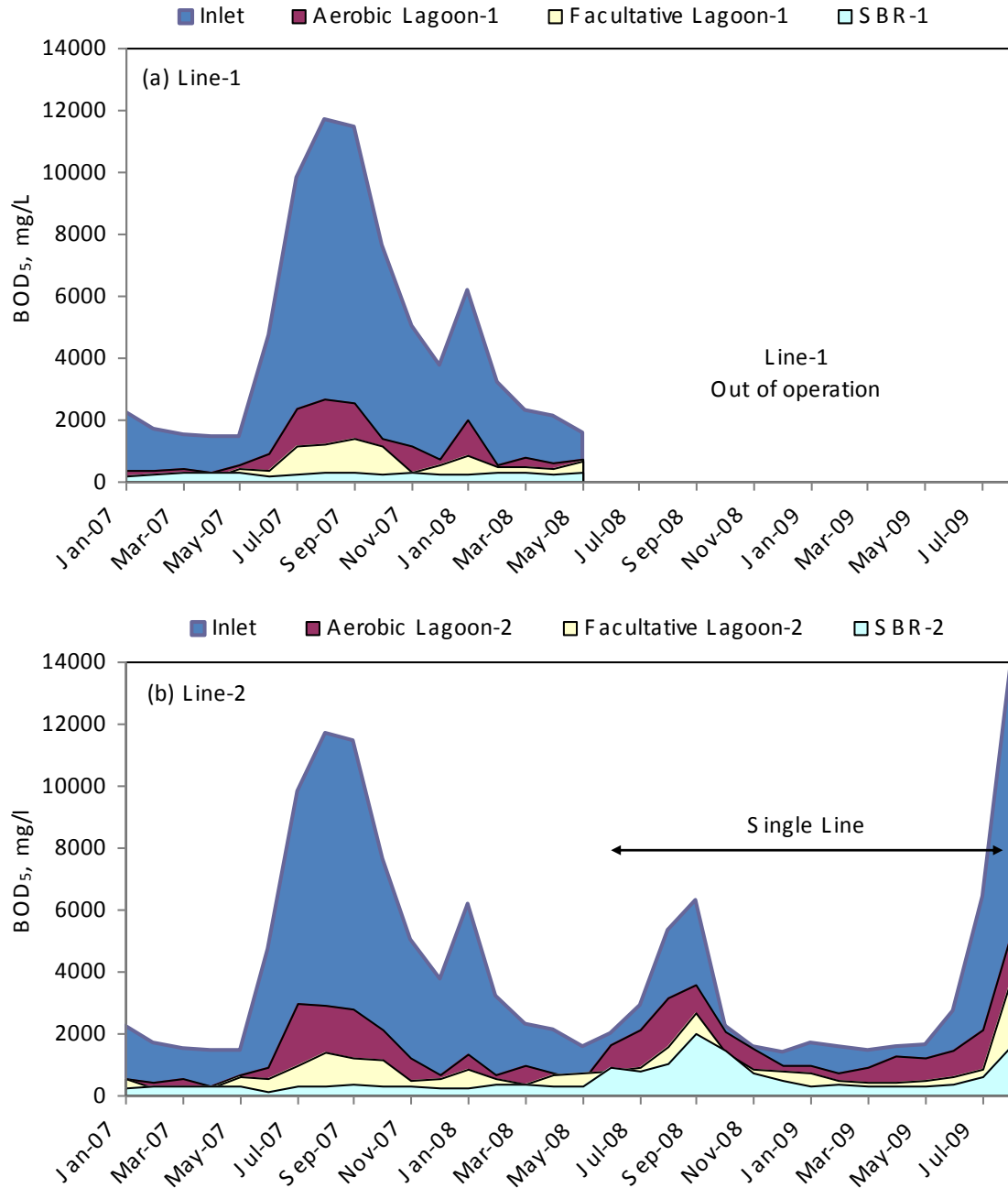


**Figure IV. 8** BOD<sub>5</sub> concentrations measured at inlet of HLTP, and outlet of aerobic lagoons, facultative lagoons and SBR's during January 2007 and August 2009; (a) Line-1 (b) Line-2

#### IV.2.1.1. BOD<sub>5</sub> removal rate constants calculated for Aerobic Lagoons

In evaluating the treatment performance of HLTP, overall first-order BOD<sub>5</sub> removal rate constants were calculated for Aerobic Lagoon-1 and Aerobic Lagoon-2 separately.

During operation of Aerobic Lagoon-1 till June 2008, influent BOD<sub>5</sub> concentration changed between 1425 and 11675 mg/L, while the effluent BOD<sub>5</sub> was in the range of 308 and 2657 mg/L (Figure IV.8a).

The incoming leachate flow rate was almost equally distributed to Line-1 and Line 2 while HLTP was operated with two parallel lines. The HRT of Aerobic Lagoon-1 was calculated as half of the incoming flow rate by using equation (III.3). As a result, HRTs in the range of 5.9 and 17.1 days were calculated for Aerobic Lagoon-1 between January 2007 and May 2008 (Figure IV.9).

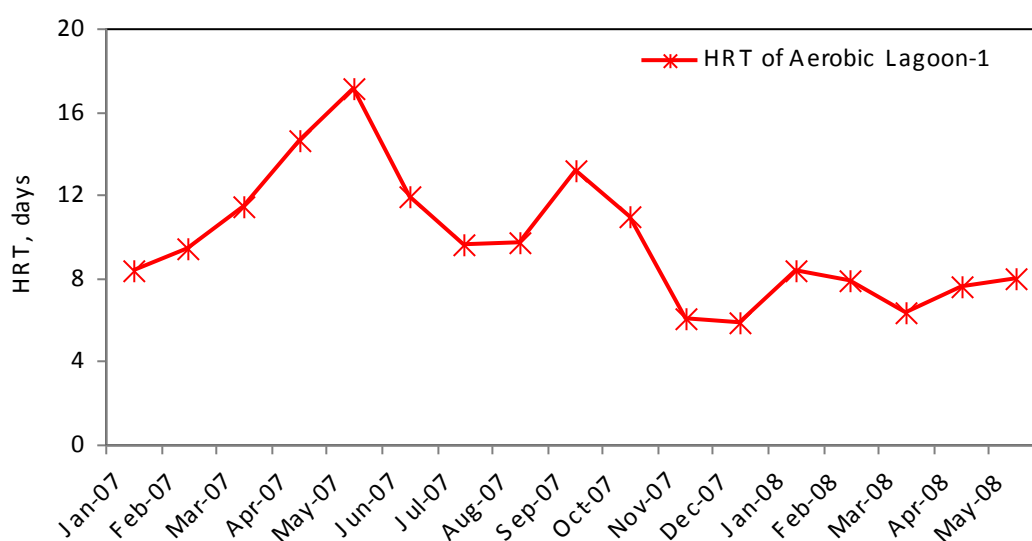


**Figure IV. 9** HRT of Aerobic Lagoon-1 between January 2007 and May 2008

After calculating HRT, first-order BOD<sub>5</sub> removal rate constants for Aerobic Lagoon-1 were calculated in accordance with influent and effluent BOD<sub>5</sub> concentrations of lagoon and HRT using the equation (III.4).

Average monthly temperatures measured in Aerobic Lagoon-1 in the period of January 2007 and May 2008 are shown in Figure IV.10. For Aerobic Lagoon-1 temperature values changed between 11.65 and 27.49 °C (Figure IV.10). Overall first-order BOD<sub>5</sub> removal rate constants were calculated in accordance with real temperature values using equation (III.5).

Influent BOD<sub>5</sub> concentrations of Aerobic Lagoon-2 were almost same with the ones of Aerobic Lagoon-1 till May 2008, and it changed between 1404 and 13829 mg/L while been operated alone after June 2008 until the end of study period August 2009 (Figure IV.8b). BOD<sub>5</sub> removal efficiency of Aerobic Lagoon-2 changed dramatically after Aerobic Lagoon-1 has been closed and therefore outlet

concentrations affected negatively after this date and increased from 332-2950 mg/L to 750-5142 mg/L (Figure IV.8b).

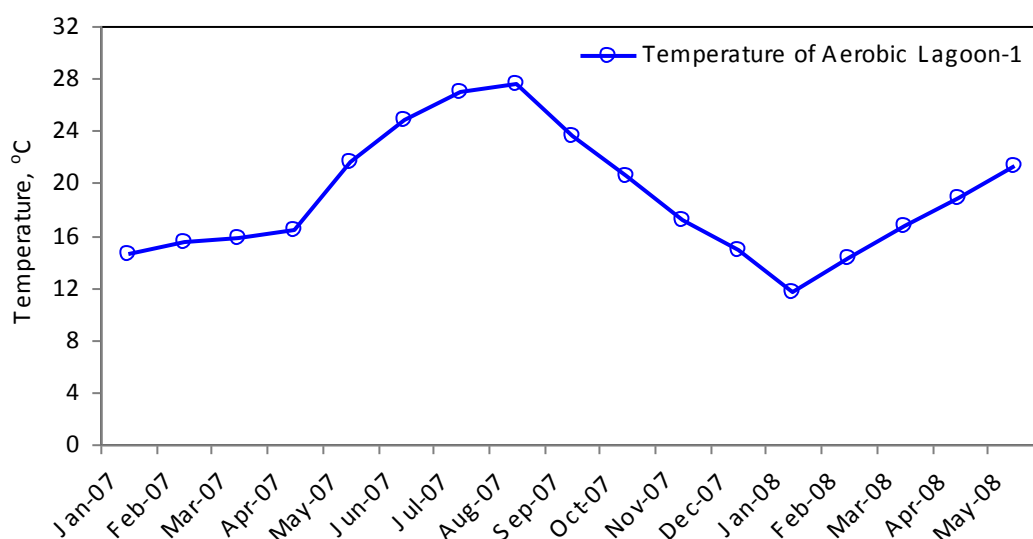


**Figure IV. 10** Temperature values measured in Aerobic Lagoon-1 between January 2007 and May 2008

HRT of Aerobic Lagoon-2 was also affected after Aerobic Lagoon-1 has been closed due to the increase in incoming flow rate to the single line. The HRT of Aerobic Lagoon-2 fluctuated between 5.9 and 17.1 days decreased to a range of 2.7-5.8 days after the operation of Aerobic Lagoon-1 has been terminated (Figure IV.11).

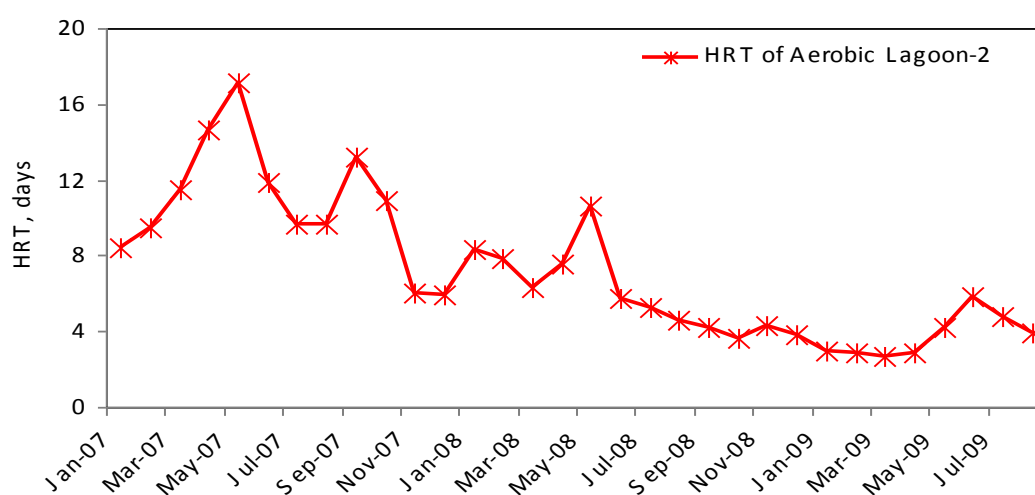


**Figure IV. 11** HRT of Aerobic Lagoon-2 between January 2007 and August 2009

The first-order BOD<sub>5</sub> removal rate constants for Aerobic Lagoon-2 were calculated using the real influent and effluent BOD<sub>5</sub> concentrations and HRTs according to equation (III.4).

For Aerobic Lagoon-2 the average monthly temperature values changed between 17.31 and 29.22 °C in the period between January 2007 and August 2009 (Figure IV.12).

Overall first-order BOD<sub>5</sub> removal rate constants for Aerobic Lagoon-2 were calculated in accordance with real temperature values using equation (III.5).

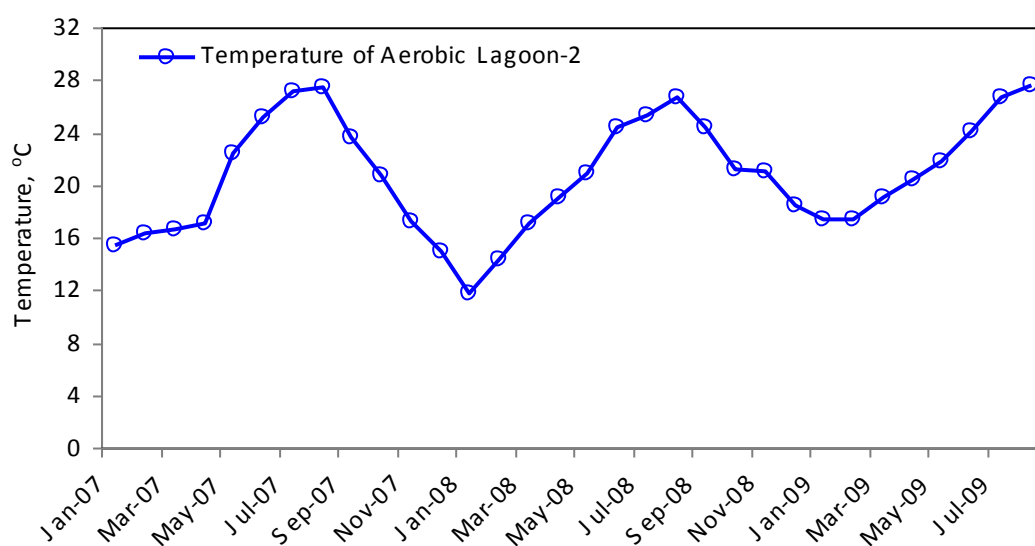


**Figure IV. 12** Temperature values measured in Aerobic Lagoon-2 between January 2007 and August 2009

The overall first-order BOD<sub>5</sub> removal rate constants calculated for Aerobic Lagoon-1 changes between 0.1 and 0.91 d<sup>-1</sup> until it has been closed. For the period when both lines were under operation, the overall first-order BOD<sub>5</sub> removal rate constants calculated for Aerobic Lagoon-2 are in the range 0.07 and 1 d<sup>-1</sup>, and decreases significantly to 0.01-0.4 d<sup>-1</sup> as a result of reduction in BOD<sub>5</sub> removal efficiency. The average BOD<sub>5</sub> removal rate constant determined for aerobic lagoons was about 0.38 d<sup>-1</sup> for the period that two lines were operated parallel and decreased to 0.16 d<sup>-1</sup> after Line-1 has been closed. This drop in BOD<sub>5</sub> removal rate constant was certainly related with the decrease in HRT. It should be also noted that, these rates are considerably lower than the value of 1.3 d<sup>-1</sup> used in the design (Figure IV.13).

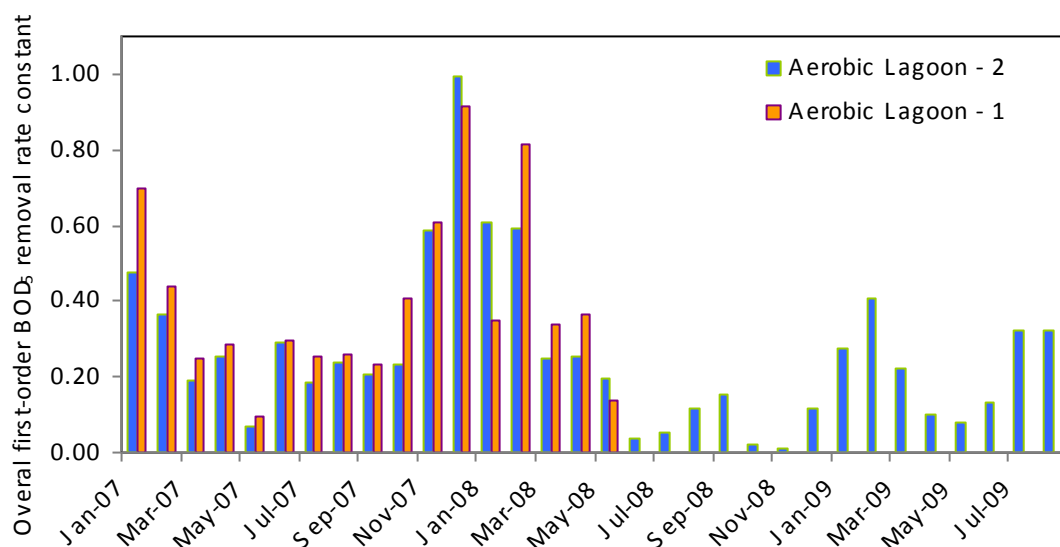


**Figure IV. 13** BOD<sub>5</sub> removal rate constants calculated for Aerobic Lagoon-1 and Aerobic Lagoon-2

#### IV.2.1.2. BOD<sub>5</sub> removal rate constants calculated for Facultative Lagoons

Overall first-order BOD<sub>5</sub> removal rate constants were calculated for Facultative Lagoon-1 and Facultative Lagoon-2 separately.

During operation of Facultative Lagoon-1 till June 2008, influent BOD<sub>5</sub> concentration changed between 308 and 2657 mg/L, while effluent BOD<sub>5</sub> was in the range of 150 and 1367 mg/L (Figure IV.8a).

HRT calculated for Facultative Lagoon-1 by using equation (III.3) changed between 8 and 22.6 between January 2007 and May 2008 (Figure IV.14).

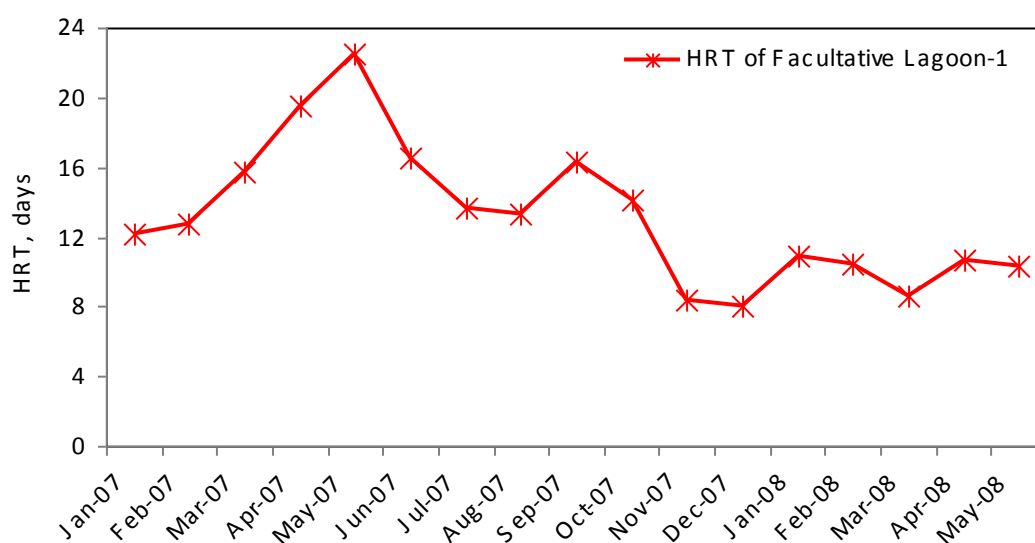


**Figure IV. 14** HRT of Facultative Lagoon-1 between January 2007 and May 2008

First-order BOD<sub>5</sub> removal rate constants of Facultative Lagoon-1 were calculated, in accordance with measured influent and effluent BOD<sub>5</sub> concentrations of lagoons, and HRT by using equation (III.4).

Average monthly temperatures measured in Facultative Lagoon-1 changed between 8.7 and 25.3 °C in the period of January 2007 and May 2008 (Figure IV.15).

Overall first-order BOD<sub>5</sub> removal rate constants were calculated in accordance with real temperature values using equation (III.5).

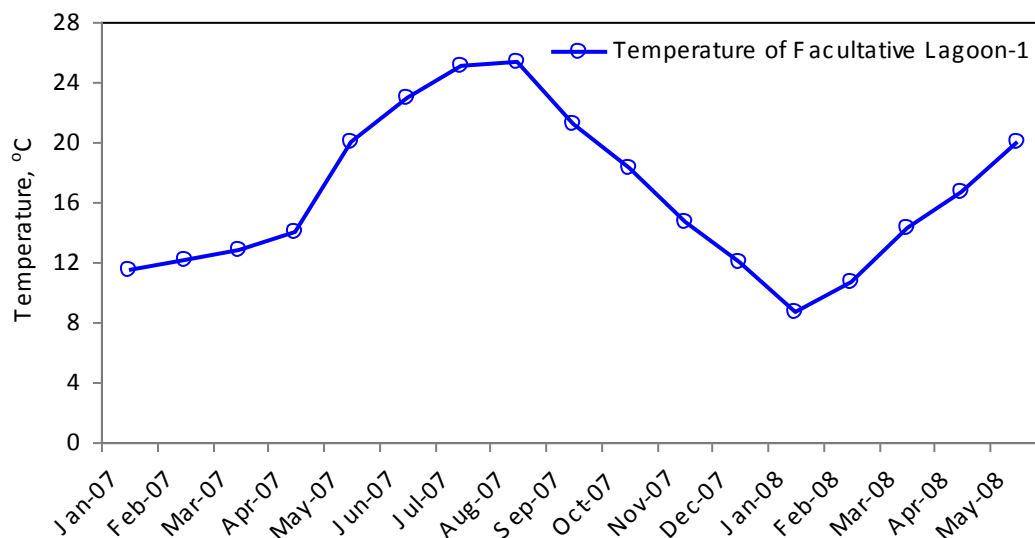


**Figure IV. 15** Temperature values measured in Facultative Lagoon-1 between January 2007 and May 2008

Influent BOD<sub>5</sub> concentration of Facultative Lagoon-2 changed between 332 and 5142 mg/L and outlet BOD<sub>5</sub> concentration of Facultative Lagoon-2 changed between 250 and 3700 mg/L between January 2007 and August 2009 (Figure IV.8b).

HRT of Facultative Lagoon-2 between January 2007 and August 2009 is given in Figure IV.16.

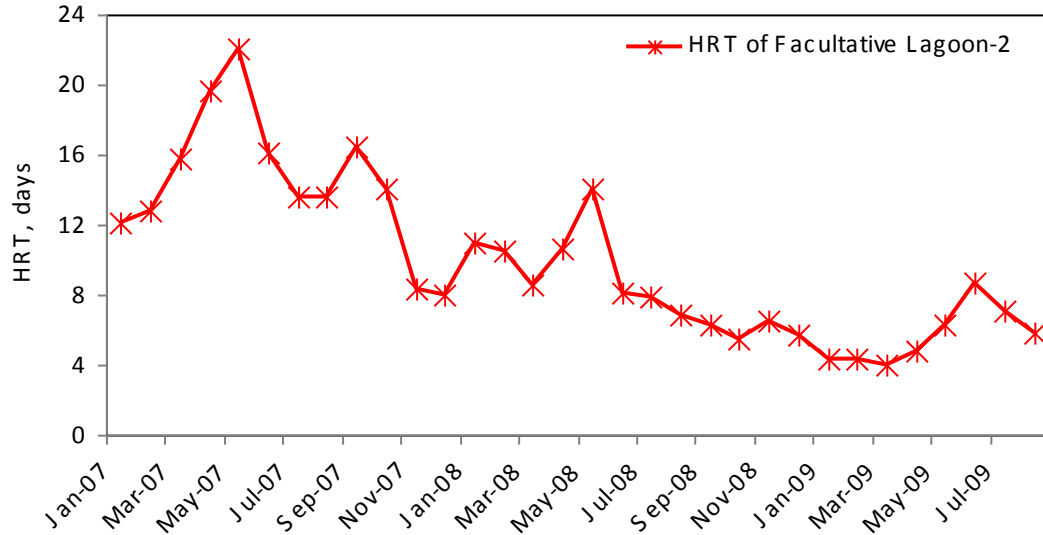


**Figure IV. 16** HRT of Facultative Lagoon-2 between January 2007 and August 2009

HRT of Facultative Lagoon-2 was also affected after Facultative Lagoon-1 has been closed due to increase in incoming flow rate to the single line. The HRT of Facultative Lagoon-2 fluctuated between 8 and 22.1 days decreased to a range of 4-8.7 days after the operation of Facultative Lagoon-1 has been terminated.

The first-order BOD<sub>5</sub> removal rate constants for Facultative Lagoon-2 were calculated using real influent and effluent BOD<sub>5</sub> concentrations, and HRTs according to equation (III.4).

For Facultative Lagoon-2 the average monthly temperature values changed between 8.62 and 25.29 °C in the period between January 2008 and August 2009 (Figure IV.17).

Overall first-order BOD<sub>5</sub> removal rate constants for Facultative Lagoon-2 were calculated in accordance with real temperature values using equation (III.5).

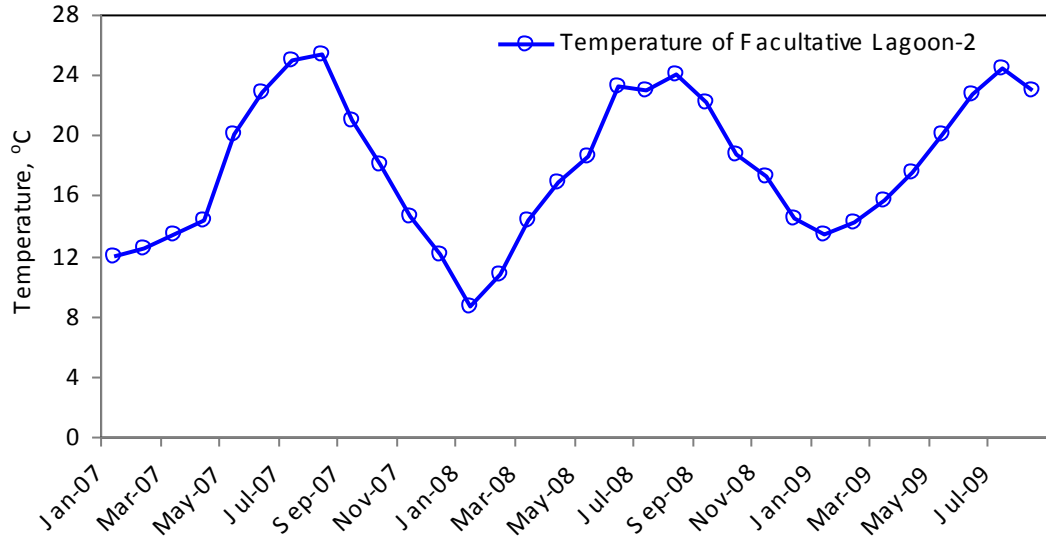


**Figure IV. 17** Temperature values measured in Facultative Lagoon-2 between January 2008 and August 2009

The overall first-order BOD<sub>5</sub> removal rate constant calculated for Facultative Lagoon-1 changes between 0.01 and 0.49 d<sup>-1</sup> until it has been closed, and overall first-order BOD<sub>5</sub> removal rate constant calculated for Facultative Lagoon-2 changes between 0.004 and 0.42 d<sup>-1</sup> during this study period. Average overall first-order BOD<sub>5</sub> removal rate constant of facultative lagoons was calculated as 0.10 d<sup>-1</sup> for the study period, which is so much lower than the design value of 0.80 d<sup>-1</sup> (Figure (IV.18)).

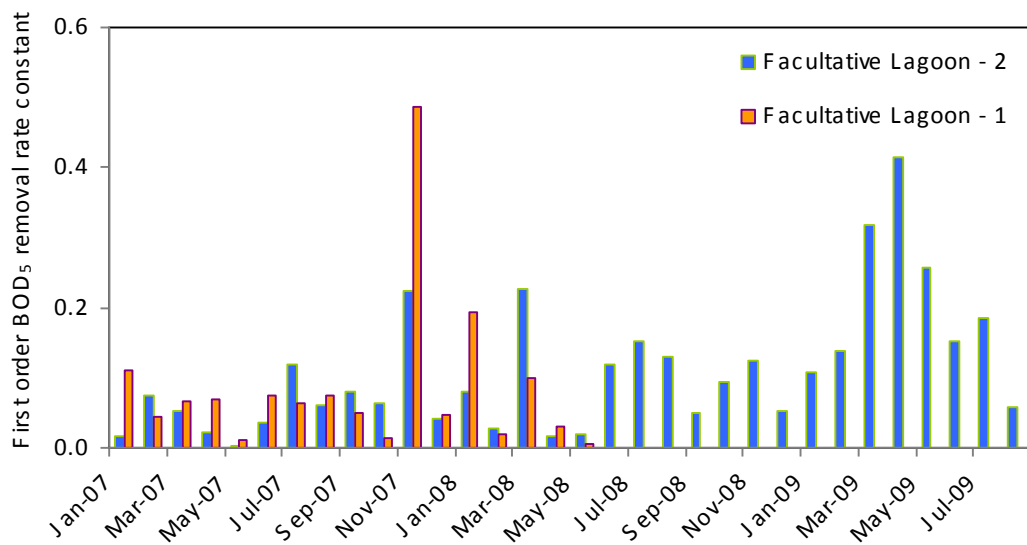


**Figure IV. 18** BOD<sub>5</sub> removal rate constants calculated for Facultative Lagoon-1 and Facultative Lagoon-2

#### IV.2.2. Ammonium Removal

It is supposed that the uncontrolled ammonium removal observed in HLTP occurs through ammonia stripping and biological nitrification/denitrification. In this section, results of the computation performed to find out the amount of ammonia that can be stripped in aerobic and facultative lagoons and SBRs of HLTP are presented. In addition to calculation results, the evidences of biological nitrogen removal were discussed considering the  $\text{NH}_4\text{-N}$  and  $\text{NO}_3\text{-N}$  concentrations measured in the influent and effluent of HLTP.

##### IV.2.2.1. Ammonia Stripping

The amount of ammonia that can be stripped from SBR's, facultative lagoons and aerobic lagoons were estimated separately by assuming that the calculated influent ammonium concentration of a unit is the effluent concentration of the previous one.

In order to compute the amount of ammonia stripped first of all free ammonia concentrations in each unit were calculated. The effluent  $\text{NH}_4\text{-N}$  concentration of HLTP was considered as the effluent ammonium concentration of SBR units. The effluent free ammonia concentration in SBR units was calculated using equation (III.7) after calculating the ionization constant of ammonium ( $\text{pK}_a$ ) according to equation (III.8).

Average temperature values measured in SBR-1 and SBR-2 units changed from 284 °K to 302 °K (Figure IV.19a) and average pH values representing pH values SBR units changed between 7.6 and 8.7 (Figure IV.19b) between January 2007 and August 2009.

The effluent free ammonia concentration calculated for SBR changed between 6 and 204 mg/L according to changes in pH and temperature during study period (Figure IV.20). The highest effluent free ammonia concentration for SBR was calculated in April 2007 when the effluent ammonium concentration was also the highest observed throughout the 32-month study period. In general, the free ammonia concentrations were found to be 2-15% of the effluent ammonium concentration in SBR units. Rising temperature and pH increase the free ammonia fraction, but change in fraction is mainly based on  $\text{NH}_4\text{-N}$  concentrations. Therefore free ammonia concentration changed parallel to the effluent  $\text{NH}_4\text{-N}$  concentrations (Figure IV.20).

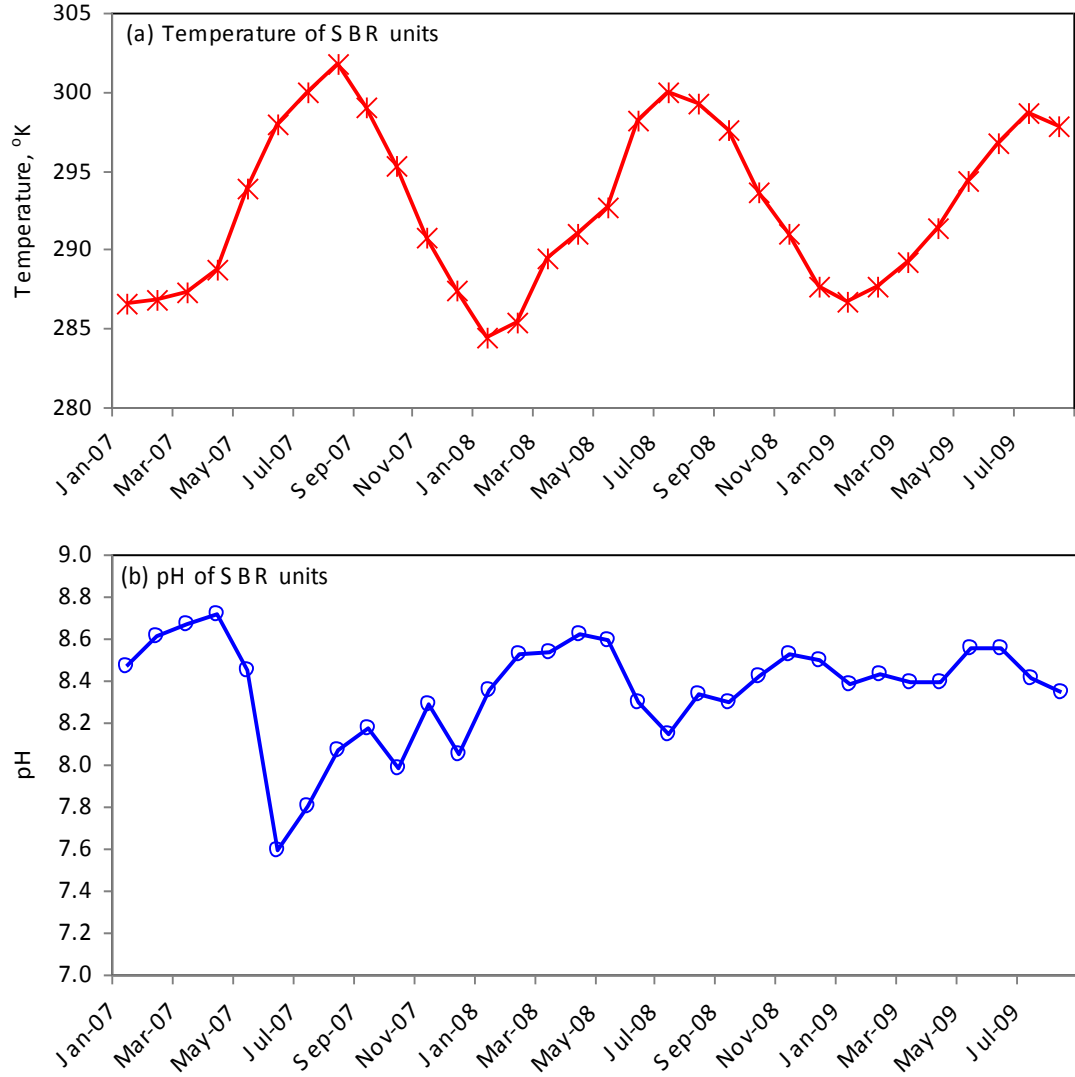


**Figure IV. 19** Average temperature (a) and pH (b) values of SBR units between January 2007 and August 2009

Each SBR unit is aerated for eight hours with a capacity of  $1075 \text{ Nm}^3/\text{h}$  and these operating parameters were accepted for the free ammonia stripping modelling of SBR units. Air input to the SBR units was calculated as  $17200 \text{ m}^3 \text{ air/d}$  during the parallel operation period. After June 2008 the air input reduced by half to  $8600 \text{ m}^3 \text{ air/d}$ .

Specific air inputs were calculated by dividing air inputs ( $\text{m}^3 \text{ air/d}$ ) to leachate flow rate ( $\text{m}^3/\text{day}$ ). Then the specific air input values were converted into gas to liquid ratio (G/L) using equation (III.13) by assuming the molecular weight of the water as  $18 \text{ g/mole}$  and volume of one mol of air as  $24.1 \text{ L/mole}$ .

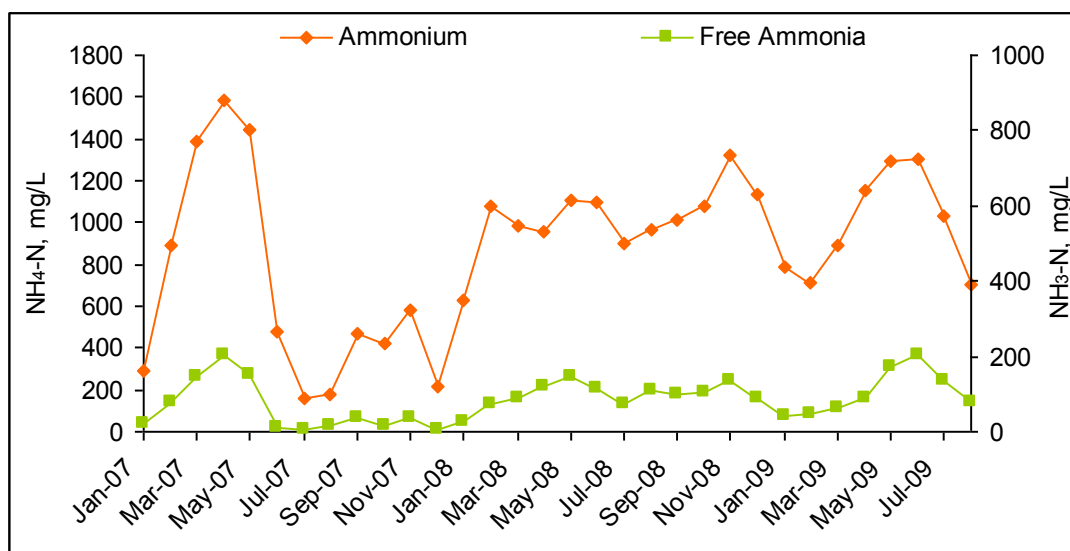


**Figure IV. 20** Effluent  $\text{NH}_4\text{-N}$  and  $\text{NH}_3\text{-N}$  concentrations calculated for SBR units between January 2007 and August 2009

Gas to liquid ratio calculated for SBR units changed from 0.010 to 0.066 mole air/mole leachate between January 2007 and August 2009 (Figure IV.21). The G/L ratio calculated for SBR units decreased significantly after closing SBR-1 in June 2008 (Figure IV.21).

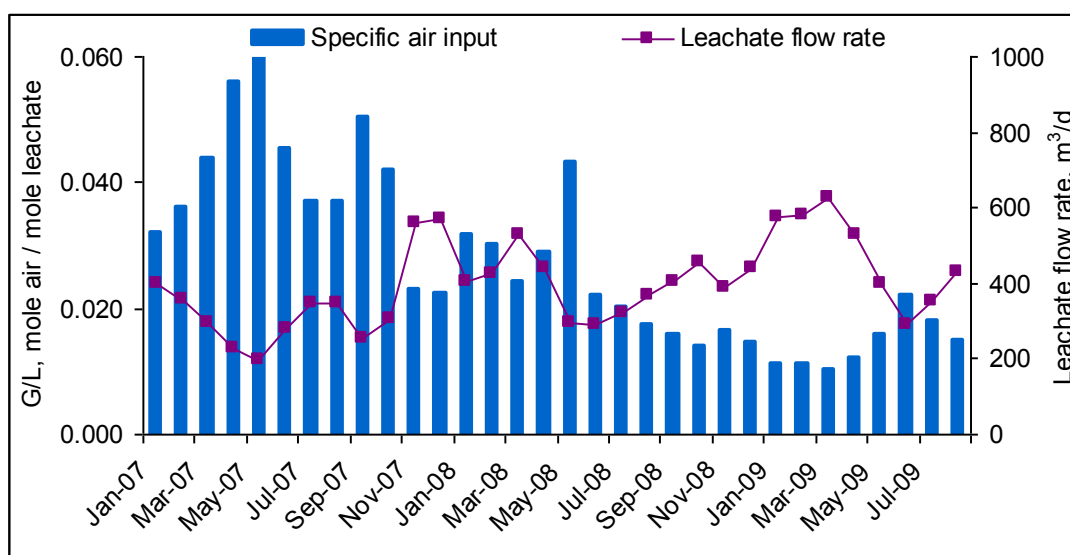


**Figure IV. 21** Gas to liquid ratio (G/L) calculated for SBR units and leachate flow rate between January 2007 and August 2009

The effluent mole fraction of free ammonia ( $C_e$ ) of SBR units was calculated by using equation (III.15) by assuming that one liter of water is 55.5 moles as calculated according to (III.16). When calculated,  $C_e$  values were used in computing the mole fraction of ammonia in the air leaving through the surface of facultative lagoons ( $y_e$ ) together with the Henry's law constant in equation (III.16).

The Henry's law constants calculated with average temperatures measured in SBR units varies between 0.48 and 1.15 atm (Figure IV.22). The values were found to increase in summer months and decrease in winters parallel to the temperature. (Figure IV.22).

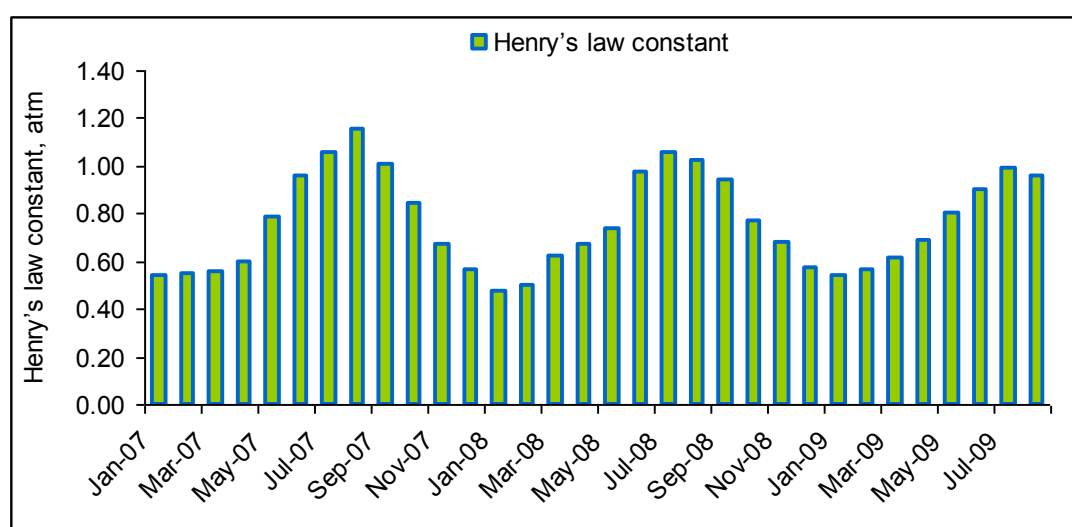


**Figure IV. 22** Henry's law constant at average temperatures measured in SBR units between January 2007 and August 2009

After calculating the mole fraction of ammonia in the air leaving through the surface ( $y_e$ ) according to equation III.14, influent mole fraction ( $C_0$ ) and then the  $\text{NH}_4\text{-N}$  concentration were computed using the equations (III.7) and (III.8), respectively.

The calculated influent  $\text{NH}_4\text{-N}$  concentrations of SBR units were in the range of 170-1689 mg/L (Figure IV.23). The highest amount of free ammonia was striped in May 2007 and in this way 9.3% of the ammonium in the influent of this unit was removed. A decrease in free ammonia stripping was observed after SBR-1 has been closed due to decrease in the amount of air introduced to the units. In other words, decrease in G/L ratio after June 2008 affected the free ammonia stripping efficiency negatively. In January 2009, free ammonia stripping efficiency decreased down to

1.2% along with the decreasing temperature and pH values. The calculations showed that, in average, 4% of the ammonium in the influent of SBR units was removed via stripping in these units (Figure IV.23). The findings clearly indicated that, the pH and temperature values and the amount of air introduced with surface aerators are not enough to strip considerable amount of ammonia from SBR units.

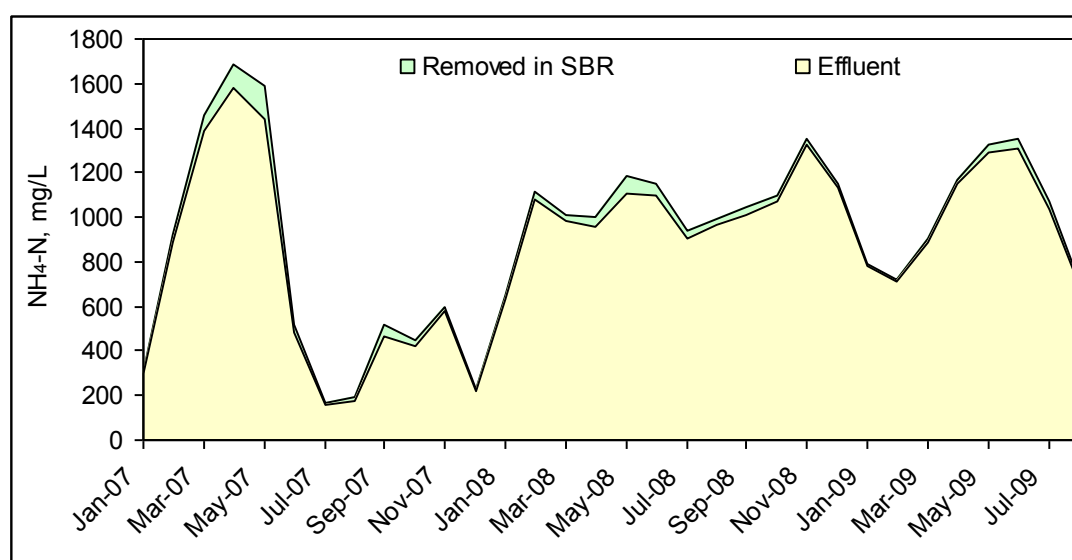


**Figure IV. 23** Effluent and influent  $\text{NH}_4\text{-N}$  concentrations calculated for SBR units between January 2007 and August 2009

The influent  $\text{NH}_4\text{-N}$  concentration of SBR units was considered as the effluent ammonium concentration of facultative lagoons. The effluent free ammonia concentration of facultative lagoons was calculated using equation (III.7) after calculating the ionization constant of ammonium ( $\text{pK}_a$ ) according to equation (III.8).

Average temperature values measured in Facultative Lagoon-1 (Figure IV.15) and Facultative Lagoon-2 (Figure IV.17) were used in the calculations of effluent free ammonia concentrations of Facultative Lagoons. The temperature values that were measured in Facultative Lagoon-2 were used after Line-1 has been closed in June 2008. pH parameter was not measured at the facultative lagoons, and pH values measured in aerobic lagoons (Figure IV.28) were used for free ammonia modeling of facultative lagoons.

The highest free ammonia concentration in facultative lagoons was calculated as 203 mg/L in May 2007 and the lowest concentration was calculated as 10 mg/L in

December 2007. Similarly in SBR units, free ammonia concentration changed parallel to the effluent  $\text{NH}_4\text{-N}$  concentrations of lagoons (Figure IV. 24).

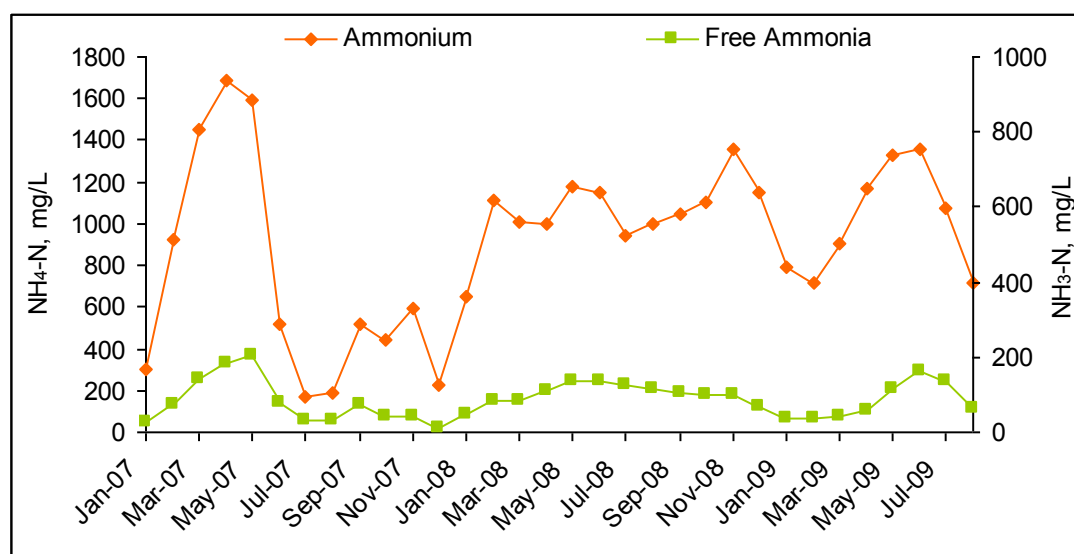


**Figure IV. 24** Effluent  $\text{NH}_4\text{-N}$  and  $\text{NH}_3\text{-N}$  concentrations calculated for facultative lagoons between January 2007 and August 2009

Each facultative lagoon has six floating mechanical surface aerators that have 7.5 kW power. 90 kW power was applied to facultative lagoons with twelve surface aerators until June 2008. This value decreased to 45 kW when Facultative Lagoon-1 and its six surface aerators have been closed. Oxygen input to the facultative lagoons was calculated as 2754 kg  $\text{O}_2/\text{d}$  during the parallel operation period. After June 2008 the oxygen input reduced by half to 1377 kg  $\text{O}_2/\text{d}$ . These values were calculated by using the specific oxygen input capacity of aerators as 1.7 and working hour of surface aerators as 18 hours. Using the fraction of  $\text{O}_2$  in air as 21% and density of air as 1.225 kg/ $\text{m}^3$ , air input was calculated as 13114  $\text{m}^3$  air/d and 6557  $\text{m}^3$  air/d for two line and single-line operation periods, respectively.

Specific air inputs were calculated by dividing air inputs ( $\text{m}^3$  air/d) to leachate flow rate ( $\text{m}^3/\text{day}$ ). Then the specific air input values were converted into gas to liquid ratio (G/L) using equation (III.13) by assuming the molecular weight of the water as 18 g/mole and volume of one mol of air as 24.1 L/mole.

Gas to liquid ratio calculated for facultative lagoons changed from 0.006 to 0.041 mole air/mole leachate between January 2007 and August 2009 (Figure IV.25). After Facultative Lagoon-1 has been closed in June 2008, the G/L ratio decreased

significantly due to reduced amount of air introduced to facultative lagoons and increase in leachate flow rate (Figure IV.25).

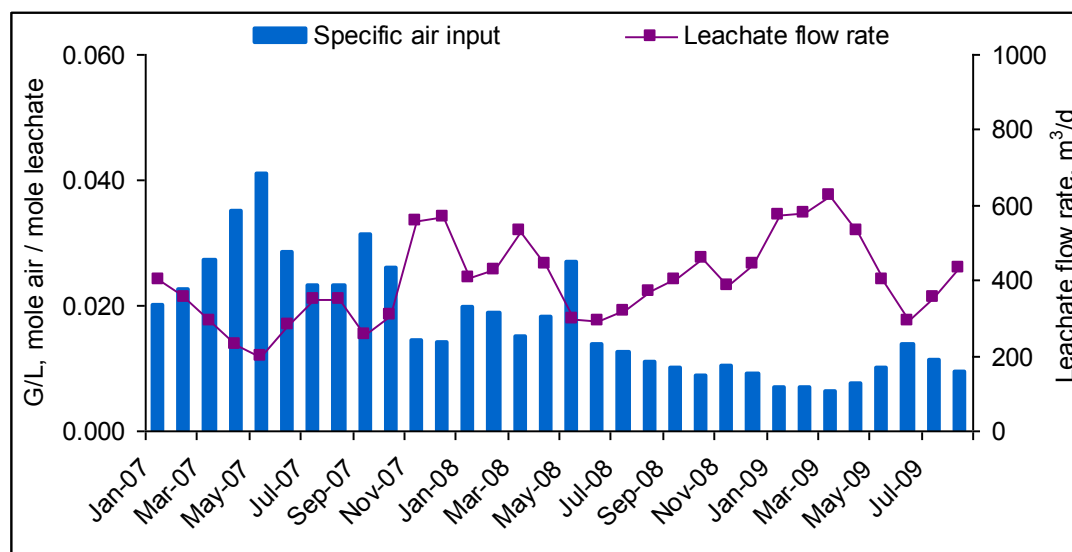


**Figure IV. 25** Gas to liquid ratio (G/L) calculated for facultative lagoons and leachate flow rate between January 2007 and August 2009

$C_e$  of facultative lagoons was calculated by using equation (III.15) by assuming that one liter of water is 55.5 moles as calculated according to (III.16). When calculated,  $C_e$  values were used in computing the mole fraction of ammonia in the air leaving through the surface of facultative lagoons ( $y_e$ ) together with the Henry's law constant in equation (III.16).

The Henry's law constants calculated with average temperatures measured in facultative lagoons varies between 0.42 and 0.98 atm (Figure IV.26). Similarly in SBR units, these values were found to increase in summer months and decrease in winters parallel to the temperature. (Figure IV.26).

After calculating the mole fraction of ammonia in the air leaving through the surface ( $y_e$ ) according to equation III.14, influent mole fraction ( $C_0$ ) and then the  $\text{NH}_4\text{-N}$  concentration were computed using the equations (III.7) and (III.8), respectively.

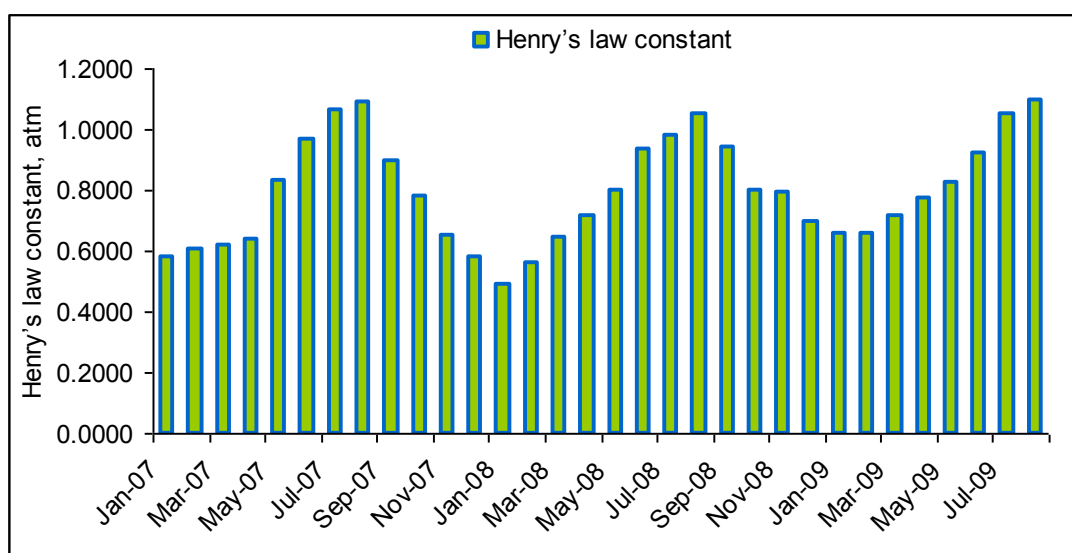


**Figure IV. 26** Henry's law constant at average temperatures measured in facultative lagoons between January 2007 and August 2009

The actual effluent and calculated influent  $\text{NH}_4\text{-N}$  concentrations of SBR units and facultative lagoons between January 2007 and August 2009 are depicted on a graph in Figure IV.27.

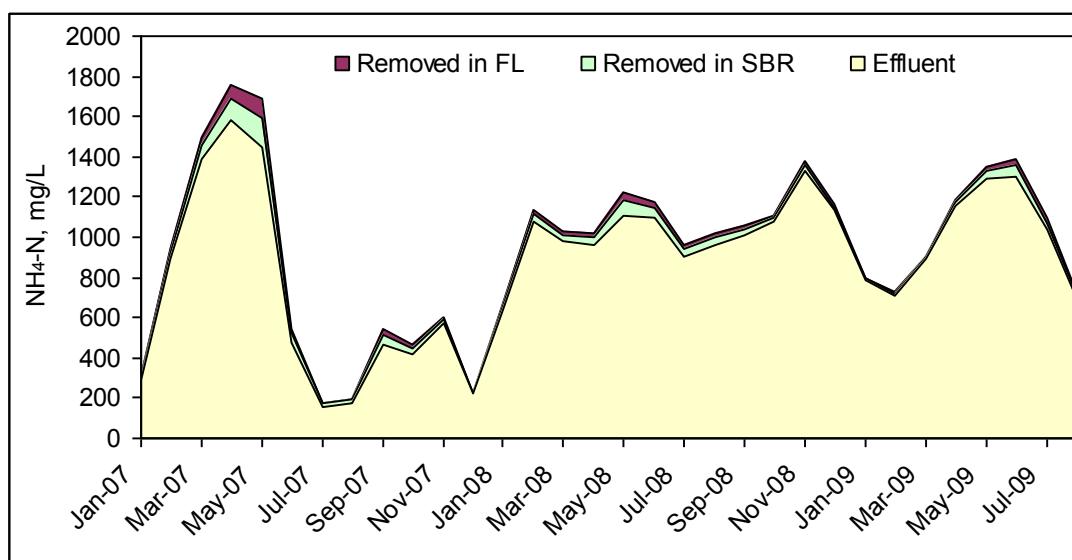


**Figure IV. 27** Effluent and influent  $\text{NH}_4\text{-N}$  concentrations calculated for SBR's and facultative lagoons between January 2007 and August 2009

The calculated influent  $\text{NH}_4\text{-N}$  concentrations of facultative lagoons were close to the effluent concentrations; i.e. it changed between 178 mg/L and 1755 mg/L at

the inlet of facultative lagoons, where outlet concentrations were in a range of 170-1689 mg/L. The highest amount of free ammonia was stripped in May 2007 and in this way 5.8% of the ammonium in the influent of facultative lagoons was removed. A decrease in free ammonia stripping was observed after Facultative Lagoon-1 has been closed due to decrease in the amount of air introduced to the lagoons. In January 2009, free ammonia stripping efficiency decreased down to 0.7% along with the decreasing temperature and pH values. The calculations showed that, in average, 2.3% of the ammonium in the influent was removed via stripping in facultative lagoons. The stripping efficiencies calculated for facultative lagoons were found insignificant and also lower than the efficiencies achieved in SBR units, because in spite of the comparable pH and temperature values, the amount of air supplied to facultative lagoons are quite less than the amount supplied to SBR units.

The influent  $\text{NH}_4\text{-N}$  concentration of facultative lagoons units was considered as the effluent ammonium concentration of aerobic lagoons. The effluent free ammonia concentration of aerobic lagoons was calculated using equation (III.7) after calculating the ionization constant of ammonium ( $\text{pK}_a$ ) according to equation (III.8). Average pH values measured in aerobic lagoons varied between 8.21 and 8.68 in this study period (Figure IV.19).

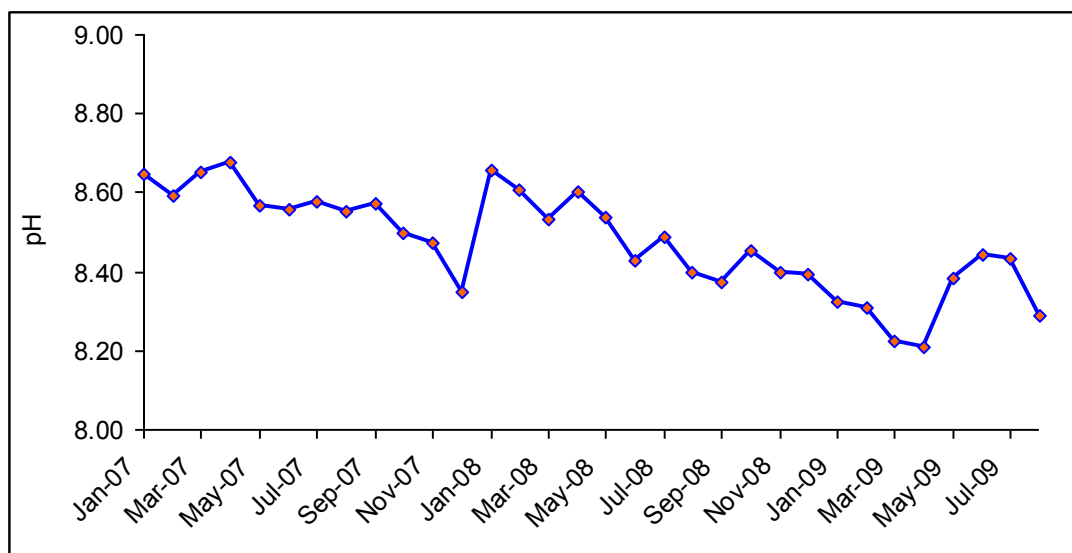


**Figure IV. 28** Average pH values measured in aerobic lagoons between January 2007 and August 2009

The pH values that were measured in Aerobic Lagoon-2 were used after Line-1 has been closed in June 2008.

Likewise, average temperature values measured in Aerobic Lagoon-1 (Figure IV.10) and Aerobic Lagoon-2 (Figure IV.12) were used in the calculations of free ammonia concentrations.

The highest free ammonia concentration in aerobic lagoons was calculated as 244 mg/L in May 2007 and the lowest concentration was calculated as 13 mg/L in December 2007. Similarly in SBR and facultative lagoon units, free ammonia concentration changed parallel to the effluent  $\text{NH}_4\text{-N}$  concentrations of aerobic lagoons (Figure IV. 29).

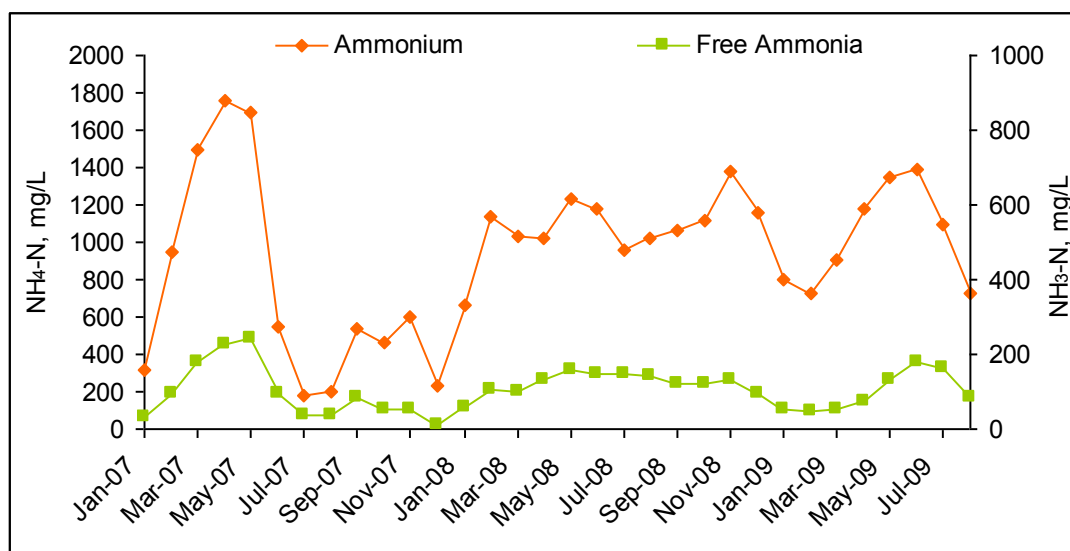


**Figure IV. 29** Effluent  $\text{NH}_4\text{-N}$  and  $\text{NH}_3\text{-N}$  concentrations calculated for aerobic lagoons between January 2007 and August 2009

Each aerobic lagoon has three floating mechanical surface aerators that have 37 kW power. Until June 2008, for aeration in total 222 kW power was applied to aerobic lagoons with six surface aerators. This value decreased to 111 kW when Aerobic Lagoon-1 and its three surface aerators have been closed. Oxygen input to the aerobic lagoons was calculated as 11136 kg  $\text{O}_2\text{/d}$  during the parallel operation period. After June 2008 the oxygen input reduced by half to 5568 kg  $\text{O}_2\text{/d}$ . This value was calculated by using the specific oxygen input capacity of aerators as 2.28 and working hour of surface aerators as 22 hours. Using the fraction of  $\text{O}_2$  in air as 21% and the density of air as 1.225 kg/ $\text{m}^3$ , air input was calculated as 43287  $\text{m}^3$  air/d and 21643  $\text{m}^3$  air/d for two-line and single-line operation periods, respectively.

Specific air inputs were calculated by dividing air input ( $\text{m}^3 \text{ air/d}$ ) to leachate flow rate ( $\text{m}^3/\text{day}$ ). Then, the specific air input values were converted into gas to liquid ratio (G/L) using equation (III.13) by assuming the molecular weight of the water as 18 g/mole and volume of one mol of air as 24.1 L/mole.

The gas to liquid ratio calculated for aerobic lagoons changed from 0.026 to 0.165 between January 2007 and August 2009 (Figure IV.30). After Aerobic Lagoon-1 has been closed in June 2008, the G/L ratio decreased significantly due to the reduced amount of air introduced to aerobic lagoons and the increase in leachate flow rate (Figure IV.30).

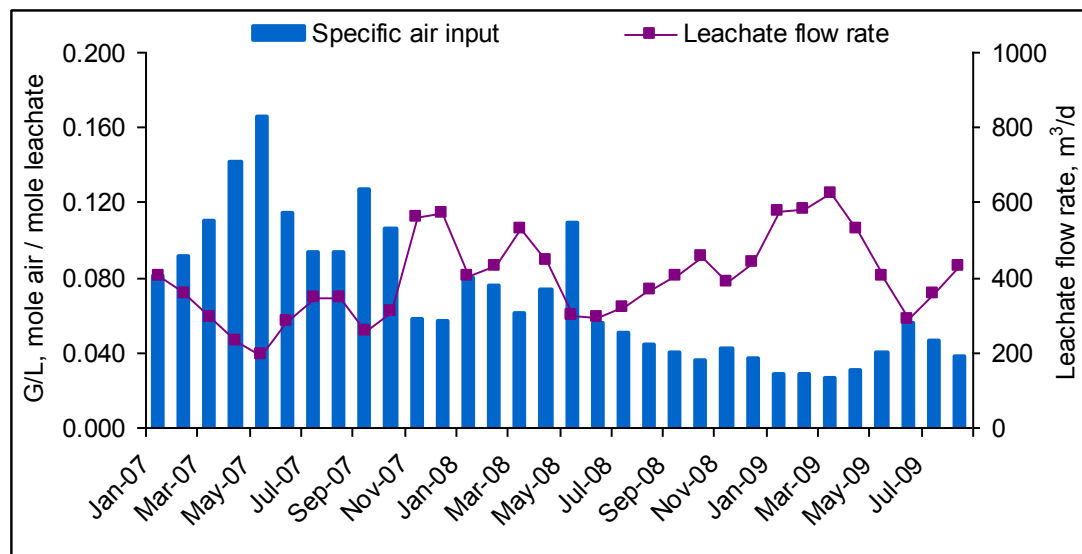


**Figure IV. 30** Gas to liquid ratio (G/L) calculated for aerobic lagoons and leachate flow rate between January 2007 and August 2009

$C_e$  of aerobic lagoons was calculated using equation (III.15) by assuming that one liter of water is 55.5 moles as calculated according to equation (III.16). When calculated,  $C_e$  values were used in computing the mole fraction of ammonia in the air leaving through the surface of aerobic lagoons ( $y_e$ ) together with the Henry's law constant in equation (III.16).

The Henry's law constants calculated with the average temperatures measured in aerobic lagoons varies between 0.49 and 1.1 atm (Figure IV.31). The values were found to increase in summer months and decrease in winters parallel to the temperature (Figure IV.22).

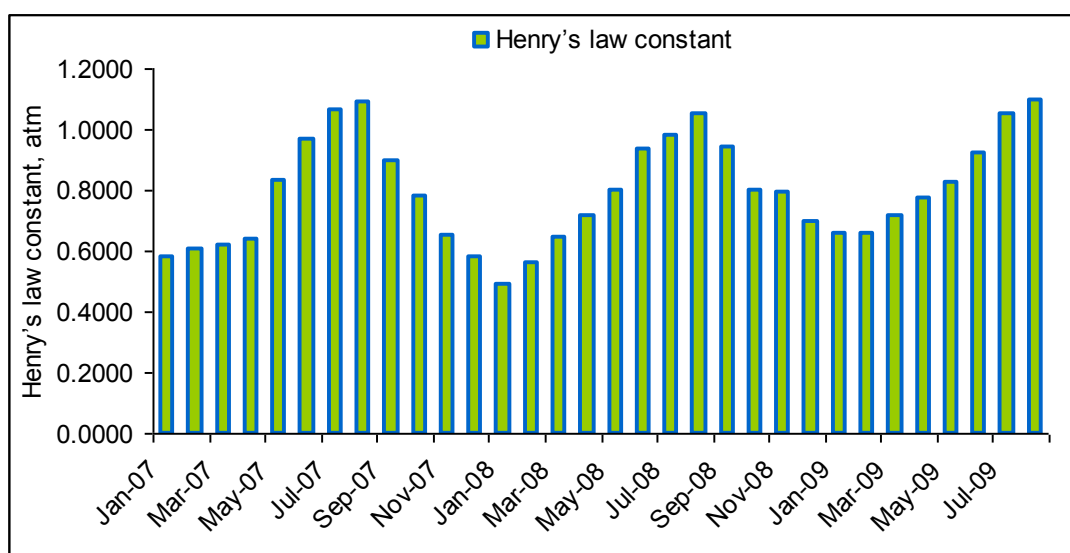


**Figure IV. 31** Henry's law constant at average temperatures measured in aerobic lagoons between January 2007 and August 2009

After calculating the mole fraction of ammonia in the air leaving through the surface ( $y_e$ ) according to equation III.14, influent mole fraction ( $C_0$ ) and then the  $\text{NH}_4\text{-N}$  concentration were computed using the equations (III.7) and (III.8), respectively.

The calculated influent  $\text{NH}_4\text{-N}$  concentrations of aerobic lagoons were in the range of 213-2154 mg/L. The highest amount of free ammonia was striped in May 2007 and in this way 21.6% of the ammonium in the influent of aerobic lagoons was removed (Figure IV.32). A decrease in free ammonia stripping was observed after Aerobic Lagoon-1 has been closed due to decrease in the amount of air introduced to the lagoons. In other words, decrease in G/L ratio after June 2008 affected the free ammonia stripping efficiency negatively. In January 2009, free ammonia stripping efficiency decreased down to 3.6% along with the decreasing temperature and pH values. The calculations showed that, in average, 9.7% of the ammonium in the influent of aerobic lagoons was removed via stripping in these units (Figure IV.32). The findings clearly indicated that, the pH and temperature values and the amount of air introduced with surface aerators are not enough to strip considerable amount of ammonia from aerobic lagoons.

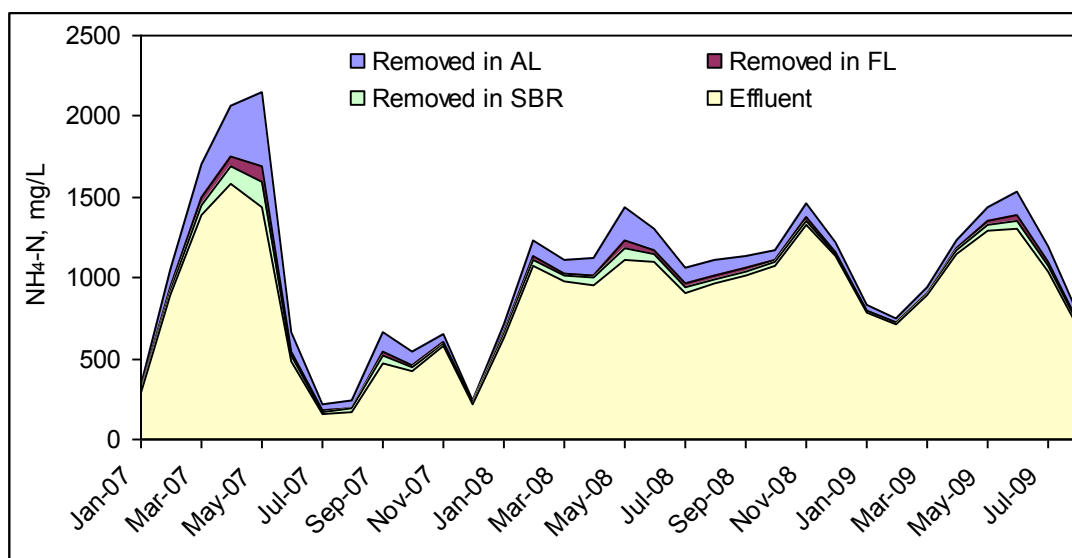


**Figure IV. 32** Effluent and influent  $\text{NH}_4\text{-N}$  concentrations calculated for SBR's, facultative lagoons and aerobic lagoons between January 2007 and August 2009

#### IV.2.2.2. Biological Ammonium Removal

Although the theoretical calculations showed that only maximum 32% of the influent ammonium could be removed via stripping in HLTP, especially in summer months quite low ammonium concentrations were observed in the effluent (Figure IV.33). This condition indicated that there has to be some other means for ammonium removal such as nitrification/denitrification processes.

In the beginning of the study in January 2007, a quite low  $\text{NH}_4\text{-N}$  concentration of 570 mg/L was measured in the influent of HLTP (Figure IV.33). This low value can be explained with the dilution effect of the rainfall (Figure IV.1) seen several days in this month. Starting from February 2007, the influent  $\text{NH}_4\text{-N}$  concentration rose sharply and remained around 2000 mg/L in the coming months (Figure IV.33). Until May 2007, average  $\text{NH}_4\text{-N}$  concentration measured in the effluent of HLTP was about 1500 mg/L. Starting with the warmer summer months, effluent  $\text{NH}_4\text{-N}$  concentrations dropped below 200 mg/L while Line-1 and Line 2 were operated together. 92%  $\text{NH}_4\text{-N}$  removal was achieved in July 2007 when at the same time leachate temperature raised up to 26 °C (Figure IV.33). Simultaneously, in average 200 mg/L  $\text{NO}_3\text{-N}$  was measured in the effluent of HLTP showing that considerable amounts of  $\text{NH}_4\text{-N}$  were oxidized to  $\text{NO}_3\text{-N}$  biologically via nitrification. It is also possible that some  $\text{NO}_3\text{-N}$  was consumed as electron acceptor in facultative lagoons where the low dissolved oxygen conditions are suitable for denitrification. Therefore

the  $\text{NO}_3\text{-N}$  concentrations measured in the effluent of HLTP could be lower than the  $\text{NO}_3\text{-N}$  produced with nitrification (Figure IV.33).

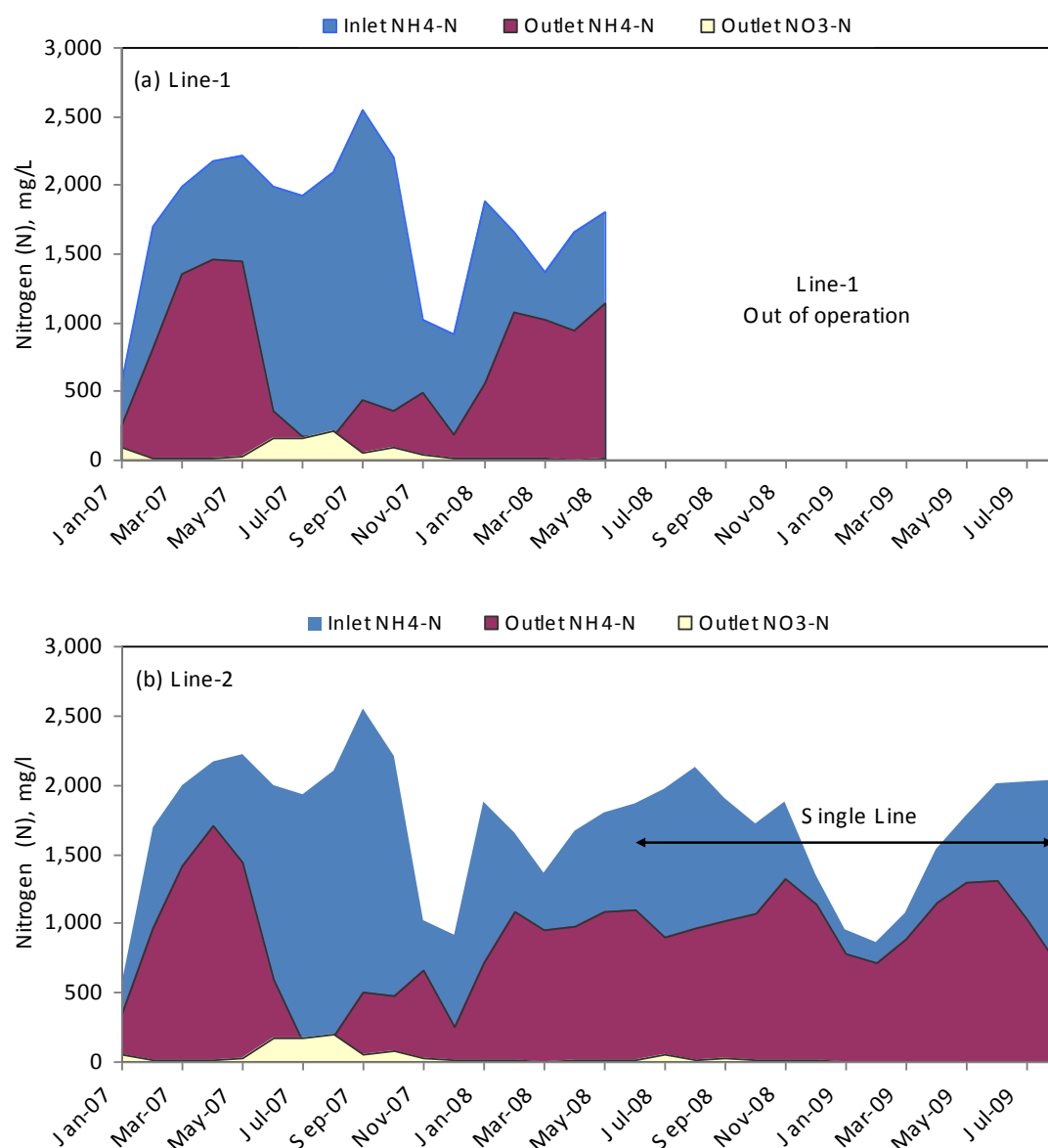


**Figure IV. 33** Influent and effluent  $\text{NH}_4\text{-N}$ , and effluent  $\text{NO}_3\text{-N}$  concentrations; (a) Line-1 (b) Line-2

61% of the  $\text{NH}_4\text{-N}$  in the influent was removed in HLTP between January 2007 and October 2007. The calculated ammonia stripping efficiency for this period was only 12% (Figure IV.32). Thus, the remaining 49% of ammonium is believed to be removed biologically. Other strong indications that support the occurrence of biological nitrogen removal in this period are the considerable amount of  $\text{NO}_3\text{-N}$  measured in the effluent of HLTP and the increasing biological activity reflected by

the biomass (MLSS) build-up in aerobic lagoons parallel to the elevation in temperature.

After October 2007,  $\text{NH}_4\text{-N}$  removal in HLTP decreased to 46%. In average the 1500 mg/L influent  $\text{NH}_4\text{-N}$  concentration reduced to 800 mg/L while almost no  $\text{NO}_3\text{-N}$  was measured in the effluent. The drop in  $\text{NH}_4\text{-N}$  removal is supposed to be caused by decreasing biological activity due to relatively lower leachate temperatures.

After Line-1 has been closed, the maximum  $\text{NH}_4\text{-N}$  removal was about 55% in Line-2 in summer 2008. In this period, only 50 mg/L  $\text{NO}_3\text{-N}$  was measured in the effluent of HLTP. It is believed that higher hydraulic loads applied on single operating Line-2 and resulting lower HRTs (average 11 days) adversely affected the nitrification in this period and reduced biological  $\text{NH}_4\text{-N}$  removal.

Between October 2008 and March 2009, heavy rain falls in winter increased the leachate flow rate from 420  $\text{m}^3/\text{d}$  to 840  $\text{m}^3/\text{d}$ . As a result, hydraulic load applied on single operating Line-2 elevated significantly and  $\text{NH}_4\text{-N}$  removal efficiency decreased below 20%. Relatively lower temperatures in the lagoons and SBRs are also supposed to be another reason of decreased nitrogen removal. Because of low temperature, the calculated ammonia stripping efficiency was also insignificant and less than 6%.

Starting with the warmer summer months, although the influent  $\text{NH}_4\text{-N}$  concentrations exceeded 2000 mg/L, the  $\text{NH}_4\text{-N}$  removal increased up to 66% in August 2009 in accordance with increasing temperature and biological activity. Unfortunately, after January 2009  $\text{NO}_3\text{-N}$  measurement in the effluent of HLTP was terminated. Therefore, it is difficult to show the contribution of nitrification on  $\text{NH}_4\text{-N}$  removal. However, the maximum ammonia stripping efficiency calculated around 8% shows that biological  $\text{NH}_4\text{-N}$  removal was again the main removal process in this period.

Although, free ammonia concentration exceeds 240 mg/L in summer 2007 and elevates up to 170 mg/L in summer 2008 and 2009 due to increasing temperature and influent  $\text{NH}_4\text{-N}$  concentrations, significant biological  $\text{NH}_4\text{-N}$  removal was experienced in HLTP. In contrast, 150 mg/L free ammonia was reported to severely inhibit nitrification of landfill leachate (Calli et al., 2005). The results given above show that when temperature is above 25 °C and the total HRT is higher 15 days, significant biological  $\text{NH}_4\text{-N}$  removal as high as 90% can be achieved in HLTP.

Under these circumstances, it can be concluded that, nitrification is the main  $\text{NH}_4\text{-N}$  removal process in HLTP. Probably it occurs simultaneously with denitrification. Ammonia removal via stripping also occurs however, the amount of ammonia removed via this way is very limited because of the low reactor pH for this purpose and insufficient amounts of air supplied. During 32-month study period, it is estimated that in average 9% of the influent  $\text{NH}_4\text{-N}$  was removed via stripping and 37% via nitrification and in average 75% of the ammonium removal is caused by nitrification and remaining 25% of the ammonium removal is caused by free ammonia stripping.

## CHAPTER V

### CONCLUDING REMARKS and RECOMMENDATIONS

In this study, the time based variation in the characteristics of Hamitler Landfill leachate and organic carbon and ammonium nitrogen removal performances of Hamitler Leachate Treatment Plant (HLTP) were evaluated for 32 months between January 2007 and August 2009. In evaluating the treatment performance of HLTP, organic and hydraulic loads applied, COD, BOD<sub>5</sub> and NH<sub>4</sub>-N removal efficiencies, kinetic coefficients of BOD<sub>5</sub> removal were calculated. In addition, the maximum amount of free ammonia that can be stripped was estimated and afterward compared with total NH<sub>4</sub>-N removal in order to explain the prevailing means for nitrogen removal.

Significant increases have been observed in incoming leachate flow during heavy rainfall periods between October and April. This condition adversely affected the performance of HLTP which was designed for a flow rate of 500 m<sup>3</sup>/day. When leachate flow rate exceeded 650 m<sup>3</sup>/day some portion of leachate was by-passed. Additionally, depending on cumulative amount of municipal solid waste landfilled, yearly average leachate flow rate gradually rose from 450 m<sup>3</sup>/day in 2007 to 550 m<sup>3</sup>/day in 2009.

High concentrations of COD and BOD<sub>5</sub> ranging from 713 to 38400 mg/L and from 240 to 19000 mg/L respectively, shows that Hamitler Landfill is a young to medium age landfill in acid formation phase. High COD and BOD<sub>5</sub> concentrations measured in summer, reduces in winter months due to the dilution effect of precipitation. The average BOD<sub>5</sub>/COD ratio of about 0.49 indicates that the leachate generated from Hamitler Landfill contains high amounts of biodegradable organic matter.

During 32 month study period, total and ammonium nitrogen concentrations fluctuated between 560 and 3916 mg/L and 105 and 3392 mg/L, respectively. The average NH<sub>4</sub>-N/TN ratio of about 0.72 indicated that a noticeable amount of the

nitrogen was in ammonium form. This ratio dropped with heavy rainfall as non-hydrolyzed organic nitrogen content increased in leachate.

Until Line-1 has been closed in June 2008, Line-1 and Line-2 were operated together with 184 m<sup>3</sup>/day average hydraulic load and 23 days hydraulic retention time (HRT). After this date, the hydraulic load on Line-2, which was operated alone, increased to 430 m<sup>3</sup>/day and as a result HRT was decreased to 11 days.

COD and BOD<sub>5</sub> removal efficiencies followed a similar profile during 32-month study period. The average BOD<sub>5</sub> removal in both aerobic and facultative lagoons and SBR's were 75%, 44% and 55%, respectively which makes 94% in total. After Line-1 has been closed, BOD<sub>5</sub> removal decreased to 45%, 42%, and 30% in Aerobic Lagoon-2, Facultative Lagoon-2 and SBR-2, respectively. As a result, the total BOD<sub>5</sub> removal efficiency decreased to 78% when Line-2 was operated alone. It is concluded that, the deterioration in COD and BOD<sub>5</sub> removal was a result of increased hydraulic load and the consequent decrease in HRT.

When both lines were under operation, the BOD<sub>5</sub> removal rate constants calculated for aerobic lagoons were in the range of 0.07-1 d<sup>-1</sup>. This range decreased significantly to 0.01-0.4 d<sup>-1</sup> after Line-1 has been closed as a result of increased hydraulic load on Aerobic Lagoon-2. It should be noted that, these rate constants are considerably lower than the design value of 1.3 d<sup>-1</sup>. The BOD<sub>5</sub> removal rate constants calculated for facultative lagoons did not change noticeably after Line-1 has been closed. The average value of about 0.10 d<sup>-1</sup> was 8-fold lower than the design value.

The results of calculations about ammonia stripping clearly indicated that, pH and temperature and the amount of air supplied are not enough to strip considerable amount of ammonia in HLTP. Although the theoretical calculations showed that maximum 32% of the influent ammonium could be removed via stripping, especially in summer months quite low ammonium concentrations were observed in the effluent. This condition indicated some other means for ammonium removal such as nitrification/denitrification processes.

Starting with the warmer summer months, effluent NH<sub>4</sub>-N concentrations dropped below 200 mg/L while Line-1 and Line 2 were operated together. 92% NH<sub>4</sub>-N removal was achieved in July 2007. Simultaneously, in average 200 mg/L NO<sub>3</sub>-N was measured in the effluent of HLTP showing that considerable amounts of NH<sub>4</sub>-N were oxidized to NO<sub>3</sub>-N biologically via nitrification. It is concluded that higher

hydraulic loads on Line-2 after Line-1 has been closed adversely affected nitrification and caused lower  $\text{NH}_4\text{-N}$  removal.

Finally, it is concluded that, although small amount ammonia is removed via stripping, the main  $\text{NH}_4\text{-N}$  removal process in HLTP is nitrification. It was estimated that in average 9% of the influent  $\text{NH}_4\text{-N}$  was removed via stripping and 37% via nitrification, and in average, 75% of the ammonium removal is caused by nitrification and remaining 25% of the ammonium removal is caused by free ammonia stripping during 32-month study period.

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# CURRICULUM VITAE

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| Institution [ Date from - Date to ]                                                         | Degree(s) or Diploma(s) obtained: |
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7. **Language skills:** Indicate competence on a scale of 1 to 5  
(1 - excellent; 5 - basic)

| Language | Reading       | Speaking | Writing |
|----------|---------------|----------|---------|
| Turkish  | Mother tongue |          |         |
| English  | 1             | 1        | 1       |

8. **Other skills (e.g. Computer literacy, etc.):**

Full computer literacy (Word, Excel, PowerPoint, Microsoft Project, Access), AutoCAD, WaterCAD, StormCAD, SewerCAD, Belebungs-Expert (ATV-DVWK-A 131), MS Water, MS Channel, Activated Sludge Model (ASM), Single Sludge Simulation Program (SSSP), Visual Basic, Fortran, Excel Stripping via Visual Basic

9. **Other relevant information (eg, Publications):**

Kaşıkçı, K.; Çallı, B.: “Investigation of Bursa Hamitler Landfill Leachate Treatment Plant”, *Solid Waste Management at Turkey Symposium (TURKAY 2009)*, Yıldız Technical University, Istanbul / TURKEY, (2009), 305 – 313.