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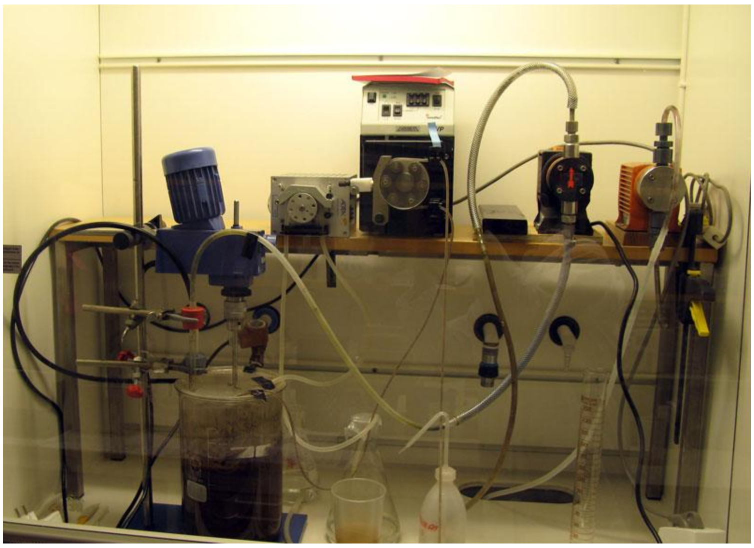
Water and Environmental Engineering
Department of Chemical Engineering

Leachate Treatment at Filborna Landfill with Focus on Nitrogen Removal



Master's Thesis by

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August 2009

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1

1. Sequencing Batch Reactor at the laboratory of the Department of Chemical Engineering, Lund University, Sweden

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Summary

Leachate is a liquid, which drains through the garbage at the landfill sites, thus being heavily contaminated by the organic matter, nutrients and toxic substances that are washed from the garbage. This master thesis is devoted to the problem of on-site leachate treatment at Filborna landfill in Helsingborg, southern Sweden, focusing on nitrogen removal from leachate. The thesis work was performed in cooperation with a Swedish landfill operator Nordvästra Skånes Renhållning AB.

Extended literature study presents number of methods, which are currently used in the industrial scale as well as developing methods that are still at the trial stage. Among these methods sequencing batch reactor (SBR) was chosen as the most perspective one for leachate treatment and tried at the lab-scale. The novel anammox process was unsuccessfully attempted to be adjusted for nitrogen removal from leachate. The current system, which allows only partial leachate treatment before withdrawal to the municipal wastewater treatment plant (WWTP), is described and evaluated.

Current leachate treatment system is a low-technique solution based on the utilization of aerated and sedimentation ponds. Though decreasing concentrations of the contaminants in the effluent, it does not provide appropriate treatment for leachate to be withdrawn directly to the environment. The additional treatment at the municipal WWTP is charged by the latter for about 2.5 millions SEK annually and the running of magnesium ammonium phosphate (MAP) precipitation plant installed at the site could be profitable only if the nitrogen concentrations in the influent are higher than existing ones. Moreover, MAP precipitation doesn’t lead to the decrease of nitrogen concentration in the effluent below discharge limits.

The SBR combined nitrification/denitrification process, which was running in the laboratory during 160 days, indicated high nitrogen removal rates, decreasing ammonium nitrogen in the effluent 99 % and total nitrogen 93 %. The process proved to be feasible for complete nitrogen removal. The main operational parameters were determined and calculation of full-scale installation performed. Different operational conditions were tried for prediction of the ability of bacterial cultures to withstand unfavorable conditions.

At the same time, the SBR treatment does not decrease phosphorous and COD concentrations in the effluent below discharge limits. The development of appropriate treatment processes is a matter of further researches.

Acknowledgements

The following report is the result of extended laboratory work and literature study at Water and Environmental Engineering, Department of Chemical Engineering. The preliminary study of nitrogen removal from leachate at Filborna landfill, which stimulated further research, has been carried out as a part of Environmental Biotechnology course in February 2008. The present MSc project has been implemented since November 2008 until August 2009.

I would like to express my keen appreciation to my supervisor Professor Jes la Cour Jansen, who was eager to support my wish to implement the project at the department. I am grateful to Professor Jes la Cour Jansen for all that knowledge, which I obtained while studying under his supervision, for his patience and understanding, and the valuable comments on my job and hints to overcome the problems, which arose during research.

I appreciate helpfulness of my examiner, Associate Professor Karin Jönsson. I am especially grateful to her for the support of my eagerness to continue working within the project on leachate treatment at the Department after finishing current master thesis.

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Andrii Butkovskyi

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Table of Contents

1 Introduction

The problem of landfill waste is an important issue for the environmental protection policy of the EU countries. Sweden, which was one of the first countries in Europe that realized the need for sustainable environmental development, makes considerable progress in environmental protection. Environmental Quality Objectives, adopted by Riksdag in 1999, aim at the solution of major environmental problems, with waste generation and handling being one of the most important of them [www.sweden.gov.se, 2009]. Thus decrease of landfill impact on the environment is a part of national policy and requires sufficient efforts from landfill operators. Though organic waste will not be stored at the landfills since year 2009, the existing facilities continue to affect surrounding ecosystems.

Leachate being the liquid that drains through a waste and consisting of both an internal moisture of the waste and precipitations that fall on a landfill, will be an important source of soil and groundwater contamination for the next several decades, even if landfilling of the organic waste is banned. Leachate contains mixture of heavy metals, large quantities of nitrogen (organic nitrogen and ammonium) and inert slowly degradable organic substances (fulvic and humic acids). Withdrawn directly to the environment, nitrogen contaminated liquids will cause eutrophication and subsequent oxygen depletion. Nitrogen is especially important when it comes to the eutrophication of saline water basins (seas and oceans), as it is the limiting nutrient for marine algae. Thus, regions that withdraw wastewater to seas (as the province of Skåne does) should put special concern to the nitrogen removal from different types of wastewater.

Currently Skåne has 10 municipal landfills according to the latest data from Avfall Sverige [Avfall Sverige, 2009]. Most of them solve the leachate problem by simple withdrawn to municipal wastewater treatment plants (WWTPs), often with preliminary sedimentation or on-site pond treatment. The latter helps to remove not more than 50% of the total nitrogen and only easily degradable organic material. It means that the subsequent removal is required. Leachate, which differs in composition from domestic wastewater, could be the source of potential operational problems at municipal WWTPs. On the other hand, landfill contractors are charged by WWTPs for the leachate, which comes from landfill facilities. Thus the construction of cheap and reliable on-site leachate treatment systems that allow effluent to be released directly to the environment will be beneficial for both landfill operator and WWTP.

1.1 Objectives

The objectives of the current master thesis are:

- to describe backgrounds of leachate formation and its characteristic together with state of the art in the leachate treatment technologies;
- to evaluate leachate treatment technologies, which are currently used for partial leachate treatment at the particular landfill in the southern Sweden;
- to check on the lab-scale the possibility of using sequencing batch reactor (SBR) for complete treatment of leachate from particular landfill;
- to determine main design parameters for SBR and preliminary design of SBR installation for leachate treatment;
- to attempt adjusting novel nitrogen removal method (Anammox process) for nitrogen removal from leachate.

1.2 Aims

The aims of the current master thesis are:

- to deepen the knowledge about on-site leachate treatment techniques;
- to evaluate and develop reliable methods for complete leachate treatment that can be used for designing of treatment facilities in the southern Sweden;
- to create scientific base for further researches concerning leachate treatment.

1.3 Methodology

For successful achievement of the objectives and aims of this thesis, different scientific approaches had to be applied, which included both studying of the theory behind the problem and practical work in the laboratory.

The extended literature study of the leachate treatment methods which are currently used both in a lab scale and an industrial scale was conducted to determine the strategy to be focused on. For description of current leachate treatment system study visits to the landfill were organized and information about the treatment facility was obtained from NSR AB. For evaluation of possibilities to use SBR and anammox process for leachate treatment and determination of main process design parameters laboratory studies were performed from November 2008 till April 2009. The obtained design parameters were used for calculation of the SBR facility according to the ATV-M 210 model from German Association for the Water Environment.

1.4. Limitations

The following thesis is focused on the treatment of leachate from Filborna landfill which already passed the pre-treatment in buffer pond at the landfill site. Leachate is derived from an old landfill site thus having characteristic properties and composition, more or less typical for landfills in Sweden, where landfilling of organic waste is prohibited. SBR technique was chosen as the most promising one for achieving aims and objectives of this study. The report is focused mainly on nitrogen removal from leachate, though COD and phosphorous removal is also evaluated.

1.5 Content of the report

The report consists of seven chapters. The current chapter is a short preface which gets the reader acquainted with the problem of leachate treatment as well as aims and scope of this report. Chapter 2 is a literature review, where leachate as a type of toxic waste is characterized and the short overview of processes, which are utilized for leachate treatment, is presented. The existing leachate treatment system at Filborna landfill is described and evaluated in the Chapter 3. Chapters 4 and 5 are an extended description of laboratory experiments with presentation of results and discussion. Chapter 4 is focused on the various aspects of lab-scale SBR process, while chapter 5 gives short description of the experiment with anammox bacterial culture, which was not successful. Chapter 6 presents the calculations of SBR facility for Filborna landfill, based on the data, obtained during laboratory experiments. The conclusions of the work are listed in the Chapter 7.

2 Leachate characteristic and treatment methods

2.1 General characteristic and composition of leachate

General term “leachate” refers to any liquid, which is percolating (“leaching”) through the layer of solids [www.leachate.co.uk, 2009]. Very often it washes out the substances that are present in the solid phase, transferring them into dissolved condition. Nowadays the term is most often used for liquid, percolating through the garbage at the landfill sites, as it is of immense importance as the source of pollution of both the soil at the landfill site and groundwater layers in the district.

2.1.1 The historical background

The need for leachate treatment at the landfill sites has been noticed not long ago, comparatively to the problem of wastewater treatment. The reason of this absence of attention to the important question of environmental protection is that the landfills’ compositions and sizes differed a lot from today’s ones even 50 – 60 years ago. Until the mid of the 20th century most of the organic substances were either composted at the gardens or burned in the open fires, which was the most common way of heating [www.leachate.co.uk, 2009]. However, the situation changed gradually with the urbanization due to the several important factors:

1) Increased amount of different types of waste produced by people and industries;
2) Widespread utilization of toxic substances and chemicals in everyday life, since chemical industry developed rapidly after 2nd World War;
3) Transition from open fire heating to centralized heating of houses;
4) Higher percentage of people living in the urbanized area.

All these factors led to both a rapid increase of amount of landfilled waste and to changes in its composition. An organic material with high content of organic nitrogen and phosphorous, which was previously burned or used as compost, ended up at the landfills. Lots of toxic waste, containing heavy metals and toxic organic compounds, has been landfilling too.

The need for landfill leachate collection and treatment was first understood in 70th. At those times the concept “dilute, attenuate and disperse” was prevailing. Landfills were placed at the sites with permeable soils to allow leachate percolation through soil. It was believed, that dilution and dispersion of leachate will minder the problem. But an uncontrolled percolation of leachate through the soil without proper evaluation of soil structure and hydrology of the place led to the contamination of groundwater below many landfills. This caused another strategy to be prevailed – to collect leachate from landfill and treat it either at the municipal wastewater treatment plant or separately.

Recent changes in the landfilling legislation in many EU countries, including Sweden, had significant impact on the issue [Naturvårdsverket. Deponeringsförbud, 2009]. Landfilling of organic waste tends to be minimized and prohibited at all, with a shift towards controlled digestion of organic material in the constructed digesters. Still, in spite of prohibition of landfilling, problem of leachate handling will be important during next 70 – 100 years, as the existing closed landfills will generate leachate during several decades. The second issue concerns the shift towards on-site treatment of leachate instead of discharging it to the municipal wastewater treatment plants – both because of appropriate legislative measures and stringent requirements of wastewater treatment plants for an acceptance of incoming industrial wastewater.

2.1.2 Landfill structure and water balance

Before presenting detailed overview of the processes, which are currently used for leachate treatment and have potential to be used in the future, the basic hydrology of landfills, processes behind leachate generation and composition of leachate should be covered.

Modern landfills are usually situated at the sites with appropriate geological understructure – impermeable soil (e.g. clay) and a deep level of groundwater is preferred to exclude groundwater contamination. The site should also have low land value, be distinct from dwelling areas and have good transport connections to the served community.

In addition to the natural low permeable soil, artificial liner made from impermeable materials (e.g. compacted clay liner, bentonite-enhanced soil, geosynthetic clay liner, textile membranes) is utilized [Williams, 2005]. Leachate is usually collected above this layer through the system of pipes. Landfill may contain gas drainage system for the collection of methane gas, which is generated during anaerobic decomposition of organic material. Methane gas could be further used at the landfill itself or sold. Above the landfill artificial sealing layer is usually required. It serves for several purposes: to prevent blowing of garbage from the landfill by wind, to minder the amount of percolated precipitations, to prevent pollution of surface runoff water, to minimize the spread of diseases by common landfill “inhabitants”, such as rats or seagulls. The impermeable liners could also be placed between different layers.

The simplified water balance of the landfill is shown on the Figure 2.1. The precipitations at the landfill site could percolate through the landfill, evaporate back to the atmosphere or end up in the surface runoff. The latter is also contaminated liquid and should be treated together with leachate. Another important source of leachate is a moisture content of the garbage itself. While being pressurized by the new layers of garbage, put above, the old one loses its water which is mixed with percolating precipitations. Most of the leachate is collected at the liner, but still some minor part of it would percolate to the ground.

The water balance of landfill and production of leachate could be described by the following equation [Björnsson, 2006]:

$$L = P - E - R - \delta\Delta M \qquad \text{(eq. 2.1)}$$

where:

L – leachate production;
P – precipitation;
E – evapotranspiration;
R – surface runoff;
$\delta\Delta M$ – changes in the water content of the landfill.

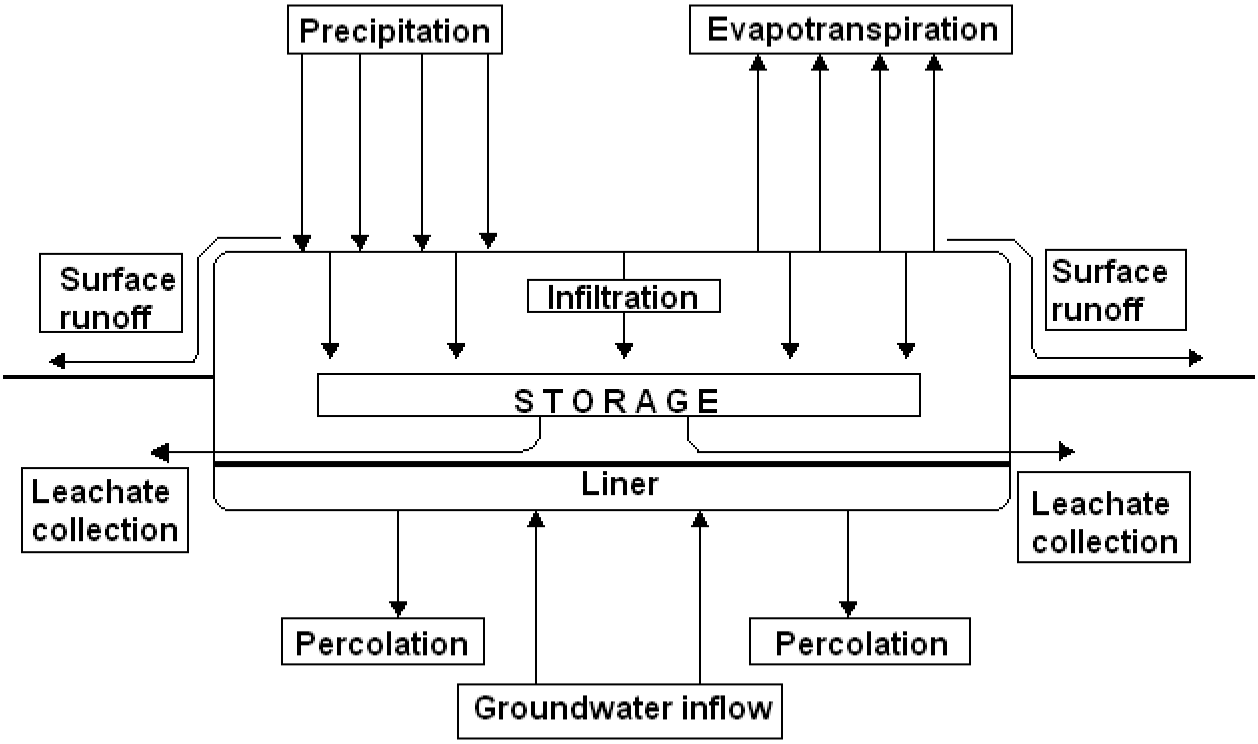


Figure 2.1 Schematic overview of the water cycle at constructed landfill with impermeable liner

Leachate is collected by a set of collection pipes and pumps, which are installed at the bottom of a landfill. Two different types of pumps could be used: borehole pumps or eductor-jet pumps [www.portfolio.mvm.ed.ac.uk, 2009]. Borehole pumps start working only when leachate is present in the collection pipes, which is detected by special sensors. They consume less energy, but are less reliable. Eductor-jet pumps are usually used for drainage purposes. They work continuously independently on whether leachate is present or not, thus, they consume more energy but are simpler than borehole pumps and more reliable. Usually it is not necessary to run them 24 hours a day, so, energy could be saved. In such way eductor-jet pumps could be a competitive solution to borehole pumps.

2.1.3 Biological processes in the landfill

The set of biological processes takes place in the operated landfill and affects leachate formation. Four phases of decomposition could be marked out:

1) Hydrolysis (aerobic decomposition). It lasts during several months at the surface layer of the landfill (50 – 80 cm) where oxidation of organic matter (usually, food leftovers) take place at pH 6.5 – 7.2. Oxidation processes facilitate corrosion of metals and acidification as the result of organic decomposition. Aerobic decomposition is an exothermic process and temperature in outer layers of landfill could rise up to 80°C. Together with high content of poisonous substances, it leads to gradual decrease of the amount of pathogenic microorganisms in leachate [Hamer, 2003].

2) Acidogenic phase. The first stage of anaerobic decomposition of organic matter. Monomers of organic compounds, which are the products of hydrolysis, are converted into volatile fatty acids (e.g., propionate, butyrate). Only part of the

organic matter (ca 20%) is converted through acidogenesis, another part is directly converted to the acetic acid, CO_2 and H_2O via acetogenic pathway.

3) Acetogenic phase. The second stage of anaerobic decomposition of biodegradable waste with acetic acid, CO_2 and H_2O as the main products of decomposition. The formation of acids leads to decrease of pH down to 4.5 – 6.5. Metals are oxidized in acidic conditions and become soluble. Thus, the COD and heavy metals concentrations are very high in this phase. Usually acidogenic and acetogenic phases lasts up to 5 years together.

4) Methanogenic phase. The third stage of anaerobic decomposition is followed by fermentation of acids, which are generated in acetogenic phase. Gases, such as methane, carbon dioxide, ammonium hydrate are produced. pH rises up to 7.2 – 8.6. Active methanogenesis lasts up to 30 years, and stable methanogenic process could last up to 100 years. COD and heavy metals content decreases during these stages with prevalence of fraction of inert organic compounds among COD and gradual decrease of BOD/COD ratio.

The simplified scheme of metabolic processes in the landfill is shown on the Figure 2.2.

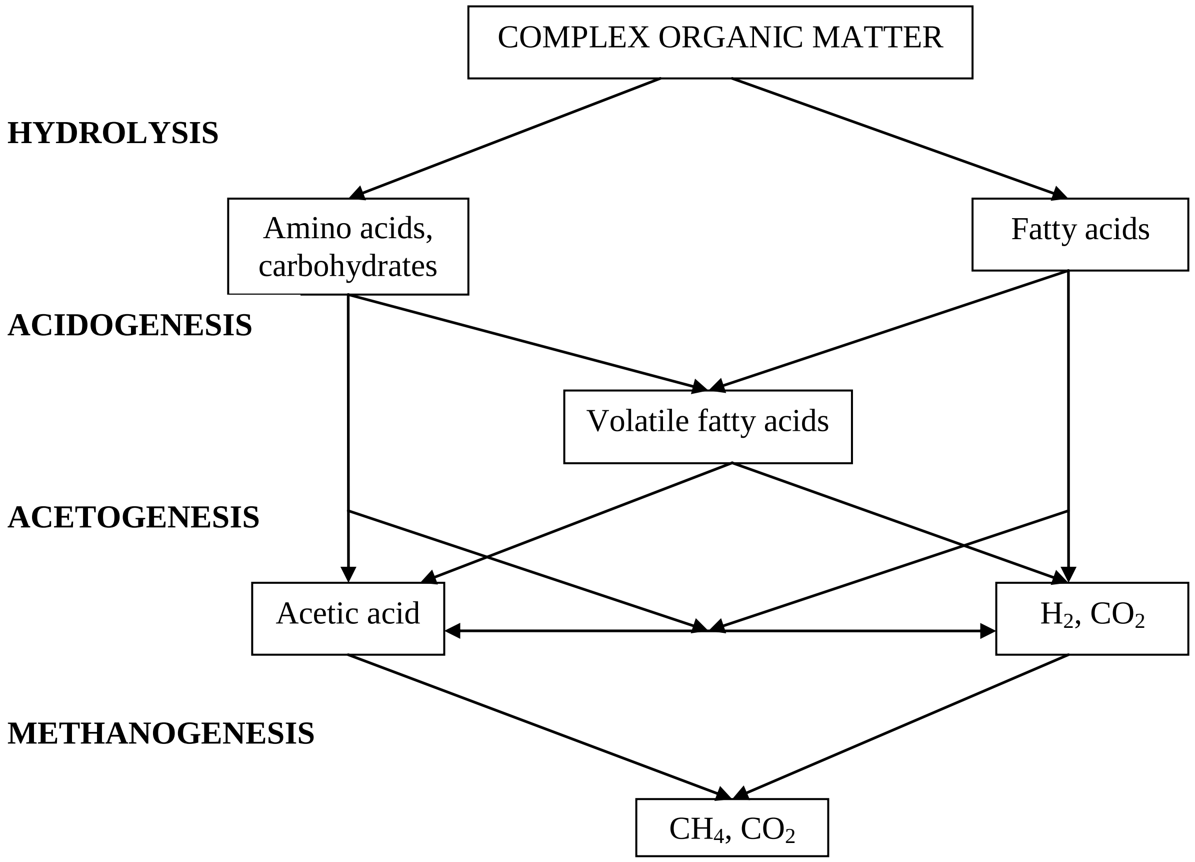


Figure 2.2 Metabolic processes in leachate

2.1.4 Leachate composition

Leachate is usually more contaminated, than conventional household wastewater that is used to be treated at the municipal wastewater treatment plants, thus it is regarded to be an industrial wastewater. It includes high concentrations of COD and nitrogen. BOD and phosphorous concentration can vary depending on the age of landfill and type of waste stored at the landfill.

As leachate is continuously produced at the landfills with various types of depositions and various conditions of their storage, it is clear that its composition should differ greatly depending on the type of landfill waste, age of the landfill, deposition conditions, climate conditions such as temperature and humidity.

Landfill waste could be classified according to the degradability into the following groups:

- biodegradable (food, paper, wood, garden waste, textile);
- chemically degradable (plastic);
- non-degradable (stones, building materials, glass).

Several fractions could be defined among the biodegradable waste depending on the level of biodegradability:

- easily biodegradable (food waste, grass, leaves);
- medium biodegradable (paper, garden waste, cellophane);
- slowly biodegradable (some types of paper, wood).

The percentage of different types of waste in the particular landfill will greatly affect the composition of leachate. Furthermore, presence of toxic metals and toxic organic materials in landfill waste will affect its biological treatability and define treatment methods for leachate.

The impact of landfill age on leachate composition becomes obvious if the succession of processes, which occur in leachate, will be observed. It is clear, that young landfill, where aerobic and acetogenic degradation is prevailing, will generate leachate with quite different composition, comparatively to the old, methanogenic landfill. In abandoned landfills usually only methanogenic processes take place, so, leachate from such landfills will also differ from operated ones. The difference of leachate composition versus its age is presented in the Table 2.1.

Table 2.1 Landfill leachate composition vs. landfill age [Renou, 2008]

Parameter	***Recent landfill***	***Intermediate landfill***	***Old landfill***
Age (years)	<5	5 – 10	>10
pH	6.5	6.5 – 7.5	>7.5
COD (mg/l)	>10000	4000 – 10000	<4000
BOD/COD	>0.3	0.1 – 0.3	<0.1
Organic compounds	80% volatile fatty acids (VFA)	5 – 30 % of VFA + humic and fulvic acids	Humic and fulvic acids
Heavy metals	Low-medium	Low-medium	Low
Biodegradability	High	Medium	Low

With increase of the age pH of leachate also tends to increase, while COD concentration decreases. It is explained by the prevailing of acidogenic “pH-decreasing” processes in the young leachate and methanogenic “pH-increasing” processes – in the old ones. As the decomposition of easily degradable organic compounds occurs, inert compounds (humic and fulvic acids) prevail in old leachate with subsequent decrease of BOD/COD ratio. Concentration of heavy metals decreases due to their leaching form landfill. The volume of generated leachate gradually decreases when the landfill is withdrawn from operation.

Depositional conditions together with technological and operational solutions at the landfill could affect generation and composition of leachate. Among these conditions are area of the landfill, density and humidity of deposition, temperature inside the landfill body [Schmoll, 2006]. Among the technological solutions at the landfill, which more or less impact leachate formation external isolation from infiltration of precipitations, solutions for leachate drainage, and methane collection systems could be mentioned.

The volume of leachate, produced at the landfill and concentrations of contaminants depend on the amount of water which is precipitated and evaporated at the landfill site. In humid climate with high precipitation rates all over the year more leachate with lower concentration of contaminants will be produced, than in dry climate with high evaporation rates. The designer of leachate treatment facility should also pay attention to the seasonal weather changes – in the areas with annual changes of dry and wet season leachate volumes and composition could vary greatly during the year.

Temperature is another important factor that affects leachate composition. In warmer climates degradation processes are faster, thus, landfill will be stabilized earlier, but with higher concentrations of contaminants at the leachate in the young landfill sites. Leachate, produced at landfills in the moderate climate usually has higher concentration of organic compounds in summer than in winter.

2.1.5 Characteristic of landfill leachate problems in Sweden

Currently there are more than 300 landfills in Sweden [Avfall Sverige, 2009]. However, only half of them are still in operation – other landfills were closed in recent ten years. According to recent governmental decisions, landfilling should be decreased in the future as much as possible with the shift towards more sustainable technologies of recycling and anaerobic digestion of waste. Figure 2.3 describes the recent decrease of amount of operated landfills since 1994 and future trends in this field. According to the ordinance SFS 2001:512, 52 from 160 landfills had to be closed until 1st January 2009 [http://www.notisum.se, 2009].

8 to 12 millions m^3 of leachate are produced at the Swedish landfills yearly. Thus, the rate of leachate production is between 1500 and 3500 m^3/ha·year, depending on the hydrogeological conditions and placement of the particular landfill site [Naturvårdsverket. Fakta: Lakvatten från deponier, 2008].

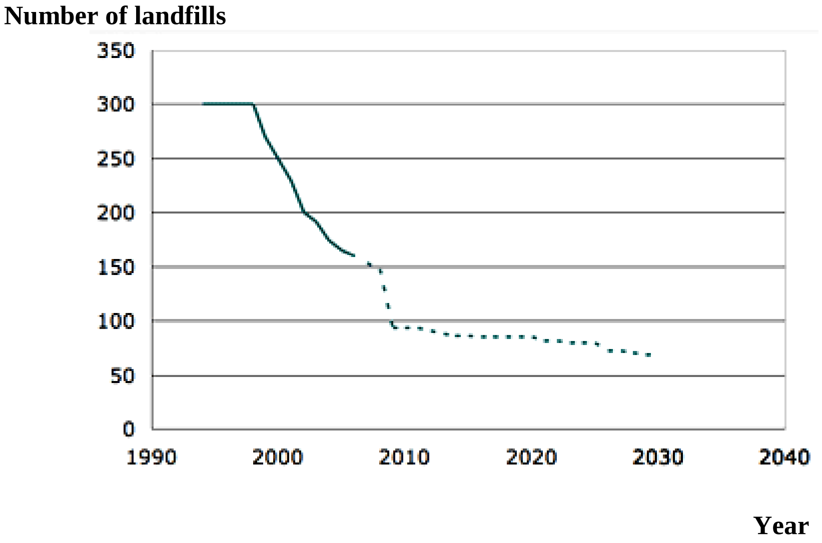


Figure 2.3 Decrease of amount of operated landfills during years 1994 – 2006 and prediction of further decrease during years 2007 – 2030 [Avfall Sverige, 2009]

The limiting concentrations for leachate discharge in Sweden are provided separately for each landfill according to the guidelines from the Swedish Environmental Protection Agency (Naturvårdsverket). Discharge limits depend on the type of receiving water basin, concentrations of pollutants in the discharged leachate and in the receiving basin etc. The permission for discharge and discharge limits for each landfill are given by the Swedish Environmental Court.

2.2 Biological nitrogen removal processes

Nitrogen is usually one of the main targeted compound in leachate treatment, as nitrogen content in leachate is very high and its removal down to the guidelines' levels requires application of sophisticated techniques, while sufficient BOD removal could be achieved by simple solutions. There are several chemical and physical processes for nitrogen removal from wastewater, but they aren't so widespread comparatively with biological nitrogen removal, which is better in terms of effectiveness and cost economy.

Combination of biological nitrification and denitrification is the most common and the most efficient solution for nitrogen removal today. It is applied in various types of reactors with attached and suspended biomass. COD and phosphorous removal is often successively combined with nitrification/denitrification process. The novel anammox and deamox processes are lab-scale approved but need further researches and development for application to the leachate treatment. Short description of these processes is provided below.

2.2.1 Biological nitrification/denitrification

Combination of these two natural biological processes, which are implemented by different consortiums of microorganisms at different conditions leads to the removal of nitrogen from wastewater as a nitrogen gas (N_2). As nitrification and denitrification requires different conditions they are divided from each other either in time or in space. Nitrification is usually the limiting process, as nitrification rates are significantly lower than denitrification ones; nitrification is also more dependent on the environmental conditions, than denitrification.

Nitrification is the process of biological ammonium oxidation with nitrate as the final product. It consists of two steps: oxidation of ammonia (NH_4^+) to nitrite (NO_2^-) and subsequent oxidation of nitrite to nitrate (NO_3^-):

$$NH_4^+ + 1.5O_2 \rightarrow NO_2^- + 2H^+ + H_2O \qquad \text{(eq. 2.3)}$$

$$NO_2^- + 0.5O_2 \rightarrow NO_3^- \qquad \text{(eq. 2.4)}$$

The total reaction of nitrification is described by the following chemical equation:

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O \qquad \text{(eq. 2.5)}$$

Nitrification is performed by autotrophic microorganisms, which means that they use carbon dioxide instead of organic compounds as a carbon source. Nitrifiers have low growth rates. As the process is aerobic, minimal required dissolved oxygen concentration in the reactor is 0.5 mg/l and optimal concentration – 2 – 3 mg/l [Gerardi, 2002]. As any biochemical process, nitrification is temperature dependent. Between 0ºC and 30ºC nitrification rate increases twice with 10ºC temperature increase. Nitrification rates are constant at the temperature 30 – 35ºC. Process is inhibited when the temperature rises up to more than 40ºC [Henze, 2000]. The most suitable pH for the process is in the range from 8 to 9. At pH lower than 7 nitrification rates are very low and complete inhibition occurs at pH 5.5 [Heander, 2007]. pH values higher than 10 are also inhibitive [Campos, 2007]. As 2 moles of H^+ are produced for 1 mole of NH_4^+ converted, the alkalinity should be high enough to prevent the drop of pH (at least 5 eqv/m^3) [Henze, 2000]. The high content of organic matter in the wastewater could also decrease nitrification rate, as the oxygen will be consumed by heterotrophic bacteria for oxidizing of organic matter instead of ammonium oxidizing by nitrifiers. The maximal nitrification rates reported for leachate treatment systems are in the range of 25 – 35 mg NH_4–N/l·h or 2.5 – 5.5 mg NH_4–N/gVSS·h [Heander, 2007; Kazuichi, 2007; Jeong-hoon, 2001].

Denitrification is the reduction of oxidized nitrogen compounds. The final product of complete denitrification process is nitrogen gas (N_2). Denitrification runs stepwise, from the most oxidized to the most reduced compound:

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2 \qquad \text{(eq. 2.6)}$$

The resulting reaction of denitrification:

$$2NO_3^- + H^+ + \text{organic matter} \rightarrow N_2 + HCO_3^- \qquad \text{(eq. 2.7)}$$

Denitrification is performed by heterotrophic bacteria, which use organic material as carbon source. It means that external organic carbon source should be added if its concentration is not sufficient in raw wastewater. Reduction occurs under strict anoxic conditions, so, dissolved oxygen concentration in the reactor should be as low as possible. The process is less dependent on the temperature conditions. The most favorable pH for denitrification lies between 7 and 9. In the recent study of leachate treatment using SBR technique maximal denitrification rate of 118 mg NO_3–N/l·h was reported [Heander, 2007].

2.2.2 Anammox process

The anaerobic ammonium oxidation (Anammox) process was discovered in the mid-90th. It is a microbiological conversion of ammonium to nitrogen gas with nitrite utilization as an oxidative agent. The overall chemical reaction is the following [www.anammox.com, 2009]:

$$NH_4^+ + 1.32NO_2^- + 0.067HCO_3^- + 0.13H^+ \rightarrow 1.02N_2 + 0.26NO_3^- + 2.03H_2O + 0.067CH_2O_{0.5}N_{0.15} \qquad \text{(eq. 2.8)}$$

The process is based on metabolism of the particular group of chemoautotrophic anammox bacteria. Anammox bacteria are difficult to grow in a lab, as the doubling time for them is about 11 days. The enrichment procedure is also difficult, as anammox bacteria always occur in the mixed cultures [www.anammox.com, 2009]. Anammox bacteria are very sensitive to oxygen concentration, which should be held less than 0.05 mg/l [Schmidt, 2003]. The excess of nitrite could be also inhibitive for the process, but for different anammox bacteria different concentrations in the range from 50 to 180 mg/l (> 100 mg/l is most common) are inhibitive [Schmidt, 2003]. The main physiological parameters of anammox bacteria and operational parameters for running of anammox process are listed in the Table 2.2.

The anammox process requires presence of nitrite in the media. As nitrite is not usually found in the wastewater at the high concentrations, partial conversion of ammonium into nitrite is required. Several alternatives have been developed for that purpose. The Single Reactor System for High Ammonium Removal over Nitrite (SHARON process) and Completely Autotrophic Nitrogen Removal over Nitrite (CANON) are most often described in the literature.

Table 2.2 Anammox bacteria characteristic and operational conditions [Lamsam, 2008]

Parameter	***Unit***	***Value***
pH		6.7 – 8.3 (optimum 8.0)
T	ºC	20 – 43 (optimum 35 – 40)
Inhibition by nitrite	mg/l	> 100
Inhibition by oxygen	% saturation	0.5
Growth rate	h^{-1}	0.003
Doubling time	d	10.6

The stoichiometry of the SHARON process is described by the following reaction:

$$NH_4^+ + 1.5O_2 \rightarrow NO_2^- + 2H^+ + 2H_2O \quad \text{(eq. 2.9)}$$

Partial nitrification is achieved due to the short retention time (1 d), comparatively low pH (6.5 – 7.5) and high temperature (35ºC). CANON process is the combination of SHARON and anammox processes in a single reactor [Third, 2001]. It has been laboratory approved, but no full-scale installation exists up to nowadays, though combination of SHARON and anammox processes performed in different reactors is already used for treatment of wastewater with high ammonium concentrations.

Anammox process is claimed to have a potential for application in treatment of wastewater with high nitrogen concentrations. It is cost beneficial comparatively with conventional nitrification/denitrification process, as no additional carbon source and twice less aeration is required. Still the process needs very long start up time (up to 6 months) and it requires additional energy input for heating up to the optimal temperature (35ºC). The latter isn't a problem for an industry which utilizes water for cooling. The second drawback of anammox process is the presence of the certain amount of nitrate in the effluent (according to the stoichiometry of reaction), which requires removal in case of strict nitrogen discharge requirements.

2.3 Leachate treatment techniques: general overview

Leachate, which is classified as toxic industrial waste, requires necessary treatment before it could be allowed to withdraw to natural water basins. In developed countries leachate is usually treated at municipal wastewater treatment plants (WWTP), though such treatment has is own disadvantages. In developing countries the situation is usually worth – landfills are organized primitively and lack leachate collection systems that results in the strong pollution of landfill sites and groundwater in the nearby district.

Specially arranged leachate treatment systems aren't widespread yet. Among the most popular solutions are aerated ponds, sequencing batch reactor (SBR), chemical precipitation, membrane bioreactors and reverse osmosis systems. Some solutions, though performing well in the lab-scale installations, turned to be unsuitable for industrial process. Other possible future solutions are on the stage of lab-scale experiments [www.leachate.co.uk, 2009].

The full-scale leachate treatment system usually includes several steps. Primary treatment serves for particle removal, such as sand and grit, by settling. During main treatment stage the largest part of the organic contaminants and nitrogen should be removed. Polishing step is designed for removal of the rest of organic material (usually inert COD), excess of phosphorous or some toxic substances, which levels are above the discharge limits.

The overview of the implementation of different leachate treatment techniques at Swedish municipal landfills is presented in the Table 2.3.

Table 2.3 Different leachate treatment methods, used at the municipal landfills in Sweden [Naturvårdsverket. Fakta: Lakvatten från deponier, 2008]

Technique for leachate treatment	***Number of landfills, which use this technique***
Aerated pond	86
Direct discharge to the municipal WWTP	84
Irrigation	37
Recirculation	29
Infiltration	28
Wetland	18
Sand filter	14
Chemical precipitation	9
Sequencing batch reactor (SBR)	4
Mechanical	4

Almost all the above mentioned techniques, such as aerated ponds, irrigation, recirculation etc. don't provide the appropriate level of treatment that allows treated leachate to be withdrawn directly to the open water basins, thus, treatment at the municipal wastewater treatment plant is necessary.

Not all the existing methods for wastewater treatment are applicable for leachate treatment. Some processes, though being widely used for treatment of municipal wastewater or different kinds of industrial wastewaters, are quite unsuitable for leachate treatment because of the peculiarities of this type of waste. Activated sludge units, which are so common at the municipal WWTPs, aren't used at the leachate treatment facilities because of their bad performance at high loads. By anaerobic digestion one could reduce BOD very efficient, but it is impossible to remove ammonium in the anaerobic bioreactor, so, this solution could not be used for leachate treatment as well [www.leachate.co.uk, 2009].

2.4 Conventional methods for leachate handling and partial treatment

2.4.1 Treatment at municipal wastewater treatment plant (WWTP)

The most often used method for leachate treatment is treatment together with domestic sewage at the municipal WWTP. It doesn't require high capital costs, as leachate is either directly pumped through the pipes to the municipal sewage system, or transported in the special trucks to the WWTP.

However, if the percentage of leachate, withdrawn to the municipal WWTP, is higher than 0.5 % by volume, it may require changes in operational mode or even reconstruction of the treatment facility [Anhert, 1992]. Leachate with high nitrogen concentrations may be a problem for operation of WWTP, where domestic sewage is used as an organic source for denitrification [Anhert, 1992]. As a consequence, municipal WWTP, which receives leachate from the landfill site, usually charges landfill operator both for the volume of incoming leachate and additional load of contaminants.

To reduce the impact of leachate on municipal WWTP before discharging, the landfill operator could dilute leachate with surface water [www.leachate.co.uk, 2009].

2.4.2 Irrigation

Irrigation of nearby wetland or abandoned parts of the landfill with leachate could be applied in some cases, particularly for the leachate from old landfills in stable methanogenic phase with low toxicity [RVF:s utvecklingssatsnings deponering, 2003:4]. The leachate is spread over the land in order to achieve the loss of volume due to the evaporation. Some types of plants could be grown on the site, e.g. energy crops, as the high nitrogen content in leachate will facilitate their growth. Organic matter and nutrients are degraded by soil microorganisms to some extent. The technical and energy requirements are very simple for such kind of system. On the other hand irrigation couldn't be considered the appropriate treatment – the soil at such sites becomes contaminated and the risk of groundwater pollution exists.

2.4.3 Recirculation

Leachate recycling through the garbage is one of the least expensive leachate handling techniques available, which is still widely used all over the world. Leachate recycling facilitates the stabilization of landfill: with enhanced moisture content of the landfill the production of methane increases and the anaerobic decomposition starts faster. It is also beneficial for the gas collection, as the methane content in the landfill gas increases [www.leachate.co.uk, 2009].

Leachate recycling is also beneficial for the further treatment processes, as it allows leachate composition to be stabilized and minder concentrations of COD and nitrogen in the young leachate. On the other hand, it helps to flush some slowly degradable compounds from waste. In some cases recirculation may be a temporally solution for the period, when the system for appropriate leachate treatment is being built and other solutions couldn't be applicable.

The main drawback of leachate recirculation is the increase of possibility of groundwater contamination which occurs together with increasing of moisture content in the landfill waste. Due to this reason leachate recirculation is often banned by environmental authorities, especially in cases, when landfill is not equipped with appropriate liner.

2.4.4 Infiltration

Infiltration is based on the old strategy "dilute, attenuate and disperse" which was developed and widely recognized in the 70th but than failed due to the lack of process management and inappropriate geological selection of the landfill sites that led to the numerous cases of groundwater contamination .

Today's concept of infiltration include careful selection of the landfill site, the development of infiltration layers and migration barriers for prevention of groundwater contamination and active management of the process, which includes monitoring of the leachate percolation through the infiltration layers and collection and recycling of the excess amounts of leachate to prevent overload of the infiltration system. The pollutants reduction during infiltration processes is reached due to the ability of some types of soil to absorb and moderate contaminants from leachate before it reaches groundwater. The main constraint, however, is the probability of groundwater contamination [Hester, 1995].

2.5 Biological methods of leachate treatment

Among the modern leachate treatment methods, which are used for complete on-site treatment of leachate without further transferring to the municipal WWTP, biological methods, particularly sequencing batch reactor (SBR) technique are the most widespread ones. Most of the biological processes were developed for the treatment of municipal wastewater, so, in many cases they have to be modified for handling of more contaminated leachate or additional treatment steps have to be used afterwards. The biological treatment techniques in the following description are grouped according to the biomass placement in the system – suspended in the whole volume of a tank or attached to the different types of carriers.

Suspended biomass systems

Suspended biomass systems are based on the cultivation of microorganisms, which are freely distributed in the whole volume of bioreactor. The utilization of suspended biomass systems gives the advantage of better interaction between solid (microorganisms) and liquid (wastewater) phases which facilitate transport processes. Another advantage is that it's quite easy to control sludge age in such systems. The drawback of suspended biomass system is that bigger volumes have to be utilized, thus, requiring more space for the treatment facility.

2.5.1 Sequencing-batch activated sludge systems

SBR is quite common solution for on-site leachate treatment. If applied together with further polishing steps, it allows significant decrease of contaminants' concentrations in the leachate. SBR process is also often applied before discharging leachate into the municipal sewer. In this case polishing steps are neglected and SBR cycle consists only from the aeration (nitrification) stage, that serves just for ammonium oxidation (nitrate is not a toxic compound comparatively to ammonium, so its discharge to the municipal WWTP will be less problematic) [www.leachate.co.uk, 2009].

Sequencing batch process is performed as a repeated operational cycle. It starts with addition of wastewater to the reactor (filling phase), usually accompanied by mixing and aeration. By introducing aeration together with mixing or just mixing one can achieve either aerobic or anaerobic (anoxic) conditions in the reactor, which favors either nitrification or denitrification processes. Easily degradable organic compounds are also removed in a sequencing batch reactor. During settling phase mixing and aeration is stopped, which allows separation of liquid phase (treated water) at the top from solid phase (sludge) at the bottom. At the draw phase treated liquid is decanted either to the further treatment step or to the final recipient. Afterwards the new cycle starts with a new filling phase. The excess of sludge could be removed during aeration phase. The simplified design principle of SBR reactor is presented on the Figure 2.4.

The widespread utilization of SBR for the purpose of leachate treatment is possible due to its several advantages [Tomaszek, 2005]:

1) Operational flexibility: the duration of different phases of the process could be simply changed if the influent concentrations are changed;
2) Easy to maintain the reactor at high contaminants concentrations: either the cycle duration could be extended or the filling phase prolonged;
3) Low probability of filamentous bacteria growth and bulking;
4) Do not require recycling of sludge: saving of energy costs for pumping;

5) Do not require secondary clarifiers: less space requirements for the plant, saving of capital investment and maintenance costs.

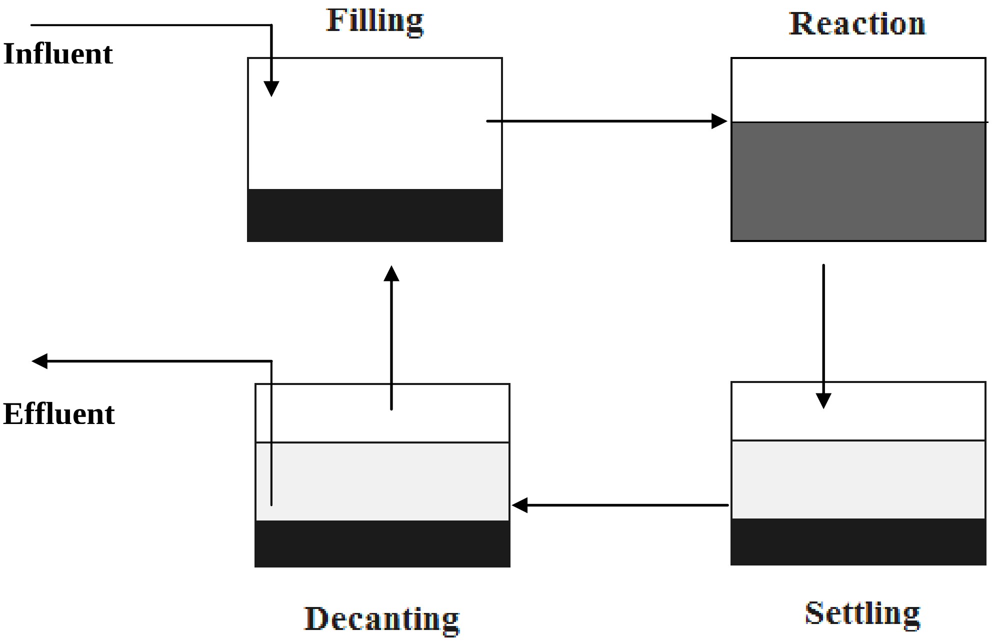


Figure 2.4 The design principle of SBR reactor

The main disadvantages of SBR include:

1) Higher level of complicity which requires more daily maintenance than plug-flow system and stricter control (more timers, valves etc.);
2) Technical problems (e.g., pump break) could lead to the complete failure of the process;
3) The process is less suitable for treatment of high amount of wastewater, as there are extended periods during each cycle (settling and decanting phases) when the treatment capacity of the unit is not utilized.

As the leachate is usually concentrated liquid with significantly lower daily flow comparatively to municipal wastewater, SBR process turns to be more preferable for treatment. Space requirements and costs are also important issues for choosing SBR as a treatment technique, especially because leachate treatment is not the main activity for landfill operators.

2.5.2 Lagooning

Lagooning is currently the most widespread method for leachate pretreatment at landfill sites before discharging to the municipal WWTP. Lagoons or aerated ponds require low capital investments and maintenance costs. Usually they are artificial ponds with impermeable bottom and walls where leachate is collected and allowed to stay during certain period, which is required for organic and nutrient removal.

The removal of contaminants in the lagoons is achieved due to the sedimentation processes and biological decomposition by the microorganisms, which are presented in the lagoons. To facilitate nitrogen removal, several aerobic and anaerobic zones may be combined in the lagoon or several ponds could be combined into the single system, modeling nitrification and denitrification conditions. Aerobic zones in treatment ponds are aerated by installation of surface or bottom aerators.

Cheap installation and maintenance costs favor utilization of treatment ponds for leachate handling, but this solution has a number of important drawbacks which don't allow reaching discharge limits with this type of treatment [Bodzek, 2005]. Treatment ponds are affected greatly by the external conditions. Temperature drop during winter affects the removal rates significantly, as the biological processes almost stop at the temperatures close to 0ºC. During summer period intensive algal growth may lead to the creation of anaerobic conditions in the ponds. Space requirements for the treatment ponds are high; they could be also the source of unpleasant odors and facilitate the spread of infectious diseases [www.leachate.co.uk, 2009].

Attached biomass systems

In attached biomass systems microorganisms are immobilized on static support material. Liquid and solid phases in the reactor are separated in such way. Other important advantages of the attached biomass systems are higher resistance to toxicity and higher metabolic activity of attached microorganisms [Björnsson, 2007]. Possible problems with nutrients transport between liquid phase and microorganisms and difficulties with controlling the sludge age in the reactor are the main disadvantages of attached biomass systems. Among the variety of technical solutions trickling filters and moving-bed biofilm reactors are used for leachate treatment.

2.5.3 Trickling filter

Trickling filter consists of the "bed" layer of material, which serves as a support for the growth of microorganisms. Stones, gravel and plastic are the materials, which are most often used in trickling filters. Wastewater is distributing with special sprinkling system above the filter and percolates through it. While percolating through the filter, it is being treated with microorganisms, attached to the filter surface.

The main drawback of trickling filter system is that the filter becomes clogged by particles in a while and has to be cleaned. It increases maintenance costs and the whole treatment process has to be stopped for the cleaning procedure. As leachate is a heavily contaminated wastewater, clogging of filters happens quite often. Though trickling filters are quite unsuitable for leachate treatment, they are still used at some plants.

2.5.4 Moving-bed biofilm reactor (MBBR)

MBBR is a novel technique for distribution of attached biomass in the reactor. The biomass is placed on small carriers, which are usually made from plastic material and have large surface to allow more biomass to be adjusted to it. With this technique one of the main drawbacks of trickling filters is overcome: as the carriers are suspended in the whole volume of the reactor, the clogging doesn't occur. The suspended carriers are easily prevented from washout from the reactor by sieves, thus, settling is not required, comparatively to activated sludge systems.

The significant nitrogen reduction (up to 85 – 90% of the nitrogen concentration in the influent) while applying MBBR systems for leachate treatment has been reported in the literature. However, COD reduction in MBBR process doesn't exceed 20% and additional treatment step (e.g., adsorption on activated carbon) is required.

2.6 Chemical and physical methods of leachate treatment

Chemical and physical methods are most commonly used in leachate treatment as a combination with biological methods for further polishing of the treated leachate and removal of specific contaminants, though ammonia stripping, precipitation and reverse osmosis could be used as a main removal step.

2.6.1 Precipitation

Precipitation is defined as the formation of solids in the solution as the result of chemical reaction. This process is often used in the wastewater treatment to change the physical state of dissolved contaminants with their subsequent removal. Precipitation is most commonly used for ammonium nitrogen reduction: up to 97 percent of ammonium can be removed. The reported COD removal is considerably lower – 40 to 50 percent [Renou, 2008].

Ammonium is removed in the mineral form of magnesium ammonium phosphate ($MgNH_4PO_4 \cdot 6H_2O$), which is better known as struvite. The magnesium compound ($Mg(OH)_2$, MgO, $MgCl_2$ etc) and phosphoric acid (H_3PO_4) have to be dosed for this reaction to be occurred, as Mg- and P-containing substances usually occur in very low quantity, comparatively to the ammonium compounds, which have to be removed [Kabdasli, 2000]. The process is described by the following reaction [Celen, 2001]:

$$Mg^{2+} + NH_4^{+} + HPO_4^{2-} + 6H_2O \rightarrow MgNH_4PO_4 \cdot 6H_2O + H^{+} \qquad \text{(eq. 2.10)}$$

The removal rates depend on many factors. Temperature doesn't significantly affect the removal rates. However, pH is important for successful implementation of the process. Alkaline conditions are necessary, with preferable pH 8.5 – 9.0 [Celen, 2001] or 10.5 [Shin, 1997]. The process should last for at least 10 minutes to achieve successful precipitation. The ratio of Mg:N:P has to be around 1.2:1:1.2, thus, magnesium and phosphorus have to be in slight excess. The type of magnesium source is also important, e.g. it's possible to reach higher removal rates with magnesium chloride comparatively than with magnesium oxide, but the latter requires less lime addition for pH maintenance.

Struvite could be applied as the slow-released additive to fertilizer because it doesn't contain any toxic substances. However, struvite precipitation is quite expensive method due to the high cost of phosphorous and magnesium salts added. Another problem is clogging of pipes and connections with precipitated struvite, which has to be removed by pressurized washing, and reduction of service life period of equipment.

Precipitation is the most commonly used technique for phosphorous removal from different types of wastewater. Aluminium, iron salts or lime could be used; preferably Al^{3+} salts which are the most effective ones for phosphorous precipitation. Phosphorous removal is not usually focused on while handling leachate, as its concentration is generally neglectable comparatively with organic and nitrogen concentrations. Still, if the leachate should be withdrawn to the environment, particularly into surface water, the discharge limits for phosphorous are strict (0.3 – 0.5 mg/l in Sweden) and phosphorous precipitation could be used.

2.6.2 Adsorption

Adsorption on the activated carbon is one of the most common techniques for polishing leachate effluent after biological treatment step [www.leachate.co.uk, 2009]. Sometimes it is used as the main treatment step without application of biological process. Activated carbon adsorbs inert COD, which cannot be removed during biological step.

Activated carbon is available in powdered and granular forms. Powdered activated carbon has to be precipitated with aluminium or iron salts and settled afterwards. Granulated activated carbon is placed in the immobilized columns and treated leachate passed by them. This method has several advantages – it doesn't require precipitation and sedimentation steps; the activated carbon in granulated form could be further reused after thermal regeneration [Stegmann, 2005]. Activated carbon couldn't be used for residual phosphorous removal from the leachate, as phosphorous is poorly adsorbed on it [Mortula, 2007].

2.6.3 Advanced oxidation

Advanced oxidation with strong oxidative agents, such as ozone or hydrogen peroxide, facilitated by UV-light is often used for the same purposes, as adsorption with activated carbon – removal of inert COD compounds. Oxidative agent is mixed with treated water in a treatment chamber. Aqueous hydrogen peroxide usually is easier to mix, than gaseous ozone, thus, ozone is often difficult to utilize effectively. Quite often it is necessary to recirculate leachate through the treatment unit several times to achieve better removal rates. As the costs for advanced oxidation are high, it is not used as a main treatment step – easily degradable organic compounds should be preliminary removed in a less expensive biological process [Stegmann, 2005].

2.6.4 Ammonia air-stripping

Ammonia stripping is driven by intensive aeration of treated leachate at high pH (10.5 – 11.5). The process is quite efficient in elimination of high ammonium concentrations. Process is running in the stripping tower, filled with aerated media, which is overflowed by leachate. Treated leachate is collected at the bottom of the tower and gases raise up to the top. The air, polluted with ammonium, has to be treated with sulphuric or chloric acids. Recirculation of treated leachate is often required to achieve discharge limits.

Ammonia stripping could be cost-efficient only at very high ammonium concentrations in the wastewater. Costs are spent on lime addition for increasing pH before the treatment and acid addition afterwards. Calcification of the tower is also widespread problem [www.leachate.co.uk, 2009].

2.6.5 Membrane techniques

Membrane treatment is a typical physical process, where removal of contaminants is based on their relative sizes. Depending on the size of membrane pores, membrane processes are defined as microfilration, ultrafiltration, nanofiltration and reverse osmosis. The last two processes usually concentrate about 25% of initial flow, which has to be either further concentrated and treated as solid waste, or returned to the contaminated leachate. To prevent clogging, membranes are treated by chemicals, such as combination of acid, caustic soda, hypochlorite solutions [www.leachate.co.uk, 2009].

The advantages of utilization of membrane techniques in leachate treatment are:

1) Small footprint;
2) Membrane processes don't require any start-up time, as biological processes do;
3) High automation of the process.

The drawbacks of membrane processes:

1) Clogging of membranes occurs often;
2) Chemicals are required for membrane cleaning;
3) Disintegration and leakage of the membrane may cause pollution of the receiving waters.

3 Current leachate collection and treatment system at NSR landfill facility

3.1 Description of NSR landfill facility

Nordvästra Skånes Renhållnings, AB (Northwest Skåne Cleaning, Ltd; NSR AB) is a municipality owned company which operates on the waste landfilling and recycling market in the north-west region of the province of Skåne (southern Sweden). The company was founded in 1982 and is jointly owned by six municipalities - Bjuv, Båstad, Helsingborg, Höganäs, Åstorp and Ängelholm, which population is ca 225 000 inhabitants.

The main recycling facility – Filborna landfill – is situated at the north-eastern outskirt of Helsingborg, between industrial district Väla and motorway E6. The landfill was put in operation in autumn 1951. It was operating as a traditional landfill until 1990th, when the waste separation approach was applied by the company.

Nowadays food and food industry waste is treated at the biogas plant which was built in 1996 and extended in 2007. It can process 80000 t of food waste per year. The produced raw gas contains 75% methane and 25% carbon dioxide. The gas could be either supplied to the municipal heating system or upgraded to the quality of vehicle fuel at the upgrading station. The upgrading station was built in 2002 and extended in 2007. Its capacity now is 1000 Nm^3 of fuel biogas per hour with 98% methane and 2% carbon dioxide. The upgraded biogas is used as a fuel by city buses, garbage trucks and private cars. The non-degradable material from biogas reactor is supplied as a fertilizer to nearby farmers.

The landfill gas, produced at Filborna landfill, is supplied to the Öresundkraft AB for heating of apartments in Helsingborg and to commercial greenhouses for heating.

NSR recycles following types of waste: waste paper, newspapers, carton packaging, corrugated cardboard, metal packaging, plastic packaging, colored and clear glass, vehicle tyres. Most of them are sent to partner recycling companies, although plastic packaging is recycled at the plastic recycling facility at Ängelholm, which is owned by NSR. Garden waste is composted at Filborna site – ca 11000 t of compost is sold as a soil product to different customers yearly. Wood waste is processed into wood chips which are further supplied to Ängelholms Energy AB as a combustion fuel.

Currently only non-degradable and non-recyclable waste, such as ash, waste from construction industry, production waste etc., is landfilling. This waste is landfilled in cells, which are separate one from another and has their own leachate and landfill gas collection systems. The old landfill was closed at the 1st of April, 2009. The new landfill is being built according to the current EU regulations to the north-east from the existing site.

3.2 Leachate treatment system

Leachate is collected through the system of perforated pipes from beneath of the landfill cells, which were used previously, before landfilling of organic waste has been banned. It is partly run by the gravity flow and partly pumped to the treatment site, which is situated in the south-eastern low-lying area of the Filborna landfill. The annual leachate production is 320 000 – 360 000 m^3.

Leachate treatment system consists of four ponds (buffer pond, aerated pond, sedimentation pond and stabilization pond) and the Reco-N precipitation plant. The latter was installed in 2006, when the clogging of leachate collection system caused the rising of water level in the landfill cells and the subsequent increase of contaminants' concentrations in the leachate. The quality of the effluent water from the treatment ponds doesn't meet the discharge requirements, so it has to be withdrawn to the municipal WWTP.

The leachate is regularly analyzed at the NSR laboratory. Samples for analyses are taken at four different points: before buffer pond (raw leachate), after buffer pond, after aerated pond and after stabilization pond.

3.2.1 Treatment ponds

The system of leachate treatment ponds is situated in the eastern part of the facility, between old and new landfill sites. The aerial photo of the treatment ponds is presented on the Figure 3.1.

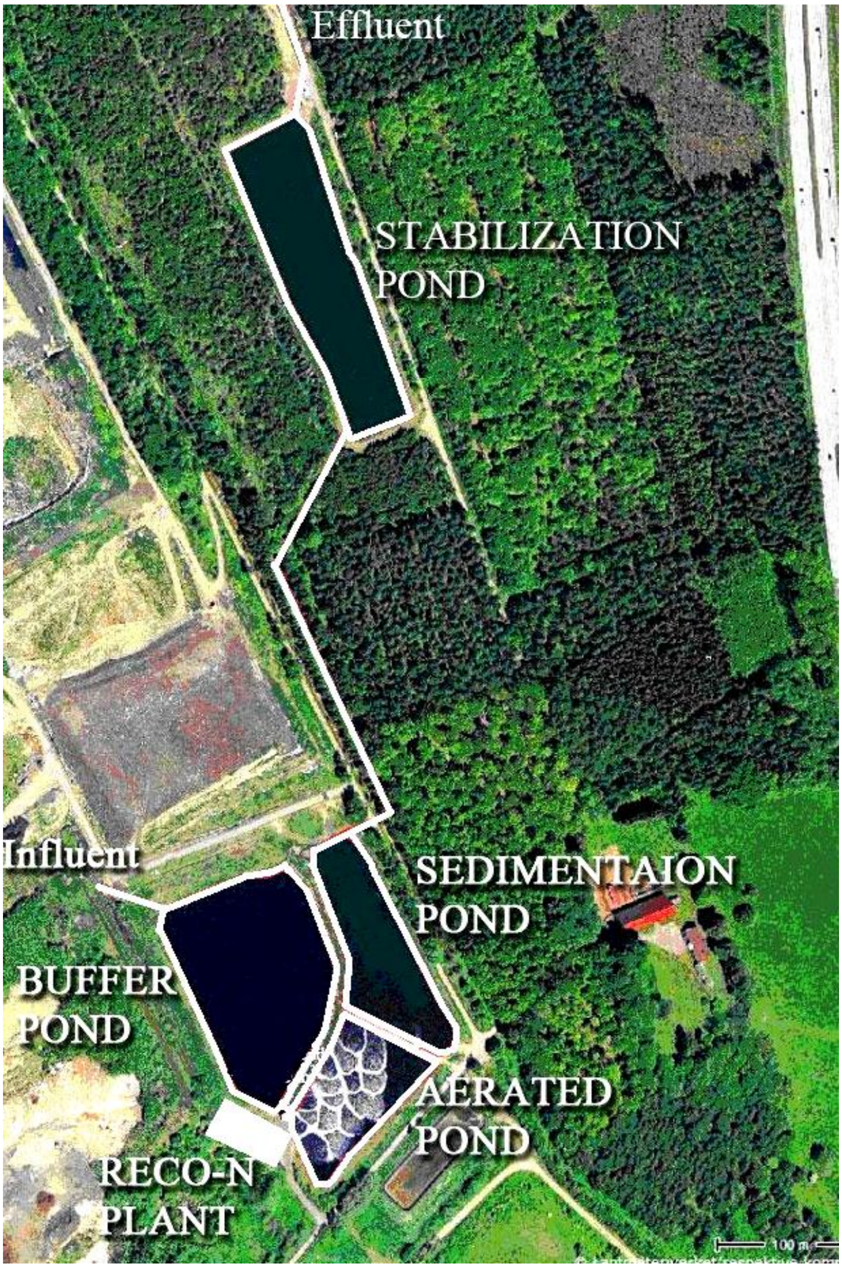


Figure 3.1 Layout of the treatment ponds (image from www.kartor.eniro.se)

The volumes of each pond and hydraulic retention time (HRT) are displayed in the Table 3.1. HRT was calculated from annual leachate production and the volume of each pond according to the formula:

$$HRT = \frac{V_{pond}}{V_{leachate, per\_year} / 365} \qquad \text{(eq. 3.1)}$$

The annual precipitation in southern Sweden is around 600 – 700 mm, while the evaporation exceeds 500 mm. Thus, evaporation and precipitation equalize each other and could be neglected in the formula above.

Table 3.1 Hydraulic retention time in the present leachate treatment system

Pond	***Volume of the pond, m^3***	***HRT, d***
Buffer pond	25 000	27
Aerated pond	9 000	10
Sedimentation pond	15 000	16
Stabilization pond	25 000	27
TOTAL	74 000	80

Buffer pond serves both for the equalization of the influent and sedimentation of suspended solids (Figure 3.2). BOD removal in the buffer pond is high, partly due to the particle sedimentation and partly because of heterotrophic conversion. Suspended solids concentration also decreases more than twice due to the sedimentation process. Phosphorous removal efficiency of the pond is moderate (33% of total phosphorous removed) and nitrogen removal efficiency – low (Table 3.2).

Figure 3.2 Current leachate treatment system at NSR AB. Buffer pond with surface aerators, January 15, 2009

Table 3.2 Removal efficiency of the buffer pond

Parameter	***Concentration in raw leachate, mg/l****	***Concentration after buffer pond mg/l****	***Removal of the contaminant, mg/l***	***% removal***
BOD_7	776	185	591	77
SS	403	175	228	57
NH_4-N	171	156	15	9
N-tot	296	255	41	14
P-tot	7.7	5.2	2.5	33

* The mean values of several measurements from 30 March 2006 to 28 August 2008. The HRT of the buffer pond (27 d) was taken into consideration for calculations

Hydrogen sulfide which is derived together with anaerobic leachate was a problem recently, as it caused bad smell around the treatment facility. To overcome the problem 12 surface aerators have been installed in the buffer pond during summer 2008. They supply air and mix water, improving organic and nitrogen removal.

Leachate from the buffer pond is pumped to the aerated pond (Figure 3.3). It is smaller than buffer pond and has 16 bottom aerators which provide both aeration and mixing. Removal efficiency of the aerated pond is low (Table 3.3).

Figure 3.3 Current leachate treatment system at NSR AB. Aerated pond with surface aerators. Two surface aerators are shown on the separate photo. January 15, 2009

Table 3.3 Removal efficiency of the aerated pond

Parameter	***Concentration after buffer pond mg/l****	***Concentration after aerated pond mg/l****	***Removal of the contaminant mg/l***	***Removal of the contaminant, %***
NH_4-N	155	150	5	3.2
N-tot	315	265	50	16
P-tot	8.9	7.5	1.4	16

* The mean values of several measurements from 18 October 2007 to 8 November 2007. The HRT of the buffer pond (10 d) was taken into consideration for calculations

After the buffer pond leachate approaches the sedimentation pond, where the rest of suspended solids are allowed to sediment. The stabilization pond, which is the last one in the sequence, is used for equalization of peak flows and storage of the treated leachate before discharging to the recipient. The overall removal efficiency of aerated, sedimentation and stabilization ponds is presented in the Table 3.4.

Table 3.4 Removal efficiency of the aerated, sedimentation and stabilization ponds

Parameter	***Concentration after buffer pond mg/l****	***Concentration after stabilization pond mg/l****	***Removal of the contaminant, mg/l***	***Removal of the contaminant, %***
BOD_7	245	130	115	47
SS	205	101	104	51
NH_4-N	150	92	58	38.5
N-tot	224	169	55	24.5
P-tot	5.5	4.4	1.1	20

* The mean values of several measurements from 22 February 2006 to 28 August 2008. The total HRT of three ponds (53 d) was taken into consideration for calculations

Comparing removal efficiency of the aerated pond with total removal efficiency of the first three ponds, it could be mentioned that removal of total nitrogen occurs mainly in the aerated pond, while ammonium removal occurs in the sedimentation and stabilization ponds, though the latter are not aerated. The reason could be the lack of carbon source in these ponds, which is consumed in the first two ponds under aerobic conditions. With the absence of carbon source heterotrophic bacteria overcompete nitrifiers, which use oxygen that diffuses through the pond surface for the ammonium oxidation.

Overall removal efficiency of the system of four treatment ponds is presented in the Table 3.5. Current system is quiet efficient in BOD and suspended solids removal, while nutrient are removed only up to 60%.

Partly treated leachate is withdrawn to the Helsingborg WWTP through the pipe that is separated from the municipal wastewater system, but is shared with several other industrial enterprises, which also withdraw their wastewater to the municipal WWTP.

Table 3.5 Removal efficiency of the whole treatment ponds system

Parameter	***Concentration in the influent mg/l****	***Concentration in the effluent mg/l****	***Removal of the contaminant, mg/l***	***Removal of the contaminant, %***
BOD_7	884	102	782	89
SS	388	94	294	76
NH_4-N	192	76	116	60
N-tot	355	159	196	55
P-tot	9.9	4.5	5.4	55

* The mean values of several measurements from 22 February 2006 to 28 August 2008. The total HRT of four ponds (80 d) was taken into consideration for calculations

3.2.2 Reco-N precipitation plant

The principle of operation of Reco-N plant is based on chemical precipitation of ammonium in the form of magnesium ammonium phosphate (struvite), which is described in section 2.6.1. The design of the plant is discussed in details below. General view of the plant is presented on the Figure 3.4, draft of the plant – on the Figure 3.5.

Figure 3.4 General view of the RECO-N precipitation plant. Precipitation tank, regeneration unit and ammonia scrubber are situated in the bottom compartment, hydrocyklon – at the top to the right, equipment for chemical addition – at the top to the left. January 15, 2009

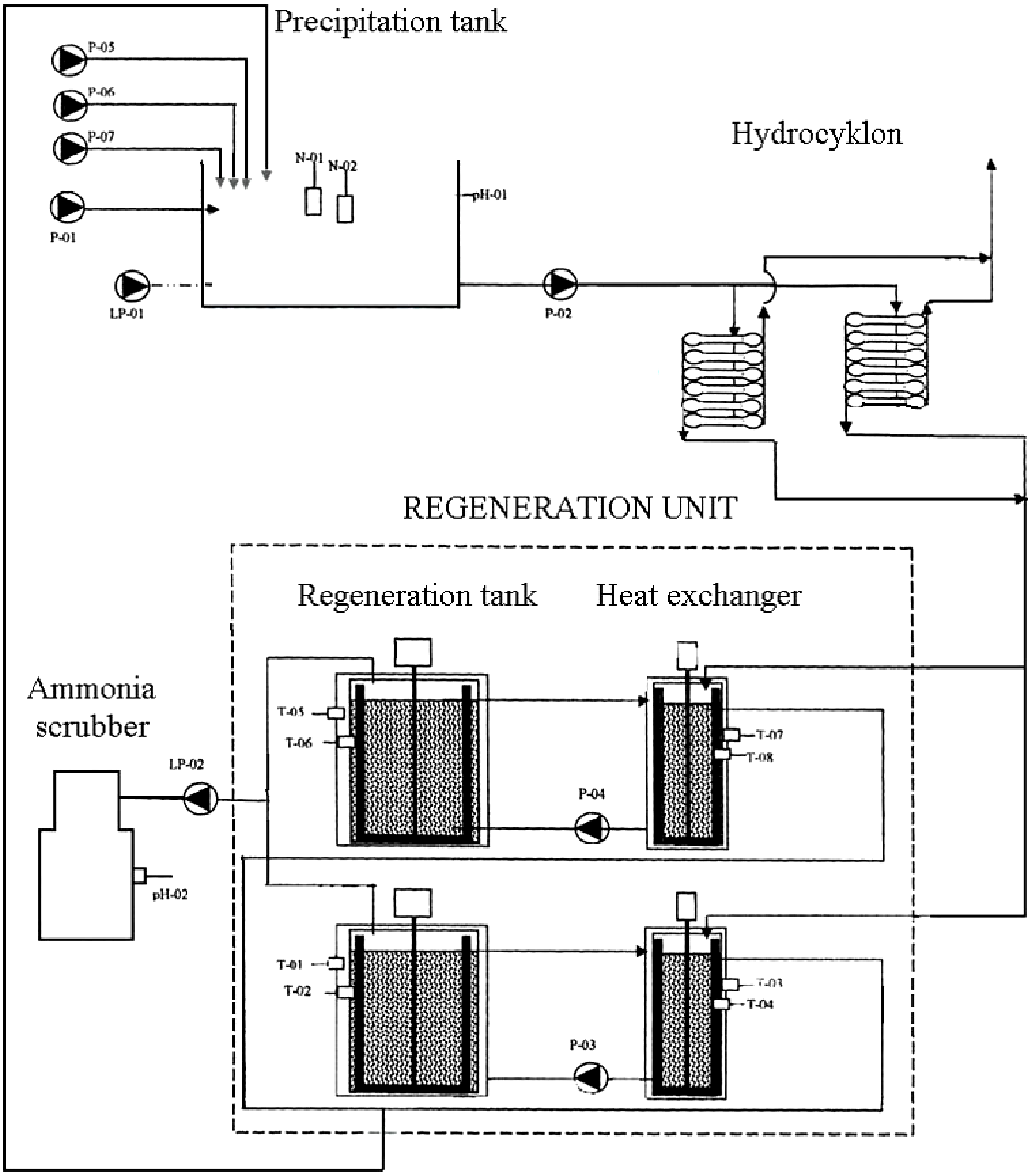


Figure 3.5 Simplified draft of the Reco-N precipitation plant [NSR AB, 2009]:

P-01 – Pump for leachate delivery;
P-02 – Pump for the transportation of MAP solution to the hydrocyclon;
P-03, P-04 – Pumps for sludge transportation between heat exchanger and regeneration tank in the regeneration unit.;
P-05 – Pump for MgO addition;
P-06 – Pump for NaOH addition;
P-07 – Pump for H_3PO_4 *addition;*
LP-01 – Aeration pump in the precipitation tank;
LP-02 – Pump for ammonia-rich air delivery from the regeneration tank to the ammonia scrubber;
N-01, N-02 – Level transmitters;
T-01 – T-08 – Temperature sensors in the precipitation unit;
pH-01 – pH sensor in the precipitation tank;
pH-02 – pH sensor in the ammonia scrubber

The leachate is pumped to the Reco-N plant from the buffer pond. It comes to the precipitation tank, where reaction occurs (see section 2.6.1). Magnesium oxide (MgO) and phosphoric acid (H_3PO_4) are used as magnesium and phosphorous sources for the reaction. The molar ratio between Mg, NH_4 and PO_4 in the reaction is 1.2:1:1.2. NaOH is added to keep the pH in the favorable range (8.5 – 9). Air for stirring is provided by an air pump. The upper and lower level of the liquid in the tank is controlled by level transmitters.

After the reaction liquid with precipitate is pumped from the precipitation tank to two hydrocyclon units. They separate particles from liquid according to rotational effect and gravity. Contaminated liquid enters hydrocyclon from its side close to the top and follows downstream by high speed rotation. The cleaned liquid is lifted upwards by gravitational flow and leaves hydrocyclon from the top, while the heavier particles fall down at the bottom. The removal efficiency is determined by the size and shapes of the unit.

After separation in the hydrocyclon water is sent to the aerated pond, while the sediments have to be treated further. The latter process is going in the regeneration unit, where nitrogen is recovered from struvite in the form of ammonia gas and the crystals of magnesium phosphate are formed. Magnesium phosphate could be present in forms of newberyite ($MgHPO_4 \cdot 3H_2O$), bobierrite ($Mg_3(PO_4)_2 \cdot 8H_2O$) and cattiite ($Mg_3(PO_4)_2 \cdot 22H_2O$), depending on the pH of solution [Bhuyian, 2008]. The raise of temperature up to 80°C is necessary for successful thermal decomposition

The regeneration unit is constructed in order to minimize heat consumption and energy requirements. The struvite-rich liquid comes to the inner coil of the heat exchanger, where it is warmed up to around 55°C by the liquid, which flows from regeneration tank. After the heat exchanger liquid is delivered to the regeneration unit, where it is further heated up to 80°C. The gaseous ammonia is collected above the liquid and pumped to the ammonia scrubber, while the heated magnesium phosphate solution is returned to the outer tank of the heat exchanger and further to the precipitation tank. In the scrubber ammonia reacts with scrubbing solution of sulphuric acid with formation of ammonium sulfate. The latter is sold as a fertilizer.

Sufficient costs are saved due to the resupply of magnesium phosphate solution from the regeneration tank through the heat exchanger to the precipitation tank. It helps to save costs both by pre-heating of MAP solution in the regeneration unit and by multiple utilizations of magnesium and phosphorous source in the precipitation tank. The latter decrease the addition of magnesium oxide and phosphoric acid to the precipitation tank by 90 to 95 percent.

3.2.3 Cost evaluation of the pond treatment system and Reco-N precipitation plant

The discharge of leachate to the Helsingborg WWTP is charged by treatment plant contractor. Currently NSR pays 3.8 SEK/m^3 of leachate withdrawn to the municipal sewage and 46 SEK/kg of nitrogen exceeding 52 g/m^3. With the mean annual volume 350 000 m^3 annual costs for leachate withdrawal are 1.33 millions SEK and for excess nitrogen – 1.1 millions SEK. The precipitation plant lowers nitrogen concentration in the effluent down to 30 mg/l, thus, it minders costs for effluent treatment of the leachate at the municipal WWTP for 1.1 millions SEK annually, but such treatment is still required [NSR AB, 2009].

To compare savings for running of the leachate treatment facility with and without Reco-N precipitation plant one could compare reduction of payments to the WWTP for extra nitrogen in the effluent together with incomes for selling of ammonium nitrate as fertilizer with costs for running of Reco-N plant. The latter include costs for chemicals, energy input and capital investments. The calculation, performed by NSR AB, showed that Reco-N plant could be profitable only with nitrogen concentrations in the effluent higher than 175 mg NH_4-N/l [NSR AB, 2009].

3.3 Current effluent concentrations and discharge requirements

Effluent from leachate treatment facility is currently discharged to the municipal wastewater treatment plant in Helsingborg mainly because it doesn't fit nitrogen discharge requirements. Discharge requirements are set in Sweden individually, dependent on the effluent and recipient water quality. In case of direct withdraw of wastewater to Öresund, discharge requirements for NSR leachate treatment system would be similar to the requirements for wastewater discharged from Spillepengs landfill. The predicted values for the latter are given by Heander [Heander, 2007]. They are compared with current concentrations in the effluent to the Helsingborg WWTP in the Table 3.6.

Table 3.6 Comparison between concentrations of contaminants in the raw leachate, leachate discharged to Helsingborg WWTP and predicted discharge requirements for Öresund.

Parameter	***Unit***	***Raw leachate***	***Leachate, discharged to the Helsingborg WWTP***	***Discharge requirements***
BOD_7	mg/l	545	**26**	**10**
TOC	mg/l	975	**134.4**	**100**
SS	mg/l	483	84	-
NH_4-N	mg/l	176	66	-
NO_2-N, NO_3-N	mg/l	0.64	3.2	-
N-tot	mg/l	323	**154**	**15**
P-tot	mg/l	7.3	**3.6**	**0.5**
As	mg/l	0.019	0.0087	0.01
Pb	mg/l	0.035	0.0036	0.05
Fe	mg/l	12	1.7	-
Cd	mg/l	0.0006	< 0.001	0.001
Cl	mg/l	701	455	-
Cu	mg/l	0.046	0.012	0.5
Cr	mg/l	0.072	0.028	0.05
Hg	mg/l	0.000016	0	0.001
Ni	mg/l	0.061	0.049	0.5
Sn	mg/l	0.01	0.0024	-
SO_4	mg/l	344	220	-
Zn	mg/l	0.67	0.13	0.5

The current system is efficient in reduction of metal ions' concentrations. However, nutrients and organic concentrations in the effluent are above the discharge requirements. These compounds have to be treated further in order to meet the effluent standards.

4 Laboratory experiments: Sequencing Batch Reactor (SBR)

The lab-scale sequencing batch reactor has been running during 160 days since 5th of November, 2008 until 15th of April, 2009 with a 30-days break during Christmas holidays (from 17th of December 2008 until 15th of January 2009). The choice of SBR technique as the foundation stone of this work was evoked by its important advantages, which are listed in the section 2.5.1 together with the maximum suitability of the SBR installation regarding the present treatment facility at the NSR site.

4.1 Laboratory installation

The simplified layout of the laboratory installation is present on the Figure 4.1 and a photo – on the Figure 4.2.

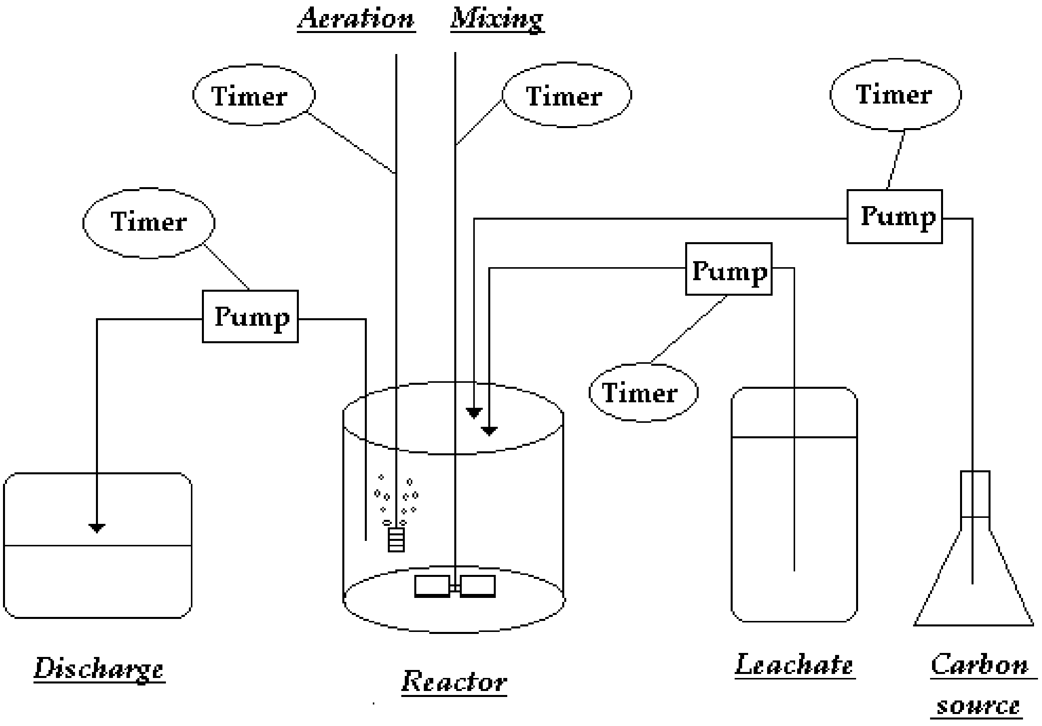


Figure 4.1 The layout of laboratory SBR installation

The laboratory installation consisted of the following parts:

- The sequencing batch reactor with total volume of 5 l and working volume of 4 l;
- Mixer;
- Aerator;
- Vessels for incoming leachate, additional carbon source and discharge of treated leachate;
- Pumps for leachate and carbon source addition and discharge of treated leachate;
- Timers for regulation of mixing, aeration, leachate and carbon source addition and discharge of treated leachate.

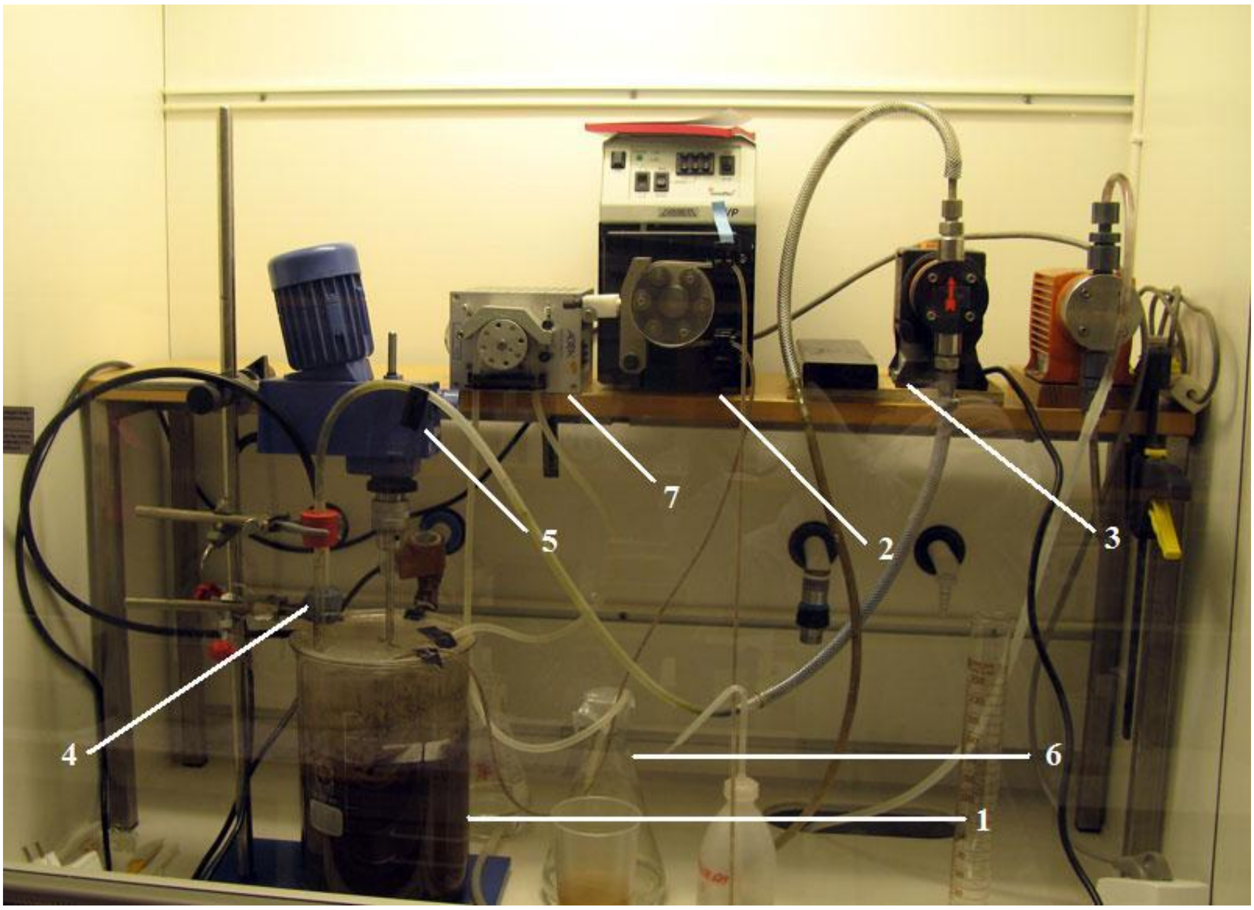


Figure 4.2 General view of the SBR laboratory installation. The following parts are marked: 1) Reactor; 2) Pump for the influent; 3) Pump for the effluent; 4) Aerator; 5) Mixer; 6) Vessel for carbon source; 7) Pump for addition of carbon source. The vessel for effluent is temporary removed

The leachate was pumped in continuously at the beginning of each cycle. The volume of incoming leachate was regulated by the speed of the pump.

An additional carbon source was pumped in continuously at the beginning of the denitrification stage. Ethanol was used as the carbon source during the process except of the last week, when it has been changed to sucrose. The amount of extra carbon source that had to be added depended on the concentration of nitrate in the reactor after nitrification step. As the pump speed was constant, carbon source addition was regulated by changing of the concentration of carbon source solution or changing of the pumping time.

The speed of the discharge pump was also constant. It has been working during 20 minutes, which was enough to pump out the maximal volume, loaded for one cycle. Excess of the sludge was removed manually by syringe to keep the sludge age constant. Timers were used to regulate the work of the pumps and mixer according to the time schedule.

Nitrifying activated sludge for the process was collected from the aeration basin of Källby wastewater treatment plant in Lund, which uses pre-denitrification process for nitrogen removal.

4.2 Treatment cycles

The duration of one treatment cycle has been set to 8 h. One cycle for complete nitrification/denitrification consisted of the following phases:

1) 1st aerobic phase. Nitrification process, which requires oxygen, took place during the first phase of the experiment. Duration of the phase varied from 2 to 5 h. The leachate was added to the reactor at the beginning of the phase during 0.8 h;

2) Anoxic phase. Denitrification proceeds during this phase. External carbon source for nitrate reduction was added at the beginning of the phase. Duration of the phase varied from 1.33 to 4.33 h;

3) 2nd aerobic phase. This phase was necessary for oxidation of easily biodegradable organic substances, which could be left in the reactor after anoxic phase because of incomplete utilization of carbon source. Duration of the phase was 20 min;

4) Settling phase. The sludge was settled for separation from the liquid. Duration of the phase was 1 h;

5) Discharge phase. The supernatant treated liquid was pumped from the vessel. Duration of the phase varied from 20 to 30 min.

While running the reactor in the nitrification mode, anoxic phase was excluded and the single aerobic phase lasted for 6.67 h. Time scales for three operation modes, which were used during the process, are presented on the Figures 4.3, 4.4 and 4.5. The SBR cycle with extended anoxic phase was used during first stages of the process (Days 14 – 42), while cycle with extended aerobic phase – at the last stages (Days 72 – 160). Total occupation of the equipment during 1 cycle and 24 hours is presented in the Table 4.1.

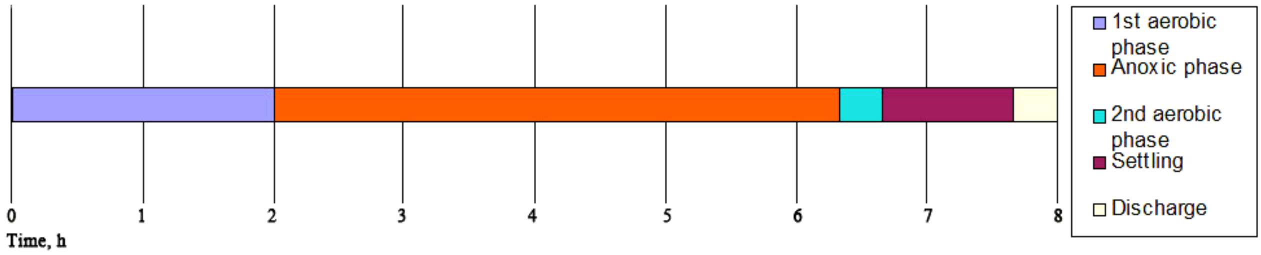


Figure 4.3 SBR cycle with extended anoxic phase

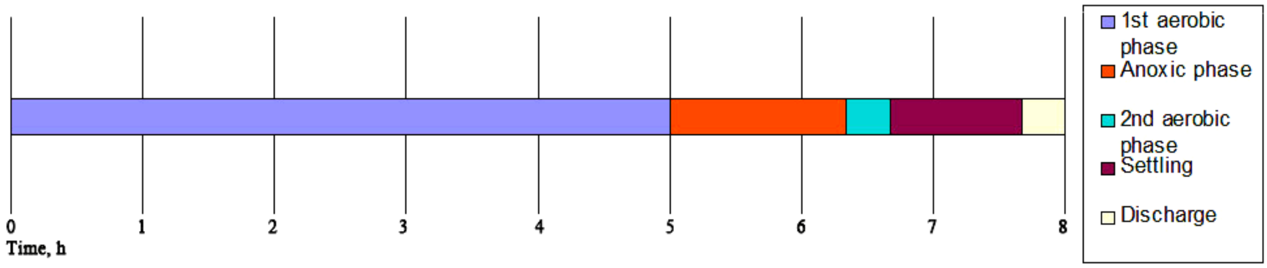


Figure 4.4 SBR cycle with extended aerobic phase

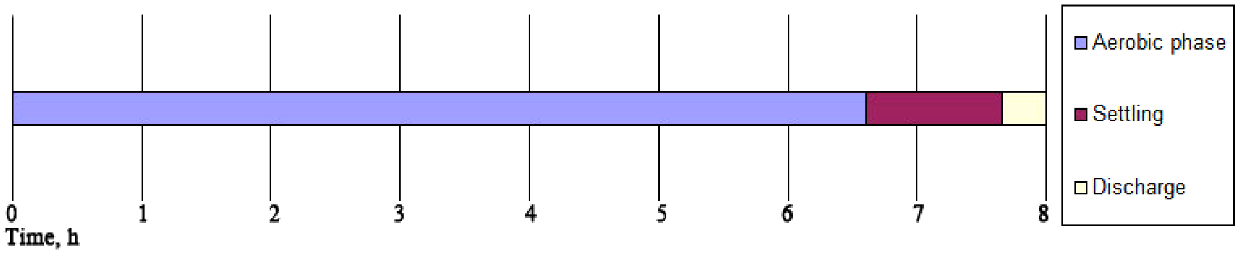


Figure 4.5 SBR cycle without denitrification process

Table 4.1 Occupation of the equipment at three different operational modes (in hours)

	Cycle with extended anoxic phase	***Cycle with extended aerobic phase***	***Cycle without denirification***
Aerator	2.33	5.33	6.67
Mixer	6.67	6.67	6.67
Pump for leachate addition	0.8	0.8	0.8
Pump for COD addition	0.17	0.17	-
Pump for discharge	0.33	0.33	0.33

4.3 Analytical methods

The following parameters were analyzed in the leachate and effluent: total nitrogen (filtered and unfiltered), ammonium (NH_4-N), nitrate (NO_3-N) and nitrite (NO_2-N) nitrogen, COD (filtered and unfiltered), total phosphorous (filtered and unfiltered), phosphate phosphorous (PO_4-P), suspended solids (SS), volatile suspended solids (VSS), pH, alkalinity, conductivity. pH, dissolved oxygen, alkalinity, SS and VSS were measured in the reactor.

4.3.1 Dr LANGE cuvette tests

COD and concentrations of nitrogen and phosphorous compounds were determined with Dr. LANGE cuvette tests (Table 4.2). The samples were filtered through the glass fiber filter and diluted before analyzes when necessary.

The interference between leachate and effluent color and 4-nitro-2,6-dimethylphenol coloration was found. Measurements of undiluted leachate in the LCK-339 cuvette without addition of 2,6-dimethylphenol gave the evidence of nitrate concentration, equal to 1.7 – 2.0 mg/l of nitrate. Thus, the correction value 1.85 was subtracted from the measured NO_3-N values to obtain the actual concentration of nitrate in the undiluted samples. In the samples, diluted 5 times or more, the correction factor was neglected.

Table 4.2 Dr LANGE analytical methods, used in the experiment

Parameter	***Dr LANGE cuvette test***	***Measuring range, mg/l***	***Measuring principle***
COD	LCK-114	150 – 1000	Oxidized substances are reduced by reaction with potassium dichromate ($Cr_2O_7^{2-} + 14H^+ + 6e- = 2Cr^{3+} + 7H_2O$). The intensity of green coloration of Cr^{3+} is measured. Silver sulphate is a catalyzing agent, mercury sulphate masks chloride ions.
NH_4-N	LCK-303	2 – 47	Ammonium ions react with hypochlorite and salicilate ions at pH 12.6. The intensity of generated inophenol blue is evaluated
NO_3-N	LCK-339	0.23 – 13.5	Nitrate ions react with 2,6-dimethylphenol with formation of 4-nitro-2,6-dimethylphenol. The intensity of yellow coloration is measured
NO_2-N	LCK-341	0.015 – 0.6	Nitrite ions react with aromatic amines with formation of diazosalts. The intensity of pink coloration of the latter substance is measured.
Total nitrogen	LCK-138	1 – 16	Ammonium and organic nitrogen are oxidized to the nitrate. Nitrate ions react with 2,6-dimethylphenol with formation of 4-nitro-2,6-dimethylphenol. The intensity of yellow coloration is measured
PO_4-P	LCK-349	0.05 – 1.5	Phosphate ions react with antimony and molibdate ions with formation of antimonyl phosphomolybdate. The latter is reduced by ascorbic acid to phosphomolybdenum blue
Total phosphorous	LCK-349	0.05 – 1.5	Organic phosphorous is oxidized to orthophosphate. Phosphate ions react with antimony and molibdate ions with formation of antimonyl phosphomolybdate. The latter is reduced by ascorbic acid to phosphomolybdenum blue

4.3.2 Other analytical methods

Suspended solids (SS) and volatile suspended solids (VSS) in the samples were determined according to the standard SS-EN 872-1. A sample of the certain volume (from 5 to 50 ml, depending on the thickness) was filtered through the pre-weighed filter. The filter was dried during at least 1 h at 105°C in the oven. The difference between the initial filter weight and the final weight after drying corresponded to the concentration of suspended solids in the sample.

The volatile suspended solids were determined by burning filter with sample in the oven for 2 h at 550ºC. The difference between the weight of dried filter and filter with ash after burning corresponded to the concentration of VSS in a sample.

pH in the leachate, reactor and effluent was measured with *WTW pH 320* pH meter, dissolved oxygen concentration – with oxygen analyzers *WTW Oxi 197S* and *WTW inoLab oxi 730*, conductivity – with conductivity sensor *WTW cond 340i*. Alkalinity of the leachate, reactor process liquid and effluent was determined by titration of 25 ml of the sample with 0.05 M HCl until pH 4.5 was reached. Alkalinity was calculated according to the formula:

$$Alkalinity = \frac{V_{HCL} \times M_{HCL}}{V_{sample}} \qquad \text{(eq. 4.1)}$$

where:

V_{HCl} – the volume of hydrochloric acid consumed for titration, l
M_{HCL} – the molarity of hydrochloric acid (50 mmol/l)
V_{sample} – the volume of the titrated sample

4.3.3 Determination of nitrification and denitrification rates

Nitrification and denitrification rates were determined by analyzing samples of the processed liquid, taken from the reactor at the certain intervals. Liquid was filtered directly after sampling to separate bacteria and stop nitrification or denitrification. Ammonium, nitrate and nitrite nitrogen were analyzed in all the samples with Dr LANGE cuvette tests (nitrite nitrogen was analyzed optionally).

The intervals for sampling were set to 10 minutes for denitrification and 15 – 30 minutes for nitrification, as nitrification rates are usually lower, than denitrification ones.

4.4 Leachate composition

The leachate for SBR reactor was being supplied from Filborna landfill. The sampling point was between buffer and aerated pond, as the new SBR reactor is most likely to be introduced there. The composition and main characteristics of treated leachate are presented in the Table 4.3.

Table 4.3 Composition of influent leachate, treated in the lab-scale SBR reactor (values for 4 complete analyzes and mean values are presented)

Parameter	***11.11.08***	***16.12.08***	***16.01.09***	***27.03.09***	***Mean value***
Total nitrogen (unfiltered), mg/l	153.8	n.m.	131.3	204	163.03
Total nitrogen (filtered), mg/l	103.2	n.m.	130	183	138.73
NH_4-N, mg/l	115	121	126	162	131
NO_3-N, mg/l	0.94	0	0.09	0.64	0.42
NO_2-N, mg/l	0.38	0.083	0.152	0.161	0.19
Total phosphorous (unfiltered), mg/l	3.75	4.15	14.6	8.37	7.72
Total phosphorous (filtered), mg/l	1.61	3.11	9.27	5.61	4.9
PO_4-P, mg/l	0.88	2.19	3.19	4.82	2.77
COD (unfiltered), mg/l	448	465	590	808	578
COD (filtered), mg/l	424	423	530	682	515
SS, mg/l	86	90	100	110	96.5
VSS, mg/l	86	90	94	100	92.5
pH	7.9	8.1	8	8.1	8
Alkalinity, mmol HCO_3^-/l	26.8	26.2	27.3	27.3	26.9
Conductivity, mS/cm	4.35	4.25	4.56	5.44	4.65

4.5 Results and discussion: the course of SBR process

The following section describes the course of the combined nitrification/denitrification process according to the time scale. The changes in concentration of NH_4-N, NO_3-N, NO_2-N, PO_4-P and COD were determined. Note, that at the beginning of the process and at the final stages NO_2-N, PO_4-P and COD were not measured regularly.

4.5.1 Hydraulic load

The hydraulic load has been increased in the course of the process to evaluate maximal possible load that can be applied at the 8-h treatment cycles. The load was changed by changing the volume of the leachate, pumped to the reactor for one complete cycle. The hydraulic load vs. timescale is presented on the Figure 4.6.

The process was started by stepwise increasing of the load from 0.07 to 0.33 m^3/m^3·cycle during the days 0 – 6, which corresponded to 1 l of leachate added to 3 l of sludge. Afterwards the process was run continuously with the same loading rate for adaptation of biomass to the new type of feed, observation of steady nitrification/denitrification process and determination of nitrification and denitrification rates. The load was increased up to 0.75 m^3/m^3·cycle during the days 35 – 41.

After the Christmas break (days 43 – 71) the process was renewed with hydraulic load of 0.25 m^3/m^3·cycle and extended aerobic phase, which allowed increase of hydraulic load up to 1.33 m^3/m^3·cycle (days 72 – 93). The increase of the load was accompanied by the wash-out of biomass because of excessive removal of sludge due to the mistake in calculations and subsequent decrease of the sludge age and ammonium removal rates (see Figure 4.8). Hydraulic load was decreased on the day 98 down to 1 m^3/m^3·cycle, but as it didn't provide the expected restoration of nitrification, leachate addition was stopped for a while and the process was started with the low hydraulic load (0.1 m^3/m^3·cycle) on the day 103. The significant drop of the pH in the reactor down to 6.1 on the day 112 was another reason to stop the process, as the nitrification completely failed.

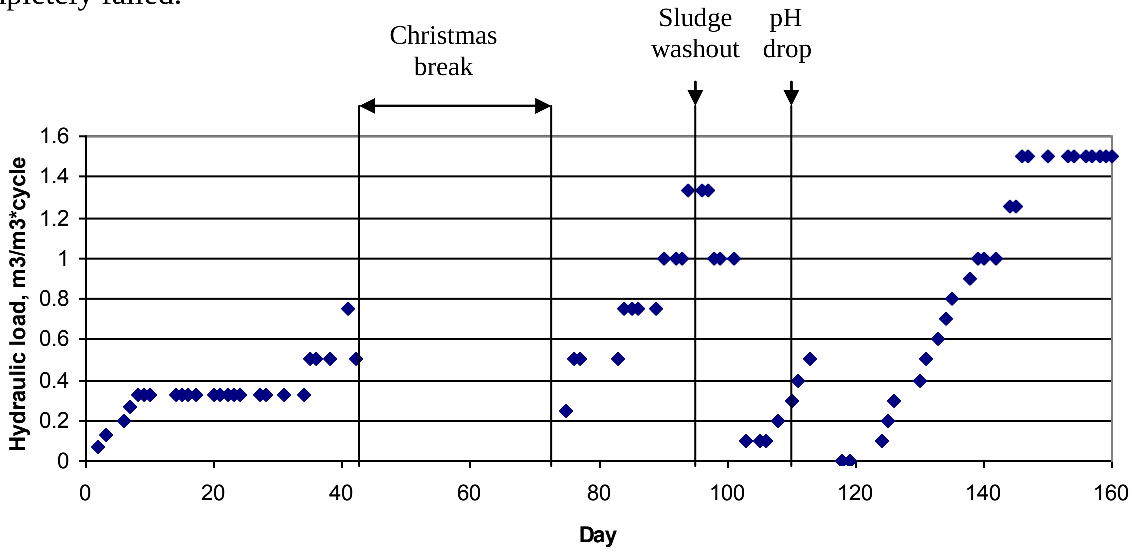


Figure 4.6 Hydraulic load in the course of the process

Beginning from the day 125 the hydraulic load of the process was increased up to 1.5 m^3/m^3·cycle which was the expected maximal load for the experiment. With the suspended solids concentration SS ≈ 10 g/l (Figure 4.7), the load could be expressed as 0.15 l/g SS. The process was run with this load until the day 160.

The experiments with increasing of hydraulic load proved the feasibility of using process load equal to 1.5 m^3/m^3·cycle in the estimation of full-scale SBR installation. The temporary faults in the process performance were caused by operational problems rather than by intrinsic properties of activated sludge or treated leachate.

4.5.2 SS and VSS in the reactor

SS and VSS concentrations increased in the course of the process (Figure 4.7). The highest SS concentration (≈ 16 g/l) was observed on the day 125. The decrease of suspended solids concentration on the day 97 was caused by the increased excess sludge removal which results in the washout of biomass.

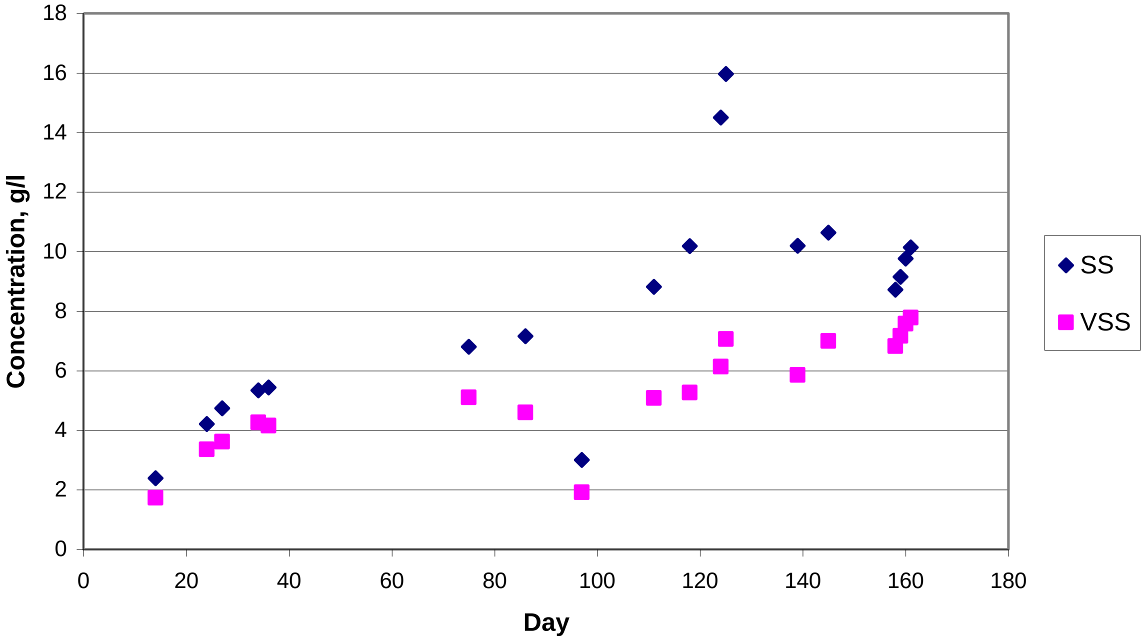


Figure 4.7 SS and VSS concentrations in the reactor in the course of the process

VSS/SS ratio of 0.75 – 0.8 was observed during the first phase of the experiment (days 14 – 42). After the day 75 this ratio decreased down to 0.4 – 0.5, which is typical for activated sludge with a high sludge age [von Sperling, 2007]. The VSS/SS ratio was restored to previous level after start up of the process on the day 125. The darting increase of SS and VSS concentrations was observed during the last week of the experiment, when the carbon source has been changed from ethanol to sucrose.

As the SS concentration is generally proportional to the concentration of biomass in the reactor, the increase of SS concentration vs time was measured to determine the growth rate of bacteria (see Figure 4.8).

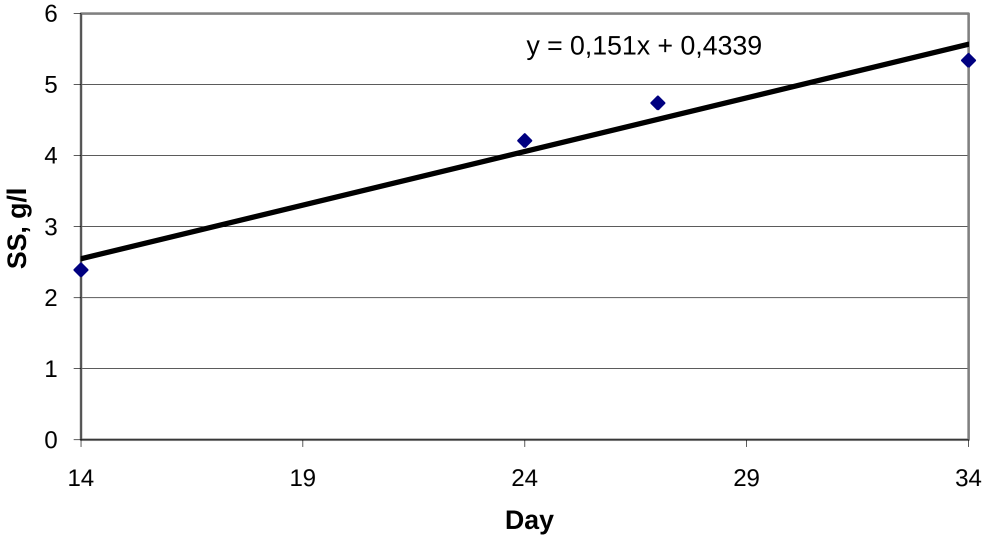


Figure 4.8 The increase of SS concentration during the days 14 – 34 (Sludge was not removed)

The growth rate of 0.15 g/l·d was found for the SBR sludge with combined nitrification/ denitrification process.

4.5.3 Sludge volume index

Sludge volume index (SVI) is the measure of settleability of solids in the clarifier of the activated sludge process or during settling phase at the sequencing batch process. SVI is the volume, which is occupied by 1 g of activated sludge after 30 minutes of settling in the 1000 ml measuring cylinder. The measuring unit is ml/g.

The SVI was determined on the day 34 of the experiment. After 30 minutes of settling the sludge occupied 450 ml in the 1000 ml cylinder. With the suspended solids concentration of 5.34 g/l the sludge volume index, which could be used for modeling of the full-scale installation was found according to the formula:

$$SVI = \frac{V_{settled} \times 1000}{SS} = \frac{450 ml/l \times 1000}{5340 mg/l} = 85 \text{ ml/g} \quad \text{(eq. 4.2)}$$

4.5.4 Nitrogen concentrations

The concentrations of three forms of inorganic nitrogen, which were measured in the effluent as the experiment progressed, are presented on the Figure 4.9. Ammonium, nitrate and nitrite nitrogen levels generally were lower than 1 mg/l after treatment in the SBR. The high NO_3-N concentrations in the effluent during days 14 – 22 depicted the slow start-up of denitrification process.

The temporary increase of NH_4-N concentrations in the effluent were observed mainly due to the technical faults, most commonly, aeration problems because of the aerator clogging or break of energy supply. Two significant failures of nitrification process were observed during the days 94 – 102 (washout of sludge) and 111 – 124 (low pH of the leachate). NO_3-N concentrations in the leachate were significantly increased several times because of the lack of carbon source (insufficient dosage or stop of pumping due to the break of energy supply).

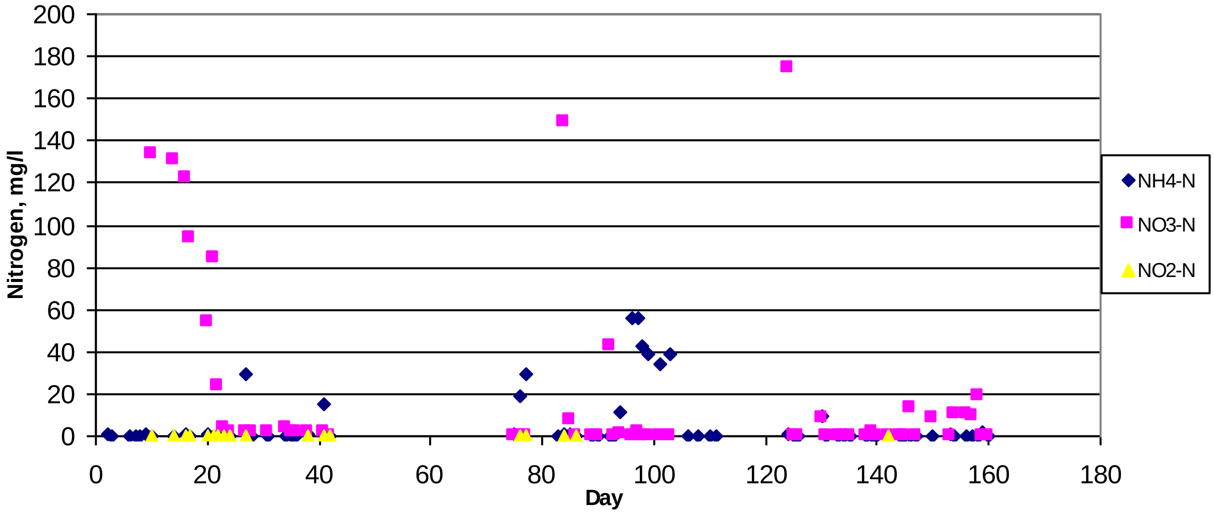


Figure 4.9 NH_4-N, NO_3-N and NO_2-N concentrations in the effluent in the course of the process

Concentrations of nitrogen compounds in the effluent in the course of the process indicate that the process could effectively decrease nitrogen concentrations in the treated leachate down to the discharge limits. The concentrations, which exceed discharge limit of 15 mg/l were observed in the effluent either during start-up of the process or because of technical faults.

4.5.5 COD, phosphorous concentrations and other parameters

COD concentrations in the effluent were in the range between 300 and 500 mg/l (Figure 4.10). The increase of COD concentration on the days 16 – 17 is explained by overdosing of ethanol for the denitrification process.

PO_4-P concentration in the effluent fluctuated in the wide range between 0.6 and 17 mg/l (Figure 4.11). The correlation between inorganic phosphorous concentration and other operational parameters was not observed.

Suspended solids in the effluent were in the range between 20 and 100 mg/l. The suspended solids consisted totally from volatile substances, as VSS concentrations were the same, as SS concentrations. Alkalinity of the effluent was determined in the range of 14.2 – 17.4 mmol HCO_3^-/l, conductivity – 3.2 – 4.2 mS/cm.

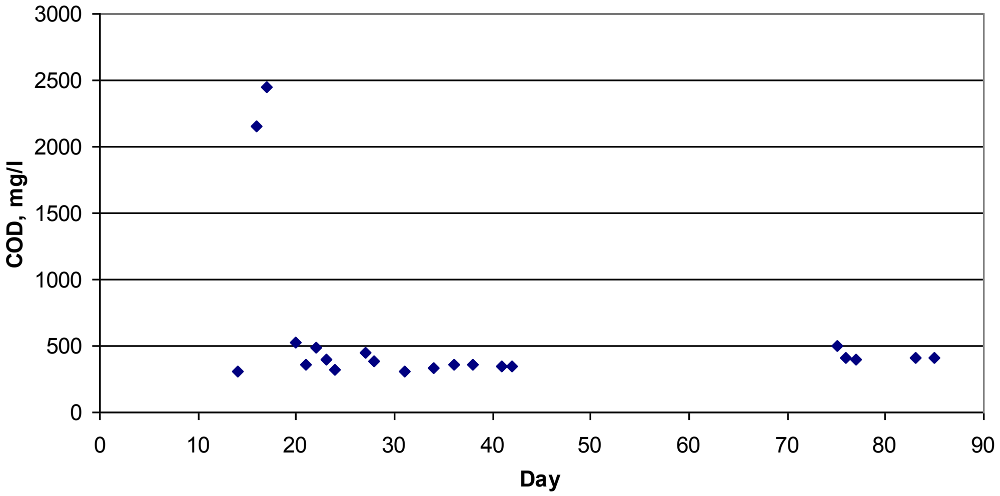


Figure 4.10 COD concentrations in course of the process (Days 14 – 85)

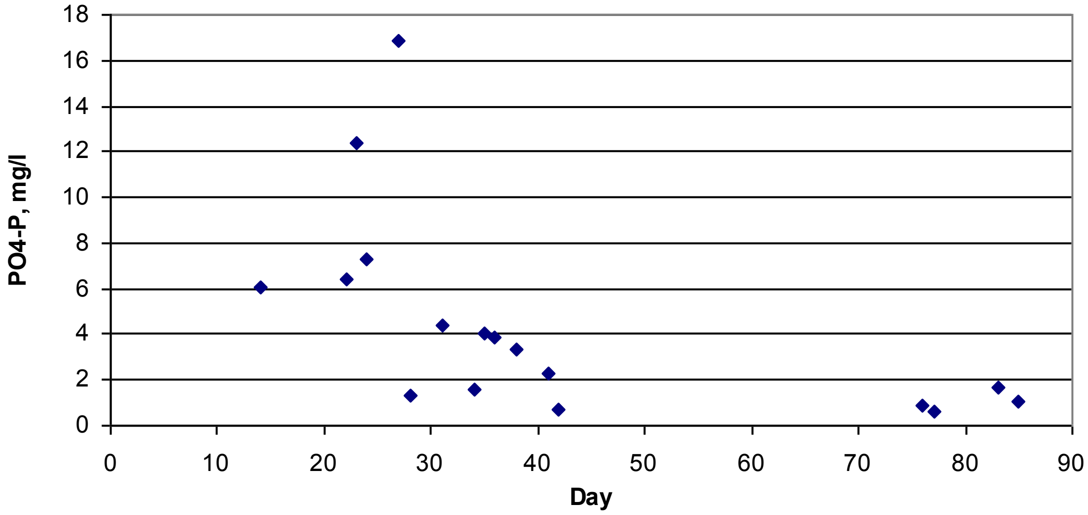


Figure 4.11 PO_4-P concentrations in course of the process (Days 14 – 85)

pH in the reactor was between 8.0 and 8.5 except of the days 75 (pH 7.3) and 111 – 124 (pH 6.1 – 6.4), In the first case low pH was observed after storage of sludge during 30 days at the low temperature (7°C) and in the anaerobic conditions. The pH was restored on its own during one day after start up of the reactor in nitrification/denitrification mode with extended aerobic phase.

Decrease of pH down to 6.1 on the day 111 resulted from the low pH of the incoming leachate. Among the possible reasons for the decrease of pH in the leachate are the mistake of operators, who deliver leachate (delivery in the dirty cans, delivery of the wrong liquid etc), or decrease of pH during storage of the leachate in the laboratory. pH in the reactor was restored on the day 124 up to 7.8 by addition of 10 ml of 2.5M NaOH. The temporary increase of alkalinity in the reactor up to 80 mmol HCO_3^-/l were observed afterwards.

4.6 Results and discussion: removal efficiency of the process

4.6.1 Reduction of COD, phosphorous and nitrogen

Removal efficiency of the process was expressed in concentrations and in percentage of removed compound from its initial concentration in the leachate. Results are presented in the Table 4.4.

The results shows, that the SBR process was efficient in terms of nitrogen removal (93% of total nitrogen and 99% of ammonium nitrogen removed). Since ammonium is the dominant nitrogen-containing compound in the influent and the target compound to be removed with the SBR process, efficiency of ammonium nitrogen removal was evaluated on a daily basis in the course of the process. Figure 4.12 presents the removal efficiency of ammonium in percent from its initial concentration in the leachate. Removal efficiency exceeded 99% during the process except of five cases, which corresponded to the certain failures of the process (overloading, low pH, oxygen limitations).

Table 4.4 Removal efficiency of the SBR process

Parameter	***Concentration in the influent, mg/l****	***Concentration in the effluent, mg/l****	***Reduction, mg/l***	***Reduction, %***
Total N (unfiltered)	163	12.3	150.7	93
Total N (filtered)	138.7	10.2	128.5	93
NH_4-N	131	0.2	130.8	99
NO_3-N	0.4	0.6	-0.2	-
NO_2-N	0.2	0.1	0.1	50
Total P (unfiltered)	7.7	6.1	1.6	20
Total P (filtered)	4.9	4.4	0.5	10
PO_4-P	2.8	3.6	-0.8	-
COD (unfiltered)	578	362	216	37
COD (filtered)	515	330	185	36
SS	96.5	54	42.5	44
VSS	92.5	44.5	48	52

*- mean values of four measurements, taken during the period November 2008-April 2009

Nitrate and nitrite nitrogen concentrations stayed below 1 mg/l both in the influent and effluent. Slightly larger concentration of nitrate in the effluent could have been observed due to the residual nitrate that was not converted into nitrogen gas during denitrification. Removal of the total nitrogen was slightly lower – 93% respectively. Still, with current concentration of total nitrogen in the leachate, total nitrogen concentrations in the effluent were below discharge limits (15 mg/l).

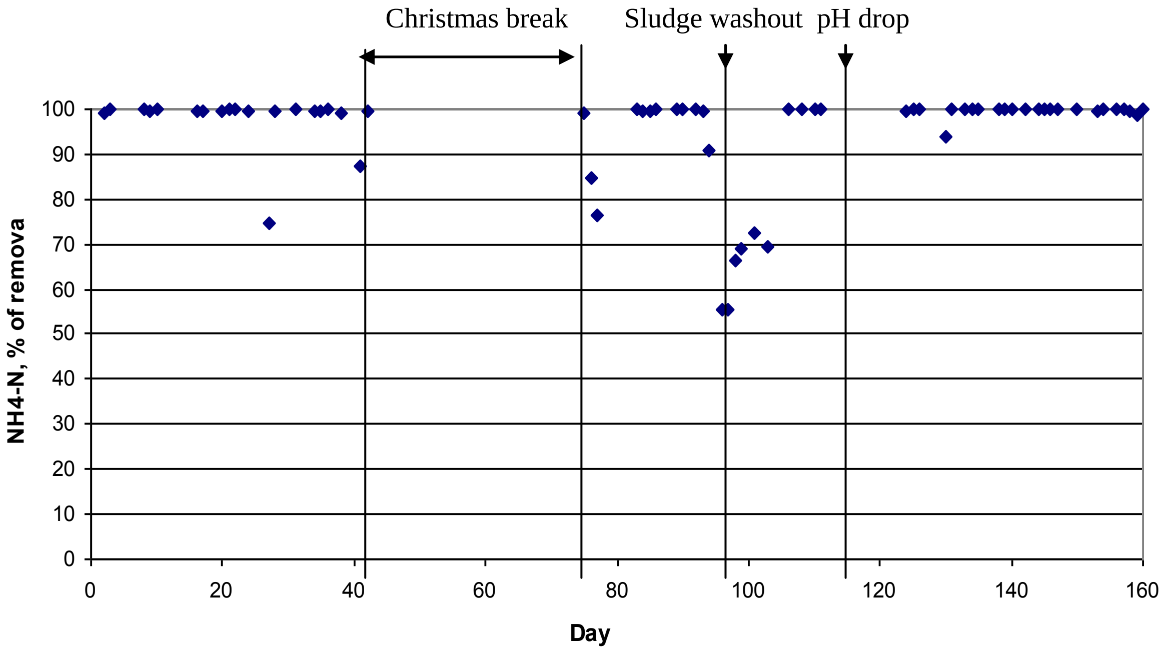


Figure 4.12 Efficiency of NH_4-N removal
(in percent of removed NH_4-N from the initial concentration in the leachate)

COD removal didn't exceed 40%, and phosphorous removal – 20%, which was an expected result as SBR process was designed only for nitrogen removal. Figure 4.13 shows the efficiency of COD and PO_4-P removal according to the time scale during first weeks of the process running. No reasonable correlations between the process conditions and removal rates have been found. It means that combined SBR nitrification/denitrification process could not be designed for simultaneous biological COD or phosphorous removal from leachate at Filborna landfill.

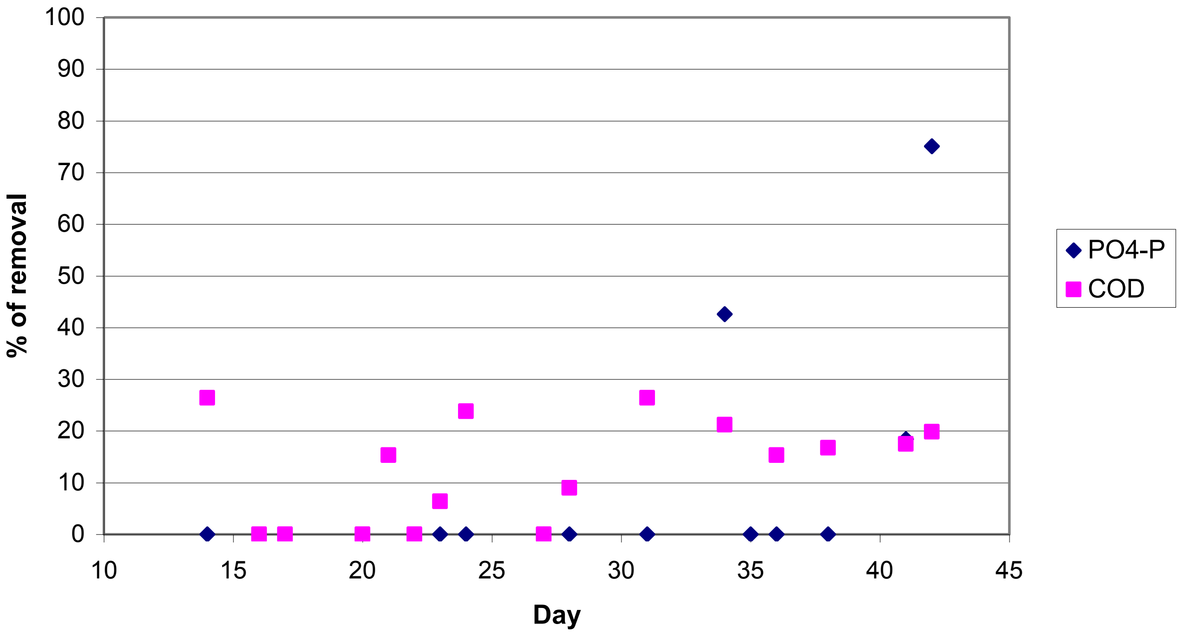


Figure 4.13 Efficiency of PO_4-P and COD removal
(in percent of removed PO_4-P and COD from the initial concentration in the leachate)

4.6.2 Changes in alkalinity and conductivity

The changes in alkalinity and conductivity between the influent and treated leachate are presented in the Table 4.5. Alkalinity was reduced with 41%. In theory, 2 moles of HCO_3^- per mole of oxidized NH_4-N are consumed during nitrification, while 0.91 moles of HCO_3^- per mole of reduced NO_3-N are produced in the denitrification process [Henze, 2002]. Thus, ≈ 50% reduction of alkalinity in the combined nitrification/denitrification process is predicted that is slightly higher than observed. Conductivity, which is the measure of ion strength of the solution, decreased in the course of the process.

Table 4.5 Changes in alkalinity and conductivity of the leachate after SBR treatment

Parameter	***Value in the influent****	***Value in the effluent****	***Reduction***	***Reduction, %***
Alkalinity, mmol HCO_3^-/l	26.9	15.8	11.1	41
Conductivity, mS/cm	4.65	3.63	1.02	22

* - mean values of four measurements, taken during the period November 2008 – April 2009

4.6.3 Nitrification and denitrification rates

Nitrification and denitrification rates were measured twice during the course of the process (Day 28 and Day 142) by direct determination of changes in NH_4-N and NO_3-N concentrations in the reactor during one cycle (see section 4.3.3). The rate, in mg of removed nitrogen per 1 l of sludge per 1 h, was determined from the slope of the obtained reduction curve, as it is shown on the Figure 4.14.

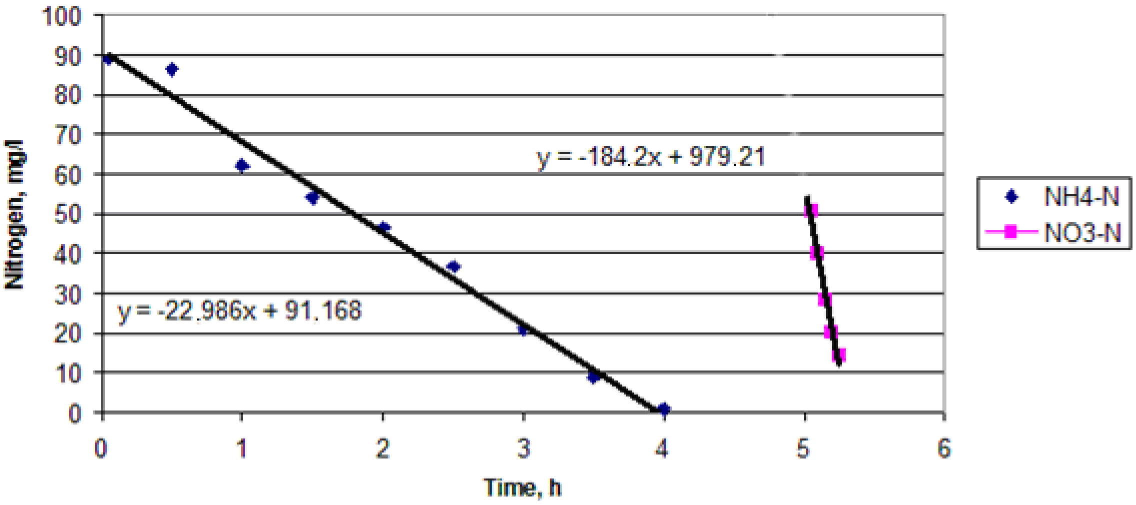


Figure 4.14 Graphical presentation of nitrification and denitrification rates

To determine the efficiency of biomass utilization in the reactor, nitrification and denitrification rates were also calculated in mg of removed nitrogen per g of VSS per 1 day (24 h). The values of nitrification and denitrification rates are presented in the Table 4.6.

Table 4.6 Nitrification and denitrification rates of the sludge in SBR process

	VSS, g/l	***Nitrification rate***		***Denitrification rate***	
		in mg NH_4-N /l·h	in mg NH_4-N/g VSS·h	in mg NO_3-N/l·h	in mg NO_3-N/g VSS·h
Day 28	3.7	23.6	6.2	178.8	48.3
Day 142	6.43	23	3.6	184.5	28.7

Process rates, determined in volumetric terms were in the same range (23 – 23.6 mg NH_4-N/l·h for nitrification respective 178.8 – 184.5 mg NO_3-N/l·h for denitrification). Significant decrease of the process rate, calculated in terms of biomass weight, with the increase of biomass concentration indicated that at higher VSS concentrations efficiency of biomass utilization both for nitrification and denitrification decrease. Nitrification rate was almost 8 times lower, than denitrification rate thus, nitrification is a limiting step in combined nitrification/denitrification process.

4.6.4 Alterations of nitrogen concentration in the reactor in the course of one cycle

The changes in NH_4-N, NO_3-N and NO_2-N concentrations during the single cycle of the combined nitrification/denitrification process are presented on the Figure 4.15. The data were taken on the Day 28 with hydraulic load of 0.33 m^3/m^3·h and cycle with extended anoxic phase applied. The increase of NH_4-N concentration during first 45 minutes of the process was observed because of the continuous addition of leachate to the reactor during first 50 minutes of the cycle, combined with nitrification process: the supply of the NH_4-N exceeded nitrification rate. Continuous lines show the actual concentrations in the reactor, while discontinuous lines – assumed concentrations for the case of immediate addition of all leachate to the reactor at the beginning of the process.

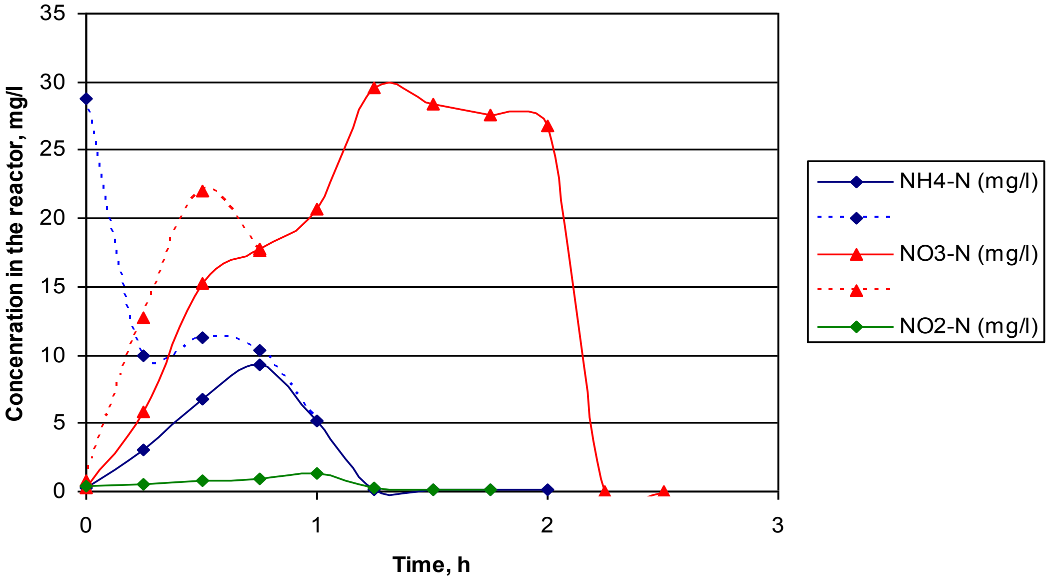


Figure 4.15 Alteration of NH_4-N, NO_3-N and NO_2-N concentrations in the reactor in the course of combined nitrification/denitrification process. Discontinuous lines show the supposed NH_4-N and NO_3-N concentrations in case of immediate addition of all the leachate volume at the beginning of the cycle. The depicted markers shows actually measured concentrations, while depicted lines – assumed changes of concentrations between measurements

Concentration of NO_3-N during the nitrification stage has been increasing continuously with decrease of ammonium concentration in the reactor. After start up of the denitrification stage NO_3-N concentration gradually decreased during 15 minutes because of high denitrification rate. Concentration of NO_2-N remained constantly low except of certain increase after 1 h of nitrification. The latter could be explained by slightly higher rates of ammonium oxidation, than nitrite oxidation in two-step nitrification process.

4.7 Results and discussion: nitrogen, phosphorous and organic fractions

Fractionation of organic compounds in the leachate according to their settleability and degradability is important for technical design of a wastewater treatment facility. E.g. biological methods of removal can’t be applicable for reduction of inert compounds in wastewater. Main fractions of nitrogen, phosphorous and organic compounds in the leachate from Filborna landfill are determined in the following section.

4.7.1 Nitrogen fractions in the leachate

Nitrogen fractions in the leachate before and after SBR treatment are presented on the Figure 4.16.

NH_4-N, NO_3-N and NO_2-N fractions correspond to the concentration of these ions in the solution. The difference between total nitrogen concentration in the filtered sample and dissolved easily degradable organic nitrogen fraction is the nitrogen, included into the dissolved inert organic material. The fraction of suspended nitrogen is determined as a difference in total nitrogen concentrations in the unfiltered and filtered samples. It could be further divided into suspended easily degradable organic nitrogen and inert organic nitrogen using the difference between suspended nitrogen fractions in the leachate before and after combined nitrification/denitrification process. The fractionation of nitrogen in the leachate is presented in the Table 4.7

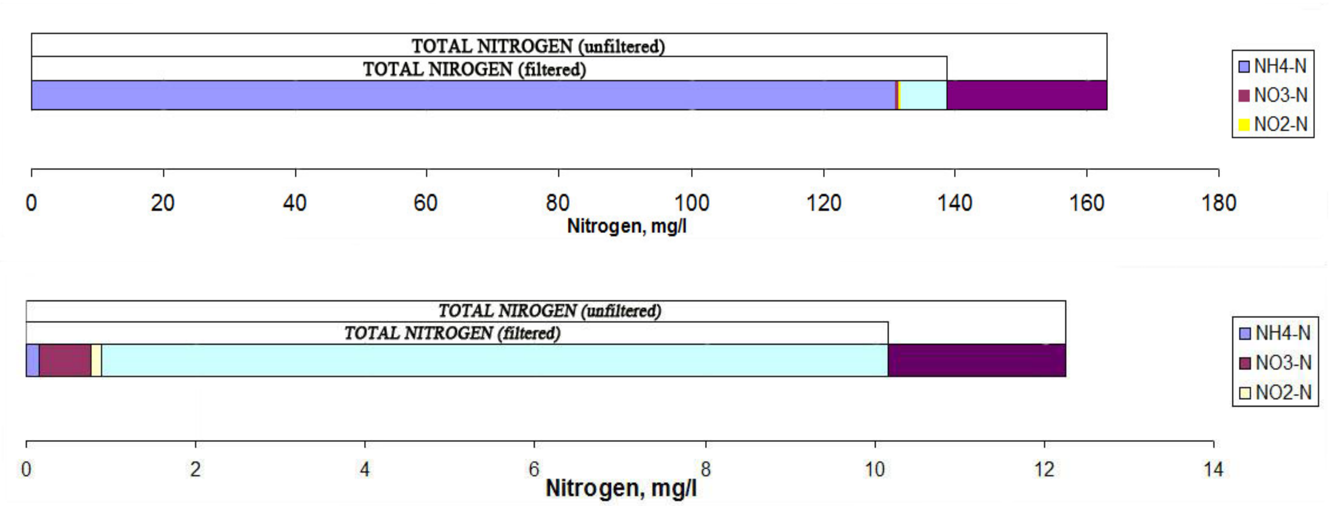


Figure 4.16 Measured nitrogen fractions in the influent (upper bar) and in the effluent leachate (lower bar)

Table 4.7 Nitrogen fractions in the leachate from Filborna landfill (after treatment in the buffer pond at the landfill site). Classification according to [Henze, 2002]

Fraction	***Concentration, mg/l***	***% of total nitrogen***
Total nitrogen (C_{TN})	163	100
Nitrite+nitrate nitrogen (S_{NOX})	0.61	0.3
Ammonium nitrogen (S_{NH4})	131	80.4
Dissolved inert organic nitrogen ($S_{I,N}$)	7.1	4.4
Suspended easily degradable organic nitrogen ($X_{S,N}$)	22.2	13.6
Suspended inert organic nitrogen ($X_{I,N}$)	2.1	1.3

The main nitrogen fraction in the raw leachate is ammonium nitrogen. This fraction is almost completely removed during combined nitrification/denitrification process. Suspended easily degradable organic nitrogen is the second largest fraction. However its concentration make up 22% of suspended easily degradable organic matter (see Table 4.8), which doesn’t correlate with the chemical composition of organic matter. Most probably, dissolved easily degradable organic nitrogen is also included in this fraction. In the latter case the fraction make up 10.3% of the total easily degradable organic matter which corresponds to the nitrogen content of the typical bacterial cell.

Dissolved and suspended inert organic nitrogen represents only 5.7% of the total nitrogen content in the leachate, but this nitrogen could not be removed by biological process. The comparison with the concentration of inert organic compounds in the leachate (Table 4.8) indicates that the inert organic matter has the nitrogen content of ≈ 2.5%. Such a low nitrogen content is common for humic and fulvic acids, which are typical inert organic compounds in leachates from old landfills [Rice, 1991].

4.7.2 Phosphorous fractions in the leachate

Phosphorous fractions in the leachate before and after SBR treatment are presented on the Figure 4.17. Three fractions could be accentuated: phosphorous of orthophosphate (PO_4-P), dissolved phosphorous (correspond to the difference between the total phosphorous in the filtered sample and PO_4-P) and suspended phosphorous (difference between total phosphorous in the unfiltered and filtered samples). PO_4-P fraction is larger in the treated leachate, while organic phosphorous fractions – in the influent leachate. It is explained by the heterotrophic conversion of degradable organic matter during SBR process, which leads to the release of orthophosphates.

Data from the Figure 4.17 could not be interpreted according to the classification of phosphorous fractions in wastewater, presented by [Henze, 2002]. It is not possible to determine the fraction of dissolved inorganic polyphosphates. Phosphorous of low molecular weight polyphosphates is included into the PO_4-P fraction, while phosphorous of high molecular weight polyphosphates – into dissolved organic phosphorous fraction.

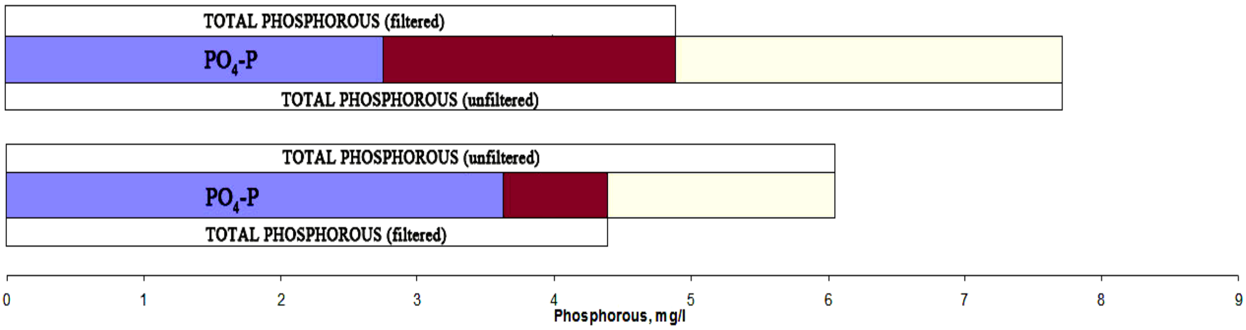


Figure 4.17 Phosphorous fractions in the influent (upper bar) and in the effluent leachate (lower bar)

4.7.3 Organic fractions in the leachate

Organic fractions in the leachate before and after SBR treatment are presented on the Figure 4.18.

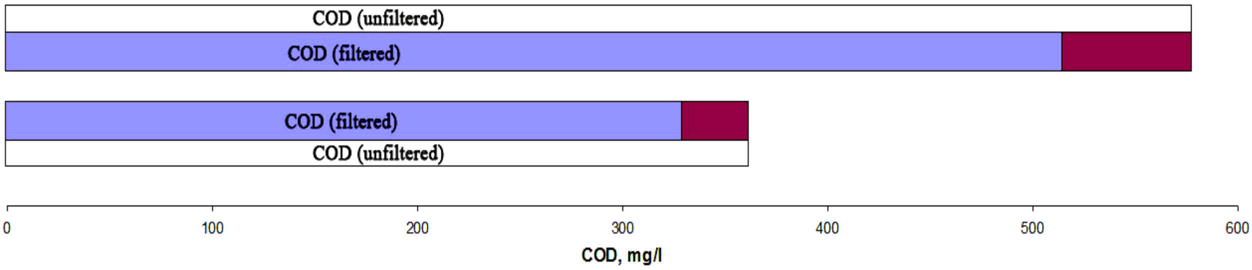


Figure 4.18 COD fractions in the influent (upper bar) and in the effluent leachate (lower bar)

The fraction of dissolved inert organic matter corresponds to the COD in the filtered sample of the treated leachate, while dissolved easily degradable organic matter – to the difference between COD in the filtered sample of the influent and treated leachate. Suspended inert organic matter is calculated as the difference between COD in the unfiltered and filtered samples, while suspended easily degradable organic matter – as the difference between COD, corresponding to the unfiltered fractions in the samples from influent and treated leachate. The fractionation of organic compounds in the leachate is presented in the Table 4.8.

Table 4.8 Organic fractions in the leachate from Filborna landfill (after treatment in the buffer pond at the landfill site). Classification according to [Henze, 2002]

Fraction	*Concentration, mg/l*	*% of total COD*
Total organic matter (COD)	578	100
Dissolved easily degradable organic matter (S_S)	185	32
Dissolved inert organic matter (S_I)	330	57
Suspended easily degradable organic matter (X_S)	31	5.4
Suspended inert organic matter (X_I)	32	5.6

The main organic fraction in the leachate is found to be dissolved inert organic material, which comprised 57% of all the organic content. It is a characteristic feature of old leachates [Renou, 2008]. Dissolved easily degradable organic matter made up 32% of organic matter of the leachate. Suspended fractions were less significant. As the concentration of inert organic matter couldn't be effectively decreased by biological treatment, development of additional chemical or physical process is required for its removal

4.8 Results and discussion: inhibition test

Inhibition test was carried out according to the screening method, presented in [Jönsson, 2001]. The test was performed on the day 104 of the process running after continuous decrease of nitrification rate during 10 days. The presence of inhibitive toxic compounds in the leachate, delivered from Filborna landfill on the day 93 (6th February 2009) was suspected. The leachate delivered previously on the day 72 (16th January 2009) was also examined. The results of the test are presented on the Figure 4.19, where the percent of inhibition of nitrifying activity in the samples with two types of analyzed leachate comparatively with nitrifying activity in the reference samples with tap water (zero inhibition on the curve) is indicated.

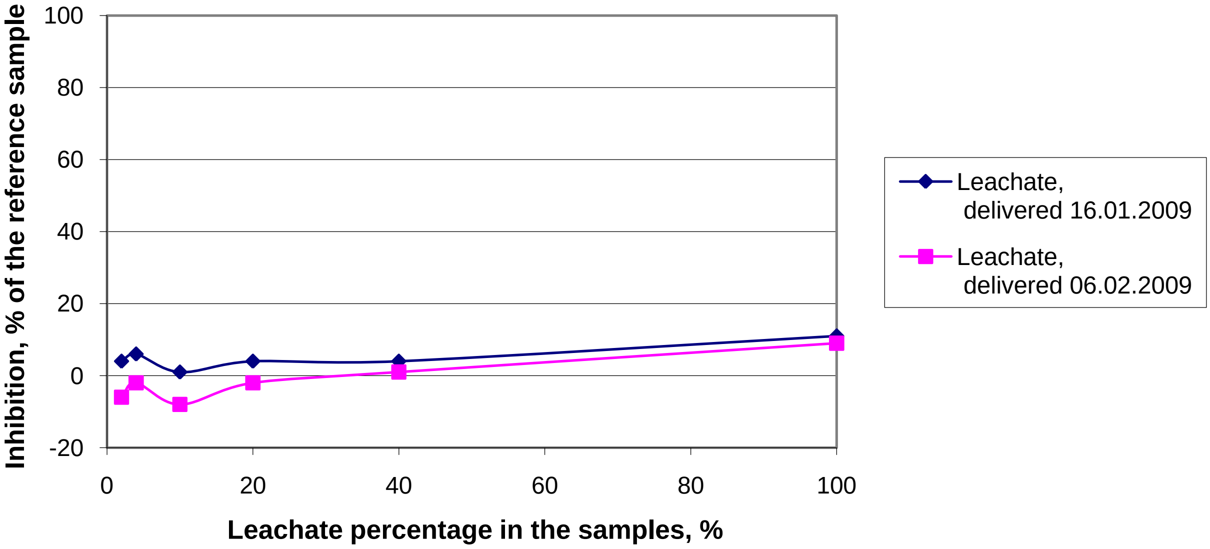


Figure 4.19 Results of inhibition test: percentage of inhibitive activity in the tested leachates from the inhibitive activity in the reference sample.

According to the results of the test, inhibitive activity of both leachates didn't alter more than 10% from the reference sample that has proved absence of inhibitive substances in the leachates.

The conclusion was made that the excessive washout of sludge was the reason for the nitrification failure.

4.9 Results and discussion: nitrification under unfavorable conditions

The impact of the environment on the nitrification process is comprehensively examined in the literature. The experiments, which are described in the sections 4.5 and 4.6, show evidently that nitrification is the process-limiting stage. Thus, more careful investigation of the factors, which most commonly affect nitrification in the full-scale wastewater treatment process, is provided below.

4.9.1 Nitrification under limitation of dissolved oxygen

The optimal concentration of oxygen, required for nitrification, is above 2 mg/l. At the lower concentrations oxygen becomes a serious process-limiting factor [Henze, 2002]. Nitrification curve for the process at the dissolved oxygen concentration of 0.5 mg/l and hydraulic load of 0.5 $m^3/m^3 \cdot d$ is presented on the Figure 4.20. Nitrification rate was equal to 13.3 mg NH_4-N/l·h (2.06 mg NH_4-N /g VSS·h), which is ≈ 2 times lower value, than nitrification rate at the oxygen concentrations above 2 mg/l.

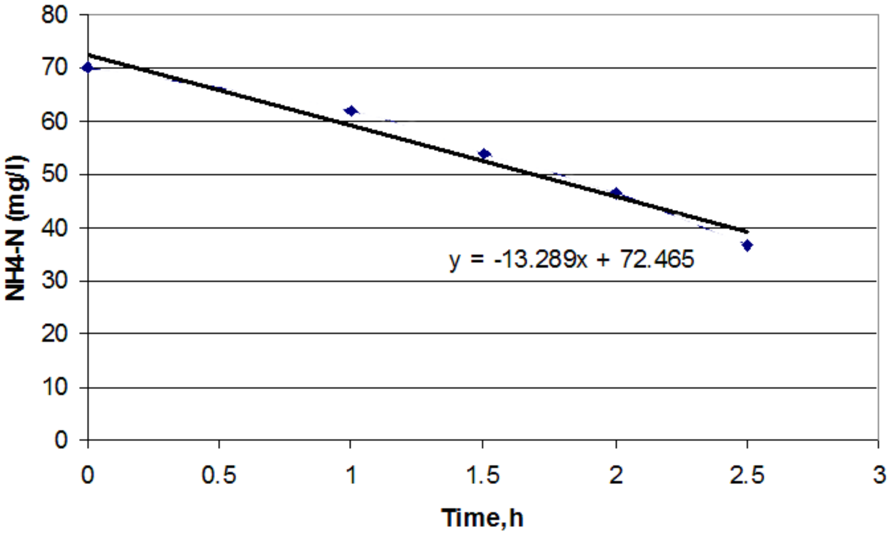


Figure 4.20 Nitrification rate at the oxygen concentration equal to 0.5 mg/l

4.9.2 Nitrification under low pH

The pH dropped on the day 111 of the process because of addition of leachate with pH 6.3. Nitrification process was examined on the day 115. The reactor was running in the continuous nitrification mode without leachate addition. To determine the nitrification rate, the certain amount of leachate with known concentration of NH_4-N was added and samples from the reactor analyzed every 30 minutes. Nitrification curve is presented on the Figure 4.21. The denitrification rate was equal to 2.6 mg NH_4-N/l·h (0.51 mg NH_4-N/g VSS·h), which is ≈ 9 times lower value than nitrification rate at the oxygen saturation conditions.

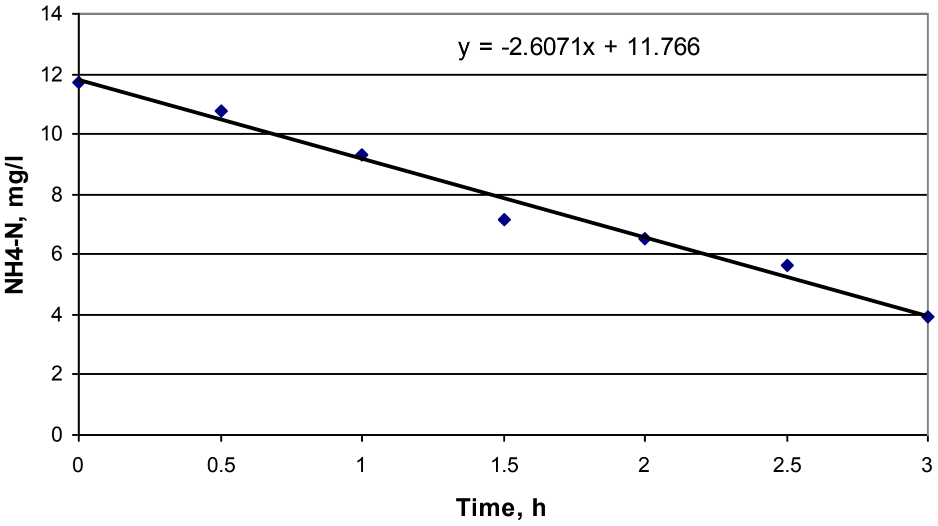


Figure 4.21 Nitrification rate at the pH 6.1

4.9.3 Nitrification under low temperature conditions

The experiment on determination of nitrification rate was carried out at the temperature 20°C (mean temperature in the lab). However, in reality SBR installation would be situated outside. Thus, the estimation of nitrification rates at the temperature range from 5°C to 25°C is necessary.

The dependence of nitrification rate on the temperature could be described by Arrhenius equation [Stark, 1996]:

$$V_{\max} = A \times \exp^{\frac{-Ea}{RT}} \qquad \text{(eq. 4.3)}$$

where:

V_{max} – relative nitrification rate (mg/kg·d)
A – pre-exponential factor of Arrhenius constant (mg/kg·d)
E_a – activation energy (kJ/mole)
R – universal gas constant (R = 8.31 J/K·mole)
T – temperature (K)

The pre-exponential factor and activation energy is determined individually for each reaction. The following values were found by Stark for complete nitrification process [Stark, 1996]:

$A = 5.12 \cdot 10^9$ mg/kg·d
$E_a = 52.5$ kJ/mole

Considering following values, nitrification rate could be estimated at different temperatures from 5°C to 25°C with 5ºC interval, referring to the nitrification rate, obtained experimentally for 20°C. These values are shown in the Table 4.9.

Table 4.9 Estimation of nitrification rate of sludge at different temperature conditions

Temperature, ºC	***Relative nitrification rate, mg/kg·d***	***Nitrification rate, mg NH_4-N/g VSS ·h****
5	0.69	1.9
10	1.03	2.9
15	1.52	4.3
20	2.21	6.2**
25	3.18	8.9

* VSS = 3.8 g/l
** Experimental value

4.10 Results and Discussion: Utilization of Different Carbon Sources

The question of the choice of the carbon source is essential in the design of wastewater treatment facilities, as external carbon source could often become one of the most cost-consuming operational factors. The universal answer for this question doesn’t exist, as the choice of carbon source depends very often on its price and availability.

In this study two possible carbon sources are considered for the denitrification step in SBR leachate treatment – ethanol and sucrose. Ethanol has been used in the course of the project, while sucrose – only during the last week of reactor operation.

4.10.1 Ethanol as a carbon source in SBR process

Ethanol is one of the most common sources of organic substrate for denitrification in the wastewater treatment processes, both lab-scale and industrial. Ethanol is considerably cheap liquid which is easy to store and deliver to the treatment tank. The drawback of the ethanol utilization is the flammability of the liquid and strict safety requirements. Ethanol is not also the cheapest carbon source among all the available.

Denitrification process with ethanol as a carbon source is described by the following equation:

$$\tfrac{5}{12}C_2H_6O + NO_3^- + H^+ \rightarrow \tfrac{1}{2}N_2 + \tfrac{10}{12}CO_2 + \tfrac{21}{12}H_2O \qquad \text{(eq. 4.4)}$$

According to the formula, the theoretical ethanol consumption in the denitrification process is 1.37 g of ethanol per 1 g of nitrate nitrogen (NO_3-N) reduced. Only 42% of ethanol in the influent is used directly for the denitrification [Gomez, 2000]. The rest of it is utilized for bacterial growth. Thus, 3.26 g of ethanol per 1 g of nitrate nitrogen has been added. The COD content in ethanol is 2.09 g COD/g ethanol. In terms of COD the ethanol requirements for the nitrate nitrogen reduction are 6.8 g ethanol COD/g NO_3-N.

4.10.2 Sucrose as a carbon source in SBR process

Sucrose was considered to be an alternative carbon source for denitrification. The main advantages of sucrose comparatively to the ethanol are lower costs, handleability and availability. The latter is important issue for the NSR site, where waste from sugar processing factories in Skåne could be used as a source of sucrose. The main drawback of sucrose utilization is the considerably larger amount of carbon source that has to be added, comparatively to the ethanol addition. If the process water from sugar industry is used, considerably large amounts may be required, which affects the size of equipment. Other problems with sucrose addition could be foaming (Figure 4.22) and growth of filamentous bacteria. The latter can be the reason for sludge bulking.

Denitrification process with sucrose as a carbon source proceeds according to the following equation:

$$0.104C_{12}H_{22}O_{11} + NO_3^- + H^+ \rightarrow 0.5N_2 + 1.248CO_2 + 1.648H_2O \qquad \text{(eq. 4.5)}$$

From this formula, the theoretical consumption of sucrose for denitrification process is 2.54 g of sucrose per 1 g of nitrate nitrogen (NO_3-N) reduced. Only 13% of sucrose is used directly for denitrification [Gomez, 2000]. In consideration of the latter, 19.5 g of sucrose was added to reduce 1 g of NO_3-N. The COD content in sucrose is 1.122 g of COD per g of sucrose, thus, 21.9 g of sucrose COD was used for each gram of nitrate nitrogen.

The determined volumetric denitrification rate was 53 mg NO_3-N/l·h (Figure 4.23) which is 3.4 times lower, than the denitrification rate with ethanol as a carbon source (Table 4.6).

Figure 4.22 Foaming in the reactor observed with the utilization of sucrose as a carbon source

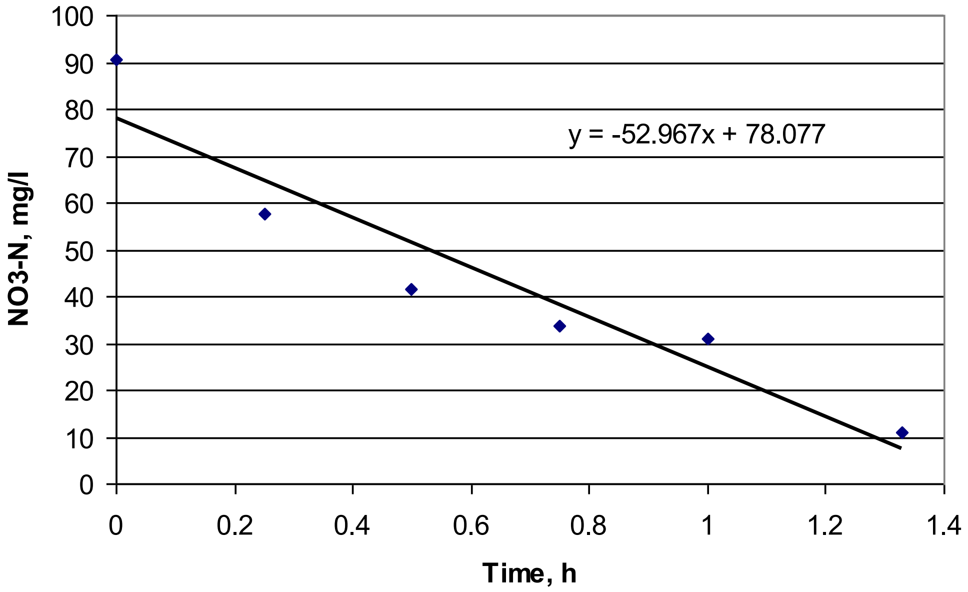


Figure 4.23. Denitrification rate with sucrose utilized as an external carbon source

The biomass growth rate was estimated from the changes of SS concentration in the reactor (Figure 4.24). The determined biomass growth rate is 3.2 times higher, than the growth rate of biomass with ethanol as a carbon source. Together with lower sucrose utilization for denitrification process it means that most of the carbon of sucrose participates in heterotrophic conversion and is included into biomass.

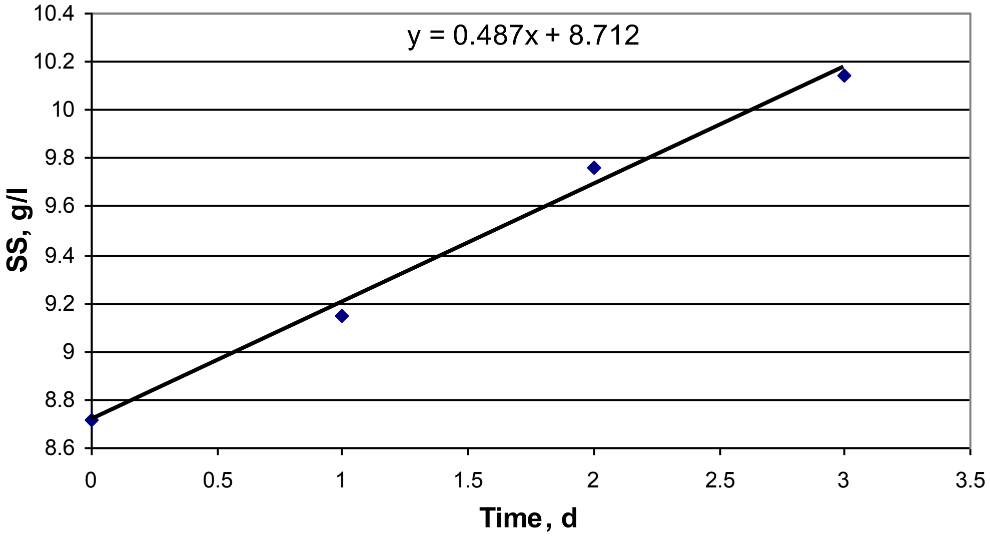


Figure 4.24 The increase of VSS concentration during the days 158 – 161 (Sludge was not removed)

The utilization of ethanol as a carbon source for SBR treatment process is advised basing on the results of experiments, as sucrose gives considerably lower denitrification rates and could cause several important operational problems, such as high growth rates of heterotrophic bacteria and foaming in the reactor. However, SBR process with sucrose addition has been running during comparatively short period of time, thus further researches may be required to evaluate adaptation of denitrifiers to the sucrose as a carbon source.

5 Laboratory experiments: Anammox process

The lab-scale Anammox reactor has been running during 20 days from 5th of February, 2009 until 25th of February, 2009. The experiment was planned to evaluate potential of Anammox process for nitrogen removal from leachate. The theory of anaerobic ammonium oxidation is presented in the section 2.2.2.

5.1 Laboratory installation

The simplified layout of the laboratory installation is present on the Figure 5.1.

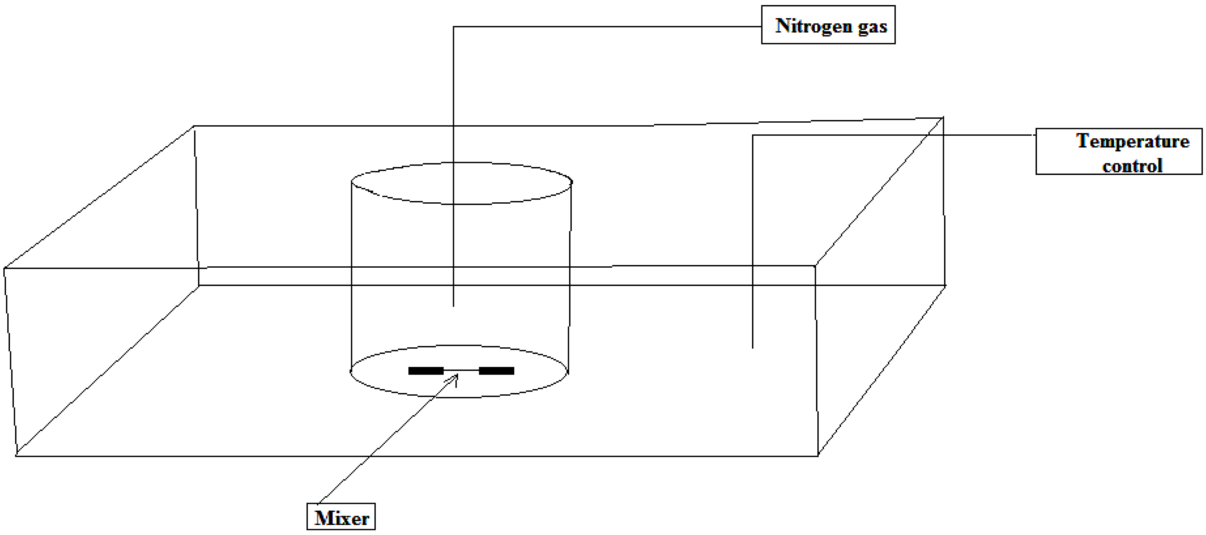


Figure 5.1 The layout of laboratory Anammox installation

The lab-scale installation consisted of the reactor, which was placed into the water bath. The reactor had the total volume of 1 l and the working volume of 0.8 l. Mixing was provided by magnetic stirrer. Elevated temperature (35°C) and strict anoxic conditions are mandatory for successful implementation of the process. The temperature was maintained by heating of water in the water bath, while anoxic conditions – by displacement of oxygen with nitrogen gas. Dissolved oxygen concentrations were kept below 0.2 mg/l.

Activated sludge for the process was taken from the full-scale Anammox reactor in Rotterdam (Netherlands) in October 2006 and was stored in the fridge at 7°C. Before start up of the process activated sludge contained 120 mg/l of NH_4-N; NO_2-N and NO_3-N concentrations were around 0 mg/l.

The reactor was fed with synthetic wastewater, prepared according to [van de Graaf, 1996] with minor changes. The composition of synthetic wastewater, used in the experiment, is given in the Table 5.1. pH of the synthetic wastewater was kept around 8.0. Nitrogen nitrate and ammonium sulphate were added to the reactor according to the stoichiometry of Anammox process (see equation 2.7). NO_2-N concentration in the feed didn't exceed 90 mg/l, as higher concentrations could be inhibitory for Anammox bacteria.

100 ml of synthetic wastewater were added once each 24 h. The appropriate volume of supernatant after 1 h settling was removed. NH_4-N, NO_2-N, NO_3-N and COD concentrations were measured in filtered samples using the analytical methods, listed in the section 4.3.1.

Dissolved oxygen concentration and pH were determined in the reactor using the methods, listed in the section 4.3.2.

Table 5.1 Composition of synthetic wastewater for Anammox process (adapted from van de Graaf et al., 1996)

Synthetic wastewater		***Trace elements solution***	
Reagent	***Concentration, g/l***	***Reagent***	***Concentration, g/l***
$NaNO_2$	Varied	EDTA	15
$(NH_4)_2SO_4$	Varied	$ZnSO_4·7H_2O$	0.43
$NaHCO_3$	1.04	$CaCl_2·6H_2O$	0.24
KH_2PO_4	0.025	$MnCl_2·4H_2O$	0.99
$CaCl_2·2H_2O$	0.3	$CuCl_2·2H_2O$	0.25
$MgSO_4·7H_2O$	0.2	$NaMoO_4·2H_2O$	0.22
$FeSO_4·7H_2O$	0.012	$NiCl_2·6H_2O$	0.19
EDTA	0.00625	H_3BO_3	0.014
Trace elements	1.25 ml/l		

5.2 Results and discussion

5.2.1 The course of the experiment

The amount of nitrogen in the feed decreased with the course of the experiment. At the beginning of the experiment synthetic wastewater which contained 90 mg/l of NO_2-N was added to the anammox sludge with 120 mg/l of NH_4-N. The addition of NO_2-N in appropriate stoichiometric concentration was not possible, as the inhibition of Anammox process at NO_2-N concentrations higher than 100 mg/l occurs. After decrease of initial NH_4-N and NO_2-N concentrations, the synthetic wastewater with appropriate concentrations of ammonium and nitrite nitrogen was added so that NO_2-N concentration did not exceed 60 mg/l. On the day 12 NH_4-N and NO_2-N addition was stopped due to the low conversion rates of ammonium and nitrite in the reactor.

pH in the reactor equaled to 9.0, which is an inhibitive level for Anammox process according to the literature [Schmidt, 2003]. pH was lowered by titration with 1M HCl down to 8.0, though previous pH level was restored in the reactor in spite of addition of acid.

5.2.2 Nitrogen reduction in the reactor

Reduction of NH_4-N and NO_2-N in the effluent was expressed in percents of reduced nitrogen from the initial concentration after synthetic wastewater addition (Figure 5.2). The reduction of NH_4-N was above 60% and NO_2-N – above 95% during first 7 days of the process running. The gradual decrease of NH_4-N removal rates was observed after the day 7, and NO_2-N removal rates – after the day 9. The reduction of nitrogen containing compounds during 24 h at the end of the process was lower than 20% from the initial concentration.

COD concentration in the reactor increased from 700 to 1350 mg/l during first 7 days of the process. On the day 8 all the supernatant with high COD concentration was removed and replaced with synthetic wastewater, which didn't contain organic compounds. The COD concentration in the reactor was lowered down to 400 mg/l in that way. Further decrease of COD was observed during last 10 days of the course of the process.

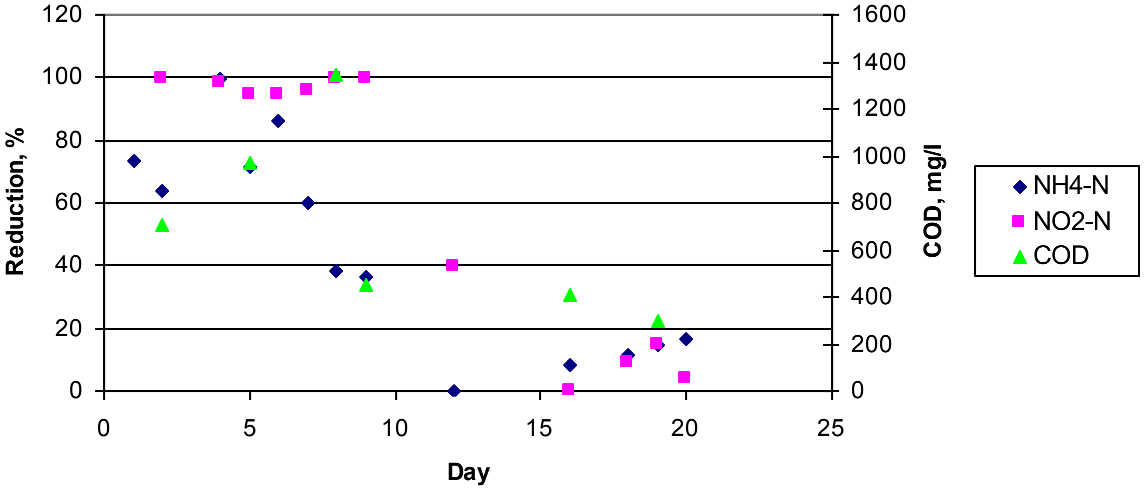


Figure 5.2 NH_4-N and NO_2-N reduction and COD concentrations in the reactor in the course of the process

The observed nitrogen reduction can't be explained by the action of Anammox bacteria. The possible reason for NH_4-N reduction in the reactor was stripping with nitrogen gas. Sparging of nitrogen gas was reduced during days 5 – 10 which was accompanied by decrease of NH_4-N reduction rates. High pH in the reactor also favored ammonia stripping.

NO_2-N reduction could be explained relative to the changes of COD concentration in the reactor. The increase of COD concentration in the supernatant at the beginning of the process happened most likely because of endogenic metabolism of bacterial culture. Organic compounds, released into the solution, were used as a carbon source by denitrifiers, which were present in the culture. Thus, NO_2-N reduction was caused by denitrification, instead of Anammox process. Directly after removal of organic-rich supernatant, reduction of NO_2-N ceased down to zero.

The difference in coloration of bacterial culture was also observed. The red color, typical for the culture of Anammox bacteria, has changed to the light-grey in the course of the process. Thus Anammox bacteria didn't meet competition with denitrifies that were also presented in the sludge and were outcompeted.

6 Designing of SBR leachate treatment facility

The data, collected during experimental phase and obtained from NSR AB allows preliminary calculation of the dimensions of the future SBR installation. The following calculations were done according to the guideline ATV-M 210, developed by German Association for the Water Environment (ATV) for construction of wastewater treatment plants based on the SBR technology [Teichmann, 2002; Teichgräber, 2001]. The basic principles of this guideline are derived in turn from the guideline ATV-A 131 for dimensioning of the activated sludge WWTP.

6.1 Input data, process configuration and parameters

The lab-scale SBR experiment, which was described in details in the chapter 4, allows choosing main design parameters for the SBR installation. 8-h cycle ($t_c = 8h$) with extended aeration phase ($t_n = 5$ h, $t_{dn} = 1.33$ h) is chosen as the operational cycle. The total reaction time (t_r) is 6.67 h, as the second aeration phase which lasts 0.33 h is also included.

The designed SBR process aims at the nitrogen removal below discharge limit (15 mg/l in the effluent). The process should be developed according to the assumptions, that leachate doesn’t contain any inhibitory substances and has sufficient pH and alkalinity for successful ongoing of combined nitrification and denitrification process without pH adjustment. The latter was proved during lab-scale experiment and by the analytical data, obtained at NSR landfill during the last three years.

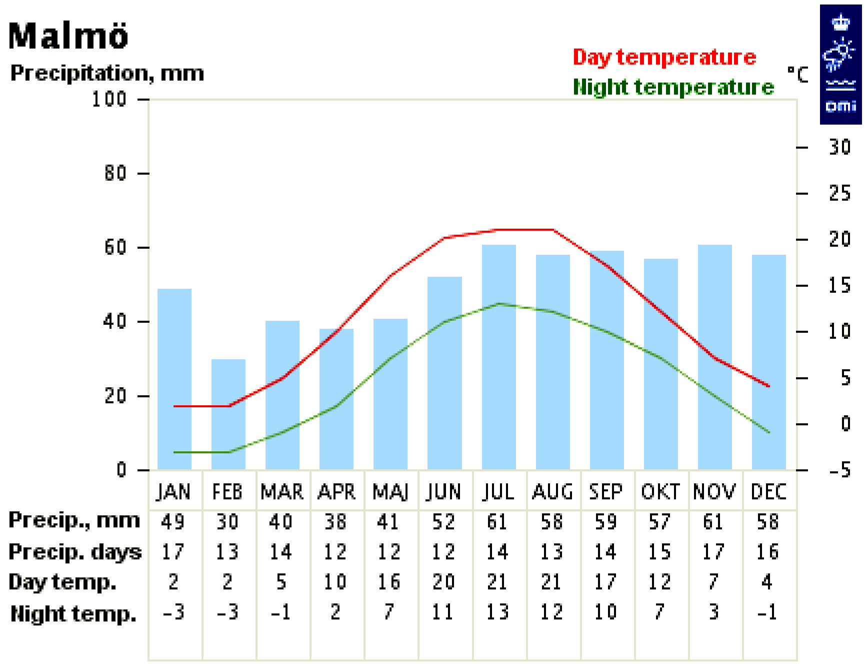
The annual volume of the leachate, collected at the Filborna landfill, is between 320 000 and 360 000 m^3. The annual volume (V_y) of 350 000 m^3 was chosen for further calculations. With this volume average daily flow will be the following:

$$Q_d = \frac{V_y}{365} = \frac{350000m^3}{365d} \approx 960m^3/d \qquad \text{(eq. 6.1)}$$

With t_c=8 h, the influent volume for one cycle (Q_c) will equal 320 m^3. The SBR is going to be placed instead of currently existed aerated pond. It means that buffer pond will be still served for equalizing of the influent and the constant volume of leachate entering the SBR could be assumed, without consideration of daily leachate influent variations.

The hydraulic retention time in the buffer pond is 27 days, thus, monthly variations of the influent, which depend mainly on the seasonal variations in precipitation, has to be considered. The annual precipitation data for Malmo, which is ca. 60 km south from Helsingborg, were taken for calculation of maximal influent to the SBR ($Q_{d,max}$). The annual precipitation in the district is 604 mm. The data on monthly precipitation are presented on the Figure 6.1. The largest precipitations (61 mm) are observed during July and November. Thus, the maximal daily flow during these months in the year with highest precipitation rates ($V_{y,max}$ = 360 000 m^3) will be equal to:

$$Q_{d,\max} = \frac{360000m^3 \times \frac{61mm}{604mm}}{30d} = 1200m^3/d \qquad \text{(eq. 6.2)}$$



	JAN	FEB	MAR	APR	MAJ	JUN	JUL	AUG	SEP	OKT	NOV	DEC
Precip., mm	49	30	40	38	41	52	61	58	59	57	61	58
Precip. days	17	13	14	12	12	12	14	13	14	15	17	16
Day temp.	2	2	5	10	16	20	21	21	17	12	7	4
Night temp.	-3	-3	-1	2	7	11	13	12	10	7	3	-1

Figure 6.1 Average monthly precipitations in course of one year [www.dmi.dk]

Two reactors will be operated in parallel because of the safety reasons. However, more reactors are not necessary for the facility of this size. The designing temperature is set to 10°C.

The maximal concentrations of suspended solids and nitrogen in the leachate, which were measured in the course of lab-scale experiments, is used for calculation of SS and nitrogen load. The BOD concentration in the influent could be assumed simply to be equal to the difference between COD concentrations in the SBR influent and effluent, which in its turn equals to the biological degradable organic matter. Biological oxidation of nitrogen compounds could be neglected in this approximation, as BOD values are used in the model for calculation of excess sludge production.

The following concentrations and loads are used in the model:

$C_{BOD,in} = 400$ mg/l → $L_{d,BOD,in} = C_{BOD,in} \times Q_d = 385$ kg/d
$X_{SS,in} = 310$ mg/l → $L_{d,SS,in} = X_{SS,in} \times Q_d = 300$ kg/d
$C_{N,in} = 160$ mg/l → $L_{d,N,in} = C_{N,in} \times Q_d = 152$ kg/d

The ratio of volume, which is required for denitrification to the total volume of aerated basin is equal to the ratio of anoxic (denitrification) phase duration and the difference between aerobic and anoxic phase durations in ATV-M210 model:

$$\frac{V_{dn}}{V_{aerated}} = \frac{t_{dn}}{t_n - t_{dn}} = \frac{1.33}{5-1.33} = 0.36 \qquad \text{(eq. 6.3)}$$

The key parameter for modeling of SBR according to ATV-M210 is sludge age (solid retention time, SRT). The calculations are derived from the ATV-DVWK-A 131 model for activated sludge facilities with combined nitrification and denitrification:

$$SRT = SF \times 3.4 \times 1.103^{15-T} \times \frac{1}{1-\frac{V_{dn}}{V_{aerated}}} = 1.8 \times 3.4 \times 1.103^{15-10} \times \frac{1}{1-0.36} = 15.6d \qquad \text{(eq. 6.4)}$$

where:

SF – safety factor (1.8 for the installations with BOD load less than 1200 kg/d);
T – designing temperature (10°C).

The production of excess sludge is also calculated according to ATV-DVWK-A 131 model for activated sludge facilities, basing on the influent BOD. The specific sludge production ($ES_{C,BOD}$) is derived from the following expression:

$$ES_{C,BOD} = 0.75 + 0.6 \times \frac{X_{SS,in}}{C_{BOD,in}} - \frac{0.8 \times 0.17 \times 0.75 \times SRT \times F_T}{1 + 0.17 \times SRT \times F_T} = 0.81 kg/kg \qquad \text{(eq. 6.5)}$$

where:

$X_{SS,in}$ – concentration of the suspended solids in the influent, mg/l;
$C_{BOD,in}$ – BOD concentration in the influent, mg/l;
SRT – solids retention time, d;
F_T – temperature factor, which is calculated according to the formula $F_T = 1.072^{(T-15)}$.
With T=10°C, F_T = 0.71.

Thus, the daily excess sludge production ($ES_{d,C}$)is obtained by multiplying specific sludge production and daily BOD load:

$$ES_{d,C} = L_{d,BOD,in} \times ES_{C,BOD} = 385 kg/d \times 0.81 kg/kg = 312 kg/d \qquad \text{(eq. 6.6)}$$

6.2 Hydraulic dimensioning

6.2.1 Determination of SBR volumes

From the ATV-DVWK-A 131 model the volume of aeration basin ($V_{aerated}$) is calculated from the sludge age and ratio of excess sludge production ($ES_{d,C}$) to the suspended solids concentration ($SS_{aerated}$) in the reactor. The suspended solids concentration in the reactor considered to be 5 g/l.

$$V_{aerated} = SRT \times ES_{d,C} / SS_{aerated} = 15.6 d \times \frac{312 kg/d}{5 kg/m^3} = 975 m^3 \qquad \text{(eq. 6.7)}$$

The volume of aerated basin, calculated for activated sludge facility according to ATV-DVWK-A 131 model is coupled to the volume of SBR facility (V_{SBR}) through the sludge age with the following relation:

$$SRT = \frac{nV_{SBR}SS_{SBR}}{ES_{d,C}} \times \frac{t_r}{t_c} \qquad \text{(eq. 6.8)}$$

or, using the eq. 6.8:

$$V_{aerated} \times SS_{aerated} = nV_{SBR}SS_{SBR} \times \frac{t_r}{t_c} \qquad \text{(eq. 6.9)}$$

where:

n – number of SBR operated in parallel (2 reactors in this particular case);
SS_{SBR} – concentration of suspended solids in the SBR, considered being 5 g/l;
t_r – reaction time, h;
t_c – duration of the cycle, h.

By rearranging eq. 6.9 the volume of one SBR reactor is calculated:

$$V_{SBR} = V_{aerated} \times SS_{aerated} / SS_{SBR} \times \frac{t_c}{t_r} \times \frac{1}{n} = 975m^3 \times \frac{5g/l}{5g/l} \times \frac{8h}{6.67h} \times \frac{1}{2} = 585m^3 \qquad \text{(eq. 6.10)}$$

The maximal exchanging volume is obtained according to eq. 6.11:

$$\Delta V_{\max} = Q_{d,\max} \times \frac{t_c}{n} = 50m^3/h \times \frac{8h}{2} = 200m^3 \qquad \text{(eq. 6.11)}$$

where:

Q – influent, m^3/h;
t_c – duration of the cycle, h;
n – number of SBR operated in parallel.

The hydraulic-associated calculations give actual volume, required for one SBR (in the installation with two parallel reactors):

$$V_{SBR} = \frac{\Delta V_{\max}}{f_{decant,\max}} = \frac{200m^3}{0.35} = 570m^3 \qquad \text{(eq. 6.12)}$$

$f_{decant,max}$ is a ratio of the influent volume (for one cycle) and total volume of the reactor at maximum filling. $f_{decant,max}$ is usually in the range between 0.2 and 0.5; The value 0.35 is chosen for these particular calculations.

The suspended solids concentration (SS_{SBR}) is acquired from the ratio between V_{SBR}, obtained from ATV-DVWK-A 131 model and hydraulic-associated calculations:

$$SS_{SBR} = SS_{SBR} \times \frac{V_{SBR}(eq.6.10)}{V_{SBR}(eq.6.12)} = 5g/l \times \frac{585m^3}{570m^3} = 5.13g/l \qquad \text{(eq. 6.13)}$$

The minimal volume after decanting stage is the following:

$$V_{\min} = V_{SBR} - \Delta V = 570m^3 - 200m^3 = 370m^3 \qquad \text{(eq. 6.14)}$$

6.2.2 Determination of SBR dimensions

Assuming the maximal water level in the reactor during reaction stages (h_w) being 5 m. The minimal water level after decanting ($h_{w,min}$) is:

$$h_{w,\min} = h_w \times (1 - f_{decant,\max}) = 5m \times (1 - 0.35) = 3.25m \qquad \text{(eq. 6.15)}$$

The sludge level in the reactor after settling (h_s) will be the following:

$$h_s = h_w \times SS_{SBR} \times \frac{SVI}{1000} = 5m \times 5.13g/l \times \frac{85ml/g}{1000} = 2.18m \qquad \text{(eq. 6.16)}$$

The distance between the minimal water level in the reactor after decanting and sludge level in the reactor after settling shouldn't be less than 10% of the total height of the reactor, which is 0.5 m:

$$h_{w,\min} - h_s = 3.25 - 2.18 = 1.07 \qquad \text{(eq. 6.17)}$$

As the value is bigger than 0.5, the dimensioning of the reactor for peak flow is correct.

The sludge settling velocity is defined according to the eq. 6.18:

$$v = \frac{q_{sv}}{SS_{SBR} \times SVI} = \frac{650 l/m^2 \cdot h}{5.13g/l \times 85ml/g} = 1.49m/h \qquad \text{(eq. 6.18)}$$

where q_{sv} is a specific sludge surface load, which is assumed to be 650 $l/m^2 \cdot h$

6.2.3 Verification of complete nitrogen removal in the designed SBR

With the hydraulic load HL = 0.15 l/g SS·cycle at 20°C (see section 4.5.1) and nitrification rates 6.2 mg NH_4-N/g VSS·h at 20°C and 1.9 mg NH_4-N/g VSS·h at 5°C (see section 4.9.3) it is possible to calculate volume requirements for complete nitrification, which is the rate-limiting step in biological nitrogen removal. The maximal possible nitrogen load on the process ($L_{c,N,max}$) considering designing value $C_{N,in}$ = 200 mg/l is:

$$\begin{aligned} L_{c,N,\max} &= HL \times C_{N,in} \times V_{SBR} \times SS_{SBR} = \\ &= 0.15m^3/kgSS \cdot cycle \times 0.2kg/m^3 \times 570m^3 \times 5.13kg/m^3 = 87.7kg/cycle \end{aligned} \qquad \text{(eq. 6.19)}$$

With the current influent for single SBR reactor ΔV=200 m^3 the actual nitrogen load for one SBR reactor is:

$$L_{c,N} = \Delta V \times C_{N,in} = 160m^3/cycle \times 0.2kg/m^3 = 32kg/cycle \qquad \text{(eq. 6.20)}$$

As $L_{c,N} < L_{c,N,max}$, the designed SBR facility can efficiently reduce nitrogen concentrations in the effluent down to the discharge requirements.

During the winter period significant drop of the temperature down to 5ºC can occur. According to the calculations, which were presented in the section 4.9.3, the nitrification rate will be 6.2/1.9 = 3.3 times lower. The $L_{c,N,max}$ will also decrease down to 87.7/3.3 = 26.6 kg NH_4-N/cycle. At this conditions $L_{c,N} > L_{c,N,max}$, thus, the actual nitrogen load should not exceed 25 kg/cycle The following increase of SBR volume for successful nitrogen removal during winter term may be required:

$$V_{SBR,new} = V_{SBR} \times \frac{L_{c,N}}{L_{c,N,new}} = 570m^3 \times \frac{32kg/cycle}{25kg/cycle} = 730m^3 \qquad \text{(eq. 6.21)}$$

However, in practice temperature effect is often overestimated while using theoretical calculations, thus, more precious estimations using experimental approach has to be accounted [Teichgräber, 2001]. Another ways to overcome temperature effect on the process during winter season could be the increase of the sludge age in the reactor or heating of the reactor (if cheap process water is available).

6.3 Removal of excess sludge

According to ATV-M210 model, sludge production (SP) in the SBR is calculated from the ratio of suspended solids concentration to the sludge age:

$$SP = V_{SBR} \times \frac{SS_{SBR}}{SRT} \times \frac{t_r}{24} = 570 m^3 \times \frac{5.13 kg/m^3}{15.6d} \times \frac{6.67h}{24h} = 52 kg/cycle \qquad \text{(eq. 6.22)}$$

where:

V_{SBR} – volume of the reactor;
SS_{SBR} – concentration of suspended solids in the reactor;
SRT – sludge age;
t_r – reaction time.

With two operated reactors and three complete cycles during one day daily sludge production is 306 kg/d. The density of sludge after sedimentation step is calculated from the sludge volumetric index:

$$SS_{ES} = \frac{1000}{SVI} = \frac{1000}{85} = 11.8 kg/m^3 \qquad \text{(eq. 6.23)}$$

With the density, obtained from eq. 6.23 and sludge production from eq. 6.22 volume of the excess sludge that has to be removed from one reactor per cycle is:

$$V_{ES} = \frac{SP}{SS_{ES}} = \frac{52 kg/cycle}{11.8 kg/m^3} = 4.4 m^3/cycle \qquad \text{(eq. 6.24)}$$

6.4 Oxygen and carbon requirements

The calculation of oxygen requirements in the combined nitrification/denitrification process in a sequencing batch reactor is done based on [Zilch, 2001].

The specific oxygen consumption for BOD oxidation (OC_C) depends on the sludge age and temperature. It is calculated according to the equation 6.25:

$$OC_C = 0.5 + \frac{0.144 \times SRT \times F_T}{1 + 0.08 \times SRT \times F_T} = 0.5 + \frac{0.144 \times 15.6 \times 0.71}{1 + 0.08 \times 15.6 \times 0.71} = 1.35 kgO_2/kgBOD \qquad \text{(eq. 6.25)}$$

where F_T is a temperature factor: $F_T = 1.072^{(T-15)}$=0.71.

The specific oxygen consumption for nitrification (OC_N) is calculated according to the eq. 6.26:

$$OC_N = \frac{4.6 \times (N_{NO3-out} - N_{NO3-in}) + 1.7 \times N_{NO3,D}}{C_{BOD,in}} = \\ = \frac{4.6 \times (0.42 mg/l - 0.1 mg/l) + 1.7 \times 160 mg/l}{400 mg/l} = 0.68 kgO_2/kgBOD \qquad \text{(eq. 6.26)}$$

where:

$N_{NO3\text{-}out}$ – nitrate concentration in the effluent, mg/l;
$N_{NO3\text{-}in}$ – nitrate concentration in the influent, mg/l;
$N_{NO3\text{-}D}$ – nitrate concentrations at the beginning of the denitrification stage, mg/l

From the values of specific oxygen consumptions multiplied by frequency factors f_C=1.15 and f_N=2.0 [Teichmann, 2002], the maximal hourly oxygen consumption (OC_h) is calculated:

$$OC_h = \frac{f_C \times OC_C + f_N \times OC_N}{24} \times L_{D,BOD,in} = \\ = \frac{1.15 \times 1.35 kgO_2/kgBOD + 2 \times 0.68 kgO_2/kgBOD}{24h} \times 385 kg/d = 46.7 kgO_2/h \qquad \text{(eq. 6.27)}$$

The oxygen supply requirements (OS) are calculated from the value of maximal hourly oxygen consumption:

$$OS = \frac{OC_h}{\alpha} \times \frac{Cs}{Cs - Cx} = \frac{46.7 kgO_2/h}{0.6} \times \frac{6.86}{6.86 - 2} = 110 kgO_2/h \qquad \text{(eq. 6.28)}$$

where:

α – oxygen supply factor, which is between 0.5 and 1.0 for aeration systems in wastewater treatment, α = 0.6 was considered for the calculations above;
C_x – oxygen concentration in the reactor, which has to be at least 2 mg/l;
Cs – oxygen saturation concentration, calculated according to the formula [www.ibfo.de, 2009]:

$$Cs = Css \times (1 - \frac{h_\varepsilon}{20.7}) = 9.02 mg/l \times (1 - \frac{4.9m}{20.7}) = 6.86 mg/l \qquad \text{(eq. 6.29)}$$

where:

Css – dissolved oxygen saturation value, Css = 9.02 mg/l at 20°C [Shun Dar Lin, 2007];
h_τ – aeration deepness, which is the distance between bottom aerator and water surface in the reactor, assumed to be 5 – 0.1 = 4.9 m in this case.

Ethanol addition for the denitrification step is calculated basing on the theoretical ethanol requirements for denitrification (3.26 g C_2H_5OH/ g of NO_3-N reduced):

$$F_{C_2H_5OH} = \frac{3.26 \times L_{c,N}}{\rho_{C_2H_5OH}} = \frac{3.26 kg/kg \times 32 kg/cycle}{809 kg/m^3} = 0.13 m^3/cycle \qquad \text{(eq. 6.30)}$$

6.5 Overview of the designed system

According to the calculations, presented in the sections 6.1 – 6.4, the summary of the SBR treatment system could be done. The main design features of the system are listed in the Table 6.1.

Table 6.1 Summary of the designed SBR treatment system

Parameter	***Value***
Cycle duration, h	8
Reaction time, h	6.67
Nitrification (aerobic) phase duration, h	5
Denitrification (anoxic) phase duration, h	1.33
SS in the reactor, g/l	5.13
SRT, d	15.6
Number of reactors	2
Volume of the single reactor, m^3	570
Maximal water height in the reactor (during reaction), m	5
Minimal water height in the reactor (after decanting), m	3.25
Sludge level in the reactor after settling, m	2.18
Excess sludge removal, m^3·reactor/cycle	4.4
Oxygen supply requirements, kg O_2/h	110
Type of carbon source	Ethanol
Carbon source supply, m^3/cycle	0.13

The proposed phases of the treatment cycle are presented on the Figure 6.2

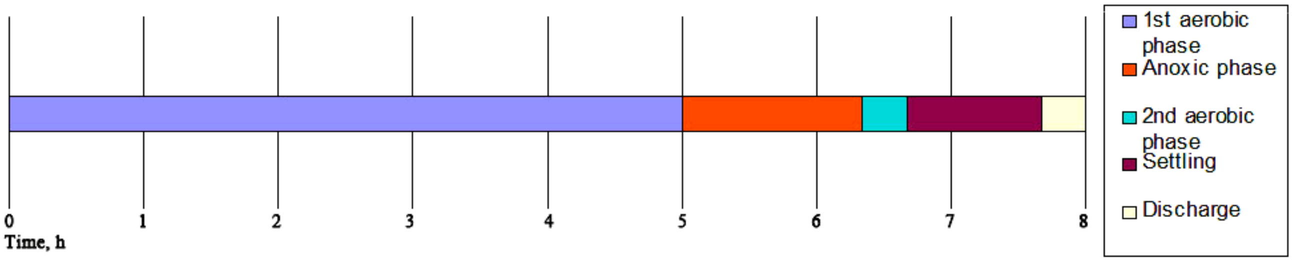


Figure 6.2 Duration of single SBR treatment cycle and separate phases

7 Conclusions

- Leachate, collected at Filborna landfill is currently treated on-site in the system of treatment ponds and withdrawn further to the Helsingborg municipal wastewater treatment plant. Current treatment system does not provide reduction of COD, nitrogen and phosphorous low enough to release effluent directly to the environment. Landfill operator is charged for withdrawal of the leachate to municipal WWTP.

- The existing Reco-N precipitation plant allows reduction of nitrogen in the effluent down to 30 mg/l, which minimize charging for additional nitrogen in the effluent to municipal WWTP. However, the system is not economically feasible as the estimated operational costs are higher than profits from nitrogen reduction with the current nitrogen concentration in the leachate.

- The lab-scale SBR reactor with combined nitrification/denitrification indicates 93% reduction of total nitrogen with the effluent values of 10 – 12 mg/l that are lower, than discharge requirements for the final recipient (Öresund).

- Two parallel SBR reactors with total working volume 1140 m^3 and cycle duration 8 h are advised to be placed after currently existing buffer pond to achieve necessary nitrogen elimination.

- COD and phosphorous concentrations in the effluent of the SBR are higher than the discharge requirements. Thus, additional removal steps are necessary for complete treatment of leachate before releasing to Öresund. Further evaluation of methods for decreasing of inert COD and phosphorous concentrations in the effluent is required.

8 Suggestions for further research

The following thesis project proves that SBR is a reliable technique for complete nitrogen removal from leachate. However, costs for external organic source for the denitrification step in SBR are quite high and can make the process economically unfeasible. Also SBR doesn’t provide efficient removal of phosphorous and inert organic material from leachate. Further researches, aiming at the development of the system for complete leachate treatment should be focused on the following issues:

- Utilization of less expensive carbon source for denitrification, e.g. molasses, could be tried in the lab-scale process to compare it’s effectiveness with ethanol ones. Molasses are the by-product of sugar beet processing industry, which is developed in Skåne. Taking into account their low price and availability, molasses could make the SBR process more profitable to run.

- Trial experiments on nitritation, which is the direct oxidation of ammonium into nitrogen gas through nitrite. This process allows saving of 25% of the energy and 40% of the organic matter [Åkerman. 2005]. However, the process conditions are stricter than for conventional nitrification/denitrification process, so potential of application of this process for nitrogen removal from leachate at the landfills in southern Sweden should be evaluated.

- Evaluation and comparison of advanced oxidation and activated carbon adsorption processes as the polishing steps for removal of inert organic material after treatment in SBR reactor.

- Development of reliable technique for phosphorous precipitation, either simultaneous precipitation in SBR or post-precipitation process. The impact of inert organics on the phosphorous precipitation and possibility of simultaneous precipitation of phosphorous with organic compounds has to be studied.

As the researches are devoted to the problem of leachate treatment in Skåne, information about leachate collection and treatment systems and leachate composition at all the landfills in the region should be collected in order to propose the best technology for complete leachate treatment. The similar climate conditions, environmental legislation and peculiarities of landfill operation by contractors will probably make it possible to apply the same strategy for leachate treatment in all the landfills in Skåne with minor changes according to the local conditions.

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Appendix A

Results of analyses of the effluent from the sequencing batch process

Table A.1 Concentration of NH_4-N, NO_3-N and NO_2-N in the effluent in the course of the process

Day	Hydraulic load, m^3/m^3·cycle	NH_4-N, mg/l	NO_3-N, mg/l	NO_2-N, mg/l	Day	Hydraulic load, m^3/m^3·cycle	NH_4-N, mg/l	NO_3-N, mg/l	NO_2-N, mg/l
2	0.07	1.02	n.m.	n.m.	**97**	1.33	56.1	2.32	n.m.
3	0.13	0.2	n.m.	n.m.	**98**	1	42.6	0.1	n.m.
6	0.2	0	n.m.	n.m.	**99**	1	57.9	0.3	n.m.
7	0.27	0	n.m.	n.m.	**101**	1	34.4	0.1	n.m.
8	0.33	0.164	n.m.	n.m.	**103**	0.1	38.6	0.05	n.m.
9	0.33	0.509	n.m.	n.m.	**105**	0.1	n.m.	n.m.	n.m.
10	0.33	0.195	134	0.096	**106**	0.1	0.14	n.m.	n.m.
14	0.33	0	131	0.113	**108**	0.2	0.01	n.m.	n.m.
15	0.33	n.m.	n.m.	n.m.	**110**	0.3	0.2	n.m.	n.m.
16	0.33	0.639	122	0.003	**111**	0.4	0.1	n.m.	n.m.
17	0.33	0.315	94	0.073	**113**	0.5	n.m.	n.m.	n.m.
20	0.33	0.534	54.5	0.15	**118**	0	n.m.	n.m.	n.m.
21	0.33	0.106	84.3	0.05	**119**	0	n.m.	n.m.	n.m.
22	0.33	0.177	23.7	0.024	**124**	0.1	0.66	174	n.m.
23	0.33	0	3.97	0.033	**125**	0.2	0.09	0.1	n.m.
24	0.33	0.263	1.69	0.035	**126**	0.3	0.11	0.02	n.m.
27	0.33	29.8	1.7	0.095	**130**	0.4	9.66	8.88	n.m.
28	0.33	0.283	1.97	n.m.	**131**	0.5	0.01	0.47	n.m.
31	0.33	0.13	1.86	n.m.	**133**	0.6	0.3	0.1	n.m.
34	0.33	0.391	3.56	n.m.	**134**	0.7	0.12	0.1	n.m.
35	0.5	0.356	1.92	n.m.	**135**	0.8	0.01	0.08	n.m.
36	0.5	0.23	2.15	n.m.	**138**	0.9	0.14	0.09	n.m.
38	0.5	1.04	2.07	0.29	**139**	1	0.1	2.04	n.m.
41	0.75	15.2	1.61	0.39	**140**	1	0.02	0.09	n.m.
42	0.5	0.276	0.1	0.266	**142**	1	0.09	0.05	0.055
75	0.25	0.923	0.13	n.m.	**144**	1.25	0.08	0.05	n.m.
76	0.5	19.4	0.3	0.234	**145**	1.25	0.08	0.13	n.m.
77	0.5	29.6	0.32	0.262	**146**	1.5	0.07	13.4	n.m.
83	0.5	0.171	n.m.	n.m.	**147**	1.5	0.157	0.17	n.m.
84	0.75	0.515	148.8	0.27	**150**	1.5	0.104	8.3	n.m.
85	0.75	0.602	7.59	n.m.	**153**	1.5	0.508	0.47	n.m.
86	0.75	0.09	0.03	0.08	**154**	1.5	0.191	10.1	n.m.
89	0.75	0.15	0.21	n.m.	**156**	1.5	0.242	10.8	n.m.
90	1	0.162	0.38	n.m.	**157**	1.5	0.101	9.9	n.m.
92	1	0.159	42.4	n.m.	**158**	1.5	0.424	19.4	n.m.
93	1	0.35	0.07	n.m.	**159**	1.5	1.99	0.044	n.m.
94	1.33	11.4	0.89	n.m.	**160**	1.5	0.04	0.42	n.m.
96	1.33	56	0.08	n.m.					

Table A.2 Results of the complete analyzes of the effluent performed on the selected days

Parameter	18.11.08	16.12.08	16.01.09	27.03.09
Total nitrogen (unfiltered), mg/l	153.8	9.65	12.9	14.2
Total nitrogen (filtered), mg/l	129.4	9.25	10.9	10.3
NH_4-N, mg/l	0	0.26	0.28	0.09
NO_3-N, mg/l	131	1.69	0.1	0.05
NO_2-N, mg/l	0.11	0.04	0.27	0.06
Total phosphorous (unfiltered), mg/l	5.22	7.89	1.47	9.66
Total phosphorous (filtered), mg/l	3.44	7.66	0.22	2.18
PO_4-P, mg/l	6.07	7.27	1.69	3.52
COD (unfiltered), mg/l	340	n.m.	345	400
COD (filtered), mg/l	312	323	340	345
SS, mg/l	58	22	28	108
VSS, mg/l	58	n.m.	n.m.	70
pH	8.5	8.4	8.3	8.4
Alkalinity, mmol HCO_3^-/l	17.4	16.4	14.2	15.4
Conductivity, mS/cm	3.8	3.3	3.2	4.3

Table A.3 COD and PO_4-P concentrations in the effluent (Days 10 – 85 of the process running)

Day	Hydraulic load, m^3/m^3·cycle	COD, mg/l	PO_4-P, mg/l
10	0.33	401	n.m.
14	0.33	312	6.07
15	0.33	n.m.	n.m.
16	0.33	2150	n.m.
17	0.33	2450	n.m.
20	0.33	527	n.m.
21	0.33	359	n.m.
22	0.33	484	6.44
23	0.33	397	12.4
24	0.33	323	7.27
27	0.33	446	16.9
28	0.33	386	1.34
31	0.33	312	4.37
34	0.33	334	1.59
35	0.5	n.m.	4.07
36	0.5	359	3.87
38	0.5	353	3.31
41	0.75	350	2.26
42	0.5	340	0.69
75	0.25	499	n.m.
76	0.5	415	0.9
77	0.5	398	0.64
83	0.5	406	1.66
84	0.75	n.m.	n.m.
85	0.75	408	1.04

Appendix B

Concentrations of contaminants in the raw leachate and in the leachate after treatment in a current system (data obtained from NSR AB)

Table B.1 Concentrations of BOD_7, SS, TOC and nutrients in the raw leachate

Date	BOD_7	NH_4-N	NO_3-N	N_{Tot}	P_{Tot}	SS	TOC
	mg/l	*mg/l*	*mg/l*	*mg/l*	*mg/l*	*mg/l*	*mg/l*
25 January 2006	520	160	9	220	7.7	370	500
30 March 2006	1200	430	7.3	610	7	140	1200
27 April 2006	1400	130	0	270	9.5	160	960
24 May 2006	1200	320	0	480	11	120	1000
29 June 2006	510	260	32	430	8.3	370	62
22 July 2006	840	260	26	450	7.3	250	850
29 August 2006	920	170	32	250	8.6	130	760
28 September 2006	580	110	0.24	360	6	200	780
26 October 2006	1000	120	0	240	5.7	270	560
30 November 2006	660	98	0	210	5.3	790	570
28 December 2006	3	190	13	250	8.2	280	650
26 February 2007	970	110	0.51	360	11	300	640
29 March 2007	1100	83	0	240	10	450	800
26 April 2007	1200	280	0	410	12	370	970
31 May 2007	1600	150	0	430	22	340	600
28 June 2007	1100	230	0	440	13	350	500
26 July 2007	790	120	0	250	10	750	650
30 August 2007	1300	230	0	400	17	400	810
27 September 2007	1600	210	0	370	14	480	240
18 October 2007	n.m.	n.m.	n.m.	340	11	n.m.	n.m.
25 October 2007	1600	290	0	560	15	800	710
01 November 2007	n.m.	270	n.m.	500	10	n.m.	n.m.
08 November 2007	n.m.	230	n.m.	370	15	n.m.	n.m.
29 November 2007	1400	110	0	450	15	720	980
27 December 2007	600	100	0	330	8	6.5	440
31 January 2008	710	190	4.4	310	5.2	130	480
28 February 2008	250	76	0	280	7.9	n.m.	380
27 March 2008	370	110	0	220	4.9	n.m.	380
29 April 2008	330	290	0.21	350	5.9	n.m.	3200
29 May 2008	380	360	0	520	5.6	200	3100
26 June 2008	740	310	0	500	16	1600	820
31 July 2008	970	220	2.8	450	8.6	530	770
28 August 2008	490	99	0	150	6.2	120	210
23 September 2008	880	140	0	320	9.1	330	570
30 October 2008	290	53	0.3	200	4.5	190	270
27 November 2008	580	87	0	250	6.5	760	550
Mean value 06’-08’	**851**	**188**	**3.8**	**355**	**10**	**397**	**787**
Mean value for 08’	**545**	**176**	**0.64**	**323**	**7.3**	**483**	**975**
Discharge requirements*	**10**	**-**	**-**	**15**	**0.5**	**-**	**100**

* - Discharge requirements for Öresund

Table B.2 Concentrations of heavy metals, Cl^- and SO_4^{2-} in the raw leachate

Date	As	Pb	Fe	Cd	Cl^-	Cu	Cr	Hg	Ni	Sn	SO_4^{2-}	Zn
	ug/l	*ug/l*	*mg/l*	*ug/l*	*mg/l*	*ug/l*	*ug/l*	*ug/l*	*ug/l*	*ug/l*	*mg/l*	*mg/l*
25 January 2006	37	19	6.7	0.23	600	34	60	0	57	n.m.	530	1.4
30 March 2006	41	25	10	0.92	1200	35	120	0	120	n.m.	400	2.7
27 April 2006	24	52	23	2.3	740	140	130	0	280	n.m.	500	2.8
24 May 2006	39	26	10	1	1200	39	82	0	93	n.m.	670	4.2
29 June 2006	33	29	11	0.86	800	34	85	0	77	n.m.	280	5.8
22 July 2006	37	210	13	1.8	750	93	72	0.13	100	n.m.	500	50
29 August 2006	39	120	14	2.3	550	140	86	0.11	110	n.m.	330	7
28 September 2006	18	12	6.6	0.46	300	7.5	89	0	70	n.m.	85	2.5
26 October 2006	27	46	8.3	0.66	700	46	130	0	70	n.m.	310	3.4
30 November 2006	19	47	11	0.82	360	73	61	0	74	n.m.	310	2.4
28 December 2006	25	21	8.8	0.49	670	53	63	0	81	n.m.	620	1.9
26 February 2007	24	19	8.8	0.55	410	27	64	0	68	8.2	280	2.5
29 March 2007	22	23	9	0.72	290	51	57	0.12	71	7.6	170	2.3
26 April 2007	23	21	14	0.73	900	40	81	0	74	12	370	1.9
31 May 2007	32	38	13	1.2	480	56	120	0.17	95	17	220	3.4
28 June 2007	25	35	14	0.84	750	65	72	0	75	11	250	1.5
26 July 2007	16	36	13	0.82	400	61	52	0.76	72	7.4	310	1.4
30 August 2007	20	22	11	0.71	730	38	76	0	72	10	370	2.4
27 September 2007	20	27	9.3	0.78	720	56	67	0.2	65	11	220	1.8
18 October 2007	11	11	n.m.	0.63	n.m.	16	62	n.m.	67	n.m.	n.m.	1.4
25 October 2007	18	17	8.2	0.65	700	31	76	0	77	13	210	1.4
01 November 2007	20	23	n.m.	0.65	n.m.	33	130	n.m.	87	18	n.m.	1.3
08 November 2007	21	38	n.m.	1.4	n.m.	89	98	n.m.	84	12	n.m.	1.5
29 November 2007	19	24	12	0.84	410	47	85	0	83	15	140	1.6
27 December 2007	13	14	9.4	0.3	530	8.1	56	0	67	3.7	200	1
31 January 2008	18	43	16	0.64	740	70	69	0	73	10	400	1.4
28 February 2008	23	110	18	1.5	290	120	79	0.17	67	14	140	1.4
27 March 2008	17	48	10	0.58	420	59	53	0	55	8	320	0.49
29 April 2008	23	10	8.9	0.29	960	21	92	0	64	12	660	0.36
29 May 2008	26	19	11	0.59	1200	24	120	0	68	16	580	0.41
26 June 2008	23	49	17	1.2	1300	46	120	0.55	78	15	440	1.7
31 July 2008	21	15	10	0.34	1300	20	86	0	67	11	400	0.69
28 August 2008	12	19	8.1	0.39	420	38	32	0	49	4	320	0.33
23 September 2008	17	7.5	9.1	0.2	520	20	43	0	48	5.6	210	0.11
30 October 2008	15	38	9.9	0.47	200	48	45	0	51	6.7	110	0.26
27 November 2008	16	22	11	0.32	360	43	48	0	47	7	200	0.2
Mean 06'-08'	**23**	**37**	**11**	**0.81**	**664**	**51**	**79**	**0.07**	**79**	**11**	**335**	**3.2**
Mean 08'	**19**	**35**	**12**	**0.59**	**701**	**46**	**72**	**0.08**	**61**	**10**	**344**	**0.67**
Discharge requirements*	**10**	**50**	**-**	**1**	**-**	**500**	**50**	**1**	**500**	**-**	**-**	**500**

* - Discharge requirements for Öresund

Table B.3 Concentrations of BOD$_7$, SS, TOC and nutrients in the effluent from the buffer pond

Date	**BOD$_7$**	**NH$_4$-N**	**NO$_3$-N**	**N$_{Tot}$**	**P$_{Tot}$**	**SS**	**TOC**
	mg/l	***mg/l***	***mg/l***	***mg/l***	***mg/l***	***mg/l***	***mg/l***
22 February 2006	300	250	0	350	6.4	110	570
24 April 2006	330	180	0.72	270	5	n.m.	31
28 August 2006	160	170	0	190	3.9	82	210
29 November 2006	220	43	0	140	3.9	86	300
26 February 2007	490	150	0	190	5.9	160	400
26 April 2007	470	170	0	210	5.8	210	300
28 August 2007	140	88	0	160	7.8	380	180
18 October 2007	n.m.	150	n.m.	310	6.6	n.m.	n.m.
25 October 2007	n.m.	160	n.m.	310	8	n.m.	n.m.
01 November 2007	n.m.	150	n.m.	320	11	n.m.	n.m.
08 November 2007	n.m.	160	n.m.	320	9.8	n.m.	n.m.
29 November 2007	150	170	0	280	7.8	310	270
27 February 2008	42	160	0	210	4	60	190
29 April 2008	20	170	0	260	3.5	60	170
27 August 2008	170	110	0	190	5.4	120	210
26 November 2008	120	120	0	170	3.4	170	190
Mean value 06’-08’	**218**	**150**	**0.06**	**243**	**6.1**	**159**	**252**
Mean value for 08’	**88**	**140**	**0**	**207.5**	**4.1**	**102.5**	**190**

Table B.4 Concentrations of BOD$_7$, SS, TOC and nutrients in the effluent from the aerated pond

Date	**BOD$_7$**	**NH$_4$-N**	**NO$_3$-N**	**N$_{Tot}$**	**P$_{Tot}$**	**SS**	**TOC**
	mg/l	***mg/l***	***mg/l***	***mg/l***	***mg/l***	***mg/l***	***mg/l***
22 February 2006	330	250	0	330	6.3	160	560
24 April 2006	260	180	0.66	290	5.1	n.m.	280
28 August 2006	78	170	0	180	3.4	95	160
29 November 2006	120	20	0	150	3.5	98	270
26 February 2007	310	130	0	200	6	120	360
26 April 2007	390	170	0	180	5.5	160	350
28 August 2007	170	88	0	150	4.2	92	230
18 October 2007	n.m.	130	n.m.	230	5.4	n.m.	n.m.
25 October 2007	n.m.	140	n.m.	210	5.8	n.m.	n.m.
01 November 2007	n.m.	140	n.m.	270	7.8	n.m.	n.m.
08 November 2007	n.m.	160	n.m.	290	9.1	n.m.	n.m.
29 November 2007	130	160	0	290	7.3	220	240
27 February 2008	27	150	0.13	240	4.1	76	160
29 April 2008	14	150	0.18	240	3.6	51	130
27 August 2008	35	74	0.75	130	4.1	91	120
26 November 2008	50	120	0.2	160	2.7	77	130
Mean value 06’-08’	**160**	**140**	**0.16**	**221**	**5.2**	**113**	**249**
Mean value for 08’	31.5	123.5	0.32	192.5	3.6	73.8	135

Table B.5 Concentrations of BOD$_7$, SS, TOC and nutrients in the effluent, discharged to the Helsingborg municipal WWTP

Date	BOD$_7$ *mg/l*	NH$_4$-N *mg/l*	NO$_3$-N *mg/l*	N$_{Tot}$ *mg/l*	P$_{Tot}$ *mg/l*	SS *mg/l*	TOC *mg/l*
20 February 2006	270	280	0	480	3.5	n.m.	660
30 March 2006	290	250	0.12	340	6.5	76	560
27 April 2006	300	97	0.18	230	4.9	n.m.	270
24 May 2006	110	160	0	220	4	n.m.	230
29 June 2006	35	96	0	180	3.6	n.m.	330
22 July 2006	120	61	0	120	4.7	n.m.	570
29 August 2006	49	66	0	110	3	46	180
28 September 2006	44	60	0	130	2.6	45	210
26 October 2006	33	110	0	110	3	65	160
30 November 2006	100	84	0	150	3.7	83	180
28 December 2006	110	110	0	180	3.6	45	200
25 January 2007	110	100	0	150	3.6	53	200
26 February 2007	220	42	0	160	3.9	72	260
29 March 2007	430	99	0	170	8.1	55	510
26 April 2007	310	73	0	150	5.4	100	320
31 May 2007	270	74	0	160	7.6	130	250
28 June 2007	310	140	0	200	5.9	130	240
26 July 2007	300	13	0	54	6	110	350
30 August 2007	140	86	0	120	6.9	110	200
27 September 2007	88	66	0	140	4.6	160	120
25 October 2007	110	76	0	190	6	200	130
29 November 2007	96	100	0	240	6	210	170
27 December 2007	68	65	0	270	7.1	150	220
31 January 2008	51	140	0	240	5.4	57	170
28 February 2008	26	56	0	230	4.3	9.1	180
27 March 2008	19	130	0.11	200	3.7	48	140
29 May 2008	34	79	1.8	130	3.4	220	140
26 June 2008	29	59	1.5	110	2.8	n.m.	1700
31 July 2008	20	24	0.74	120	3.6	n.m.	130
28 August 2008	30	39	23	110	3	n.m.	130
23 September 2008	29	29	3.9	100	3.1	n.m.	100
30 October 2008	11	32	0.86	130	3.7	n.m.	110
26 November 2008	15	68	0.67	170	3.4	n.m.	110
Mean value 06'-08'	**127**	**90**	**0.93**	**176**	**4.6**	**99**	**286**
Mean value for 08'	**26**	**66**	**3.2**	**154**	**3.6**	**84**	**291**
Discharge requirements*	**10**	**-**	**-**	**15**	**0.5**	**-**	**100**

* - Discharge requirements for Öresund

Table B.6 Concentrations of heavy metals, Cl^- and SO_4^{2-} in the effluent, discharged to the Helsingborg municipal WWTP

Date	As	Pb	Fe	Cd	Cl	Cu	Cr	Hg	Ni	Sn	SO_4	Zn
	ug/l	*ug/l*	*mg/l*	*ug/l*	*mg/l*	*ug/l*	*ug/l*	*ug/l*	*ug/l*	*ug/l*	*mg/l*	*mg/l*
20 February 2006	32	12	4.1	0.44	850	21	84	n.m.	86	n.m.	8.9	1
30 March 2006	28	15	4.4	0.44	750	26	71	0	79	n.m.	130	1
27 April 2006	28	12	4.3	0.44	310	31	60	n.m.	73	n.m.	58	0.81
24 May 2006	22	9	3.2	0.33	730	22	52	n.m.	78	n.m.	130	0.46
29 June 2006	17	5.6	1.8	0.21	640	16	48	n.m.	98	n.m.	42	0.29
22 July 2006	18	2.5	1.3	0.15	910	12	50	n.m.	90	n.m.	33	0.16
29 August 2006	17	3	1.2	0.15	610	11	47	0	66	n.m.	60	0.34
28 September 2006	14	3	1.3	0.21	420	11	48	0	68	n.m.	37	0.32
26 October 2006	12	3.2	1.6	0.14	630	11	43	0	62	n.m.	68	0.32
30 November 2006	12	7.7	2.5	0.2	410	26	40	0	55	n.m.	54	0.67
28 December 2006	10	8.7	2.8	0.26	450	33	36	0	54	n.m.	130	0.7
25 January 2007	15	12	3.4	0	420	29	40	0	59	4.2	170	0.73
26 February 2007	18	10	4.3	0.28	160	32	35	0	52	4	72	0.87
29 March 2007	21	15	6	0.42	360	36	43	0	58	5.5	140	1.7
26 April 2007	16	6.8	3.2	0.22	310	15	31	0	46	3.5	3.5	0.65
31 May 2007	16	4.7	2.2	0.21	340	12	41	0	50	3.6	2.6	0.42
28 June 2007	16	3.6	1.7	0.14	730	11	48	0	52	3.2	12	0.33
26 July 2007	12	2.7	1.6	0	520	12	30	0	42	2.4	3.7	0.19
30 August 2007	9.5	2.3	1.2	0.1	420	11	23	0	43	1.6	16	0.12
27 September 2007	8.1	2.1	1.2	0	370	12	24	0	42	1.7	42	0.13
25 October 2007	7.7	2	1.1	0	330	12	27	0	40	2.2	59	0.15
29 November 2007	8.5	2.8	1.6	0.12	420	11	35	0	48	3.2	110	0.22
27 December 2007	8	5.6	2.3	0.15	280	13	35	0	48	2.6	73	0.32
31 January 2008	7.3	6.5	2.5	0.16	560	18	34	0	47	4.3	220	0.31
28 February 2008	9.3	7.6	2.5	0.12	210	14	40	0	52	4.3	83	0.33
27 March 2008	6.5	5.4	2	0.13	540	12	33	0	53	3.5	290	0.25
29 May 2008	8.9	3.5	2.2	0.12	600	9.7	30	0	49	2.5	340	0.13
26 June 2008	12	2.6	1.5	0	710	12	28	0	52	1.8	400	0.08
31 July 2008	12	3.4	2	0	390	12	28	0	56	1.7	170	0.08
28 August 2008	7.8	1.6	0.83	0	750	11	19	0	52	0.93	390	0.06
23 September 2008	7.5	0.94	0.81	0	280	12	18	0	44	1.2	130	0.04
30 October 2008	7.0	0.92	1	0	180	8.2	21	0	40	1.3	62	0.03
26 November 2008	9.0	3.2	1.6	0	330	11	25	0	40	2.5	110	0.04
Mean 06'-08'	**14**	**5.7**	**2.3**	**0.13**	**482**	**17**	**38**	**0**	**57**	**2.8**	**335**	**0.4**
Mean 08'	**8.7**	**3.6**	**1.7**	**0.05**	**455**	**12**	**28**	**0**	**49**	**2.4**	**344**	**0.13**
Discharge requirements*	**10**	**50**	**-**	**1**	**-**	**500**	**50**	**1**	**500**	**-**	**-**	**500**

* - Discharge requirements for Öresund

Appendix C

Leachate treatment at Filborna landfill with focus on nitrogen removal

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Abstract

A lab-scale Sequencing Batch Reactor (SBR) with combined nitrification and denitrification was run for 160 days to evaluate the possibility of treatment of the leachate from Filborna Landfill (Helsingborg, Sweden). The influent leachate contained high concentrations of ammonium (160 mg/l) and COD (580 mg/l). At 20ºC the 93 % removal of total nitrogen, respective 99 % removal of ammonium nitrogen was achieved. Oxygen concentrations were about 0.1 mg/l at anoxic phases and above 2 mg/l at aerobic phases. The 93 % removal of total nitrogen, respective 99 % removal of ammonium nitrogen was achieved. However the process was insufficient for removal of organic substances, and the latter are supposed to be present in slowly biodegradable forms.

Keywords: leachate, sequencing batch reactor, nitrification, denitrification,

Introduction

Leachate is the liquid that drains through a waste. It consists of the internal moisture of the waste and precipitations that fall on a landfill. Leachate will be an important source of soil and groundwater contamination for the next several decades, even if landfilling of organic waste is banned. Leachate contains a mixture of heavy metals, large quantities of nitrogen (organic nitrogen and ammonium), inert slowly degradable organic substances (mostly fulvic and humic acids) and phosphorous [Renou, 2008]. Withdrawn directly to the environment, nitrogen contaminated liquid causes eutrophication and the subsequent oxygen depletion. Nitrogen is an especially important source of eutrophication of saline water basins (seas and oceans) as it is the limiting nutrient for marine algae. Thus, regions that withdraw wastewater to a sea (as the province of Skåne does) should put special concern on the nitrogen removal from different types of wastewater.

Today the most widespread solution in leachate handling for landfill operators in Sweden is to withdraw leachate directly or after pretreatment to municipal wastewater treatment plants (WWTP) [Naturvårdsverket. Fakta: Lakvatten från deponier, 2008]. However municipal WWTPs charge landfill operators both for the amount of received leachate and for the high concentrations of certain contaminants in it. The leachate itself could create problems in operation of municipal WWTPs if the percentage of leachate in the inflow is higher than 0.5 % by volume [Anhert, 1992]. Thus the development of reliable and efficient systems for leachate treatment at landfill sites is required.

The present examination is focused on the possible use of a SBR with combined nitrification and denitrification as an on-site treatment system of leachate at Filborna landfill, which is situated in the province of Skåne in southern Sweden. Leachate, collected at Filborna landfill is currently treated in a system of ponds and withdrawn further to the Helsingborg municipal WWTP. Current on-site treatment system does not provide sufficient reduction of organic compounds, nitrogen, and phosphorous for direct release to the environment (Table 1).

The landfill operator is charged for withdrawal of the leachate to the municipal WWTP. The effluent nitrogen concentrations (159 mg/l) exceed three times the maximal concentration allowed to be withdrawn without additional payment (52 mg/l).

Table 1 Influent and effluent concentrations of organic compounds, suspended solids, nitrogen and phosphorous in the leachate collected at Filborna landfill comparatively with the discharge requirements for the Oresund region

Parameter	Concentration in the influent, mg/l*	Concentration in the effluent, mg/l*	Discharge requirements, mg/l	Removal of the contaminant, %
BOD_7	884	102	10	89
SS	388	94	-	76
NH_4-N	192	76	-	60
N-tot	355	159	15	55
P-tot	9.9	4.5	0.5	55

To decrease concentration of nitrogen in the effluent, a plant for struvite precipitation was installed at the site in 2006. It’s principle of operation is based on the chemical precipitation of ammonium in the form of magnesium ammonium phosphate (struvite). The precipitation plant allows reduction of nitrogen in the effluent down to 30 mg/l, which removes charging for additional nitrogen in the effluent to the municipal WWTP. However, the system is not economically feasible with the current nitrogen concentration in the leachate as the estimated operational costs are higher than costs for nitrogen reduction by the municipal WWTP.

To solve the problem, a simple and reliable system for leachate treatment at Filborna landfill must be proposed. The main focus has to be put on nitrogen removal. A SBR with a combined nitrification/denitrification process has already been proved to be an efficient way to treat leachate [Heander, 2007]. The following study includes lab-scale trials of a SBR system aimed at the adjustment of the process for leachate treatment, evaluation of its efficiency, and determination of main operational parameters for the design of on-site installation.

Materials and methods

The layout of the lab-scale reactor is shown as Figure 1. The leachate was pumped at the beginning of each cycle to the reactor with a total volume 5 l and working volume 4 l. Nitrification was facilitated by aeration in the extended aerobic phase which lasted 5 h. Denitrification (anoxic) phase with duration of 1.33 h followed the aerobic phase. Ethanol was used as a carbon source for the denitrification. It was pumped in the reactor at the beginning of the anoxic phase. The second aeration phase with duration of 0.33 h was introduced for oxidation of organic matter which could be left after the denitrification phase. The sedimentation phase lasted 1 h and decanting phase 0.33 h.

The reactor was operated at 20ºC during 160 days. The analyses of total nitrogen, ammonium, nitrate and nitrite nitrogen, COD, total phosphorous and phosphorous of orthophosphates were performed in the influent and effluent liquid using Dr. LANGE cuvette tests. Suspended solids (SS) and volatile suspended solids (VSS) were measured according to the standard SS-EN 872-1, pH – with *WTW pH 320* pH-meter, dissolved oxygen – with oxygen analyzers *WTW Oxi 197S* and *WTW inoLab Oxi 730.* These parameters were measured in the influent and effluent as well as in the reactor.

Nitrification and denitrification rates were measured by analyzing ammonium nitrogen (for nitrification) and nitrate nitrogen (for denitrification) in the samples derived from the reactor at certain time intervals. The increase of SS concentration vs time was measured to determine the growth rate of bacteria.

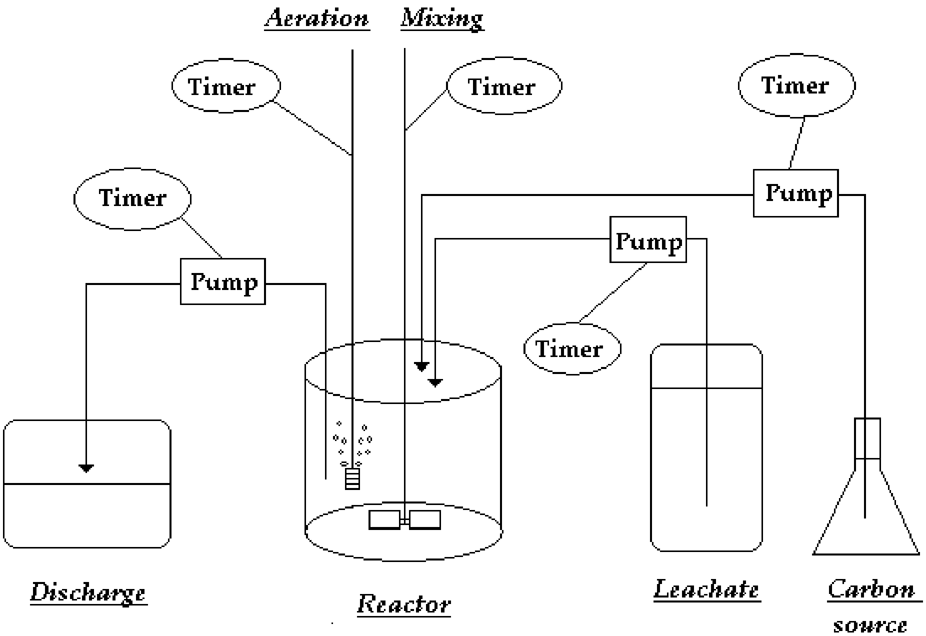

Figure 1 The layout of laboratory SBR installation

Results and discussion

Ammonium, nitrate and nitrite nitrogen concentrations in the effluent were about 0.1 mg/l as the experiment progressed (see Figure 2).

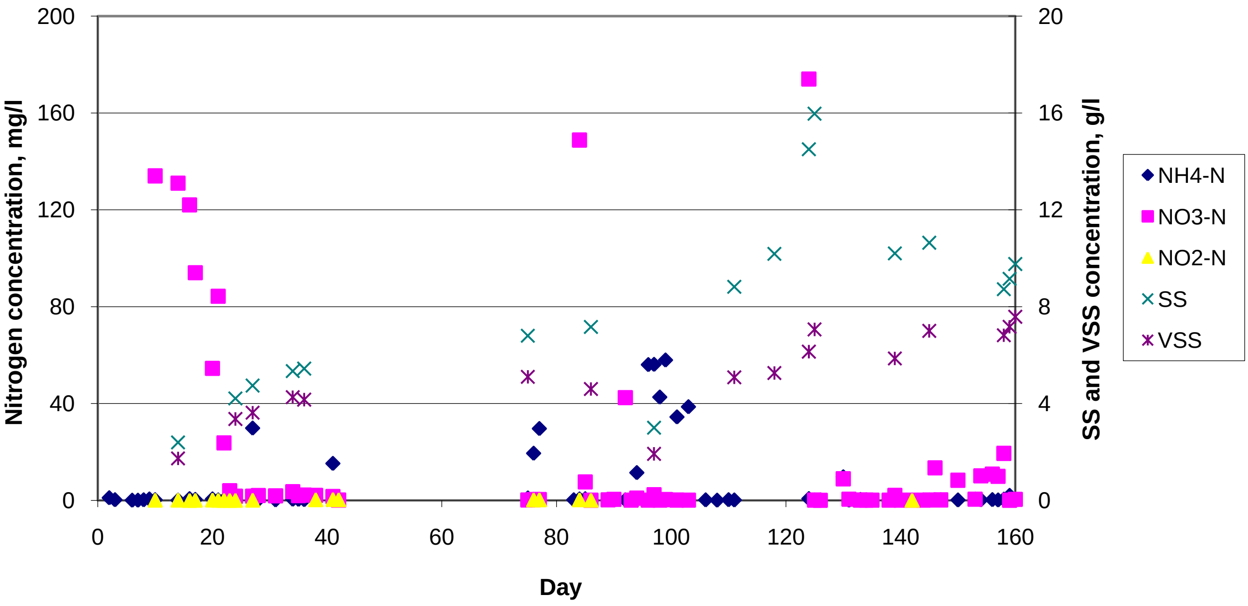

Figure 2 NH_4-N, NO_3-N and NO_2-N concentrations in the effluent in the course of the experiment

The few exceptions in Figure 2 refer to the start up of the denitrification process during the days 14 – 22 and the number of operational problems of a rather technical character: insufficient oxygen supply due to the aerator clogging, insufficient carbon source addition, cut of electricity supply etc. Thus the combined nitrification/denitrification process was generally successful in the lab-scale trials and could be applied for almost complete removal of the inorganic nitrogen from leachate at Filborna landfill.

SS and VSS concentrations increased as the experiment progressed (Figure 2). The highest SS concentration (≈ 16 g/l) was observed on the day 125. VSS/SS ratio of 0.75 – 0.8 was observed during the first phase of the experiment (days 14 – 42). After the day 75 this ratio decreased down to 0.4 – 0.5, which is typical for activated sludge with a high sludge age [von Sperling, 2007]. The temperature in the reactor was 20ºC, pH was in the range from 7.9 to 8.5 and dissolved oxygen concentrations around 0.1 mg/l at anoxic phases and above 2 mg/l – at aerobic phases

The growth rate of 0.15 g/l·d was found for the SBR sludge with combined nitrification/ denitrification process (Figure 3).

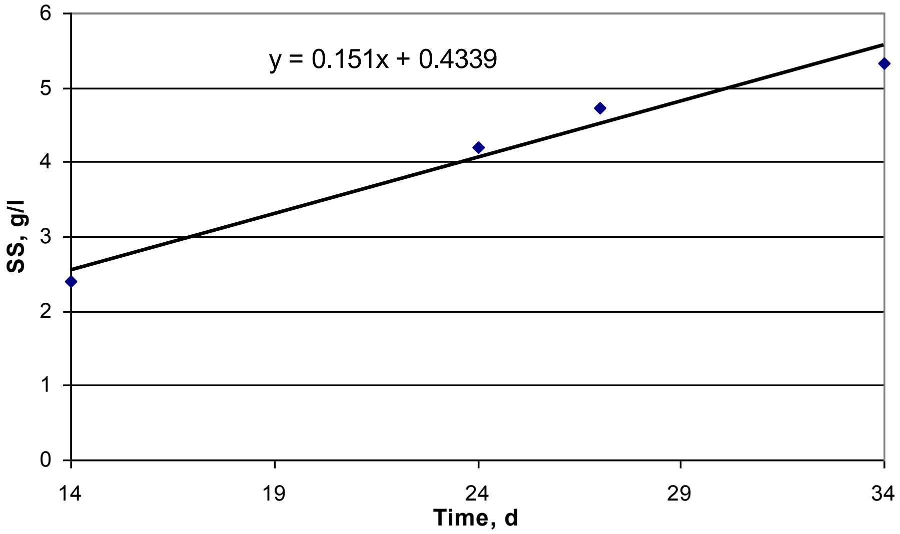

Figure 3 Graphical presentation of bacterial growth rate

Nitrification and denitrification rates were measured twice in the course of the experiment – on the days 28 and 142 (Table 2). At the first measurement the VSS concentration in the reactor was half of the concentration of the second. The volumetric rates were in comparative ranges for both measurements, while rates calculated in terms of biomass weight were significantly lower at higher VSS concentrations, meaning that efficiency of biomass utilization both for nitrification and denitrification decreased with the increase of biomass concentration. Denitrification rate was almost 8 times higher, than nitrification rate.

Table 2 Nitrification and denitrification rates of the sludge in SBR process

	VSS, g/l	**Nitrification rate**		**Denitrification rate**	
		mg NH_4-N/l·h	mg NH_4-N/g VSS·h	mg NO_3-N/l·h	mg NO_3-N/g VSS·h
Day 28	3.7	23.6	6.2	178.8	48.3
Day 142	6.43	23	3.6	184.5	28.7

The removal of nitrogen in the combined nitrification/denitrification SBR process was efficient, since the total nitrogen concentrations decreased 93 % and ammonium nitrogen concentrations 99 %. However the process was inefficient in terms of organics and phosphorous removal – only 37% of COD and 20% of phosphorous were removed (Table 3).

The fractions of organic matter measured as COD and nitrogen in the raw leachate were determined (Figure 4).

Table 3 Removal efficiency of the SBR process

Parameter	Concentration in the influent, mg/l*	Concentration in the effluent, mg/l*	Reduction, %
N-tot	163	12.3	93
NH_4-N	131	0.2	99
P-tot	7.7	6.1	20
COD	578	362	37

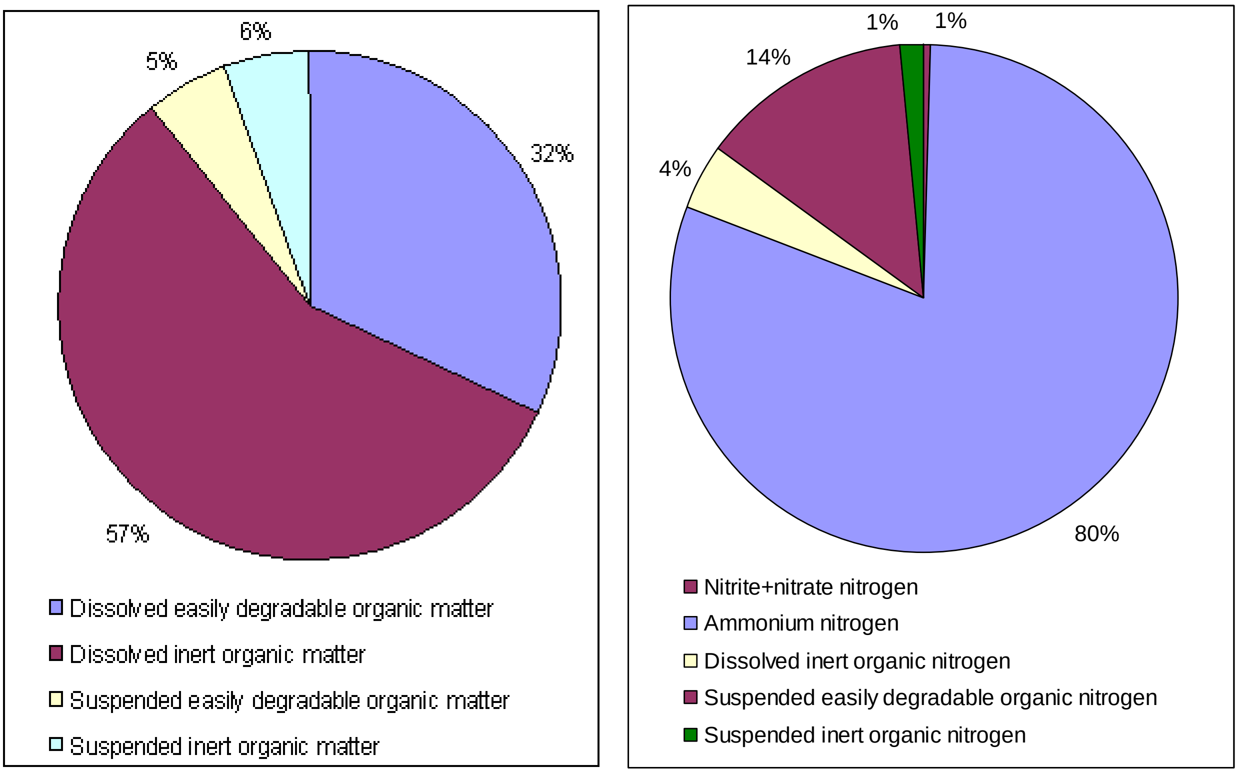

Figure 4 Organic and nitrogen fractions in the treated leachate

The organic matter and nitrogen that were measured in the effluent from the reactor corresponded to the slowly biodegradable organic matter, while the difference in concentrations of the compounds in the raw and treated leachate corresponded to the biodegradable fractions.

According to the obtained results, 63% of organic matter consisted of non-degradable organics, while only 5 % of total nitrogen corresponded to the non-degradable organic nitrogen fraction. These data correlated with the removal efficiency of the SBR process, presented in the Table 3, thus explaining the high nitrogen removal and low organic removal efficiency of the process.

Conclusions

- Current on-site pond treatment system at Filborna landfill doesn’t provide enough reduction of COD, nitrogen and phosphorous to meet the discharge requirements for the Oresund region.
- The SBR combined nitrification/denitrification process proved to be an efficient solution for nitrogen removal, as 93% reduction of total nitrogen was achieved during lab-scale trials.

- The efficiency of organic compounds' removal during the SBR process is low because they are present in the raw leachate mainly in the form of slowly biodegradable organic substances.

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