



CHALMERS
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Treatment of Landfill Leachate

Removal of Organic Contaminants, Metals and Sludge Characterisation

Master's Thesis in the Master's Programme Infrastructure and Environmental Engineering

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Department of Civil and Environmental Engineering
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CHALMERS UNIVERSITY OF TECHNOLOGY
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Cover:

A photo from the Brudaremassen site. The pilot treatment plant is located inside the tent and container.

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ABSTRACT

Leachate from landfills is known to contain contaminants harmful for the ecosystem. Therefore leachate needs to be treated before it is discharged to the environment or transported to a municipality wastewater treatment plant (WWTP). In order for a WWTP to improve the sludge quality, point-sources such as polluted landfill leachate needs to be disconnected and treated on-site.

In this thesis project the treatment efficiency of a pilot leachate treatment plant was evaluated, with a focus on removal of toxic metals and polycyclic aromatic hydrocarbons (PAH), polychlorinated biphenyls (PCB) and petroleum hydrocarbons (PHC). The removal efficiencies were also correlated to general parameters such as total and dissolved Fe, dissolved organic carbon (DOC) and total suspended solids (TSS). The sludge from the pilot treatment plant was evaluated with regard to chemical and physical properties, such as capillary suction time (CST), zone settling and water content. The studied leachate originates from a closed landfill in Gothenburg and contains high concentrations of Fe, PAHs, PCBs and nutrients. The pilot leachate treatment plant consists of chemical precipitation, moving bed biofilm reactor (MBBR), sand filters and activated carbon filters.

The results showed that chemical precipitation, sand filter and activated carbon filter are sufficient to reduce the studied contaminants. Low, medium and high molecular weight PAHs were reduced with 84, 98 and 26 % respectively. Total PCB was reduced with 96 %. Toxic metals had varying removal efficiencies, however all were below threshold values in the effluent. No strong correlations between the toxic metals and the general parameters were found. The correlation between Fe and TSS to organic compounds was improved with an increased lipophilicity. Surprisingly, none of the pollutants had a correlation above 0.6 to DOC. The sludges from the pilot plant had varying dewatering qualities. The chemical sludge had good thickening and dewatering properties (CST of 5.6 s, 1100 % increase in solid content after settling), while poor properties were measured in the biological sludge (CST of 80 s, 23 % increase in solid content). Suggested sludge treatment is gravitational settlement followed by mechanical dewatering. The chemical analysis of the sludge had low concentrations of contaminants, which was unexpected and should be further studied.

Key words: Landfill leachate treatment, organic contaminants, toxic metals, sludge, dewatering

Rening av lakvatten från deponier

Avskiljning av organiska föroreningar, metaller samt slamkaraktärisering)

Examensarbete inom masterprogrammet Infrastructure and Environmental Engineering

MATHIAS ANDERSSON

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SAMMANFATTNING

Lakvatten från deponier innehåller föroreningar som är skadliga för miljön. Följaktligen måste lakvattnet behandlas innan det släpps ut i naturen eller transporteras till ett kommunalt avloppsreningsverk. Punktkällor som lakvatten från deponier bör kopplas bort och renas lokalt för att öka kvaliteten på slammet.

I detta examensarbete har reningseffektiviteten hos en pilotanläggning för rening av lakvatten utvärderats, med fokus på avskiljning av metaller och polycykliska aromatiska kolväten (PAH), polyklorerade bifenyler (PCB) och petroleumkolväten (PHC). Reningseffektiviteten korrelerades till allmänna parametrar, så som totalt och löst järn, löst organiskt kol (DOC) och totalt suspenderat material (TSS). Slammet från pilotanläggningen utvärderades med hänsyn till kemiska och fysiska egenskaper, så som kapillärsugningstid (CST), sedimentering och vatteninnehåll. Det undersökta lakvattnet kommer från en stängd deponi i Göteborg och innehåller höga koncentrationer av järn, PAH, PCB och näringsämnen. Pilotanläggningen som har konstruerats vid deponin består av kemisk fällning, biofilmssystem med rörliga bärare (MBBR), sandfilter och aktivt kolfilter.

Resultaten visade att kemisk fällning, sandfilter och aktivt kolfilter är tillräckligt för att reducera de undersökta ämnena. Avskiljningen av PAH med låg, medelhög och hög molekylvikt var 84, 98 respektive 26 %. Koncentrationen av PCB minskade med 96 %. Avskiljningen av metaller varierade, men koncentrationerna i utgående vatten var under riktvärden. Inga starka korrelationer hittades mellan metaller och allmänna parametrar. Korrelationen mellan Fe och TSS och organiska ämnen ökade med ökad lipofilitet. Överraskande fanns ingen korrelation mellan DOC och undersökta miljögifter. Slammet från pilotanläggningen hade varierande avvattnings-egenskaper. Kemslammet hade bra förtjocknings- och avvattnings-egenskaper (CST på 5.6 s, 1100 % förtjockning vid sedimentering) medan sämre egenskaper uppmättes i det biologiska slammet (CST på 80 s, 23 % förtjockning). Föreslagen slambehandling är sedimentering följt av mekanisk avvattnings. Den kemiska analysen av slammet visade på låga koncentrationer av föroreningar i slammet, vilket var oväntat och bör undersökas vidare.

Nyckelord: Behandling av lakvatten, organiska föroreningar, metaller, slam, avvattnings

Table of contents

ABSTRACT	I
SAMMANFATTNING	II
TABLE OF CONTENTS	III
PREFACE	V
1 INTRODUCTION	1
1.1 Aim and goals	2
1.2 Limitation	2
1.3 Method	3
2 THEORY	4
2.1 Landfill design and decomposition phases	4
2.2 Landfill leachate	6
2.2.1 Organic contaminants	6
2.2.2 Toxic metals	8
2.2.3 General composition of landfill leachate	11
2.3 Current leachate treatment options	12
2.3.1 Discharge to wastewater treatment plant	14
2.3.2 Physical-chemical treatment	14
2.3.3 Biological treatment	19
2.4 Sludge	23
2.4.1 Water fractions in sludge	24
2.4.2 Treatment of sludge	25
2.4.3 Management of treated sludge	26
3 LEACHATE TREATMENT PLANTS AT OTHER LOCATIONS	28
4 BRUDAREMOSSEN LANDFILL	31
4.1 Brudaremossen background	31
4.1.1 Protective measures	32
4.1.2 Modern protective measures - leachate treatment	33
4.2 Environmental criteria and guidelines	34
4.2.1 Revaq – certification of sludge for agricultural reuse	34
4.2.2 City of Gothenburg – guidelines for emissions to water	35
4.2.3 Environmental quality criteria from the Swedish EPA	35
4.2.4 European legislation – prioritised pollutants	35
4.2.5 Canadian guidelines for water quality	35
4.2.6 United States recommended water quality criteria	36
4.3 Previous leachate studies at Brudaremossen	36
4.3.1 Ramböll investigations	36

4.3.2	Adsorption filter studies at Brudaremossen	37
4.3.3	Organic compounds in municipal landfill leachates	38
5	BRUDAREMOSSEN PILOT TREATMENT PLANT DESIGN AND SAMPLING-TAKING	39
5.1	Measurements at Brudaremossen pilot plant	41
5.1.1	Sampling locations for analysis	42
5.1.2	Sample storage	44
5.1.3	Analysis of data	44
6	LABORATORY ANALYSIS OF SLUDGE	45
7	RESULTS AND DISCUSSION	49
7.1	Results from analysis of leachate quality	49
7.1.1	Removal of toxic metals	51
7.1.2	Removal of organic contaminants	52
7.1.3	Removal efficiency of the biological treatment line	54
7.1.4	Correlation between measured parameters	55
7.2	Results from sludge analyses	57
7.2.1	Solids content in the sludge	57
7.2.2	Total suspended solids	59
7.2.3	Water content and distribution	61
7.2.4	Dewaterability	62
7.2.5	Chemical composition of the sludge	65
7.2.6	Handling of sludge	66
7.3	General discussion	67
7.4	Comparison with other treatment plants	68
7.5	Further studies	69
8	CONCLUSIONS	71
9	REFERENCES	72

Preface

During the spring of 2015 we have had the opportunity to be part of an exciting project. It has been very fun to study a treatment facility close-up. The practical work and the sampling have deepened our understanding in leachate treatment.

Some people played an important role in this project and we would like to express our appreciation. We would like to thank our supervisor Ann-Margret Strömvall for wise comments and guidance throughout the project. Her support and encouragement have been very valuable. We would also like to thank Britt-Marie Wilén, who have supervised us in the sludge analyses, and Mona Pålsson, from Water Environment Technology at Chalmers, for support with laboratory work. Lars-Ove Sörman has been helping us with sampling at Brudaremassen and has always taken time to answer our questions, we are grateful for this.

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Mathias Andersson and Hanna Säterskog

1 Introduction

Waste is material that is no longer useful or valuable for society. Looking back in time, the waste management first became a problem in densely populated areas, such as villages and towns (Ludwig et al., 2003). A poor waste management lead to sanitary and aesthetical issues. One commonly used waste management method is to dispose the waste in a landfill (Chandrappa and Das, 2012).

There are environmental downsides of disposing waste in a landfill, even if it is constructed with modern practice. The waste in the landfill is degraded during the production of greenhouse gases, such as methane (Lehr and Keeley, 2005). Percolation of precipitation and groundwater flowing into the landfill create leachate, contaminated water (Chandrappa and Das, 2012). Well-designed landfills have collection and management systems for leachate and gas. No regulations regarding protective measures for closed landfills in Sweden existed before 2001 (Gårdstam, 2015). Since then stricter rules have been introduced. Release of untreated leachate can contaminate groundwater, cause eutrophication in nearby waters, or spread toxic compounds (Lehr and Keeley, 2005).

The leachate can be treated locally or in a municipal wastewater treatment plant (Renou et al., 2008). The treatment techniques for leachate can be divided into biological and physical-chemical techniques. Biological techniques use bacteria to reduce the content of organic material and nitrogen in the leachate. Physical-chemical techniques are often used as pre-treatment for biological steps or to remove specific compounds. Examples of biological treatment methods are sequencing batch reactors, moving bed biofilm reactors and wetlands. Examples of physical-chemical techniques are chemical oxidation, precipitation and adsorption. The different treatment techniques often produce contaminated sludge, which must be treated and disposed (Gray, 2010). Generally, a reduction of the water content in the sludge is necessary before it is further treated and eventually disposed.

The biggest landfill in the Gothenburg region is called Brudaremossen. It is located in a recreational area in the eastern part of Gothenburg municipality. Brudaremossen is now inactive; it was in operation between 1937 and 1978 (Nielsen, 2007). All types of wastes were disposed in the landfill, including household waste, industrial waste and hazardous waste such as chemicals.

Brudaremossen emits 110 000 m³ of leachate annually, which is much for a landfill of its size (Lundh, 2015a; Swedish EPA, 2008a). The leachate is collected and led to an oil separator and a pond located north of the landfill, before it is ultimately discharged to the municipal wastewater treatment plant (WWTP) Ryaverket (Nielsen, 2007). The oil separator and the pond partly remove contaminants from the leachate (Krewer and Moona, 2012). However, the removal of nitrogen, iron and organic contaminants are not sufficient.

The leachate from Brudaremossen affects the sludge quality at the Ryaverket WWTP (Porse, 2013). Recently, a certification system named Revaq has been implemented to ensure that sludge from WWTP have a sufficient quality for being reused in farming (Svenskt Vatten, 2014). According to the rules of Revaq, point sources such as Brudaremossen should be disconnected from the treatment plants.

Previous studies of the leachate from Brudaremossen have shown that it is too contaminated to be released directly into the environment (Krewer and Moona, 2012). Therefore a local pilot leachate treatment plant was constructed at Brudaremossen at the end of 2014. The pilot plant will provide information that will be used to optimise the final design of the leachate treatment plant.

The pilot plant will be continuously evaluated by the consultancy firm Ramböll Sverige AB on behalf of the municipal water and sanitation administration Kretslopp och Vatten. This report will be a complement to their investigations by focusing on organic contaminants and toxic metals. This thesis work will also include a sludge characterisation and suggestions for its treatment. The results from this study will be valuable when deciding the design of the final treatment plant.

1.1 Aim and goals

The aim of the study is to evaluate the removal efficiency of specific contaminants and to characterise the sludge at the pilot treatment plant at Brudaremossen landfill. On-site measurements and laboratory results are used in combination with a literature study to evaluate the performance of the treatment plant, both for separate treatment steps and for the facility as a whole.

The specific goals are to:

- Compile a literature study describing problematics with leachate and leachate treatment.
- Investigate and study the removal efficiency of organic contaminants, such as petroleum hydrocarbons (PHCs), polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs) at the pilot plant facility.
- Determine if toxic metals (Zn, Cu, Pb, Cr, As, Hg, Ni) are sufficiently removed.
- Study the pollutant removal efficiencies for specific treatment steps and for the facility as a whole.
- Correlate the removal efficiencies of the contaminants to total suspended solids (TSS), total (TOC) and dissolved organic carbon (DOC) and total and dissolved iron.
- Chemically and physically characterise the quality of the sludge from the chemical treatment and the MBBR, and compare to sludge from other facilities.
- Suggest suitable options for handling the sludge.
- Compare the results in this study with results from other landfill leachate treatment facilities by performing a literature survey.
- Suggest recommendations for further research.

1.2 Limitation

The leachate samples are obtained during a limited time, February to May 2015. The operation of the treatment plant continues after the end of the thesis project. Analyses of organic contaminants are costly and therefore this study will focus on studying

groups of major contaminants instead of specific compounds. The number of measurements of organic contaminants and toxic metals are limited and variations will affect the results. Some contaminants are in very low concentrations in the leachate. It is a risk that these contaminants will be found below detection limit, restraining the possibility to draw conclusions further.

The physical analyses on sludge are only performed twice and the chemical analysis once. Therefore it is not possible to see trends in the sludge composition, such as an improved removal of contaminants, during the project time.

It is known that the composition of landfill leachate varies over a long time. The report focuses on the current situation and therefore no investigations for a future change in pollutants are made.

1.3 Method

The aim of this thesis is accomplished by performing a literature study and by studying the Brudaremossen pilot treatment plant. The literature study is important in the evaluation of the pilot treatment plant and serves as a background to the analyses performed on leachate and sludge at the treatment plant. The study at Brudaremossen evaluates the removal efficiency and sludge characteristics at the pilot plant. Different analyses are performed on leachate and sludge by commercial laboratories to determine the chemical composition. The evaluation of the removal rates will be based on results from these analyses. Additional analyses to determine physical properties of the produced sludge are performed at the Chalmers Environmental Chemistry laboratory as part of the project.

The result will be compared with Revaq, national and international regulations. In the literature study a review of leachate treatment methods are made. Other leachate treatment plants are also presented with regard to removal of organic contaminants and toxic metals. This will be useful when evaluating the treatment facility's efficiency.

2 Theory

The leachate produced from landfills needs to be treated before released to recipient. This can be done with many different methods, depending on the composition of the leachate. This chapter aims to present an overview of leachate characteristics and leachate treatment. Connected topics such as handling of by-products will also be included.

2.1 Landfill design and decomposition phases

Landfilling is one of the most common methods of waste management (Chandrappa and Das, 2012). In 2012, Sweden had about 80 operating landfills and several thousand closed landfills (Swedish Waste Management, 2012; Gårdstam, 2015). Historically waste was dumped or burned in the cities, since the transportation of waste was made easier when the distance was short (Chandrappa and Das, 2012). Naturally, the air quality and hygiene was affected by the poor waste management. By time, greater understanding of risks connected to landfilling has been gained. The location is of high importance, landfills needs to be placed away from waterways, earthquake zones, and areas with risk of flooding to reduce environmental hazards. The landscape also has to be considered; the landfill should not be too noticeable in the topography.

A well designed landfill should be placed on top of a liner, an impervious material that prevents the leachate from contaminating the groundwater. A liner often consists of many layers (Kreith and Tchobanoglous, 2002). Commonly used materials are geomembrane, clay and sand. Since a well-functioning liner system does not release any leachate a drainage system needs to be installed to reduce the risk of flooding (Lehr and Keeley, 2005). A cover should be placed on top of the landfill to prevent infiltration of precipitation. The cover is usually made of similar materials as the liners; geomembrane, geotextiles and clay. An illustration of a well-designed landfill is seen in Figure 1.

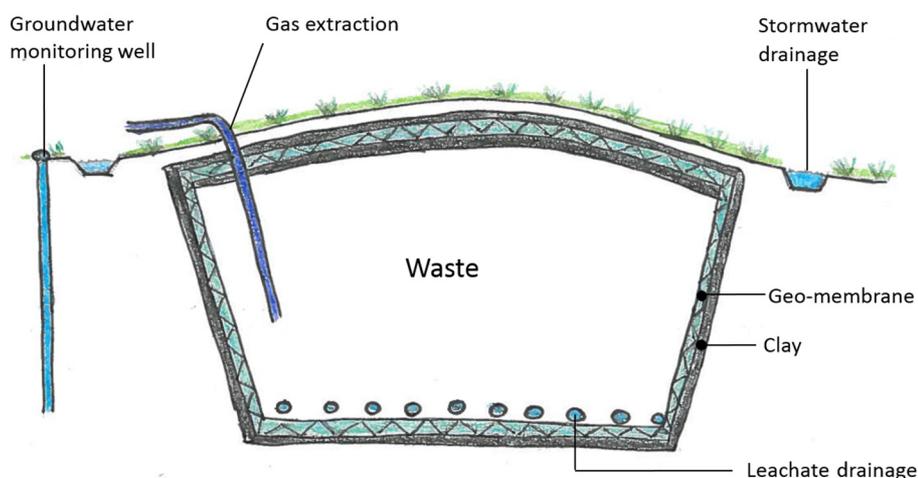


Figure 1 Important design components in a well-functioning landfill.

Gas and leachate are produced when the material in the landfill degrades. For safety and environmental reasons the release of landfill gas needs to be controlled. The gas is commonly collected in pipes and then used for energy or flared (Fischer, et al. 1999). Leachate can be treated on-site, recirculated or transported to a WWTP. In 2012 a survey was sent out to overview the leachate treatment situation in Sweden at operating and closed landfills (Swedish Waste Management, 2015); about 60 % of the contacted answered the survey. The result showed that 20 % of the landfill leachate sites directly discharged to a municipality WWTP, 40 % of the leachate was pre-treated before discharge to a WWTP and 40 % was treated on-site. Recirculation of leachate was not a common technique used in Sweden according to the survey.

The waste in the landfill will undergo a degradation process in several steps. The degradation phases are classified differently depending on which literature is consulted. However the processes remain the same. Here the degradation process will be divided into four phases, where the last phase is only a speculation. The degradation processes in the landfill depends on many different factors, e.g. type of waste, climate, pH and moisture content (Schmid et al., 2000). The chemical composition of the landfill is varying between different phases; hence the leachate quality will also vary over time. The four phases are presented below.

Aerobic Phase. The first stage of degradation only lasts a couple of days and starts immediately as the waste is placed in the landfill (Kjeldsen et al., 2002). During the aerobic phase oxygen is still present. Microorganisms consume the oxygen while they degrade organic material and produce carbon dioxide, water, heat and alcohol (Baird and Cann, 2008; Ludwig, et al. 2003).

Acid Phase. When the landfill is covered there is no inflow of oxygen and the environment inside the landfill becomes anoxic (Kjeldsen et al., 2002). In the anoxic environment fermentation processes starts and carboxylic acid and alcohols are produced. During this step the pH is lowered which causes metals to dissolve, creating a toxic leachate (Baird and Cann, 2008; Lehr and Keeley, 2005). The degradation continues and the acids and alcohols degrade to acetate, hydrogen and carbon dioxide (Kjeldsen et al., 2002). The BOD and COD increases when the cellulose is degraded.

Initial and Stable Methanogenic Phase. The acetate, hydrogen and carbon dioxide continues to be degraded by methane producing bacteria in the initial stage of methanogenic phase (Kjeldsen et al., 2002). When the acids are consumed the pH becomes more neutral and metals can no longer stay dissolved (Baird and Cann, 2008); this results in a less corrosive leachate. As the food, acetate, hydrogen and carbon dioxide, is consumed the methane producing bacteria will reduce the production of methane (Kjeldsen et al., 2002). The stable methanogenic phase starts when the peak production of methane occurs.

Predicted Phases. The first phases goes on for a long time, often 10-50 years for landfills with sufficient water content, and ends gradually (Kjeldsen et al., 2002). As the methane production decreases there is a possibility that the landfill gets intruded by air, as a result of pressure changes between landfill and surrounding air. Air can also be intruded into landfill by diffusion and/or wind pressure. Since closed landfills are covered by soil and liners, it takes a very long time before the landfill gets aerobic. The air that enters the landfill will oxidise the methane that still is present or produced

into carbon dioxide. Other oxidation reactions may also occur to sulphur and nitrogen. Since carbon dioxide is produced the pH might be affected if there is no buffering capacity. The air intrusion will mainly occur in the cracks and near the surface, resulting in local degradation processes. Models made in the end of the 1990's showed that the air intrusion and depletion of all organics can take up to 500 000 years if the waste is saturated by water (Bozkurt et al., 1999). A landfill without liner and with low water content may take around 600 years to be intruded with air.

2.2 Landfill leachate

Leachate may be defined as water that has percolated through a landfill (Kreith and Tchobanoglous, 2002). The main part of the water generally originates from infiltrated precipitation or from groundwater flowing through the landfill. If the landfill has been constructed with proper lining the main part of the leachate is precipitation. Water in the landfill can also be created from degradation processes in the waste. The composition of the leachate is, as stated previously, affected by the current phase of the landfill. It is also dependent on the type of waste disposed in the landfill (Swedish EPA, 2008a).

The amount of available water is, apart from the already mentioned inflows, affected by leakage, storage capacity of the landfill and evaporation (Swedish EPA, 2008a). The amount of leachate varies between seasons, since the precipitation is the main source of leachate. The leachate from landfills is often polluted with organic contaminants and toxic metals. These two groups of pollutants will be presented below.

2.2.1 Organic contaminants

Organic contaminants are organic compounds that are hazardous for the ecosystem (Brandt and Gröndahl, 2000). They can be carcinogenic, toxic or mutagenic for organisms. The organic contaminants accumulate in organisms because they are lipophilic and can enter the cell membrane. This can cause severe problems for organisms and biomagnification can occur in the food chain. In the 1970's this was witnessed when the sea eagle in the Baltic Sea almost got extinct in the area due to lipophilic organic contaminants that stopped the birds' ability to reproduce (Lundberg, 2014).

Organic content in leachate can be measured by several different global parameters, such as TOC and DOC. However, all organic content is not contaminants e.g. humic substances. In order to evaluate if organic contaminants are present more specific analyses need to be performed.

Organic contaminants are mainly quantified by using mass spectrometric methods (Benfenati, 1996). Gas chromatography–mass spectrometry (GS-MS) is the most common method and can identify hundred thousands of spectra. Liquid chromatography–mass spectrometry (LC-MS) is also used for more high-molecular weight and polar organic contaminants (Barceló, 2012; Petrovic et al. 2010).

There are a countless number of organic contaminants and therefore they are divided into groups. The known toxic organic contaminants are believed to only be the top of

the iceberg (Brandt and Gröndahl, 2000). Some common groups of organic contaminants in leachate are presented below.

Petroleum hydrocarbons

Petroleum hydrocarbons, PHCs, are the main components in crude oil and many oil based products (Brecher, 2006). The group consists of hundreds of different compounds with varying chemical properties. Larger petroleum hydrocarbons are more persistent and less soluble in water than petroleum hydrocarbons with lower weight. The toxicity and carcinogenicity is varying within the group of petroleum hydrocarbons. The compounds may negatively affect organisms' ability to reproduce. More harmful chemicals with similar properties as petroleum hydrocarbons can possibly dissolve in the petroleum hydrocarbons, allowing for further transport (Walker, 2009).

One group of petroleum hydrocarbons is aliphatic hydrocarbons. This group contains non-aromatic and non-cyclic hydrocarbons (Patniak, 2007). Their properties depend on the number of carbon atoms in the molecule. Aliphatic hydrocarbons with a low number of carbon atoms, for example ethane and ethyne, are gases and very flammable. Compounds with more carbon atoms are liquids or solids. Presence of aliphatic hydrocarbons in a leachate could possibly imply a chance for other harmful organic components as well, since they often coexist due to their similar lipophilic properties.

Polycyclic aromatic hydrocarbons

The polycyclic aromatic hydrocarbons, PAHs, are a group of compounds that are found in landfill leachate. The PAHs consist of aromatic rings made by hydrogen and carbon. They exist naturally, however they also origin from human activities (US EPA, 2008). PAHs are created from incomplete combustion and can origin from point sources, such as power plants, and diffuse sources, such as traffic. Forest fires are a major natural source of PAHs. Usually, a number of PAHs are found together (Samanta et al., 2002). PAHs are present in oil waste, which are sometimes disposed in landfills. This might have a local impact on the environment (Walker, 2009). They also are present in other wastes, such as coal tar and wood impregnated with preservatives (Lundstedt et al., 2007).

Polycyclic aromatic hydrocarbons are persistent and also lipophilic (Samanta et al., 2002). They are considered as environmental pollutants and may accumulate in organisms. The harmfulness varies between different PAHs. The compounds may be toxic, mutagenic and carcinogenic. Generally the PAHs themselves are not very toxic; however their degradation products can be very harmful. Transformation can occur when the PAHs are oxidised, for example by exposure to ultraviolet light (Walker, 2009). Synergy effects may also occur between different PAHs. Studies have shown connections between exposure to PAHs and reduced learning capacity among children (Perera et al., 2012). PAHs may also have harmful effects on the ecosystems (Samanta et al., 2002).

Today, PAH are commonly divided into three categories depending on their molecular weight (Swedish EPA, 2009). Compounds with high molecular weight are

called PAH H. Medium molecular weight compounds are called PAH M and low weight molecular compounds are called PAH L. The three groups have different physical and chemical properties. The carcinogenicity of PAH increases with weight.

One compound into which PAHs can be transformed are oxygenated PAHs, commonly abbreviated as oxy-PAHs. These compounds are PAHs with one or more attached carbonyl groups. Lately, researchers have increased their concern for the presence of oxy-PAHs in the environment (Lundstedt et al., 2007). PAHs can be transformed into oxy-PAHs by microorganisms. Transformation can also be made by chemical oxidation or photo oxidation. Oxy-PAHs are believed to be more persistent than regular PAHs. Studies indicate that they are the end-products in many transformation paths, and will therefore not be transformed further in nature. The carbonyl group in the oxy-PAHs partly change the chemical properties of the molecules. They become more polar and therefore more likely to be transported in water. Some oxy-PAHs are acute toxic to microorganisms, on contrary to most regular PAHs. Lundstedt et al. (2007) proposed that oxy-PAHs should receive more attention in studies of sites contaminated with PAHs. They also stated that remediation techniques for sites containing PAHs could possibly transform them into oxy-PAHs.

Polychlorinated biphenyls

Polychlorinated Biphenyls, PCBs, are chemical compounds with a number of chlorine atoms attached to two fused phenyls. The compounds are man-made and do not occur naturally (Borja et al., 2005). PCBs are easily transported in nature and can be found at almost any location.

PCBs were widely used until the 1970's due to their stable chemical properties (Borja et al., 2005). The compounds are fire resistant, do not conduct electricity very well and are not easily degraded (Walker, 2009). Their usage has been restricted in Sweden since the beginning of 1970's, however usage was not restricted as early in other parts of the world. Due to their diverse properties, the compounds were used in electrical equipment, insulation and plastics (US EPA, 2013). PCBs are liquids or resins, depending on their content of chlorine. Molecules with more chlorine atoms are less viscous and have a lower solubility in water (Borja et al., 2005). Overall, PCBs are lipophilic, bioaccumulating and biomagnifying. They can be toxic from chronic exposure, are proven to be carcinogenic and can cause liver damage in humans. PCBs reduce birds' ability to reproduce.

The release of PCBs into the environment from industries and human activities is low at present (Walker, 2009). Instead, PCBs are spread from sites where large amounts of PCB have been disposed previously, such as landfills. Since PCBs were used in a wide variety of products, in Sweden until 1970, their presence could be expected in leachate from older landfills. Their properties and ability to biomagnify means that they are harmful compounds to the environment, if present.

2.2.2 Toxic metals

Metals occur naturally in the environment in soil, rocks and water (Swedish EPA, 2014a). Many metals are essential for plants, animals and humans; such as Fe, Mg

and Cu (Swedish National Food Agency, 2012). Only low doses of metals are needed daily and if high concentrations are found in the environment they become toxic (Swedish EPA, 2014a). High metal concentrations often originate from human activities such as mining and industries. Since metals are natural elements they cannot be degraded; therefore they stay in high concentration for long before dilution occurs (Swedish EPA, 2014a). The only way metals can be reduced in the environment is by sedimentation and dilution (Bydén, et al. 2003). However, landslides, human activity or change in pH or oxygen concentration can release the metals again.

Metals are divided into different metal groups based on their density, molecular weight etc. The group with the highest density and often hazardous is many times labelled *heavy metals*. Since there are heavy metals that are not hazardous, such as iron, heavy and hazardous metals will instead be referred to as *toxic metals* in this report.

Toxic metal concentrations in landfill leachate are often low or moderate but large local variations exist (Kjeldsen et al, 2002). The toxic metal concentration in leachate is higher during the acid phase and reduces when reaching the methanogenic stage (Baird and Cann, 2008). Studies in the metal leakage from landfills reveal that only a small fraction, 0.02%, of the metal in landfills end up in the leachate during the first three decades (Kjeldsen et al, 2002). The main reason for the slow mobilisation is the neutral to basic pH, which enables sorption and precipitation. The carbonates and sulphides in the landfill can bind to, precipitate and immobilise toxic metals such as Cd, Cu and Pb. In most landfills, sulphide that precipitates with metals is the main reason for low concentrations of toxic metals in leachate. Toxic metals are present in the leachate if toxic metal waste is disposed in the landfill and it mobilises.

It has been seen from studies by Jensen and Christensen (1999) that there is a correlation between toxic metals in leachate and the concentration of colloids. If the leachate contains colloids it is a greater risk that toxic metals, such as Cd, Cu, Pb and Cr, are transported out from the landfill with the colloids. Below follows an introduction to some toxic metals often analysed in leachate.

Cadmium, Cd

Cadmium is a dangerous metal that can be found in leachate. It exists naturally in the ground and can be brought up by volcano eruption, 62% of the Cd comes from volcanic eruptions (Swedish EPA, 2014b). Industries also emit Cd, for example from metal, electronic and plastic production. Cd is easily transported in air, and inadequate combustion of waste or fossil fuels spread Cd far, often affecting other countries. Leachate with high Cd concentration needs to be treated to reduce the risk of affecting the recipient. Low concentrations of Cd can be hazardous for animals and humans (Swedish EPA, 2014b). By being exposed to only a small dose, the risk of bone fractures, decreased kidney function and cancer are increased.

Chromium, Cr

Chromium is used in paints, fungicides, chrome alloys and in the industries for leather and glass production (WHO, 2003). The environmental effect is dependent of the oxidation state, which ranges from Cr^{2+} to Cr^{6+} . The common states in leachate are

Cr^{3+} and Cr^{6+} (Öman, et. al 2000). The most toxic form is the hexavalent, Cr^{6+} , which is toxic at low concentrations (Swedish EPA, 2011). The hexavalent Cr causes various threats to humans and animals such as cancer and mutations. It is also easily transported out to the environment due to its ability to react and dissolve in water. The balance between Cr^{3+} and Cr^{6+} is dependent on pH, redox potential and oxygen concentration (Öman, et. al 2000). At high oxygen concentration, the concentration of hexavalent Cr is elevated in the leachate. If there is iron in the leachate the solubility of Cr^{3+} is reduced since they bind and precipitate.

Copper, Cu

Copper is essential to almost all animal and plants (US EPA, 2007). It is for example used in chlorophyll biosynthesis for plants (Bydén, et al. 2003). A danger with Cu is its ability to accumulate in organisms. Even if Cu is found in low concentrations in surrounding water, organisms can contain high concentrations, as much as thousand times higher than in water. Some fishes in Swedish waters are harmed when concentration exceeds 2 $\mu\text{g/l}$. A high dosage exposure for humans leads to irritation of the stomachic-intestinal canal and a long and lower exposure can give liver damages for humans (Swedish National Food Agency, 2014). Copper have been used as a pesticide, in paint, pipes and for electric cables (Swedish EPA, 2014c). Copper is many times used in coatings and pesticides because of its ability to create an environment where algae and fungi cannot survive.

Lead, Pb

Lead is poisonous for plants, animals and humans (Swedish EPA, 2014d). High concentrations can affect the nervous system in animals and humans. Children and infants are especially sensitive to exposure. The main sources of Pb used to come from petroleum but today the petroleum is lead free. Today sources of lead are electronics, ammunition and fish sinkers. The usage of Pb has been reduced since 1990's and it has been forbidden in electronics since 2006 (Swedish EPA, 2014d; Swedish EPA, 2015a). Despite that, the surface water concentrations have not recovered since the peak Pb pollutions. This is due to the slow spreading from point sources to surface waters. Lead can mobilise by creating strong complex with humus substances (Öman, et al. 2000). It has also been shown that Pb concentration can increase after the acidic phase in a landfill, opposite to many other metals.

Mercury, Hg

Mercury is one of the most toxic metals (Linderholm, 2014). It occurs in many different forms and most of them are very poisonous to humans and animals. The most toxic form is methylmercury. If exposed to Hg, there is a risk of damage to the central nervous system and children are more vulnerable than adults. Mercury binds to fat tissues and accumulates in the body. Methylmercury therefore transfer from the mother to the baby during pregnancy and breastfeeding. The biggest source of Hg is gold mines. Coal and waste combustion also spread Hg. Landfill leachate may contain Hg (Kjeldsen et al., 2002; Linderholm, 2014).

Nickel, Ni

Like other toxic metals, Ni is persistent and can bio-accumulate (Eisler, 1998). Nickel irritates lungs, skin and is carcinogenic (Eisler, 1998; Swedish EPA, 2008b). Nickel exists naturally in the environment and is crucial, in low concentration, for growth for many living organisms (Eisler, 1998). Elevated concentrations of Ni can be found around mining, metal, waste and incinerating industries. Around these sources Ni is accumulated in plants and animals. Nickel can be transported in the ecosystem by chemical, physical and biological processes.

Zinc, Zn

Zinc is an essential metal for organisms but elevated concentrations can harm aquatic life (Swedish EPA, 2014e). It exists naturally in relatively high concentrations in the environment and is not as toxic as many of the other toxic metals. Zinc has been widely used for galvanising metals and also in alloys and electronics. The Zn has then been released to environment through combustion, erosion and waste. The concentration of Zn is highest in the landfill leachate during the acidic phase (Öman, et. al 2000).

2.2.3 General composition of landfill leachate

The initial aerobic phase in a landfill is short; therefore the composition of the leachate in this phase is not of interest. The acid phase, which is characterised by low pH and toxic leachate, and the methanogenic phase, in which methane are produced, lasts longer and therefore more studies of the composition in these phases have been made. Kjeldsen et al. (2002) have made a comprehensive study of landfill leachate and phases; parts of their results can be seen in Table 1. Apart from the pH, the most significant difference is the lower organic content in the methanogenic phase, as well as the lower ratio between BOD and COD.

Table 1. Differences in leachate composition of pH and oxygen demand for landfills in the acid and the methanogenic phase. Based upon data from Kjeldsen et al. (2002).

Parameter	Acid Phase		Methanogenic Phase		Unit
	Average	Range	Average	Range	
pH	6.1	4.5-7.5	8	7.5-9	
BOD ₅	13000	4000-40000	180	20-550	mg/L
COD	22000	6000-60000	3000	500-4500	mg/L
BOD ₅ /COD	0.58		0.06		

Landfill leachate contains a wide variety of compounds. Kjeldsen et al. (2002) compiled results from studies of a few metals in leachate, see Table 2. Overall, the concentrations of these metals are significantly higher in the acid phase than in the methanogenic phase.

Table 2. Differences in leachate composition of a few selected metals for landfills in the acid and the methanogenic phase. Based upon data from Kjeldsen et al. (2002).

Parameter [mg/L]	Acid Phase		Methanogenic Phase	
	Average	Range	Average	Range
Iron	780	20-2100	15	3-280
Sulphate	500	70-1750	80	10-420
Calcium	1200	10-2500	60	20-600
Magnesium	470	50-1150	180	40-350
Manganese	25	0.3-65	0.7	0.03-45

Different organic compounds can also be found in the leachate from a landfill (Kjeldsen et al., 2002). Measured concentrations of a number of organic compounds are shown in Table 3. The measurements were mostly performed in leachate from municipal landfills, containing solid waste. Note that the number of studies is not the same for all compounds. Therefore it might seem as the range of concentration for a certain compound is very small, compared to other. In general, the range of the concentration spans over several magnitudes.

Table 3. Selected organic compounds found in leachate from, mostly, municipal solid waste landfills. The data is based on a compilation of several studies by Kjeldsen et al. (2002).

Type of compound	Compound	Range [µg/L]
Aromatic hydrocarbons	Benzene	0.2-1630
	Toluene	1-12300
	Xylene	0.8-3500
Polycyclic aromatic hydrocarbons	Naphthalene	0.1-260
Phenols	Phenol	0.6-1200
	Cresols	1-2100
Phthalates	Diethyl phthalate	0.1-660

2.3 Current leachate treatment options

There is no universal treatment suitable for all landfill leachates. Specific solutions are required for each leachate, due to the wide variety in the composition (Swedish EPA, 2008a). The volume of the leachate produced is also affecting the design of a landfill leachate treatment plant (Renou et al., 2008).

Apart from the variations in leachate flow and composition the environmental regulations many times set design criteria for the leachate treatment and discharge (Renou et al., 2008). It is important that the environment in the recipient is not negatively affected by the discharge from leachate treatment. The water volume and environmental status of the recipient therefore plays a key role when setting treatment goals for a leachate treatment plant. More information about environmental regulations can be found in Chapter 4.2. With modern treatment options it is possible to reach very high quality of the effluent, however it is not always feasible to apply

these options because of economic reasons (Renou et al., 2008). A leachate treatment plant is costly and it is important to consider both treatment efficiency and cost.

An extensive number of studies have been made regarding landfill leachate treatment methods. However, most of them were applied in a pilot or laboratory scale. The number of studies focusing on organic contaminants such as PAHs and PCBs in landfill leachate is not as extensive. Table 4 compiles information regarding the different treatment alternatives. The ratings in the table are based upon the data found in the literature study. For further details, see the individual chapters for each treatment method below. For removal efficiencies, very high corresponds to a removal above 90 %, high corresponds to above 70 %, medium corresponds to above 50 % and low corresponds to below 50 %. The complexity is taking into account how complex and difficult the treatment techniques are. For example, processes that are simple and require no continuous addition of chemicals or equipment are said to have a low complexity. The costs for the treatment systems is mostly based upon Swedish Waste Management (2012) where a compilation of treatment techniques are given. Information regarding the cost is also obtained from Renou (2008) and Lundh (2015c).

Table 4 A comparison of different treatment options based on their complexity, cost and removal efficiency.

Treatment technique		General removal	Removal of toxic metals	Removal of organic contaminants	Cost	Complexity
Physical - chemical	Adsorption	High	High	Very High	Medium	Low
	Coagulation/ Flocculation	Medium	High	Low	Medium	Medium
	Precipitation	Low	Very High	Low	Medium	Medium
	Dissolved air flotation	Low	Very High	Low	N/A*	Medium
	Chemical oxidation	Very High	Low	Very High	Medium	High
	Fenton oxidation	High/Very High	Low	Very High	Medium	Medium/High
	Reverse osmosis	Very High	Very High	High	High	High
Biological	SBR	High	Medium	Low	Medium	Medium
	MBBR	High	Medium**	High	Medium	Medium
	Constructed Wetlands	Medium	High	Very High	Medium	Low
	Aerated lagoons	Very High	High	Low	Low	Low

*Dissolved air flotation has not been applied to leachates to a large extent. **No literature regarding removal of toxic metals was found in the literature survey. The medium rating is based upon the removal efficiency of metals in SBR.

2.3.1 Discharge to wastewater treatment plant

A commonly used alternative for leachate treatment, both in Sweden and in the rest of the world, is to transfer the leachate to a WWTP (Eriksson, 2005; Renou et al., 2008). When the Swedish Environmental Research Institute performed an evaluation of treatment techniques in 2007, transferring the leachate to a WWTP was still the most common method (Cerne et al., 2007). The leachate can also be pre-treated on-site before discharge to a wastewater treatment plant.

Treating the leachate in a WWTP is generally less expensive and requires less maintenance, compared to a local treatment (Ahn, et al., 2002). The high concentrations of nitrogen in the leachate can in some cases improve the efficiency of the biological treatment at the treatment plant (Renou et al., 2008). However, there are issues related to co-treating the leachate with sewage. Organic contaminants and toxic metals in the leachate might harm the biological treatment in the plant. These compounds may have an inhibitory effect on the growth of microbial organisms and thereby lower the overall treatment efficiency. Furthermore, the components in the leachate may eventually accumulate and end up in the sludge at the WWTP (Wiszniewski et al., 2006). They may also pass untreated through the treatment plant and affect the recipient (Olofsson, 2012).

2.3.2 Physical-chemical treatment

There are a number of available physical and chemical techniques for treatment of landfill leachate. They are generally used as pre-treatment before a biological treatment step or to protect subsequent treatment steps from harm (Wiszniewski et al., 2006; Renou et al., 2008).

Some of the physical-chemical treatment methods are widely used and has a long history for water treatment, such as coagulation-flocculation. Other methods, such as membrane technologies, have more recently gained popularity for water treatment purposes (Renou et al., 2008). In the text below some available physical-chemical techniques for treating leachate are presented, with a focus on removal of organic contaminants and toxic metals.

Coagulation - Flocculation

Treatment with coagulation and flocculation has successfully been used for treating landfill leachates (Renou et al., 2008). In this method, a coagulation chemical is added to the leachate and this creates large flocs that are more easily removed. Commonly iron or aluminium salts are used as coagulation chemicals. A number of scientific studies compiled by Renou et al. in 2008 showed that iron salts were more efficient for treating leachate than aluminium salts. The coagulation and flocculation process is more efficient at certain pH. Amokrane et al. (1997) did a study to determine optimal doses of coagulants and optimal pH for treating leachate from a landfill. A pH of 5 was deemed optimal for ferric chloride and a pH of 5.5 was the most suitable when using aluminium sulphate. Improved removal and quality of the effluent can be obtained if a polymer is used in the process, or if aluminium and iron is used in combination.

Studies report a COD removal above 50% when using iron salts for coagulation-flocculation treatment of landfill leachate (Renou et al., 2008; Wiszniowski, 2006). The removal efficiency of organic contaminants in landfill leachates, in a coagulation-flocculation process, is not as widely studied. Shaber et al. (2014) reported a 21–38 % removal of PAHs when using aluminium as a coagulant in a laboratory setting with a synthetic wastewater. The same authors reported 89–100 % removal efficiency when also adding nano-clay minerals to the water. A similar result was obtained when treating real contaminated water, however not from landfills.

Wastewater from soil washing, containing high concentration of petroleum hydrocarbons, was treated in a coagulation-flocculation process in a study by Torres et al. (2009). When using ferric chloride at pH 5 in combination with a flocculent, a COD removal of 97 % was obtained. No measurements of individual hydrocarbons were made in the study. However, the authors deemed the process suitable for treating the studied type of water. Toxic metals can be removed in a coagulation flocculation process as well. Using industrial wastewater, Amuda et al. (2006) reached removal efficiencies of 91 % and 72% respectively for Cr and Zn in their study. The removal of the metal Fe was lower.

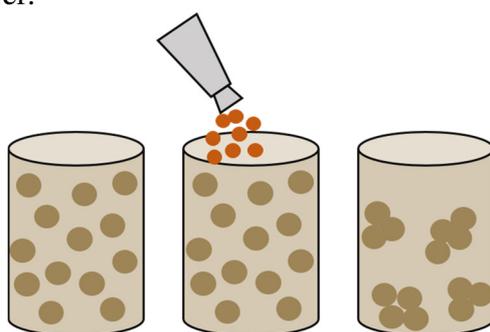


Figure 2 An illustration of chemical precipitation. By adding chemicals the leachate particles coagulate and settle.

If the main purpose of treatment is to reduce the content of metals in the leachate, the greatest removal is obtained at alkaline pH above 9 (Amokrane, et al., 1997). This process is more commonly referred to as precipitation, and is illustrated in Figure 2. This may lead to higher turbidity and additional problems in subsequent treatment steps. Toxic metals are effectively removed by chemical precipitation at alkaline pH.

A compilation made by Fu and Wang (2011) showed that it is possible to obtain removal efficiencies above 99 % for Cu, Zn, Pb, Cr and Hg when using chemical precipitation. Toxic metals can be removed at more neutral pH as well, if sulfide is used as precipitant. However, using sulfides, might create problems with separation and dangerous fumes.

Adsorption processes

Some materials have a large surface area in comparison to their volume (Schröder et al., 2007). Such materials are used in adsorption treatment processes, where the contaminants in the leachate attaches to the surface of the adsorption material and is thus removed from the liquid. The adsorbing material is either added to the water in powder form or packed in a column. If the latter alternative is used the water filters through the column and the contaminants are removed. The most common adsorption material is activated carbon (Renou et al., 2008). It is created from organic material such as coconut shells or coal (Schröder et al., 2007).

In treatment of landfill leachate activated carbon can achieve a removal efficiency of 85 % for COD (Renou et al., 2008). It should be noted that even though the pollutants are removed from the leachate they are still present in the filter. Therefore the filter material ultimately has to be handled appropriately.

An issue using activated carbon is a comparably high cost for the material (Kängsepp et al., 2008). If activated carbon filters are used for adsorption they have to be regenerated regularly to maintain their removal efficiency. If powder is used instead, it is continuously consumed in the process. Therefore there is an interest of using other materials than activated carbon for the adsorption process. Studies have shown that other materials can reach removal efficiency comparable to activated carbon. Minerals such as zeolite, keolinite and illite are alternative materials, as well as ash from incineration (Amokrane et al., 1997). Biological materials can also be used for adsorption.

Leachates from landfills were treated using alternative filter materials in a laboratory set up by Kängsepp et al. (2008). The study suggested that a serial set-up of peat and ash with remains of carbon was the most efficient alternative for treating leachate. Kalmykova et al. (2013) studied removal of organic contaminants from landfill leachate and compared the efficiency between activated carbon filters and filters using sphagnum moss. The conclusion was that the most efficient treatment would be obtained when using these two materials, in series.

Kalmykova et al. (2013) also studied the removal efficiency of organic contaminants in landfill leachate using the mentioned adsorption filters. The activated carbon filter had removal efficiency near 100 % for a number of organic contaminants. Petroleum hydrocarbons were completely removed in 50 % of the tests when using activated carbon, and in 35 % of the tests using peat moss.

Metals can be removed from water by using adsorption techniques. It is however dependent on the pH of the water or leachate, which should be 4 – 5 (Kadirvelu et al., 2001). Using industrial wastewater and activated carbon made from coconuts, Kadirvelu et al. (2001) achieved a 100 % removal of Hg, Pb and Cd, as well as 73 %

removal of Cu and 92 % removal of Ni. Modin et al. (2011) performed a study of real landfill leachate and metal removal. The conclusion was that a combination of filters were necessary in order to remove all the different metals.

Dissolved air flotation

Dissolved air flotation is a treatment method where air bubbles are introduced to the water. The bubbles attach to the suspended particles in the water and bring them to the surface, where they may be removed (Palaniandy et al., 2010). The dissolved air flotation process is not widely used in landfill leachate treatment (Renou et al., 2008). However a few recent studies have been made to evaluate its performance for treating leachate. Palaniandy et al. (2010) states that dissolved air flotation have several benefits, including low land usage, good sludge quality and high effluent water quality. In a laboratory study Palaniandy et al. obtained a COD removal of 79 % when treating leachate. However this required that the dissolved air flotation was preceded by a coagulation step using aluminium sulphate. Without coagulation the removal of COD was 33 %. Application of flotation techniques for removing oil and organic contaminants from process water from industries have been studied to a greater extent (Rubio et al., 2002). It is regarded as an appropriate technique for this type of wastewater. Experiments evaluating dissolved air flotation and removal of metals were made by Lazaridis et al. (1992). Removal efficiencies near 100 % were obtained for Cu, Zn and Ni when the pH was optimal. The experiments were however not conducted on real wastewater or leachate.

Chemical oxidation

When using chemical oxidation the contaminants in the water are oxidised to less harmful compounds. This is traditionally done by addition of a strong oxidising agent such as hydrogen peroxide or ozone (Renou et al., 2008; Wiszniowski et al., 2006). Organic matter and organic contaminants can be broken down by oxidation, ultimately into carbon dioxide and water. The removal efficiency varies depending on the type of oxidising agent. Ozone may reach removal efficiencies generally between 50–65 % for COD while other agents reach slightly lower efficiencies (Wang et al., 2003; Wiszniowski et al., 2006).

Recently, an increased interest has been reported for more advanced oxidation methods, such as using a combination of oxidants and/or irradiation. Examples of possible combinations of oxidants are ozone and hydrogen dioxide, hydrogen dioxide and ultraviolet radiation, or ozone and hydroxide ions (Renou et al., 2008). When combining ozone with another oxidant it is possible to achieve higher removal rates than for ozone alone. Wang et al (2003) reported that ozone and hydrogen dioxide in most cases gave a COD-removal above 90 %. Ozone combined with UV gave lower removal (50–60 % of COD) and hydrogen dioxide gave a removal generally between 60–100 % for COD.

Renou et al. (2008) reported that the main drawbacks of advanced oxidation processes are their high cost and energy demand. Also, intermediate products caused by incomplete oxidation may be more harmful than the original compounds. Certain combinations of oxidants might be problematic to use in practice (Wang et al., 2003).

Even the less advanced chemical oxidation processes require energy and chemicals to function efficient.

Chemical oxidation should be effective for treating organic contaminants (Ramakrishnan et al. 2015). There are a limited number of studies made on removal of organic contaminants from landfill leachate using oxidation processes. Ramakrishnan et al. (2015) state that advanced oxidation processes have the possibility to treat organic contaminants if designed properly. In a laboratory study by Vollmuth and Neissner (1995) on synthetic leachate a removal efficiency above 90 % of both PAHs and chlorinated phenols was obtained. However, no significant changes in concentration were achieved for PCBs. The study used UV irradiation in combination with ozone.

Fenton oxidation

A common oxidising agent is a combination of hydrogen peroxide and iron salt, commonly referred to as Fenton's reagent (Lopez et al., 2003). Fenton's reagent has several advantages compared to other reagents; it is nontoxic and comparably cheap. The process is technologically simple. However, it requires pH adjustment before the reagent is added. Afterwards, the pH needs to be neutralised again. Studies have determined the optimal pH to be around 3 for the Fenton process in order to achieve a good removal of organic matter (Kilic, et al. 2014). COD removal above 80 % has been measured. The Fenton process can be combined with other oxidising agents, for example UV (photo-Fenton) (Renou et al., 2008). Fenton oxidation has proven to be efficient for removal of PAHs in polluted water (Beltrán, et al. 1998). This was also concluded in a study made by Wu et al. (2011) where a complete removal was achieved for some PAHs and phthalates. In this study the Fenton process was preceded by biological treatment and coagulation.

Membrane treatment

An emerging method is the usage of membranes for landfill leachate treatment (Renou et al., 2008). Membranes only allow particles smaller than their pore size to pass through, see schematic picture in Figure 3. Therefore larger contaminants are removed when leachate is pumped through the membranes. Smaller pore sizes give a cleaner effluent, however they require a higher pressure to operate.

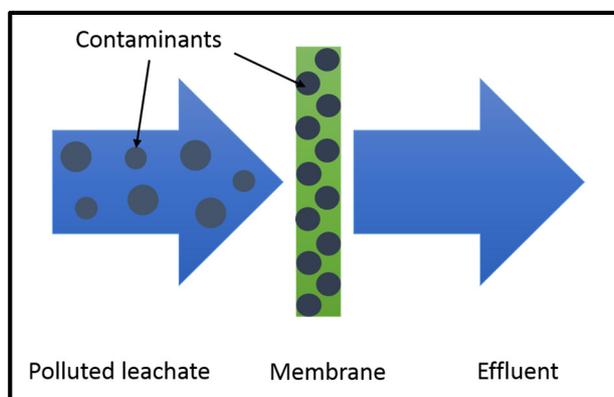


Figure 3 Membrane treatment removes particles/molecules larger than its pore size. Only water and small particles can pass through the membrane.

A disadvantage with membranes is the fouling effects (Renou et al., 2008). This is when solids stick to the membrane surface and increase the pressure required to pass through leachate. The fouling is more problematic if the pore size is smaller. A second issue is the handling of material which does not pass through the membrane; the concentrate. The concentrate will contain all the components which are removed.

Four different classes of membrane treatment exist; microfiltration, ultrafiltration, nanofiltration and reverse osmosis (Renou et al., 2008). These four classes are divided by pore size. Microfiltration has the largest pore size while reverse osmosis has the smallest. Understandably the different types of filters also have different removal efficiency.

Renou et al. (2008) established that microfiltration alone was not sufficient for landfill leachate treatment whilst the other options were plausible. The main usage of ultrafiltration is for removal of solids (Ehrig and Robinson, 2010). Nanofiltration is seldom used for leachate treatment. The pores are not small enough to remove all unwanted contaminants. Naturally, reverse osmosis is the most potent membrane type, having the smallest pore size. It is also the most common membrane treatment process for leachates.

Pilot scale treatment plants using reverse osmosis have reached removal efficiencies near 100 % of COD, independent of the incoming concentrations (Chianese et al., 1999). Toxic metals are also effectively removed. Pre-treatment of the leachate to avoid fouling is necessary in full scale plants (Swedish EPA, 2008a).

According to Ramakrishnan et al. (2015) studies have shown high removal efficiencies (>75 %) for a few selected organic contaminants when using membrane technology. However, the high content of contaminants in the concentrate was seen as problematic. Oxidation processes was suggested as after-treatment for the concentrate.

Metal ions are too small to be removed in membranes coarser than nanofiltration. If a polymer is used in combination with a membrane, it is however possible to remove metals using ultrafiltration (Fu and Wang, 2011). Ortega et al. (2008) studied the removal of metals from leachate from soil washing using nanofiltration and reached 100 % removal of Cu and 99 % removal of Zn.

2.3.3 Biological treatment

Biological treatment methods are primary used to reduce the amount of organic material and nitrogen in the leachate (Renou et al., 2008). The biological treatment methods take advantage of microorganisms capable to convert the unwanted compounds into gases and biomass. The treatment can occur in either designated treatment plants or in wetlands constructed for this purpose.

The main reason for using biological treatment methods is not to remove organic contaminants. However, since biological treatment removes organic material, parts of the contaminants might be removed at the same time. Biological processes may be hampered or damaged by the presence of organic contaminants and toxic metals, affecting the overall performance of a treatment facility.

Traditional activated sludge processes are not well-suited for treatment of leachate (Lin et al., 2000). This is because ammonia and refractory organic contaminants reduce the efficiency of the biological processes. Furthermore, variations in flow is problematic for traditional activated sludge processes.

This chapter describes the existing biological treatment techniques for leachate, with focus on the effect of organic contaminants. The chapter presents active techniques such as sequenced batch reactors and moving bed biofilm reactors, as well as passive techniques as wetlands. The chosen techniques are those seen as suitable for treatment of landfill leachate.

Sequenced batch reactor

The sequenced batch reactor (SBR) is an activated sludge process in which nitrification, denitrification and sedimentation occur in the same tank (Singh and Srivastava, 2011). A SBR operates in cycles. The tank is repeatedly filled and emptied with wastewater or leachate, while the sludge remains within the tank throughout the cycles. By changing the conditions within the tank the different biological processes can take place. An operation cycle of a SBR consists of five stages; fill, react, settle, draw and idle.

Bacteria in a SBR requires a suitable environment (Heander, 2007). The nitrification bacteria are more efficient at higher temperatures, optimal is a temperature between 22-30°C. Denitrification bacteria can run at a wider temperature range but are more efficient at higher temperatures. The nitrification bacteria requires proficient oxygen concentration, pH, temperature, sludge age and substrate concentration. By giving time for bacteria to grow when starting up a SBR, the facility will be less sensitive if changes in important parameters occur.

Sequenced batch reactors have several advantages which makes them suitable for treating landfill leachate (Diamadopoulos, 1997). Their flexibility allows them to be adjusted to handle varying flow and composition of leachate. The performance is easily controlled. Since they only are filled with leachate during a part of the cycle, a SBR should be preceded by a storage pond, or similar (Swedish EPA, 2008a).

The primary purpose of a SBR is generally to remove ammonia from the leachate (Swedish EPA, 2008a). However, it will also reduce the content of BOD, COD and organic compounds. Singh and Srivastava (2011) states that the SBR shows potential for removal of toxic organic contaminants, however this requires the sludge to be in a granular form. If activated carbon is added to the SBR the removal of organic contaminants is also improved. In a lab study on phenol contaminated groundwater it was seen that phenol was efficiently removed with a removal rate of 99% using a sequencing batch biofilm reactor, SBBR (Farabegoli, et al. 2008). Phenol has also been sufficiently removed (80 %) in synthetic wastewater, with high concentrations of phenol (Singh and Srivastava, 2011).

Removal of metals is also possible using SBR. A study of Pb and Ni reported 80-86 % removal of Pb and 77-83 % removal of Ni using an SBR (Sirianuntapiboon and Ungkaprasatcha, 2007). When activated carbon was added the removal efficiency was

slightly lower, 76–84 % for Pb and 74–79 % for Ni. However, according to a compilation of Swedish studies made by Cerne et al. (2007), toxic metals are only removed to some extent in a SBR. The same compilation also included a few measurements of organic contaminants. Removal of aliphatic hydrocarbons varied between 20 – 100 %.

Moving Bed Biofilm Reactor

The moving bed biofilm reactor, MBBR, is an attached growth treatment method, in which microorganisms grow on porous carrier material (Loukidou and Zouboulis, 2001). The carrier material is submerged in tanks where the leachate is in constant movement to ensure good mixing. The carriers can be made from a wide variety of materials (Dong et al., 2011). Plastic material can be used, as well as inorganic carriers or an adsorbing material such as granular activated carbon. Nitrification and denitrification bacteria both grow on the carriers but separated in different tanks with different conditions.

An advantage of the MBBR compared to activated sludge processes is that the former is less sensitive to toxic compounds in leachate (Loukidou and Zouboulis, 2001). The MBBR also allows for higher concentration of biomass (Renou, et al., 2008).

Studies report a removal rate of 85–90 % for nitrogen and 60–80 % for COD when using MBBR treatment (Aziz et al., 2014). Organic compounds could effectively be removed if the carrier material is granular activated carbon (Renou et al., 2008). Dong et al. (2011) studied the usage of MBBR for treating oily wastewater containing large amounts of petroleum hydrocarbons. Their study used a ceramic carrier and compared it to regular wastewater sludge. The removal of PAH in the oily wastewater was varying between 65–70 % for the setup using ceramic carriers. Aliphatic hydrocarbons had a removal efficiency between 70–85 %. The setup using wastewater sludge had a 52 % removal of PAH and 55–60 % removal of aliphatic hydrocarbons.

Constructed Wetlands

Constructed wetlands use natural processes to treat leachate (Kandasamy and Vigneswaran, 2009). Wetlands can be the only treatment step, or used in combination. By constructing a well-designed wetland, natural processes are speeded up. A constructed wetland has a bottom liner that protects the groundwater (Ehrig and Robinson, 2010). The wetland is filled with gravel, sand and/or soil. For leachate wetlands, one-sized gravel smaller than 10 mm is most frequently used. The selected plants in the wetland should promote natural processes, such as bacterial oxidation and filtration. Leachate from a landfill can flow and infiltrate through the constructed wetland while BOD, COD, nutrients and particles are removed, see Figure 4 (Kandasamy and Vigneswaran, 2009). The removal efficiency is varying between seasons (Nivala, et al. 2007). Still, COD removal above 50 % is possible. In Sweden, at low temperature seasons constructed wetlands can be problematic for treatment of leachate (Swedish Waste Management, 2015).

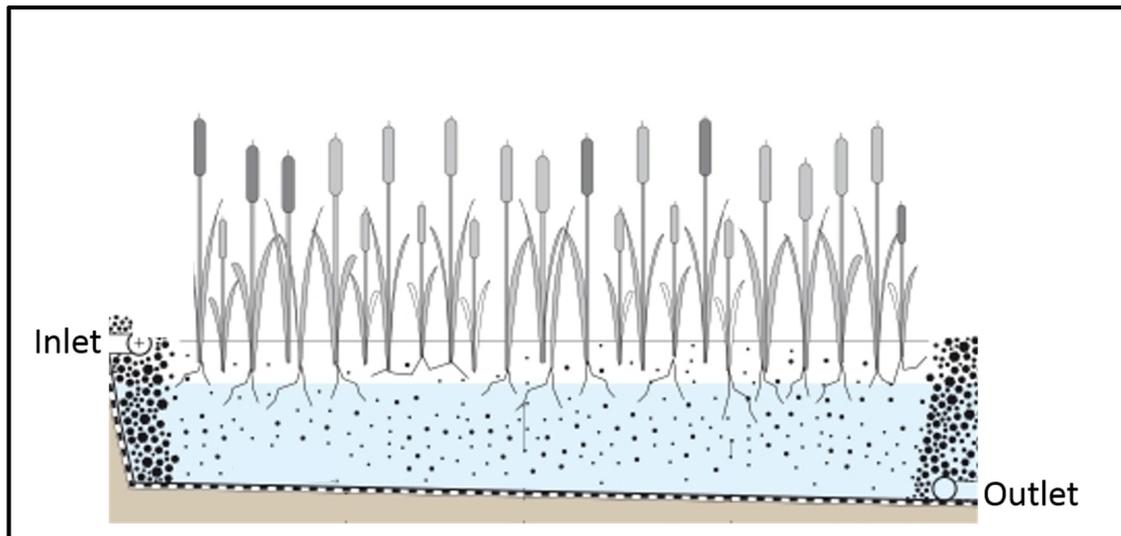


Figure 4 A constructed wetland. The leachate filters through the plant roots and pollutants can settle to the bottom (Modified from Tilley et al., 2014).

The BOD and COD are degraded by biological processes such as photosynthesis (Kandasamy and Vigneswaran, 2009). The nutrients in the leachate are taken up by the plants. When the plants die the nutrients becomes sediments in the wetland. The plants create big surface areas for biofilm to grow and are therefore very important for the function of the wetland. Nutrients can also be directly absorbed by the soil. The bigger particles in the leachate are often caught by the plants and roots and filtered away.

Metals can be removed by different processes (Kandasamy and Vigneswaran, 2009). If the pH is high metals will precipitate and settle. Metal ions can bind to plants and soil by adsorption or it can oxidise at the oxygen rich surface (Kandasamy and Vigneswaran, 2009; Bydén et al., 2003). Trace metals are mainly removed by precipitation and adsorption and also by creating complexes with organic material. A complex is a metal (or ion) surrounded by oppositely charged molecules or ions. Studies report that while the removal of metals may be efficient, the accumulation in the wetland is problematic. From landfills in the United States removal rates of toxic metals in constructed wetlands have been high: 94 % Pb, 89 % As, 88 % Ni, 93 % Zn, 91 % Mg, 75 % Hg (Vigneswaran and Sundaravadivel, 2001). For Fe the removal rate was 97 %.

Organic contaminants are also partly removed in wetlands (Kandasamy and Vigneswaran, 2009). From a study by Wojciechowska, (2013) it was confirmed that PCBs and PAHs from leachate was mainly reduced by adsorption and deposition to sediments in the constructed wetland. With a removal rate of 94 % for PAHs and 91 % for PCBs. The most important parameter for removal of organic contaminants is to have a long retention time (Zahraeifard and Deng, 2011; Wojciechowska, 2013).

The downside of using constructed wetlands to treat leachate contaminated with organic contaminants and/or metals is the accumulation of toxic compounds in the sediments (Engin et al., 2015). To maintain the wetland and reduce risk of wash-out of contaminants the sediments needs to be remediated when polluted (Peng et al., 2009). If the sediments contain toxic or carcinogenic compounds, or has some other

hazardous effect, it needs to be handled according to the waste conventions and placed in a hazardous landfill if needed (Swedish Waste Management, 2007). A sustainable treatment of highly contaminated leachate can therefore be difficult to achieve with constructed wetlands. It has been seen that leachate discharged to lakes have affected wildlife negatively (Swedish EPA, 2008c). Therefore another downside of using constructed wetlands is that animals that have their habitat in or nearby the wetland will likely be exposed to pollutants.

Aerated lagoons

An aerated lagoon is a simple treatment method often used for treating leachate (Ehrig and Robinson, 2010). It requires low maintenance and can treat varying flows. An aerated lagoon is simply a constructed pond, with water-tight sides and bottom (Swedish EPA, 2008a). The lagoon is aerated using surface aerators or by diffusing bubbles into the water (Mehmod, et al., 2009). The aerated condition enables removal of material by chemical and biological oxidation. Organic material, as well as nitrogen, is removed in a well-functioning aerated lagoon.

The efficiency of an aerated lagoon is dependent on the temperature of the leachate (Ehrig and Robinson, 2010). A long term average temperature above 15°C is necessary for a good removal rate. If the temperature is below 10°C, removal of nitrogen will not be efficient. Removal of organic matter is significantly hindered below 5°C. During good conditions a long term removal rate of 97 % for COD was reported in a full scale facility (Robinson, Grantham, 1988). Lower removal of COD is generally reported, ranging from 55–75 % (Renou et al., 2008; Mehmod et al., 2009); and a nitrogen removal of 80 % is possible to obtain.

Studies have reported an 80–88 % removal of phenols when treating leachate using an aerated lagoon (Renou et al., 2008). From a study made at a WWTP, consisting of aerated lagoons and sedimentation ponds, it was seen that metals was precipitated and adsorbed to the sediments (Chabir et al., 1997). The concentration of toxic metals was 1000–7000 times greater in the sediments than in the wastewater.

The Swedish EPA states that aerated lagoons are most suited as pre-treatment for leachate (Swedish EPA, 2008a). From the literature study it can be concluded that their temperature dependency make them unsuited for Swedish conditions. Proper treatment should only be expected during the summer months.

2.4 Sludge

During the treatment of leachate, particles, organic contaminants and metals are removed. These particles and contaminants constitute the sludge, which is a byproduct from the treatment processes. Before the sludge has been further treated the water content is high, usually around 95% (Werther and Ogada, 1999). In some cases the water content can be even higher, around 99 % (Chen et al., 2002). The sludge resembles slurry. If the leachate treatment is efficient the sludge contains high concentrations of unwanted compounds. For example, metals precipitate with the particles and bind to the sludge. Studies show that over 90% of the metals eventually end up in the sludge (Gray, 2010). This is also the case for many organic contaminants, as they are lipophilic and concentrate in the sludge; in a study at a

WWTP some PCB compounds had a high sludge adsorption rate, over 90% (Olofsson et al., 2012; Olofsson et al., 2010). The polluted sludge needs to be collected and treated to reduce the risk of spreading to the environment. The composition of the sludge is dependent on the type of treatment process that it originates from (Gray, 2010). Sludge from activated sludge processes have lower water content and are more stable than sludge from primary settling processes.

Sludge can be characterised to gain a better understanding of its attributes (Carberry and Englands, 1983). The main reason for characterisation is to understand the water binding capacity. This is important when designing for an optimal sludge handling and treatment process. Sludge can be characterised chemically, biologically and physically. For landfill leachate sludge less focus is on biological characterisation since it does not contain as much pathogenic microbial as wastewater sludge does.

2.4.1 Water fractions in sludge

Sludge created in treatment processes contains large amounts of water. The water is bound to the sludge in different ways, therefore not all water can be removed with the same methods. There is no standard classification of the water in sludge; different literature use different names and methods (Lee and Hsu, 1995).

One system for classifying water in sludge was proposed by Vesilind and Martel in 1990. The system has been adapted by other authors as well (Lee and Hsu, 1995; Chen et al., 2002). Vesilind and Martel (1990) divided the water in the sludge into four different groups, which are shown in Figure 5 and presented below.

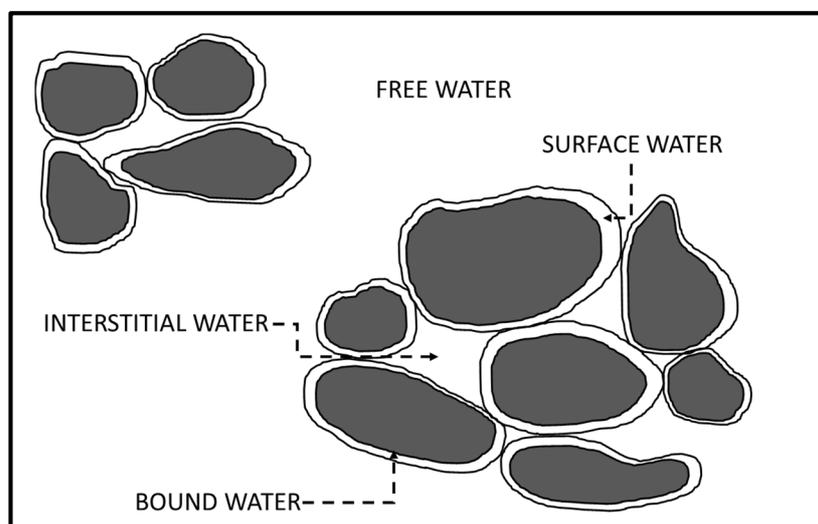


Figure 5 Illustration of different water fractions in sludge. Based upon original by Vesilind and Martel (1990).

Free water. The water surrounding the sludge particles is called free water (Vesilind and Martel, 1990). This water is the easiest to remove and does not follow the sludge particles. Therefore, methods based on settlement can be used to remove large parts of the free water.

Interstitial water. Vesilind and Martel (1990) defined interstitial water as water contained within the sludge flocs. It follows the sludge particles unless the flocs are broken down mechanically. Interstitial water is the theoretical limit of what can be removed by mechanical dewatering. Studies made by Tsang and Vesilind (1990) tested different dewatering methods, such as gravity thickening and centrifugation. Their results showed that none of the mechanical methods were able to completely remove all of the free and interstitial water.

Surface water. Surface water cannot be removed by mechanical dewatering (Vesilind and Martel, 1990). It is water that is bound to the surface of the sludge particles by surface forces. Its existence is discussed, and often the term vicinal water is used instead of surface water (Novak, 2006).

Bound water. The bound water is water that is chemically bound to the sludge particles (Vesilind and Martel, 1990). The only way to remove the bound water is to destroy the sludge particles completely by heating or addition of chemicals.

Literature reports that sludge from sewage generally contains 70–75 % free water, 20–25 % interstitial water, 1 % surface water and 1 % bound water (Werther and Ogada, 1999). Different numbers can be seen in Gray (2010), where the free water is said to constitute 70 %, interstitial water 20 %, surface water 1 % and bound water 8 %. Of course, different authors have slightly different definitions of the types of water, meaning that the numbers should be interpreted carefully.

2.4.2 Treatment of sludge

The sludge from a treatment plant comes from different treatment steps but is often collected and stored in one tank (Gray, 2010). The sludge from the different steps, such as chemical precipitation/coagulation or biological treatment, has different composition. For optimisation the sludge should be treated in adjusted ways. Mainly of economic reasons the sludge is collected, mixed and treated as one. The treatment generally consists of stabilisation and dewatering, both which will be described in more detail below.

Stabilisation of sludge

The sludge can be stabilised in order to reduce water content, odour and to improve its characteristics (Werther and Ogada, 1999). The most common method of stabilisation for sludge from WWTP in the European Union is digestion. Digestion uses anaerobic microorganisms to degrade organic matter to water and carbon dioxide. In the process methane is released. For sludge with high content of pollutants, digestion can be problematic since the digesting bacteria suffer in the toxic environment. Lime stabilisation is another method used, in which the pH is raised in the sludge by addition of lime. At highly alkaline pH the micro-organisms in the sludge will perish. A third stabilisation option is heat treatment, where the sludge is heated in two steps to stabilise it.

Dewatering of sludge

Sludge with high water content occupies a lot of space and is laborious to transport (Werther and Ogada, 1999). It is therefore necessary to reduce the water content in the sludge before it is disposed or reused. Water can be removed by different dewatering methods; mechanically, by heat treatment or by thermal drying. The dewatering can be improved by chemical additives, this is called chemical conditioning. Which dewatering method is the most suitable is largely depending on the properties of the sludge studied (Novak, 2006). The different methods are described below.

Firstly, the simplest way to reduce the water content in sludge is by using gravity thickening. In this method, the sludge is allowed to settle during quiescent conditions. The solids concentration can increase by a factor of 2–5, depending on the type of sludge and the initial solids concentration (Gray, 2010). The thickening can be improved by stirring the sludge slowly as it settles.

Additional reduction of the water content can be obtained by using a drying bed. The sludge is placed on top of a bed where the water is allowed to drain and evaporate (Gray, 2010). During good conditions it is possible to obtain a solids content of 40 % in a drying bed. A downside of the method is that it causes odour, since the sludge is openly left to dry. It is also restricted to seasons during which the weather conditions are suitable, i.e. not during the winter months.

Mechanical dewatering is commonly used for sludge. Different mechanical dewatering methods can reduce the water content to a level where the sludge behaves as a solid instead of a liquid. Usually, conditioning is used to improve the mechanical dewatering process (Werther and Ogada, 1999). One method that is used is a belt press, which consists of two moving belts, one impermeable and one porous. The sludge is continuously added to the device where it is compressed by the belts. Modern equipment can increase the solids content in the sludge to 35 % (Gray, 2010).

A filter press is similar to a belt press, in the sense that they both use pressure to compact the sludge. However, filter presses are not continuous. They consist of metal plates forming chambers into which the sludge is pumped. The pressure within the chambers is increased and water is extracted until the water flow ceases. The solids concentrations are between 30–45 % when using filter presses (Gray, 2010).

Centrifugation is another method used for mechanical dewatering of sludge. The sludge is entered into a rotating drum, where the liquids are separated from the solids (Novak, 2006). The dewatering capacities of centrifugal filters are similar to those of belt presses (Werther and Ogada, 1999).

2.4.3 Management of treated sludge

Stabilised and dewatered sludge can be recycled to agriculture, placed in a landfill, combusted or reused on e.g. road embankments (Gray, 2010; Ferrer, et al. 2011). The preferred option is to place the sludge on agricultural land (Svenskt Vatten, 2014). This way phosphorus and nitrogen can be recirculated back to the environment. This is not always possible since sludge from leachate often contains high concentrations of toxic metals and organic contaminants (Allmyr and Sternbeck, 2014). There are

directives stating the quality the sludge needs to reach in order to be used as a fertilizer in agriculture. In the past, sludge was sometimes dumped into the sea (Gray, 2010). This was stopped 1998 after a directive from the European Union. Alternative usages for sludge are described below.

Landfilling. Sludge can be placed in a landfill, the downside is the storage space required and a possible non-sustainable recirculation of pollutants to the leachate (Cyr, 2012). Sludge from leachate treatment contains more hazardous pollutants than wastewater sludge (Allmyr and Sternbeck, 2014), and therefore leachate sludge may need to be placed in a landfill for hazardous goods if the sludge has toxic, carcinogenic or in other ways hazardous properties (Swedish Waste Management, 2007).

Incineration. Incineration of sludge generates energy (Gray, 2010). Dried sludge has relatively high energy content. The bonds in the organic compounds break during the combustion. The metals are then the only hazardous content in the ashes. The volume of the sludge is reduced significantly. However, because of toxic metals the ashes needs to be placed in a special landfill for hazardous goods. There are research in metal separation from ashes, leachates and contaminated soils. One developing method of metal separation is electrolysis from ashes. Tao et al. (2014) have in laboratory tests reached high removal rates (>95 %) of Cu, Zn and Pb.

Reuse in materials. Reuse of sludge is mainly possible when the sludge contains a low concentration of pollutants (Svenskt Vatten, 2013). The dried sludge can then be used for cover of road embankments (Ferrer, et al. 2011). It is also possible to use the ashes from incinerated sludge in concrete and other building materials (Cyr, 2012).

3 Leachate treatment plants at other locations

The removal rate for organic contaminants varies between different leachate treatment plants. Factors that affect the treatment capacity are incoming leachate quality and quantity, temperature and treatment plant design, among others. The leachate pilot treatment plant at Brudaremissen is relatively advanced with many treatment steps, see Chapter 5. By comparing the removal rate achieved at Brudaremissen with other leachate treatment plants the degree of efficiency can be analysed. The leachate treatment plants used for comparison are presented below. One treatment plant is from Japan and two are from Sweden.

Landfill M, Japan

A study of two plants located in Japan, similar to the pilot plant constructed at Brudaremissen, was made by Kim and Oshako in 2002. The plants treated leachate from two different landfills, named M and N, of which M is of interest for this study. The treatment plant at landfill M consisted of biological treatment, coagulation, sand filtration and adsorption using activated carbon (Kim and Oshako, 2002). Measurements of the concentrations of PAHs and PCBs were made between each treatment step. Also, measurements of common parameters such as carbon content were made.

The PCB concentrations were measured both in the dissolved phase and in the adsorbed phase, while PAHs were only measured in total (Kim and Oshako, 2002). PCBs were mostly adsorbed to particles in the leachate. The content of PCBs in the influent was 0.058 ng/L in the dissolved phase and 0.17 ng/L in the adsorbed phase. The particulate adsorbed PCBs were effectively removed using biological treatment. Dissolved PCBs had a lower removal rate in the biological treatment, however they were removed in the coagulation step. The total removal of PCBs was approximately 93 % in the plant. The removal of PAHs varied; many PAHs had a removal rate above 97 % in the plant while some were less efficiently removed. The measurements showed an increase in some compounds between the sand filters and the adsorbed carbon step. The removal efficiencies obtained in the treatment plant are shown in Table 5. No differentiation is made between adsorbed and dissolved PCBs.

Table 5. Concentrations and removal efficiencies in Landfill M in Japan. Based on information from Kim and Oshako (2002).

Parameter	Unit	Before	After	Removal [%]
COD	mg/L	42	1.7	96
TSS	mg/L	9.0	0.50	94
PAH L	ng/L	3700	300	92
PAH M	ng/L	680	13	98
PAH H	ng/L	16	4.0	75
PCB	pg/L	230	17	93

Vankiva, Hässleholm

The large waste management facility Vankiva is located in the municipality of Hässleholm in southern Sweden. Waste at Vankiva is sorted and handled appropriate depending on its characteristics (Hässleholm Miljö AB, 2015). Hazardous and nonhazardous waste are treated in separate parts of the facility. The facility is modern, however it also contains an old landfill which is currently being covered. Leachate from the facility is treated in a plant consisting of aerated lagoons, SBR and sand filters (Hässleholm Miljö AB, 2015). Prior to the sand filters, iron chloride is added to precipitate phosphorus from the leachate. The treated leachate is led to a nearby stream, Prästabäcken, which flows into the larger river Almaån. Almaån is an important and sensitive recipient, however studies have concluded that the landfill leachate has a limited impact on the river (ProVAb, 2015). The treatment facility has occasionally had problems when experiencing large leachate volumes (ProVAb, 2015). The SBR did not function properly at these times, leading to exceedance of the environmental limits for phosphorus, nitrogen, BOD and TOC. The treatment plant has been part of a large project carried out by the consultancy firm Sweco and the Swedish EPA¹. The study evaluated and measured contaminants at six different landfills in Sweden. Leachate samples were obtained at Vankiva before and after the SBR and they were analysed for a wide range of compounds, including organic contaminants. A fraction of the result from the analysis is shown in Table 6.

Table 6. Parameters at Vankiva landfill, before and after treatment. Leachate samples were obtained by Sweco and the parameters were measured by an accredited laboratory during 2011-2012.

Parameter	Unit	Before	After	Removal [%]
TOC	mg/L	200	58	71
N	mg/L	84	12	86
TSS	mg/L	93	7.6	92
Aliphates >C16-C35	µg/L	39	0.99	97
PAH L	µg/L	0.34	0.0031	99
PAH M	µg/L	0.54	0.0008	100
PAH H	µg/L	0.05	0.00	100
Fe	µg/L	12000	1400	88
Pb	µg/L	28	3.9	86
Cd	µg/L	2.4	0.23	90
Cu	µg/L	110	19	83
Cr	µg/L	69	53	23
Ni	µg/L	89	50	44
Zn	µg/L	360	39	89

According to the results from the study, the treatment plant at Vankiva efficiently removes organic contaminants such as PAHs and aliphatic hydrocarbons. The removal of toxic metals is generally 80 – 90 %, however the treatment is not as

¹ Sara Svärd (Miljösamordnare, HKC) phone interview by the authors 2015-03-11.

efficient for Cr and Ni. The degree of removal is 71 % for TOC. The treatment removes 92 % of the total suspended solids in the leachate.

Sofielund, Huddinge

Sofielund landfills are located close to Huddinge in Sweden (Sernstad, 2014). There are three landfills in Sofielund. The oldest landfill was active 1939 – 1977, the second one 1978 – 2006 and the third landfill is active today and opened in 2005. The landfills contain mixed waste for example ashes and asbestos. The leachate from the three landfills are collected and pre-treated on-site (Sernstad, 2014). The pre-treatment consists of equalization tanks, drum screen, aerated pond, sedimentation pond, chemical precipitation and sand filter (Sernstad, 2014; Stockholm Vatten, 2014). The obtained sludge from pre-treatment is placed on reed beds.

After pre-treatment the leachate is discharged to Henriksdal wastewater treatment plant (Sernstad, 2014). The pre-treatment removes contaminants sufficiently and thereby Henriksdal WWTP is Revaq certified. In Table 7 some measured toxic metals and organic contaminants are shown. The data is taken from measurements in 2012 - 2014. For the organic contaminants the removal rate is sufficient and the effluent has satisfactory quality. For the toxic metals the removal is varying. For Hg, As, Pb and Cd the treatment removes enough and discharge levels are reached, see Appendix 1. For Cu, Cr, Ni and Zn the discharge concentrations are higher. If these are compared to Swedish EPA's environmental classes for lake and freshwater the classes would be ranging from moderate to very high, see Appendix 1. However the leachate is further treated at Henriksdal wastewater treatment plant.

Table 7 Removal efficiencies at Sofielund landfill before and after treatment. Values obtained from internal work material from Stockholm Vatten.

Parameter	Unit	Before	After	Removal [%]
Phenol	µg/L	25	6.4*	74
Sum PCB 7	µg/L		Below detection (<0.01)	
DEHP	µg/L	2.5	1.2	51
Nonylphenol	µg/L	0.77	0.18	77
Aliphates >C16-C35	mg/L		Below or close to detection (< 0.050)	
Oil Index	mg/l	1.1	0.24*	77
PAH L	µg/L	0.72	<0.2	72
PAH M	µg/L	0.51	<0.3	45
As	µg/L	5.3	3.7	30
Pb	µg/L	3.3	1.2	64
Cd	µg/L	0.35	0.093	73
Cu	µg/L	35	87**	-150**
Cr	µg/L	35	27	21
Ni	µg/L	62	60	3,7
Zn	µg/L	150	54	64

*One or more measurements were under detection limit but still included as the detection limit value

**A leakage of Cu is occurring in the treatment facility

4 Brudaremossen Landfill

The landfill that serves as a case study in this report is called Brudaremossen. From previous studies made at Brudaremossen, high concentrations of organic contaminants and iron have been measured in the leachate (Lundh, 2015a). The leachate has previously been discharged to Ryaverket wastewater treatment plant. Ryaverket is working with the Revaq certification and as a step, point-sources needs to be disconnected. Brudaremossen is one of the point-sources affecting the sludge quality at Ryaverket WWTP (Porse, 2013). A new pilot treatment plants have therefore been built at Brudaremossen, aiming to achieve a satisfactory removal of iron, organic contaminants, nutrients and toxic metals.

In this chapter background history of Brudaremossen landfill, protective measures, previous leachate treatment and a summary of previous studies are presented.

4.1 Brudaremossen background

The closed landfill Brudaremossen is located in the municipality of Gothenburg, in the region of Västra Götaland. Brudaremossen is situated east of the city, in the recreational area of Skatås and the Delsjö Lakes. The landfill is in the outskirts of a natural reserve (County Administrative Board of Västra Götaland, 2015) and also close to the raw water source for Gothenburg municipality (City of Gothenburg, 2015b). The recreational area is a popular location for, among others, running, biking, hiking and swimming. Winter time it is possible to go downhill skiing on Brudaremossen landfill. A map of the landfill is seen in Figure 6.

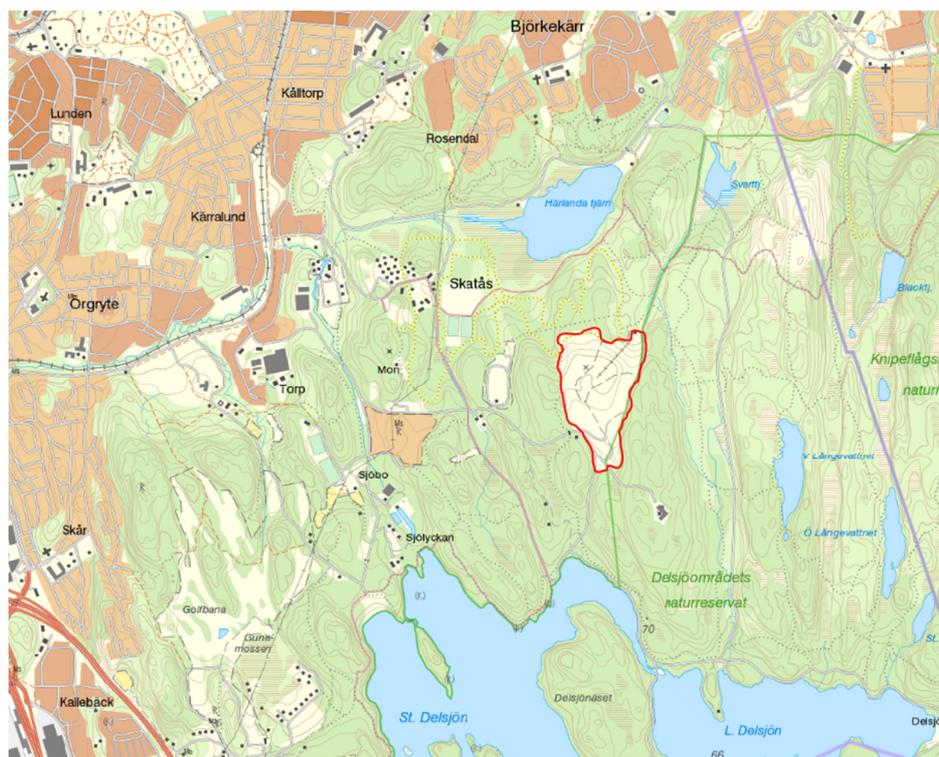


Figure 6 Map over Skatås recreational area and Brudaremossen landfill, marked out in red. (Modified from Lantmäteriet, 2015).

The landfill stretches over 20.5 ha (Jansson et al., 2012). The maximum depth of the disposed waste is estimated to 40 m and a hill exists in its western part, at 130 m above sea level according to the reference system used by the City of Gothenburg. Brudaremossen was originally constructed on top of a large bog (Nielsen, 2007). The original surface has compressed due to the mass of the landfill.

The landfill is covered by low vegetation and the surrounding area consists of a mix of foliferous and coniferous trees (Swedish EPA, 2015b). Smaller bog areas and wetlands can be found in its surroundings (Nielsen, 2007). The soil layers in the Brudaremossen area are mostly thin, and dominated by outcrops of rock and patches of postglacial sand (SGU, 2015). The bedrock is pre-Cambrian and consist mainly of gneiss (Nielsen, 2007). The climate in southern Sweden is cold and the precipitation in the area is around 800 mm annually (SMHI, 2014). The average temperature in the area of Brudaremossen is - 2°C in winter and 15°C in summer.

Brudaremossen landfill was in use between 1937 and 1978 (Nielsen, 2007). Various wastes were disposed in the landfill during its active years. It is known that household waste, industrial waste and construction waste was put in the landfill, however detailed information does not exist. Aerial photos reviewed in a previous study revealed oil lagoons on top of the landfill in the end of its active years. Pits containing alkali material can also be seen on old photos. Due to the mixed waste in the landfill and its age it is likely that hazardous materials and chemicals are present in the landfill. Eventually the harmful substances end up in the leachate.

4.1.1 Protective measures

A number of protective actions have been taken at Brudaremossen to reduce risks of contamination and accidents caused by the material in the landfill. A concrete wall was installed in the beginning of 1940's to reduce the spreading of leachate towards the Delsjö Lakes (Fersters, 2003). The landfill was originally drained to a nearby stream, in 1962 the drainage system was instead connected to the wastewater system of Partille municipality. A decade later the leachate started to be led to the recently constructed WWTP Ryaverket. The collection system allowed a maximum leachate flow of 10 l/s (Nilsson, 2001). Additional leachate volumes were lead to a nearby stream, Finngösabäcken. An improvement of the drainage system in the southern part of the landfill was made in the 1960's, when its height had become larger than the concrete wall.

It is reasonable to assume that the landfill is layered (Nielsen, 2007). During its lifetime the surface was occasionally flattened and covered by impermeable materials. This might have created different conditions for the groundwater flow. After the landfill was closed in 1978 it was covered. A clay layer was put on top of parts of the landfill in the beginning of 1990. The layer of clay does not cover the entire landfill. The clay reduces the amount of water which infiltrates into Brudaremossen, and therefore also the produced leachate. Prior to the coverage, the drainage system in the landfill was improved (Nilsson, 2001). Approximately 1800 m drainage pipes were dug down and improvements were made to surrounding ditches. The landfill itself is a natural water divider (Lundh, 2015a). However, the drainage systems installed collects the water which otherwise would flow to the south and transports it to the northern pond.

4.1.2 Modern protective measures - leachate treatment

Due to the way the drainage system is constructed the majority of the leachate from Brudaremassen is released in the northern part of the landfill (Jansson et al., 2011). A local treatment consisting of an oil separator and a pond exists, however the water is ultimately led to Ryaverket. The location of the oil separator and the pond can be seen in Figure 7.



Figure 7 Aerial picture of the Brudaremassen landfill. The pond is marked out in red. (Modified from City of Gothenburg, 2015a).

The oil separator was constructed as part of a rearrangement of the drainage system in 1996 (Nilsson, 2001). An upgrade to the oil separator was made in 2007 (Lundh, 2015b). A sludge separator was installed as well. The main purpose of the oil and sludge separator is to reduce the content of oil and sludge in the leachate, which is beneficial for the upcoming treatment steps. Previous studies show that the oil separator is functioning properly.

After the oil separator the leachate is led to a pond. The pond was originally constructed in 1996 and a major upgrade was made in 2005 (Nilsson, 2001; Nielsen, 2007). The bottom of the pond was covered by textiles to ensure that no leachate could infiltrate the groundwater (Lundh, 2015b). Drainage was installed beneath the pond to decrease the risk for bottom heave. The pond reduces a few of the pollutants in the leachate, mostly organic contaminants. It is also serving as a detention pond and is levelling out the flow of leachate to the wastewater system. The capacity of the pond is approximately 500 m³ (Lundh, 2015a). The pond was emptied and the sediment was removed during the fall of 2014².

² Lars-Ove Sörman (Responsible for plant operations) phone interview by the authors 2015-05-19.

From the pond the leachate is led to the municipal WWTP. The capacity of the system is limited (Nielsen, 2007). In case of large leachate volumes it is overflowed into a nearby wetland. The wetland is located north of the treatment pond. Previous investigations have shown that the state of the wetland and its vegetation is varying between seasons. It is plausible that the quality of the treatment of the leachate within the wetland is also varying. The previous studies also stated that overflows occur frequently during the year.

4.2 Environmental criteria and guidelines

The discharge from Brudaremossen needs to reach guideline quality values in order to be released into a recipient. A number of different guidelines and legislations exist regarding the quality of water. This chapter will present a few of the sources for guideline values applicable for the studied project at Brudaremossen. Firstly, the Revaq certification is presented. Afterwards different guidelines from the City of Gothenburg, the Swedish EPA and the European Union are introduced. Thirdly, Canadian guidelines for water quality are presented. The Canadian guidelines are useful in those cases when European guidelines do not exist. The climate of Canada is similar to that of Sweden, and therefore the guidelines should be applicable. Lastly, if no other reference values were found, American guidelines were used for comparison. The compilation of different guidelines are found in Appendix 1.

4.2.1 Revaq – certification of sludge for agricultural reuse

As described previously, there is a desire to reuse the sludge from the wastewater treatment plants in agricultural fields. By doing this the nutrients in the wastewater are recirculated. In Sweden, sludge from treatment plants can receive a Revaq certification, which shows that the sludge is approved for agricultural reuse (Sternbeck et al., 2014). The Revaq certification system is a cooperation between water authorities, food authorities and farmers associations.

The requirements to obtain a Revaq certification exceed the regulations in Swedish legislation. Transparent work and constant improvement of the treatment plants are demanded, as well as good sludge quality (Svenskt Vatten, 2015). Before certification, a number of trace elements must be analysed in the sludge and their possible origins should be investigated. One of the most important parts of Revaq is the handling of upstream sources of contaminants (Svenskt Vatten, 2014). Point sources such as landfills should be disconnected from municipal WWTP if it is technically and economically manageable. Exceptions are allowed if it can be shown that the leachate from a landfill has a limited impact on the quality of the sludge.

According to the rules of Revaq a compound originating from leachate should not constitute more than 1 % of a low risk level of the compound in the sludge (Allmyr and Sternbeck, 2014). If that is not the case, the leachate must be disconnected or treated before it is sent to the plant. Not all compounds must be investigated; it is enough to study twelve prioritised organic compounds. The compounds and their corresponding low risk level are shown in Appendix 1. However, if there is known that the leachate contains other compounds which easily bind to sludge, they should also be investigated.

4.2.2 City of Gothenburg – guidelines for emissions to water

The environmental office at the City of Gothenburg has created guideline values for emissions of contaminated water to recipients and stormwater (Miljöförvaltningen, 2013). The purpose of the guidelines is to simplify for companies and owners who handle contaminated water. The guidelines contain concentrations for metals, organic compounds and a few other parameters. The concentrations are valid for the emissions at the source, e.g. not in the recipient. The guidelines should be followed; unless it can be shown that the emissions are harmless to the recipient. The environmental office decides whether this is the case or not.

4.2.3 Environmental quality criteria from the Swedish EPA

The Swedish Environmental Protection Agency has created a series of reports containing criteria for evaluation of different environmental sectors (Swedish EPA, 2000). One of these reports deals with surface water quality. The Swedish EPA compares the water to its natural state when evaluating the quality. Guideline values are given for a number of parameters, including nutrients, physical parameters and metals. The parameters are divided into five different classes, normally ranging from low concentration to extremely high concentrations.

The Swedish EPA also has guideline values for contaminated sites (Swedish EPA, 2009). These guidelines contain values for maximum concentrations recommended for sites with sensitive land usage and sites with less sensitive land usage. According to the Swedish EPA, the values can be used for a first step when determining the status of an area. In this project the values are used when evaluating the quality of the sludge produced at Brudaremossen.

4.2.4 European legislation – prioritised pollutants

The European Union have identified pollutants threatening the water quality in the union (European Union, 2013). There are a total of 45 of these prioritised pollutants. The pollutants are often persistent in nature. Guidelines exist regarding the concentrations of pollutants in water, sediment and biota. The guidelines contain values for both annual average and maximum concentration for a number of prioritised pollutants.

4.2.5 Canadian guidelines for water quality

The Canadian Council of Ministers of the Environment, CCME, have extensive guidelines to protect the quality of water and aquatic life (Canadian Council of Ministers of the Environment, 1999). These guidelines are not part of the Canadian legislation, however it is recommended to follow them. The guidelines provide limits for a number of different compounds, including organic contaminants such as PAHs. Limits do not exist for a few compounds, such as PCBs and DDT. This is since there are scientific uncertainties regarding these compounds in water, and it is stated that limits would have little practical use. Everything in the guidelines is based upon scientific studies, mostly from Canada. The CCME states that the guidelines should be adjusted to local conditions when evaluating water quality. That is, if the water quality is excellent it should be kept that way.

4.2.6 United States recommended water quality criteria

The United States Environmental Protection Agency have compiled different studies and investigations to create national recommended water quality criteria (US EPA, 2014). The compilation contains information regarding a wide variety of compounds, including trace metals and specific organic contaminants. Concentrations causing acute and chronic issues for aquatic life are given for every compound. The criteria are meant as a guidance for states in the U.S. when they are creating their individual water quality guidelines. These criteria are used in this study when no suitable value is found in other legislation or guidelines.

4.3 Previous leachate studies at Brudaremossen

A few studies have previously been performed on the leachate from Brudaremossen landfill. Some of them evaluated the performance of old treatment facilities, and some of them characterised the composition of the leachate in general. Four studies are presented here; one which was made by Ramböll to evaluate possible recipients for the leachate, two which evaluated a treatment plant using sorption filters and one which measured a large number of organic trace contaminants in the leachate.

4.3.1 Ramböll investigations

The company Ramböll Sverige AB have been performing studies of the leachate quality at Brudaremossen (Jansson, 2011; Lundh, 2015a). Measurements have been performed four times per year of important leachate parameters such as toxic metals, groups of organic contaminants and standard parameters (alkalinity, pH, oxygen and nitrogen content). The measurements have been made by Ramböll and Sweco Environment AB (Jansson, 2011). The measured parameters' concentrations have been stable over time but with temporary variations, e.g seasonal changes. The measurements showed that the leachate from Brudaremossen contained high concentrations of iron, organic material and nitrogen (Lundh, 2015a). Organic contaminants such as PAHs and PCBs were also in high concentrations. Details from the measurement results at Brudaremossen from the years 2008 – 2012 are found in Appendix 2.

When the Revaq certification disallowed untreated leachate to be led to the WWTP, the City of Gothenburg decided to investigate alternative solutions (Lundh, 2015a). Ramböll was assigned the investigation, which will be finished in 2015. Rambölls investigations are based on available data and previous studies, as well as new investigations. The planning for the disconnection started in 2014 and Ramböll was responsible of:

- Evaluating alternative recipients and finding discharge criteria
- Pre-study of the leachate treatment
- Pre-study of possible discharge route

Ramböll investigated the possibility to release the leachate into the environment or to Ryaverket, after a local treatment (Lundh, 2015a). Three nearby recipients, apart from Ryaverket, were considered in the study. Furthermore, Rambölls study considered different discharge routes to the recipients.

The recipients require different treatment of the leachate since the natural environments vary in sensitivity. The chosen pilot treatment plant was designed to enable an evaluation of discharge to all recipients. Ramböll compared different treatment options and evaluated that chemical precipitation, MBBR, sand and active carbon filters would be a suitable treatment. For the biological treatment, a MBBR was compared to a SBR. The MBBR was chosen because of its small areal footprint, low maintenance and high stability. The oil separator and pond pre-treated the leachate before reaching the pilot treatment plant since it reduced the organic contaminants to some extent (Lundh, 2015a). However it did not reduce the concentration of nitrogen.

4.3.2 Adsorption filter studies at Brudaremsen

A pilot leachate treatment plant was constructed locally at Brudaremsen in 2012. The treatment consisted of two different combinations of filter columns, which were set up after the oil separator and the pond. The filter combinations were: 1) two granular activated carbon, GAC, filters in series, and 2) a peat filter followed by a GAC filter. All filter columns had gravel placed in the bottom. The peat moss was chosen as sorption material since it was viewed as a low cost, environmentally friendly material. The pilot plant was evaluated in two different studies; the first one was a master thesis written by Krewer and Moona (2012) and a published paper by Kalmykova et al. (2014), which was a prolongation of the first study. Both studies focused on removal and degradation of organic contaminants in the pilot plant.

The pilot plant was running for two years. Water samples were collected with the highest frequency for PHCs, metals and water parameters (1/week) and with the lowest frequency for PAHs, phenols and phthalates (1/month). Krewer and Moona (2012) evaluated the results from the first 120 days of the pilot plant. The construction of it was a part of their master thesis work.

During the initial run, the filters were clogged by iron and suspended particles, diminishing their removal efficiency and capacity. Backwashing of the columns using acid was done and a sand filter was installed as pre-treatment to improve the situation. The constant issues with the performance of the treatment plant did not allow for an extended evaluation of the removal rates for different organic contaminants in the master thesis by Krewer and Moona (2012).

The study by Kalmykova et al. (2014) was performed during a longer period of time and therefore it was possible to do detailed studies of the pilot plant. Results showed that the sand filter was not efficiently removing hydrophobic compounds despite that they were expected to be bound to particles. The peat filter had a high efficiency of removing PHCs and PAHs. Phenols were removed by 50 – 80 % in peat filters. The GAC filters removed PHCs and PAHs completely in 50 % of the samples. Also the toxic oxy-PAHs were well removed and were after GAC filter below detection limit. The GAC was more efficient than peat filter in removing organic contaminants. Low molecular weight molecules were removed most efficient in the GAC and the peat filter was more efficient in removing high molecular weight molecules.

The two studies shows that the best combination of filters is to first have a sand filter to remove particles, secondly a peat filter for high molecular weight organic

contaminants and finally a GAC filter for removal of low molecular weight organic contaminants. Furthermore, the column filters alone were not sufficient to treat the leachate without pre-treatment.

4.3.3 Organic compounds in municipal landfill leachates

Detailed analyses of the leachate from Brudaremossen were performed as part of a project by Nicklas Paxéus (1996). Analyses of more than 200 different organic compounds were made on leachate samples obtained in March 1996. Paxéus made comparisons to other landfill leachates and stated that the Brudaremossen leachate had high concentrations of hydrocarbons, likely deriving from oil contaminated waste. Benzenes and substituted benzenes were common in the leachate as well. Their origin was assumed to be varnish and degreasers respectively.

The presence of substances deriving from styrene and PAHs was explained as impurities in other products (Paxéus, 1996). Solvents were proposed as the origin of aliphatic alcohols and esters and short chained ketones. A few compounds found in the study had not previously been found in landfill leachates, namely the cyclic ethers dioxine and dioxilane.

5 Brudaremossen pilot treatment plant design and sampling-taking

The pilot treatment plant evaluated in this study was constructed in the end of 2014 and the beginning of 2015. In January 2015 the first leachate samples were collected, to evaluate the water quality and the treatment efficiency. The pilot treatment plant was designed to only treat a part of the leachate. The remaining fraction was led to Ryaverket wastewater treatment plant, without any further treatment.

The pilot plant consisted of four different treatment steps, in addition to the already existing oil and sludge separator and pond (Lundh, 2014). The new treatment steps in the plant were:

- Chemical precipitation and lamella sedimentation
- Moving Bed Biofilm Reactor (MBBR)
- Sand filters
- Activated carbon filters

The leachate could flow in two parallel lines. Either all four treatment steps were used, or the water bypassed the biological treatment. The capacity for the chemical treatment exceeded the capacity of the MBBR and filters. Therefore parts of the chemically treated water were led to Ryaverket without further treatment. Since different recipients were considered for the final plant, it was of interest to evaluate the water quality both with and without MBBR. Sampling points were located in between all treatment steps. A schematic layout of the plant can be seen in Figure 8. Since this was only a pilot treatment plant the water was not released to the environment. Instead the outgoing leachate from the activated carbon filters was collected in a tank before it was led to the municipal wastewater collection system.

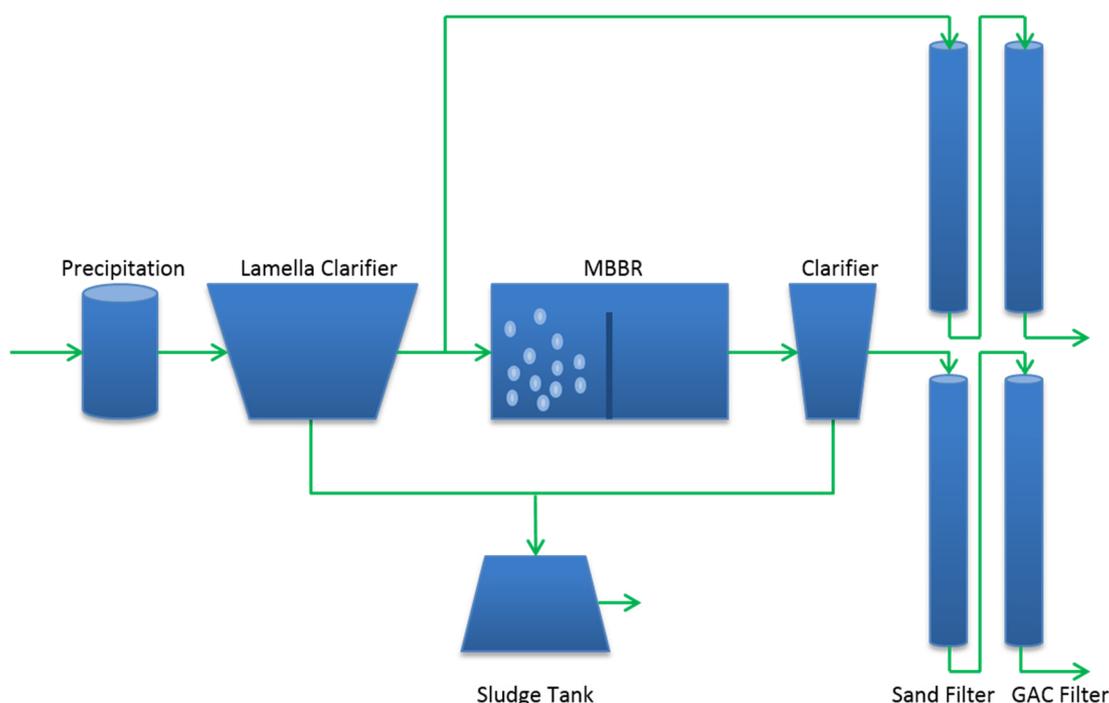


Figure 8 Schematic picture of the different treatment steps at the Brudaremossen pilot plant.

In addition to the treatment facility a high rate clarifier was installed on site during the spring of 2015. The clarifier was of the brand Actiflo and used microsand in combination with polymers to create rapidly settling flocs. According to the manufacturer the system is very compact and allows for low retention times (Veolia Water Technology, 2015). The Actiflo clarifier was used as a complement to the treatment facility. The water treated by it was not recirculated to the regular treatment. No tests made on the Actiflo clarifier were included in this project.

The treatment facility was constructed next to the pond north of Brudaremassen. The treatments steps were kept within a tent and two containers. The different treatment steps, and their purposes in the facility, are described in further detail below.

Chemical treatment and lamella sedimentation. The water was pumped from the pond into a chemical treatment unit located within the tent next to the pond. The main purpose of the chemical treatment was to remove iron, which was a major problem in previous studies (Krewer and Moona, 2012). The chemical treatment had a capacity of 3.5 m³/h, which for comparison was equal to approximately 30 % of the mean flow during the year. The unit consisted of a tank where chemicals were added followed by a lamella sedimentation unit (Lundh, 2014). In the tank aluminium sulphate was added as precipitation chemical. Laboratory studies had previously showed that the precipitation was more efficient at alkaline pH. Therefore the pH in the leachate from the pond was adjusted by addition of caustic soda, NaOH. A polymer was also added to improve the precipitation process. The mixing of chemicals into the leachate was improved by two stirrers operating at different speeds. The amount of chemicals could be adjusted manually.

The residence time in the tank was 14 minutes at the dimensioning flow (Lundh, 2014). Afterwards the water was led to a lamella sedimentation tank with an inner surface of 8 m². A distribution tank was placed after the lamella sedimentation unit, to distribute the water into three different pipes; one leading to the MBBR, one to the filter unit and one going directly to the wastewater system.

Moving bed biofilm reactor. The moving bed biofilm reactor consisted of three treatment tanks and one equalization tank (Lundh, 2014). In the first one nitrification occurred during aerated conditions. In the second tank denitrification occurred during anoxic conditions. The biofilms were growing on suspended carrier material in both tanks. In order to obtain proper removal methanol and phosphorus were added to the leachate. The nitrification and denitrification tanks were followed by an equalization tank and a sedimentation tank.

The biological treatment was studied in further detail by Porsgaard and Söderström (2015) in parallel with this study. Porsgaard and Söderström performed a comparison to an alternative treatment using a laboratory scale sequenced batch reactor.

Sand filter. Plastic tubes were filled with sand to approximately 2.5 m height. The water entered the filter at the top and passed through a layer of anthracite, followed by a layer of fine-grained sand. Bottom layers made of differently sized gravel and stones were placed below the sand layer. The filters were dimensioned for a loading of 4 m/h, which was equal to a flow of 0.25 m³/h. From the sand filters the water was

pumped to the filter with activated carbon. The design and capacity of the sand and carbon filters were the same for both treatment lines.

Activated carbon filters. The activated carbon filters were constructed similarly to the sand filters, with plastic tubes packed with the filter material and a bottom layer consisting of sand and gravel. The granular activated carbon used in the filters had a depth of 1 m. After the filters the water was led to a storage tank before it was transported to the municipal wastewater treatment plant.

Sludge disposal. The chemical precipitation and the MBBR produced sludge during the operation of the plant. At frequent intervals, the sludge was pumped from the sedimentation tanks to a sludge tank located outside of the tent and the containers. The sludge from chemical precipitation was pumped to the sludge tank once every hour. Sludge from MBBR was pumped once per day. Therefore the sludge in the sludge tank consisted mostly of chemical sludge². The sludge settled in the sludge tank, and the supernatant was recirculated to the pond. When the sludge tank was full it was emptied by a sludge suction vehicle.

Back-flushing of the sand filters were made regularly, using the water in the storage tank located after the activated carbon filters. The water from the back-flushing was led to the wastewater system without further on-site treatment.

5.1 Measurements at Brudaremossen pilot plant

Leachate samples from the pilot treatment plant were regularly collected and analysed during the spring of 2015. Occasionally samples of the sludge were collected and analysed for chemical and physical properties. The sampling of the leachate and the sludge for chemical analysis were taken by Ramböll. The analyses were performed by commercial laboratories. The physical analysis of the sludge and the sampling for this was carried out as a part of this study. Detailed descriptions of the physical analyses are found in Chapter 6.

The schedule of the samplings can be seen in Table 8. The chemical analyses of the leachate varied between the weeks. The simplest analysis consisted of basic parameters such as total nitrogen and total organic carbon. The most comprehensive analysis included toxic metals, organic contaminants and compounds required for Revaq certification. Not all available sampling points were sampled every week and all parameters were not analysed in every sampling point.

² Lars-Ove Sörman (Responsible for plant operations) phone interview by the authors 2015-05-19.

Table 8. *Sampling schedule for Brudaremossen, spring 2015. Analyses of leachate have been performed by the commercial laboratories Alcontrol and ALS.*

Date		Analysis		
Month	Week	Alcontrol	ALS	Sludge
February	7	Basic parameters		
	8	Basic parameters, Fe, Mn, DOC, Toxic metals, PCB 7	Aliphatic hydrocarbons, PAH	
	9	Basic parameters		
March	10	Basic parameters, Fe, Mn, DOC	Aliphatic hydrocarbons	
	11	Basic parameters		
	12	Basic parameters, Fe, Mn, DOC, Toxic metals, PCB 7	Aliphatic hydrocarbons, PAH, Revaq	Physical properties
	13	Basic parameters		
April	14	Basic parameters, Fe, Mn, DOC	Aliphatic hydrocarbons	Chemical composition
	15	Basic parameters		
	16	Basic parameters, Fe, Mn, DOC, Toxic metals, PCB 7	Aliphatic hydrocarbons, PAH, Revaq	Physical properties
	17	Basic parameters		
	18	Basic parameters, Fe, Mn, DOC	Aliphatic hydrocarbons	
May	19	Basic parameters		
	20	Basic parameters, Fe, Mn, DOC, Toxic metals, PCB 7	Aliphatic hydrocarbons, PAH	

The sludges were sampled for chemical analysis during the first week of April 2015. Before the samples were sent for analysis they were allowed to settle and the supernatant was removed.

The physical properties of the sludges were analysed twice during the spring of 2015; 16-17 March and 14-15 of April. The sludge was stored and analysed at Chalmers Technical University's Environmental Chemistry Laboratory, Civil Engineering Department. The analysis is described in further detail in Chapter 6.

5.1.1 Sampling locations for analysis

The locations that were used to collect samples are shown in Figure 9. The red sampling points were for leachate analyses while the orange sampling points were used when taking sludge samples. The leachate analyses varied and thereby samples were not collected at every sampling point every week. Leachate sampling points were located in between treatment steps to enable evaluation of removal rate of the different treatment steps, as well as for the treatment as a whole.

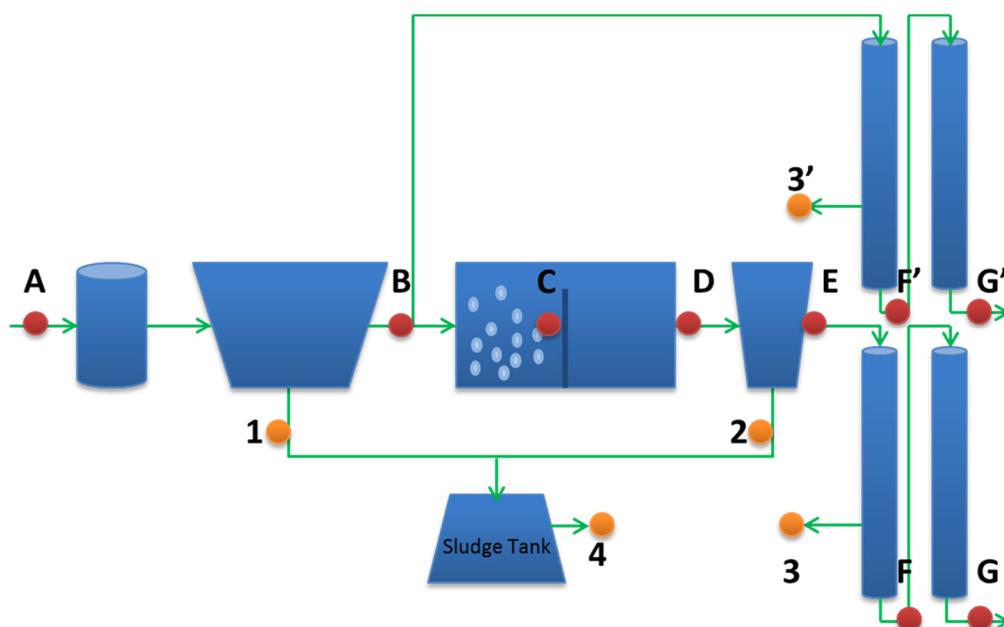


Figure 9 Schematic figure of the sampling points for sludge (orange) and leachate (red). A is incoming leachate, B is after chemical precipitation and sedimentation, C is after nitrification, D is after denitrification, E is after biological sedimentation, F is after sand filter in the biological line, G is after activated carbon filter in the biological line, F' is after the sand filter in the chemical line and G' is after the activated carbon filter in the chemical line. Sampling point 1 is sludge from the chemical sedimentation, 2 is sludge from the biological sedimentation, 3 and 3' are back-wash sludge from sand filter for biological and chemical line respectively and 4 is sludge from the mixed tank.

The red sampling points are located afterwards different treatment steps, namely: the pond (A), lamella sedimentation (B), nitrification (C), denitrification (D), biological sedimentation, (E), sand filters (F and F') and activated carbon filters (G and G').

Sludges for chemical analyses were sampled from the lamella sedimentation (marked as 1 in Figure 9) and the biological sedimentation tank (2). The physical sludge tests did also include sludge from the sand filters (3 and 3') and mixed sludge from the sludge tank (4). The sludge from the sand filters were collected during backwashing of the filters. For comparison of physical properties, sludge from Ryaverket was collected. These were taken from the treatment plants aerated active sludge process.

Sludges from point 1, 2, 3 and 3' were collected by judgmental sampling. The sludges from all these steps were ejected from the treatment steps at different intervals. Backwashing of the sand filters occurred two times per day. The volumes of the ejected or backwashed sludge had a too big volume to be collected and also the quality was varying, see Figure 10. The first volume that was ejected contained old sludge from the previous ejection and would thereby give an incorrect result if analysed. Likewise the last part of the ejected sludge volume would give a false result since the sludge then was diluted. The sludges were therefore collected in the middle of the flush-out period, by judgement.

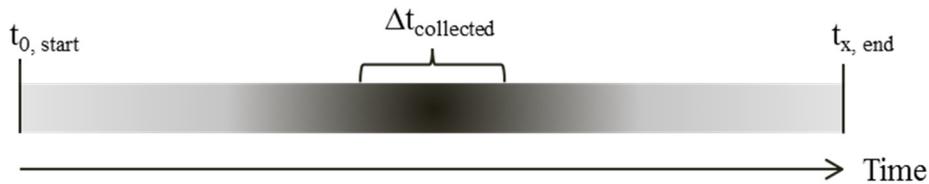


Figure 10 Illustration of how solid concentration and quality varies over time during ejection of sludge. Darker colour indicates a reliable sample.

For the mixed sludge (4) a pump was used for collecting the sample. The pump was placed on the bottom of the tank and a random sample was collected. Since the sludge in the tank had time to settle, the collected mixed sludge was concentrated. The tank had a big volume and could therefore not be stirred before collection of sludge. The Ryaverket activated sludge was collected as a grab sample in the end of the big aerated active sludge tank at Ryaverket.

5.1.2 Sample storage

Leachate chemical analysis samples were directly after sampling placed in a thermal bag with ice packs. The bottles used were obtained from the commercial laboratories and therefore suitable for the analyses. The thermal bag with samples was handed over to postal service for a fast delivery to laboratories.

The sludge samples, taken for analysis of physical properties, were stored in plastic 10 liter cans. The analyses were performed within 24 hours of collection. The samples were stored overnight in a cold room ($\sim 11\text{ }^{\circ}\text{C}$) before analysis took place the day after. The samples for chemical analysis of the sludge were also stored in plastic cans. They were sent for analysis by the postal service.

5.1.3 Analysis of data

The raw data from the leachate analysis was compiled for evaluation. The spreadsheet software Microsoft Excel 2013 and the programming software MATLAB were used for the compilation. Relative concentrations of different compounds in the different treatment steps were calculated. The data was visualised using diagrams and tables.

A test was made to find possible correlations between the different measured compounds in the leachate. This was done by using MATLAB. The data had to be slightly altered in order to perform an analysis. Sampling occasions where a wide variety of compounds were measured were used for the analysis. Three different sampling occasions fulfilled the criteria. If the concentration of a compound was below the detection limit it was assumed to be half the detection limit. This was often the case for organic contaminants in the latter treatment steps. Sometimes measurement values were missing, such data points were assumed to be the mean value of the other measurements of the specific compound in the specific measurement point. No study of correlation was made for the biological line, due to lack of data. For detailed theory regarding correlation, and the MATLAB code used for the analysis, see Appendix 3.

6 Laboratory analysis of sludge

Laboratory tests were performed to determine physical properties of the sludge produced in the Brudaremsøen pilot treatment plant. The tests aimed at determining the relation between water and solids in the sludge, as well as its dewatering properties. This information was then used to evaluate how the treatment of the sludge could be properly designed.

The studied sludge types and how they were sampled are described in Chapter 5.1. All studied sludge parameters and methods used are shown in Table 9. The methods used were either standard or adapted standard methods. The adapted methods are described in more detail in this chapter, and for further information regarding the rest, see Rice et al. (2012). All sample weights necessary for the test were obtained using a Sartorius Analytic A210P scale.

Table 9. *Physical tests performed on the sludge from Brudaremsøen. The focus of the tests were to evaluate the solid content, water content and dewatering capacity.*

	Parameter	Method	Equipment	Volume [ml]
Solids	Total solids	2540 B	Thermo Scientific HERATerm	5
	Volatile solids	2540 E	Carbolite Furnaces CSF 1200	5
	Total suspended solids	2540 D*	Vacuum pump, oven	10-50
Water	Free water	2710 H*	Vacuum pump	5
	Interstitial water	8080 C.1*	Centrifuge Sigma 4-16	50
	Surface and bound water		Calculated	-
Dewatering	Capillary suction time	2710 G	CST Triton Electronics 304M	5
	Zone settling velocity	2710 E*	Plastic cylinders	1000

* The test is an adaption from a standard method.

All tests were made on grab samples from the stored sludge containers. The containers were shaken, not stirred, before the grab samples were extracted. Three tests, capillary suction time and total solids and volatile solids were also measured on sludge that had first been thickened by sedimentation. Each test was replicated to obtain a more reliable result. The number of replications varied between the different tests, see detailed results in Appendices 5 – 10. For the second sampling occasion 3 – 4 replicates were made.

Total solids and volatile solids. The total solids, TS, in the sludge were obtained by heating a sample to 105 °C until the water in the sample had evaporated. The sample was then heated further, to 550 °C, to ignite the organic material and thereby obtaining the fixed and volatile solids. The total solids, fixed and volatile solids described the content and composition of the solid material in the sludge. By subtracting the percentage of solids from the total content, the water content of the sludge could be obtained, see equation X. The information was useful to evaluate the sludge, as well as the performance of the treatment plant.

According to Rice et al. (2012) content of volatile solids can be used as an estimation of organic content in a sample from wastewater. Therefore it was seen as suitable for determining the organic content in the sludge samples. Most organic contaminants e.g. PAH, PCB and PHC have a boiling point <550°C and should therefore leave the samples as volatile solids (European Communities, 2001; Erickson, 1997; US EPA, 2001). Metals will remain after the 550°C oven and are fixed solids.

Total suspended solids. The solid material present in the supernatant after the sludge was allowed to settle, during 24 hour, was characterised by measuring the total suspended solids. The solid material in the supernatant was separated from the water by filtration, using a vacuum pump and glass fiber filter. The filter containing the solid material was then heated in two steps, to 105 °C and 550 °C respectively.

Filter paper made of glass fiber have a very low weight loss when exposed to heat. To ensure no weight loss occurs the filters shall be pre-filtered with deionized water and then dried in ovens twice before used. This preparation was not performed during the laboratory work, instead three blank filters was weighed, filtered with deionized water, placed in 105°C oven, weighed, placed in 550°C oven and then weighed again. From these blank filters average weight loss for each oven was calculated. The weight loss was used to compensate for deviation from the standard method.

Free water. The method to measure the free water in the sludge was adapted from standard method 2710 H (Rice, et al., 2012). By using a vacuum pump, a sludge sample of known volume was filtered through a 0.45 µm cellulose acetate filter. The volume shifted between different sludge samples because the solid concentration varied and clogged the filters. The weight of the passing water was measured and the free water ratio was calculated according to Equation 1. The test was performed in duplicates or triplicates to obtain a more reliable result.

$$u_{free\ water} = \frac{m_{pass}}{m_{sample}} \times 100 \quad (1)$$

$u_{free\ water}$ = content of free water [%]

m_{pass} = mass of water passing the filter [mg]

m_{sample} = mass of 5 ml sludge [mg]

Interstitial water. Interstitial water was measured by centrifugation, using an adapted version of standard method 8080 C.1 (Rice, et al., 2012). Sludge samples of 50 ml were poured into test tubes of known weight. The weight of the sample was measured. The samples were then centrifuged for 5 minutes in 3400 G in the centrifuge Sigma 4-16. Afterwards the sample had separated into a compacted sludge cake and a supernatant. The supernatant was removed from the test tubes, and its weight was determined by weighing the remaining sludge cake. The supernatant equalled the free and interstitial water in the sample. By using Equation 2 and 3 the percentage of only the interstitial water was obtained.

$$u_{free,interstitial} = \frac{m_{supernatant}}{m_{sample}} \quad (2)$$

$u_{free,interstitial}$ = content of free and interstitial water [%]
 $m_{supernatant}$ = mass of supernatant [mg]
 m_{sample} = mass of 50 ml sludge [mg]

$$u_{interstitial} = u_{free,interstitial} - u_{free} \quad (3)$$

$u_{interstitial}$ = content of interstitial water [%]
 u_{free} = content of free water [%]

Surface and bound water. The surface and bound water can be calculated by subtracting the interstitial and free water from the water content, see Equation 4.

$$u_{surface, bound} = u_{total} - u_{interstitial} - u_{free} \quad (4)$$

$u_{surface, bound}$ = content of surface water and bound water [%]

Capillary suction time. The capillary suction time, CST, is a simple parameter used to evaluate the dewaterability of sludge (Chen et al., 1996; Jin, et al. 2004). The CST for sludge is defined as the time needed for the water in sludge to travel a fixed distance in a specific filter paper. A large CST indicates that the sludge is difficult to dewater. The equipment used to measure the CST for Brudareossen sludge is shown in Figure 11.

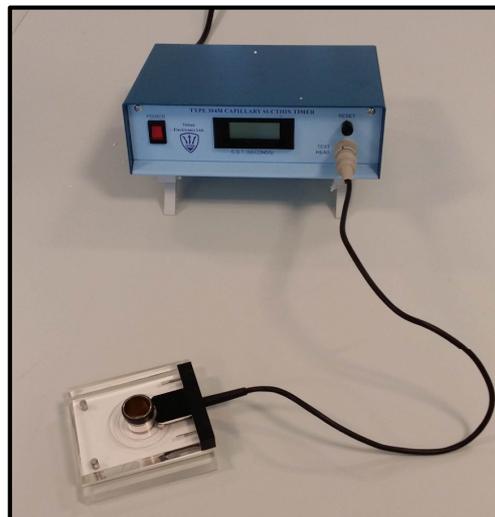


Figure 11 Picture of a CST apparatus used for measuring the dewaterability of a sludge.

Zone settling velocity. The zone settling velocity test was adapted from standard method 2710 E (Rice, et al., 2012). In the test, sludge was poured into 1 litre plastic cylinders and allowed to settle. A picture of the settling cylinders is shown in Figure 12. The cylinders were not stirred during the settling. A distinct interface between sludge and supernatant was seen, and the distance from the bottom of the cylinder to this interface was measured. The distance was monitored continuously for a six hour period, and a final reading was made 24 hours after the start of the test. After plotting

the distance versus the time the settling velocity could be obtained as the slope in the obtained graph. The velocity was then plotted against time. Duplicates of the settling tests were performed to reduce errors. At the first occasion the tests were carried out with one day in between, while at the second occasion they were carried out simultaneously.

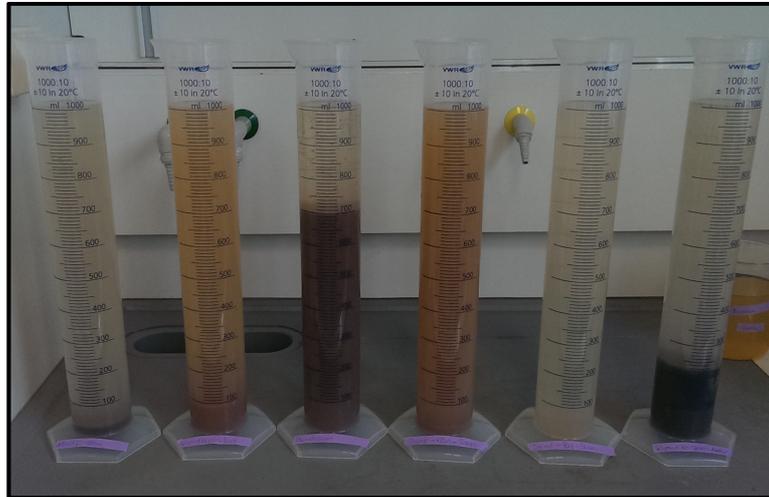


Figure 12 Picture of a zone settling velocity test. The different sludges are placed in cylinders and the distance between bottom and sludge interface is measured.

The sludge volume index, SVI, is a parameter useful for evaluating the settling characteristics of a sludge. It is obtained by measuring the volume occupied by 1 gram of sludge after 30 minutes of settling. The SVI was calculated for the sludges in this study and is found in Appendix 12.

7 Results and discussion

This chapter presents and discusses the results from the measurements and tests made on the leachate and sludge from Brudaremassen landfill. At first the results from the leachate analysis are presented and discussed. This is followed by a presentation and discussion of the sludge quality. Afterwards a general discussion is presented and the Brudaremassen plant is compared to other leachate treatment plants. Finally, possible further studies are discussed.

7.1 Results from analysis of leachate quality

In Table 10 the removal rates of some important pollutants are shown. A brief overview shows there is no big difference in removal rate between the two treatment lines (biological and chemical line) concerning metals and groups of organic contaminants. The biological line has somewhat higher removal of Cu, Ni, Zn and PCB 7. The PCB 7 was found under the detection limit after the treatment for both lines except for the measurement in February 2015, in the chemical line. Since PCB 7 only is measured once in the biological lines effluent it is not accurate to consider the biological line more efficient in treating PCBs. When calculating the removal rates and plotting the corresponding results all measurements below the detection limit were included as half the detection limit.

Table 10 Removal rates in the two different treatment lines at the Brudaremassen pilot treatment plant. Measurements are taken during the spring 2015.

Compound	Before	Bio. Line		Chem. Line		Unit
		After	Removal [%]	After	Removal [%]	
Total Fe	11	0.25	98	0.086	99	mg/L
Fe-filtered	0.11	0.047	59	0.037	67	mg/L
COD	90	110	-27	43	52	mg/L
TOC	31	40	-29	17	45	mg/L
DOC	29	31	-6.9	16	45	mg/L
TSS	35	13*	63	0*	100	mg/L
Pb	0.10	0.10	0	0.10	0	µg/L
Cd	0.01	0.01	0	0.01	0	µg/L
Cu	0.88	0.25	71	0.36	59	µg/L
Cr	3.3	2.9	11	2.5	22	µg/L
Hg	0.00	0.00	0	0.00	0	ng/L
Ni	6.9	5.4	22	6.0	13	µg/L
Zn	9.1	1.5	84	2.1	77	µg/L
PAH L	0.47	0.01	98	0.008	98	µg/L
PAH M	0.58	0.01	98	0.01	98	µg/L
PAH H	0.03	0.02	26	0.02	26	µg/L
Aliphatic >C16-35	10	5.0	52	5.0	52	µg/L
PCB 7	0.014	0.00005	100	0.0005	96	µg/L

*The total suspended solids are measured after the sand filters.

The chemical line reaches marginally higher removal efficiencies for total Fe, dissolved Fe, COD and Cr. The COD has a negative removal in the biological line, meaning it adds organic material. An explanation could be an excess of methanol in the MBBR step or that the biomass does not settle appropriately after the MBBR.

Figure 13 shows the change in the concentration of general parameters such as COD and TOC between the different treatment steps. The concentration is compared to the initial concentration in the incoming leachate, which is set as 100 %. The mean concentrations from all measurements are shown in the figure. Step A is incoming leachate, B is after chemical treatment and lamella sedimentation, F' is after sand filter (chemical line) and finally G' is the effluent from the activated carbon filter (chemical line).

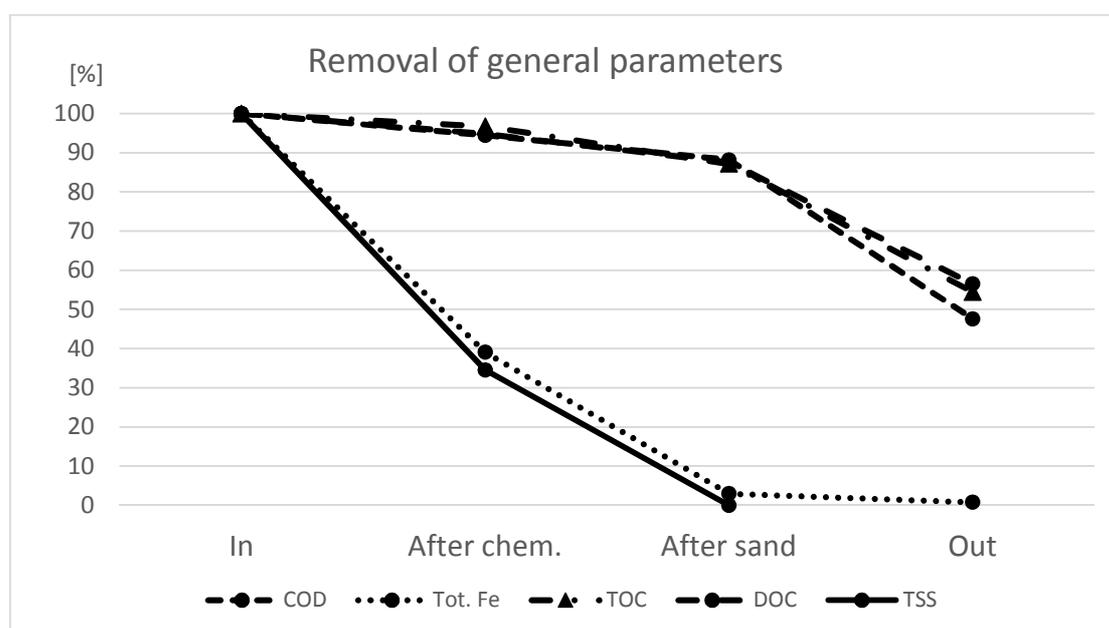


Figure 13 Removal of compounds in the chemical treatment line. The x-axis shows the treatment steps. The y-axis shows the concentration in percent in comparison to the incoming concentration.

The mean removal values match well among the different carbon measurements: TOC, DOC and COD. Also the removal rate for total Fe and TSS correlates, this is expected since iron is ~30 % of the content in the TSS. The removal rate is ~99 % for TSS and Fe. The greater part is removed in the chemical precipitation and sand filter. The chemical precipitation was designed to remove iron from the leachate, the results shows this was accomplished.

For the carbon parameters the highest removal was obtained in the activated carbon filter. According to literature, activated carbon filters could reach a removal of 85 % for COD (Renou, 2008). However this was not obtained at the pilot plant where 46 % of the COD was removed after the chemical treatment and sand filters. The removal of TOC does not achieve the recommended effluent concentrations that Gothenburg city has as a goal (Appendix 1). The chemical line was built for pre-treatment of toxic metals and organic contaminants to evaluate the recipient alternative Ryaverket WWTP, where further treatment would follow. High concentrations of TOC, COD and DOC would most likely not contaminate the sludge at Ryaverket, if the organic

contaminants are removed, and therefore high concentrations in the chemical line effluent can be acceptable. However, previous studies by Kalmykova et al. (2013) showed that organic contaminants can be linked to colloids, DOC. Therefore it is still important to remove as much carbon and colloids as possible.

The data have some small variations in concentration for the different compounds in each step. The variations are expected for a newly started pilot treatment plant. The biggest variations are for COD over sand and activated carbon filters, however the removal rate is not affected.

7.1.1 Removal of toxic metals

The average concentrations of toxic metals in the different treatment steps are shown in Figure 14. For detailed information from all measurement, see Appendix 4.

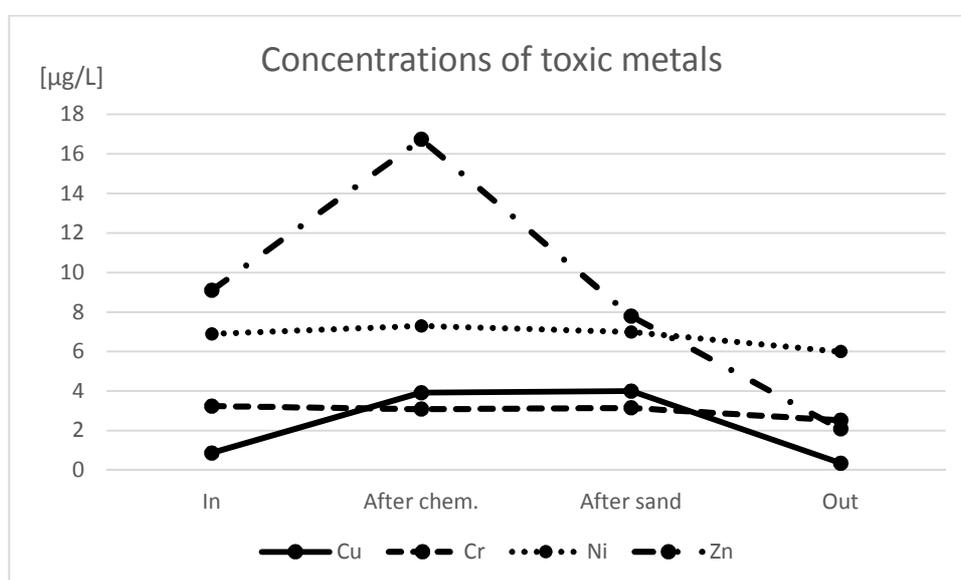


Figure 14 Removal of toxic metals in the chemical treatment line. The x-axis shows the treatment steps. Measurements below detection limit is added as half the detection limit.

For the toxic metals in the chemical line low removal was achieved. However the incoming leachate had low concentrations of Cu, Cr, Ni and Zn and the guideline values from Gothenburg city was still accomplished in the effluent. For low incoming concentrations of toxic metals a high removal rate is more difficult to reach, this could be an explanation to the obtained results.

The plotted graph (Figure 14) comes from data measured at three occasions with exception of the incoming leachate, A, that was only measured twice. The measurements varies slightly for each step but the removal rate is relatively stable over each treatment step. The Zn concentration has an unexpected peak after chemical treatment. The incoming concentration in A was only measured twice and it is possible that the concentration were exceptional low at both occasions. From old measurements at Brudaremsen it was shown that the average concentration at the outlet of the pond was 19.2 µg/L between 2008 – 2012. This supports that Zn was

measured at a low-point in concentration at the inlet A in this study, and that it is less likely that the chemical precipitation releases Zn in B.

Research shows that metals can be well removed with chemical precipitation if the pH is high (>9) (Fu and Wang, 2011). Removal rates of 99 % has been achieved for Cu, Zn, Pb, Cr and Hg. In order to improve the efficiency at Brudaremossen an increase of the pH can be an alternative.

The only toxic metal removed is Zn which was removed ~50 % over the sand filter. From a study by Modin (2007) similar results were obtained; Zn was the only toxic metal removed from the leachate using a sand filter with the removal efficiency of 50 %.

The concentrations of the toxic metals Pb, Hg and Cd are presented in Figure 15. For detailed information, see Appendix 4.

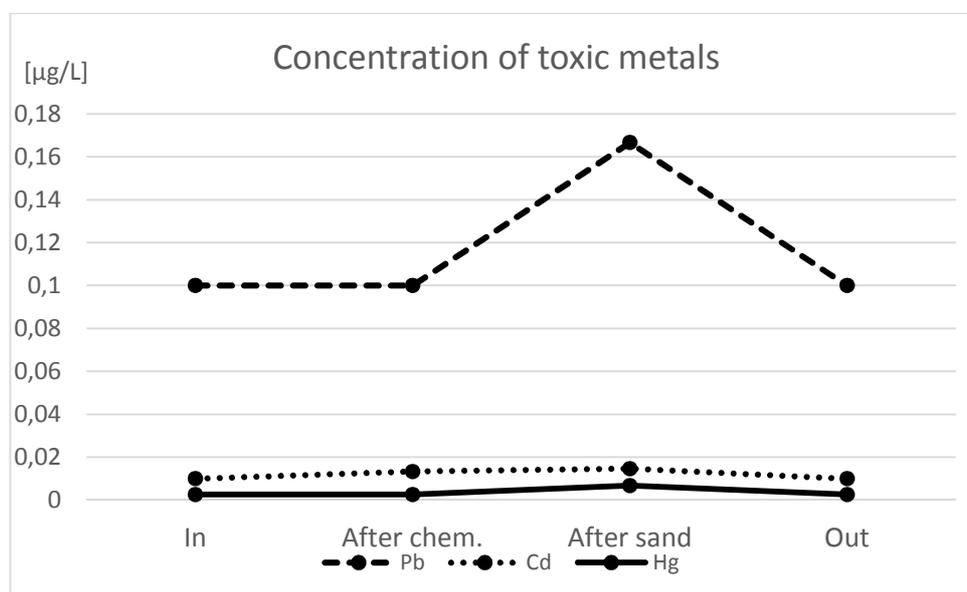


Figure 15 Removal of toxic metals in the chemical treatment line. The x-axis shows the treatment steps. Measurements below detection limit is added as half the detection limit.

The toxic metals Pb, Cd and Hg were found in lower concentrations and all of them were below the guideline value for effluent from the Gothenburg City recommendation. Most of the values plotted in Figure 15 was only found above detection limit 1-2 times (Pb one time in F', Cd once in B and one time in F' and Hg was found once in F'). For all the points below detection limit, half of the detection limit is added to Figure 15. Therefore the figure should not be interpreted as the concentration of Pb increased after the sand filter but rather as this was the only point where Pb was found. Since the metals were below detection limit it is not possible to evaluate the removal rate.

7.1.2 Removal of organic contaminants

The concentrations of a few selected organic contaminants: PAH L, M and H, PCB 7 and aliphatic hydrocarbons >C16-C35, are shown in Figure 16. The concentrations

are compared to the concentration of the contaminants in the incoming leachate. For detailed information regarding the concentrations, see Appendix 4.

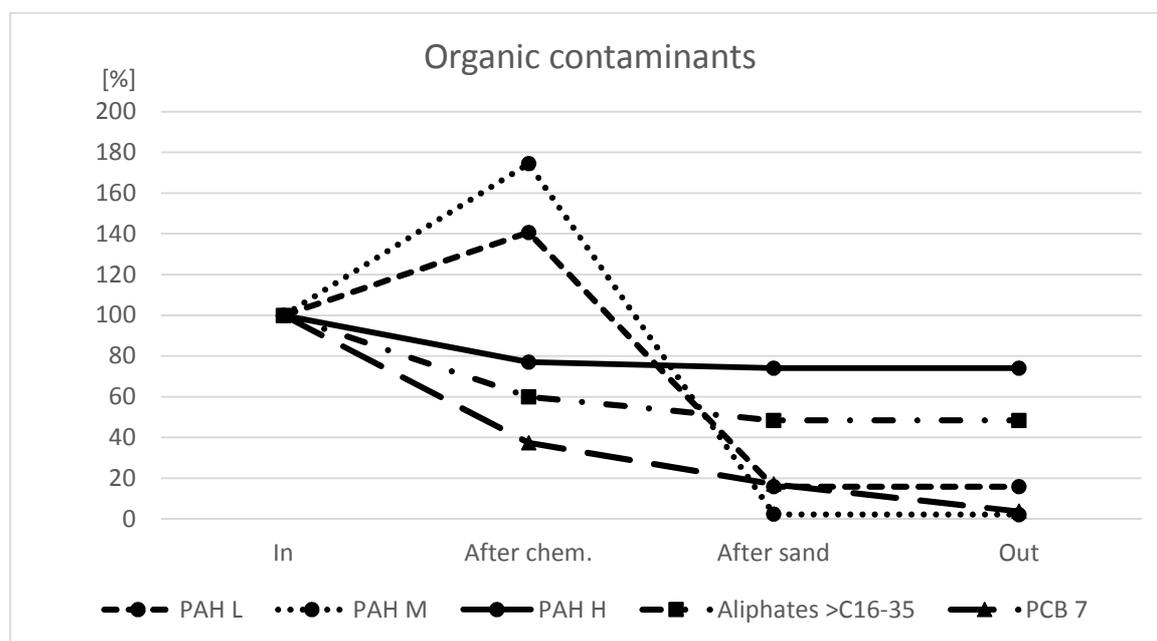


Figure 16 Removal efficiency of organic contaminants for the chemical treatment line. The x-axis shows the treatment steps. The y-axis shows the concentration in percent in comparison to the incoming concentration. Measurements below detection limit is added as half the detection limit.

The concentration of the compounds were on the same level among individual measurements for each treatment step. The number of samples obtained for each compound in each sampling point varies between 2 and 5. There is an increase in PAH L and PAH M between the incoming water and after the chemical treatment step, similar to the increase in zinc. There is no obvious explanation for this increase. Since the samples obtained were grab samples it is possible that they simply contained a different amount of PAH. By taking more samples or continuous sampling this difference in concentration might disappear. The PAH L and M are removed efficiently in the sand filter. This is in contrary to what Kalmykova et al. (2014) found in their study of leachate at Brudaremossen. Kalmykova stated that the sand filter was inefficient for improving the water quality, including PAH. However, with this new set-up including chemical precipitation the concentration of Fe is reduced and the sand filter might be more efficient in removing PAHs. No traces of PAH L or M were found in the effluent samples. From the guideline values, mainly from Canada, it seems as if the concentration of PAH L and M would be the same as the detection limit they would still be below the guideline values, see Appendix 1. Oxy-PAHs were not detected in any measurement, apart from one of the samples from the incoming leachate.

PCB 7 is also efficiently removed in the treatment plant. It is mostly removed in the chemical treatment step. PCBs are lipophilic and stay attached to surfaces of particles, and is evidently removed in the same step as them. PCB 7 was only detected in one of the samples from the outgoing leachate. This was the first sample obtained, and the

concentration was far below the guideline values from the City of Gothenburg, see Appendix 1 or Miljöförvaltningen (2013).

The treatment efficiency is lower for PAH H than the other PAH, according to Figure 16. However the concentrations were below the detection limit in all samples in point F' and G'. In reality, it is difficult to state how efficient the treatment actually is for PAH H. Still, the assumed concentration in the effluent is below applicable guideline values. The case is similar for aliphatic hydrocarbons >C16-C35. They are below the detection limit in the outgoing effluent samples as well as the samples after the sand filter. This makes it difficult to evaluate their removal efficiency. No appropriate reference values for comparison have been found for aliphatic hydrocarbons.

7.1.3 Removal efficiency of the biological treatment line

The removal of general parameters such as COD and TOC for the biological treatment line is shown in Figure 17. The concentration of the parameters are compared to the average initial concentration, which is set as 100 %. There are some variation in the individual measurement results, however the percental removal is fairly consistent.

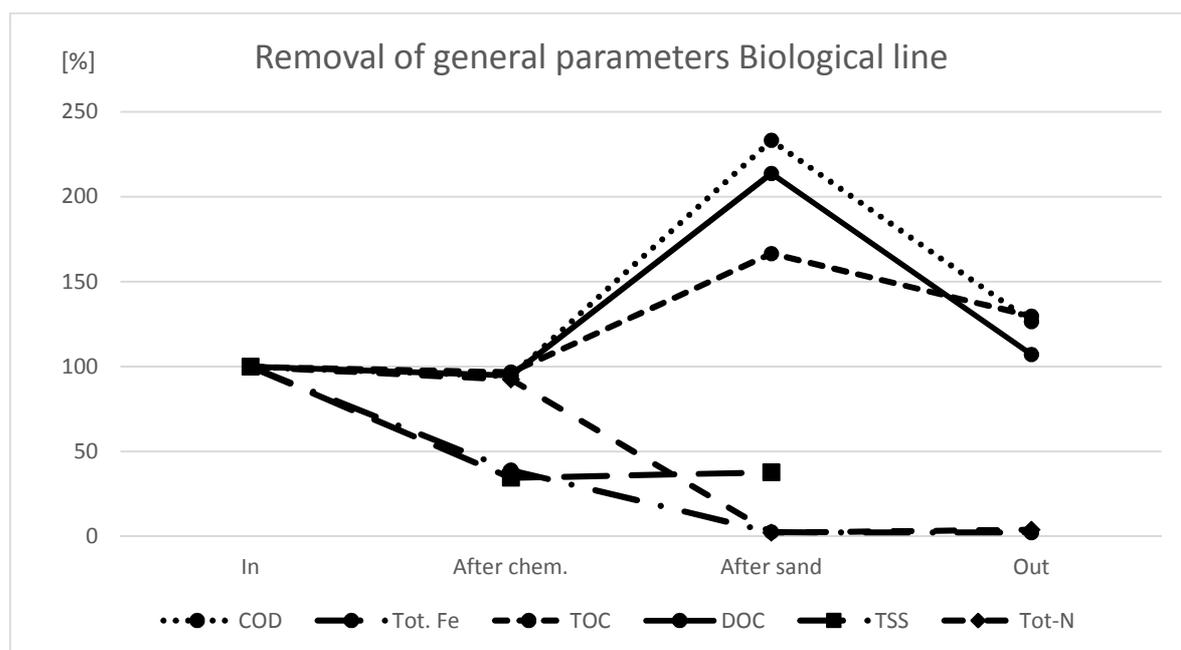


Figure 17 Removal of general parameters in the biological treatment line. The x-axis shows the treatment steps. The y-axis shows the concentration in percent in comparison to the incoming concentration.

The concentration of nitrogen in the effluent from the biological line is above the guideline value used by the City of Gothenburg (Miljöförvaltningen, 2013). The reduction in percent is high, however it is still not sufficient. Of the performed samplings, two out of three exceeded the limit. It is possible that the MBBR needs more time for adjustment before a sufficient removal is achieved. Just as for the chemical line, the TOC in the leachate is above the regulations. As explained earlier, the reason for the increase in TOC, DOC and COD could be the addition of methanol or biomass leaving the sedimentation tank after the biological treatment. Literature

states that MBBR units can remove 60 – 80 % of the COD in leachate (Aziz, et al., 2014). Possibly this removal will be obtained when the plant is more adjusted. Since the effluent from the biological line is supposed to be released into a recipient, it is important to fulfill the regulations. A good removal of iron is also obtained in the biological treatment line.

7.1.4 Correlation between measured parameters

The results from the correlation test, from the chemical line, are presented in Figure 18 and Figure 19. The first figure shows the correlation between a number of compounds and total suspended solids (TSS), total iron and dissolved (filtered) iron. As seen, the correlation is always positive, except for Cu, which is slightly negative. A very strong correlation (0.993) is found between iron and TSS. The correlation is also strong between TSS/iron and PCB and aliphatic hydrocarbons. A slightly lower positive correlation is found between TSS/iron and the different PAHs.

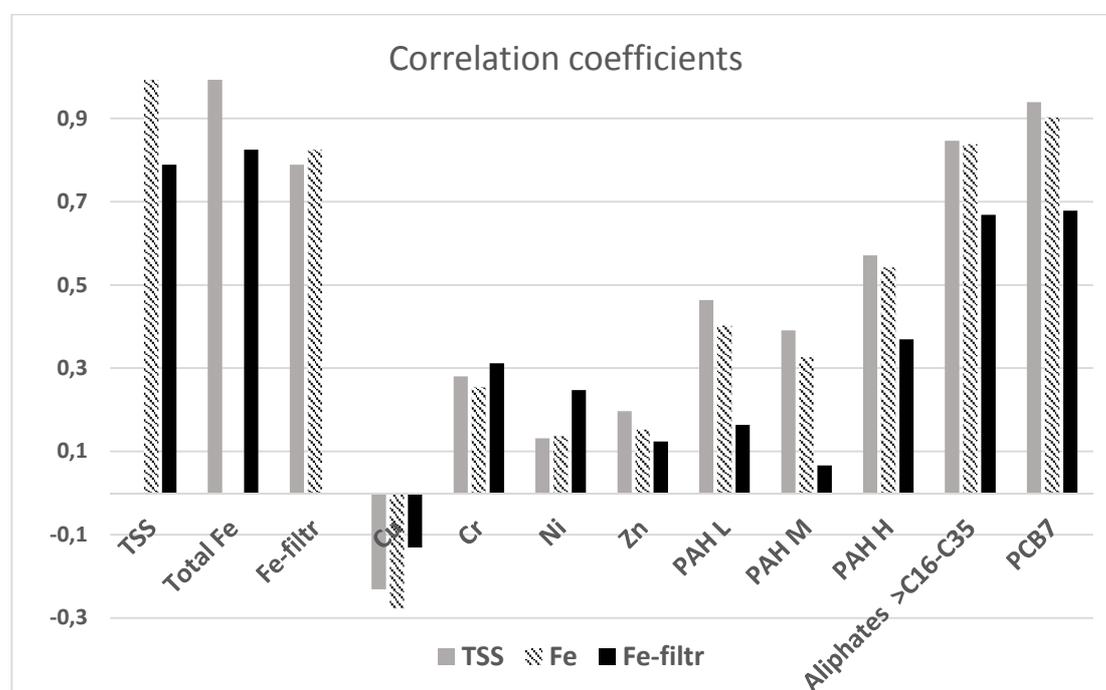


Figure 18 Correlation coefficients for total suspended solids, iron and filtered iron to metals and organic contaminants in the chemical treatment line.

When looking at Figure 19, the correlation between TOC, DOC and COD and the different compounds it can be seen that the correlations are generally less strong than when comparing to TSS and iron. The metals Cu, Cr, Ni and Zn are exceptions and have a better correlation when comparing to TOC, DOC and COD.

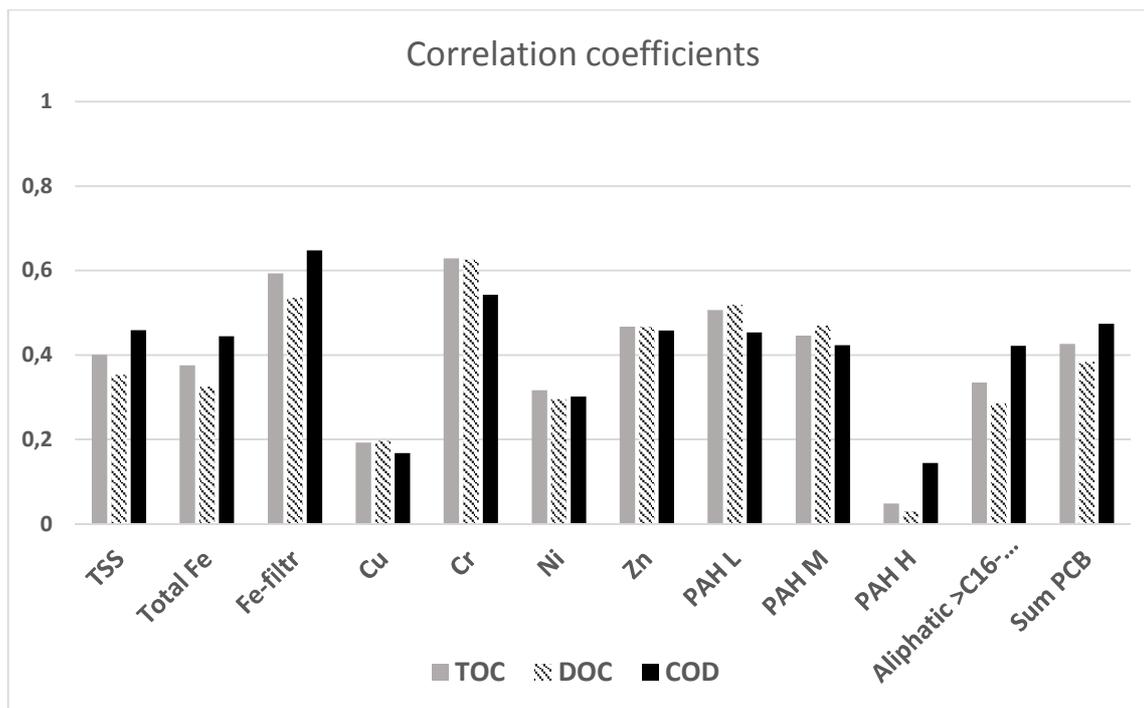


Figure 19 Correlation coefficients for total organic carbon, dissolved organic carbon and chemical oxygen demand to metals and organic contaminants in the chemical treatment line.

The correlation is made between compounds in all treatment steps at the same time. Since it is a treatment facility the aim is to reduce contaminants. In a successful treatment plant most compounds are reduced and the correlation might appear stronger than they are.

The compounds within the different types of contaminants mostly correlate within their respective groups. Low and medium molecular weight PAHs correlate well to each other, however the high molecular weight PAHs do not. This is not unexpected since the different groups of PAHs have different properties (Swedish EPA, 2009). Instead PAH H correlate stronger to the PCBs. The PCBs and the PAH H have a high molecular weight and they have similar lipophilicity and high K_{ow} values, which explains why they are removed in a similar way. PCBs have a strong correlation to each other. The aliphatic hydrocarbons >C16-C35 correlates to TSS, iron, PCB and also total phosphorus. Previous studies by Kalmykova et al. (2013) showed a connection between DOC and organic contaminants. No strong correlation to DOC was found in this study.

As seen in Figure 18 and 19 the metals have mostly weak correlations. The case is the same towards other compounds and within the group of metals itself. The best correlation obtained between metals is Cu and Zn (0.74). The weak correlations could be caused by the low incoming concentrations and the low removal rate.

The results of the correlation analysis agrees to the results presented in the previous chapter. For example, compounds removed in the same steps, such as TSS and iron, also have a strong correlation.

7.2 Results from sludge analyses

This chapter presents the results from the sludge tests. First the physical parameters of the sludge are shown and discussed, and afterwards the chemical composition is presented. Only the results from the second physical sludge sampling, in April, are presented in this chapter. The results from the first sampling in March can be seen in Appendix 5. These results are excluded from this chapter since there were problems with the testing equipment and the pilot plant had not been operating for very long.

7.2.1 Solids content in the sludge

Due to variations in the concentration of solids during the sampling in April, the results are shown separately in two charts in Figure 20. Both charts show fixed and volatile solids (\approx organic content) separately. The values displayed in the charts are averages, for detailed information see Appendix 6.

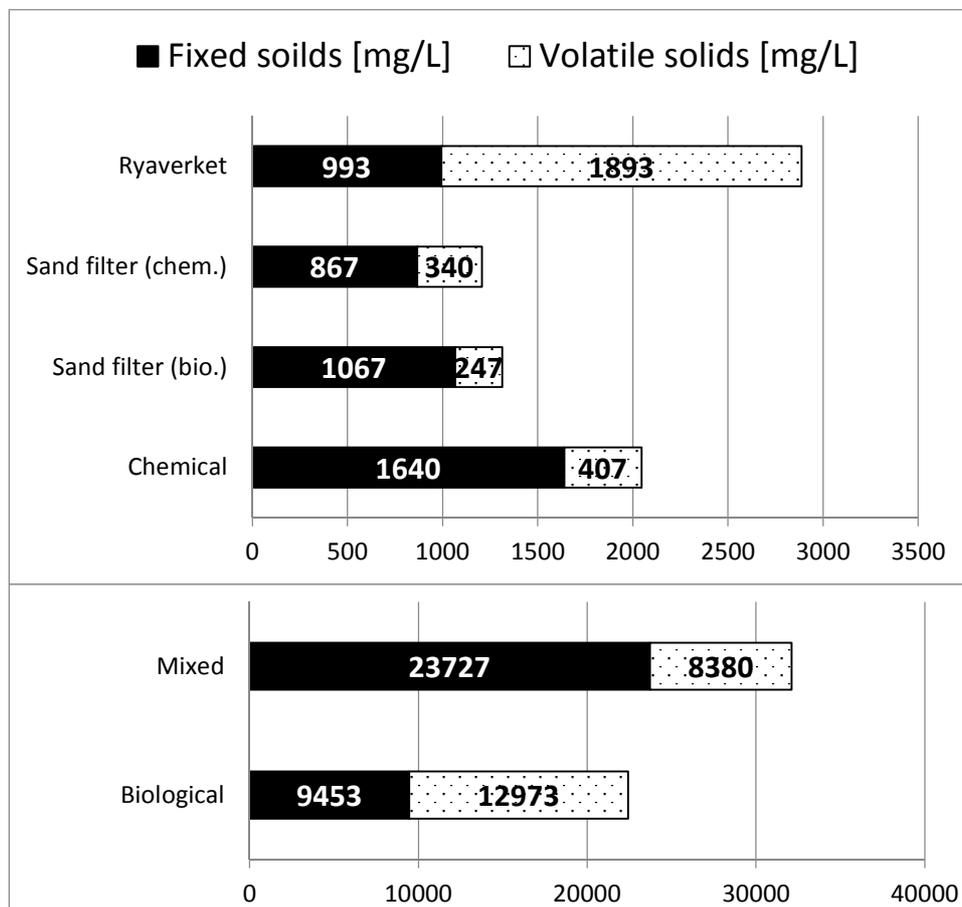


Figure 20 Average concentrations, in mg/L, of fixed (black) and volatile (white patterned) solids in the sludges from April 2015. Observe the different scales on the x-axes.

In Figure 20 the average solid concentrations are within the same range to each other and close to reference sludge, Ryaverket. The trend of later treatment steps indicates that more solids are taken out in the early treatment steps and that the solid loads on the sand filters are less. The sand filter from the biological treatment line has slightly higher solid concentration compared to the chemical line sand filter. It is a small

difference in concentration and it most likely originates from some troubles in sampling; the pipe used for sampling had visible activated carbon particles that ended up in the backwashing sludge from sand filter biological line. This could explain the low fraction of volatile solids.

For the biological and mixed sludge the solid concentration is more than 8 times higher than for the sludge seen in the upper chart in Figure 20. For the mixed sludge this result was expected since it had settled and the supernatant had been decanted from the sludge tank previously. The biological sludge also had very high solid concentration and it was a clear visible change between sludge tests ran in March compared to the one in April. The increased thickness of the biological sludge can be connected to the longer start-up time for biological treatment. After the initial start-up weeks the MBBR might be functioning better hence more biofilm is created. However the increased thickness of the biological sludge can also have been affected by a poorly functioning pump that should pump out the sludge from the MBBR sedimentation tank. If this is the case, the biological sludge could have been given time to compact further before discharged to the sludge tank, resulting in a thicker sludge.

The distribution between fixed solids and volatile solids is matching well between the sand filters, chemical and mixed sludge. For the two biological sludges, Ryaverket and the biological sludge, the greater part of the solids are volatile. This is expected, since the amount volatile solids approximately is the same as organic content (Rice et al., 2012).

The aim with the chemical treatment is mainly to reduce the amount of iron in the leachate. The solid distribution indicates that fixed material, such as iron, have been removed by the chemical treatment. The greater part of the solids from the sand filter sludges is also fixed solids, possibly remaining iron that has been transported by colloids.

The mixed sludge consists of sludge from the chemical precipitation and from the MBBR. It can therefore be expected that the result should show a mix of the chemical sludge and the biological sludge composition. The sludge that enters the sludge tank is <10 % biological sludge and >90 % chemical sludge. This is seen in the distribution of solids in the mixed sludge where the distribution of volatile and fixed solids more resembles the chemical sludge.

The TS concentration for wastewater treatment sludge is commonly 3 – 8 % for primary sludge and 0.5 – 2.5 % for activated sludge (Chen et al., 2002; Gray, 2010). The volatile solid concentration is usually 60 – 80 % of the TS in primary sludge and 50 – 60 % in activated sludge (Chen et al., 2002). The reference wastewater (activated) sludge used in this study, Ryaverket, has a TS concentration of ~0.3 % and a volatile solid concentration of ~65 %. This is close to the values literature presents.

From the studied leachate treatment plant Sofielund, see Chapter 3, the recorded TS concentration is 0.5 – 1 % (Sernstad, 2015). At Brudaremossen the TS concentration varies between 0.13 – 2.25 % for the tested sand filter sludges, chemical and biological sludge. No literature for TS in leachate sludge have been found. However the results is reasonable and similar to reference leachate treatment plants and

literature for activated sludge. The mixed sludge is however not possible to compare since it has been decanted many times.

Total solids in thickened sludge were also measured. The average results from this test are shown in Figure 21. Detailed results from each sample is found in Appendix 6.

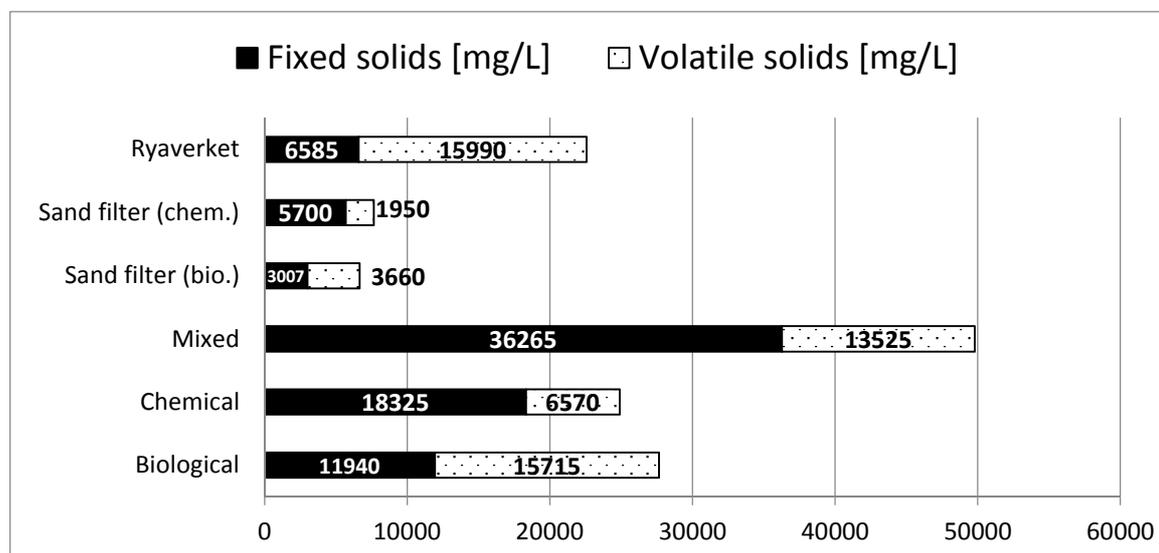


Figure 21 Average concentrations, in mg/L, of fixed (black) and volatile (white patterned) solids in the thickened sludge from April 2015. The sludge was allowed to thicken for 24 hours before measurement.

After sedimentation the solid concentration (TS) increased between 23 – 1116 % for the sludges at the pilot plant. The reference sludge, Ryaverket, increased the solid concentration 680 %. The most increased concentration was the chemical sludge that was 1116 % thicker after sedimentation, the second most increased was the sand filter chemical line (534 %). The mixed sludge and the biological sludge had the lowest increase in solid concentration, 55 % and 23 % respectively. The low increase of mixed sludge is probably due to that pre-sedimentation that already occurred in the sludge tank, hence most of the water from the sludge is already decanted from the mixed sludge. The biological sludge did not settle properly, partly as a result of biological activity in the sludge during settling indoors (20 °C). The results show that sedimentation is an efficient method for increasing the solid content.

The distribution between volatile and fixed solids was similar before and after settling except for the sludge from sand filter, biological line. For the sand filter bio line sludge the mass balance between supernatant, thickened sludge and original sludge does not add up. The traces of activated carbon, that was mention above, might have affected the result for sand filter bio. The chemical sludge have a small variation in the solid distribution between sludge and thickened sludge but still acceptable.

7.2.2 Total suspended solids

The total suspended solids in the supernatant from the settling test are shown in Figure 22. Detailed results are found in Appendix 7. The total suspended solids in the supernatant from the biological sludge were significantly higher than the rest and are

therefore not shown in Figure 22. Biological sludge had a total suspended solid content of 532 mg/L, of which 361 mg/L were volatile and 171 mg/L were fixed. When performing the tests, the supernatant from the biological sludge was less clear than the other.

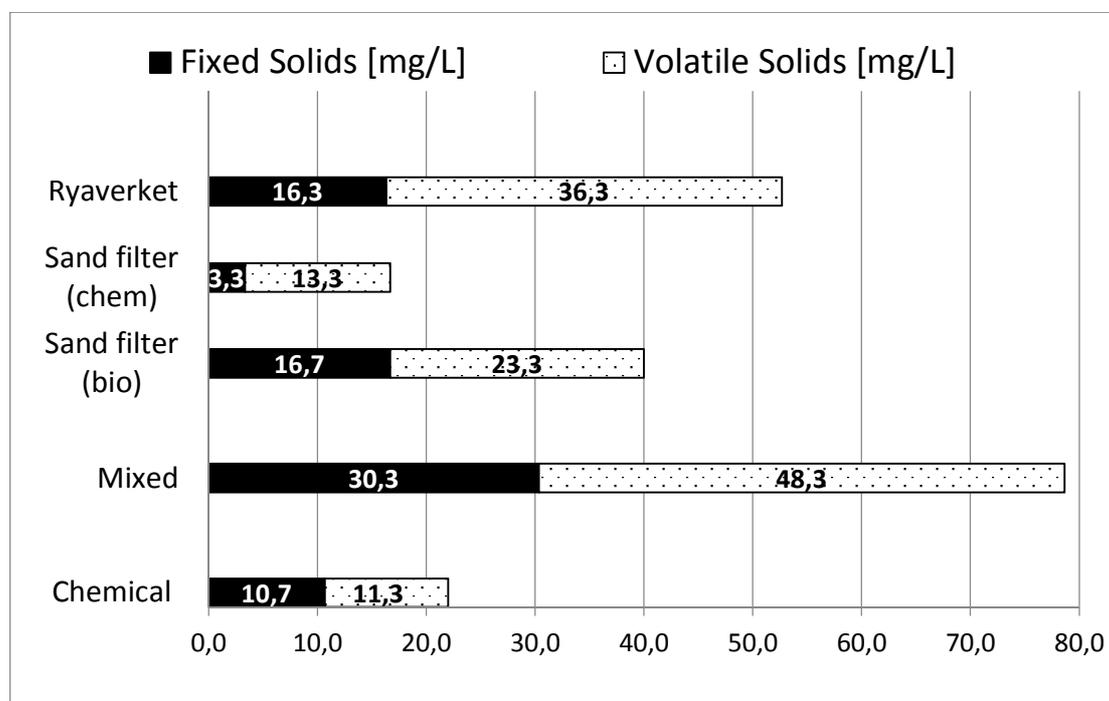


Figure 22 Average concentrations, in mg/L, of total suspended solids in the supernatant from the settling test of the sludge obtained in April 2015. Volatile solids are shown in white patterned and fixed solids in black. Biological sludge is omitted in the figure.

The supernatants was filtered for the TSS measurements and the average filter weight loss was added. Despite taking the filters weight loss into account some supernatants got negative concentrations of solids and had to be removed from the results. The weight loss of the three blank filters varied slightly and in combination with very low weights of the TSS on the filters, negative values were not unexpected. Therefore all results from the TSS measurements needs to be analysed with judgment; the total concentrations of solids are only approximations.

The distribution between fixed and volatile solids have shifted from the TS measurements. In TS the greater part of the solids were fixed solids (\approx inorganic) to in TSS having a majority of the solids being volatile (\approx organic). It is reasonable that the organic fraction has poor settling properties and therefore remains in the supernatant.

Common concentrations of TSS for leachate is around 40 mg/l (Öman, et al., 2000). Comparing the TSS leachate value to the measured TSS concentrations for the sludges one can see that the concentrations are within the same range (if excluding the biological sludge). The methods for measuring TS and TSS differs, regardless, a clear reduction of solids occurs for all sludge except the biological sludge. The biological sludge did not have a clear interface between sludge and supernatant and the TS concentration for the sludge was very high.

7.2.3 Water content and distribution

The percentages of different types of water in the sludge are shown in Figure 23. The percentages were calculated with the results from tests of free water, solids content and centrifugation. Detailed information of the charts and the results are found in Appendix 8.

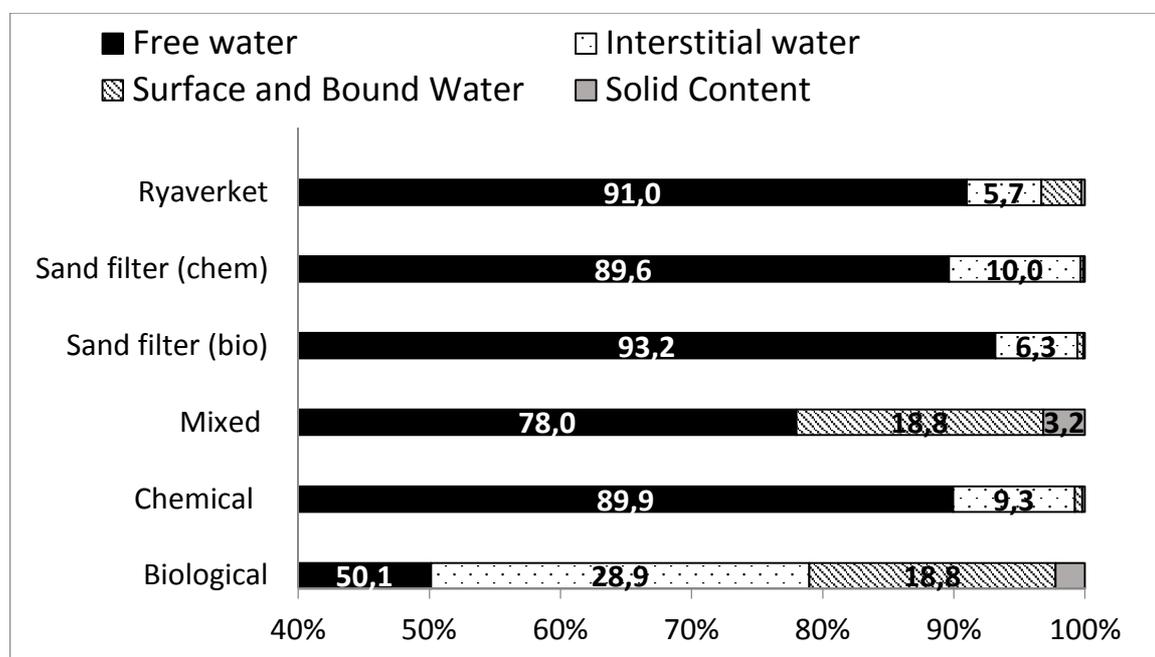


Figure 23 Different types of water in the sludge from April 2015. Black is free water, white patterned is interstitial water, striped is surface and bound water and grey is solid content.

The distribution of the water in Figure 23 was obtained using a variety of independent methods. It is possible that the sludge concentrations varied in the samples for the different tests. This could give rise to strange results. Therefore the distribution of the water must be interpreted carefully. Furthermore, the execution and the reliability of the different methods could affect the result.

As seen in Figure 23 the free water content in the sludge is generally high. This is good, since the free water is the easiest to remove and can be removed using gravimetric settlement (Vesilind and Martel, 1990). The biological sludge has a low content of free water, compared to the rest. All three performed measurements to determine the free water content in the biological sludge were consistent. Still, the thickness of the sludge could have caused problems to the equipment. Biological sludge has the largest share of interstitial water, 29 %. This is due to the low content of free water. None of the other sludges has an interstitial water content above 10 %. The content of surface and bound water is largest in mixed and biological sludge.

Graham-Jones (2014) performed a study, using similar methods, to evaluate the water distribution in sludge from two drinking water plants in Gothenburg. The free water contents in the drinking water sludge varied between 22 – 38 %, interstitial water varied between 41 – 44 % and surface and bound water varied between 17 – 35 %. Drinking water sludge is different from wastewater and leachate sludge. This sludge

is also difficult to handle and process. Therefore these higher interstitial and bound water contents are expected. According to literature, free water content in wastewater sludge is around 70 – 75 % and interstitial water is 20 – 25 % (Werther and Odaga, 1999). The Ryaverket sludge has a free water content of 91 %, which exceeds the literature. Overall, every sludge except the biological sludge has a free water content above literature values while the interstitial water is below. The combination of free and interstitial water corresponds fairly well to what is found in literature.

Determining the free water content by vacuum filtration is not a standardised method. In general, there is no standard definition of water distribution in sludge, making testing problematic (Lee and Hsu, 1995). An alternative to the selected laboratory tests could be to perform a drying test of the sludge, as described in Kopp and Dichtl (2000). However, this type of test would require equipment not available in this study. Furthermore, determination of bound water can be made by freezing a sludge sample and measuring the expansion. This would also require equipment not available.

7.2.4 Dewaterability

The dewaterability of the sludge was evaluated using the capillary suction time and the zone settling velocity. The average capillary suction times for the different sludges are shown in Figure 24. Average CST for the thickened sludge are shown in Figure 25. More detailed information of the tests are found in Appendix 9.

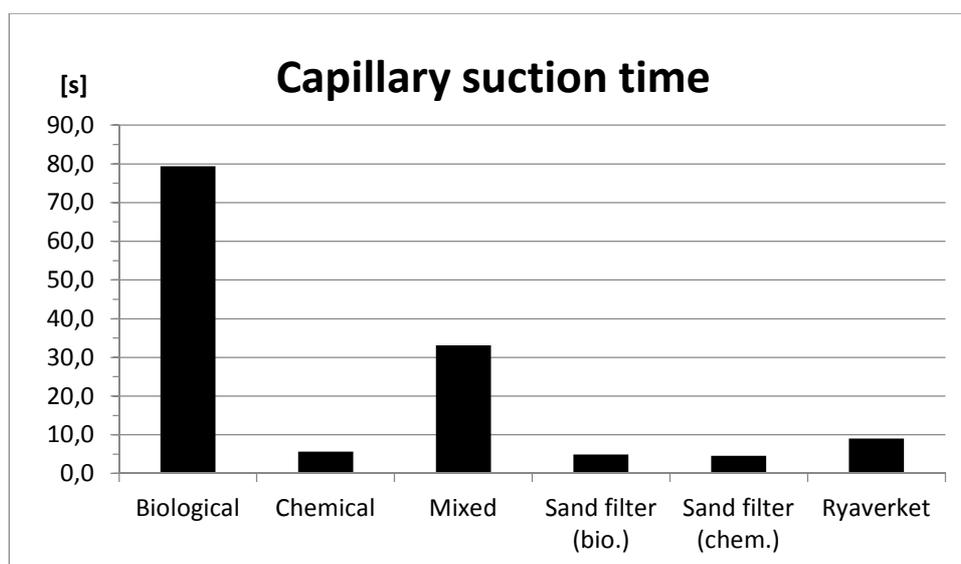


Figure 24 Average capillary suction times for the sludge sampled in April 2015. Capillary suction time is a measurement of dewaterability in sludge.

It is seen in Figure 24 that there is a significant difference in CST between biological and mixed sludge compared to the rest. Biological sludge has a CST close to 80 s, mixed sludge above 30 s, and the rest below 10 s. The difference resembles the total solids test, where biological and mixed sludge had larger content of solids than the other. However, in that test the mixed sludge had the most particles, while the biological sludge has the largest CST. Nevertheless, the CST tests imply that the mixed sludge and the biological sludge are more difficult to dewater than the rest.

Inconsistency in the results when testing CST has been reported in the literature (Sawalha and Scholz, 2006). However, in this study only the biological sludge had measurements deviating more than 10 % from the mean value for that sludge.

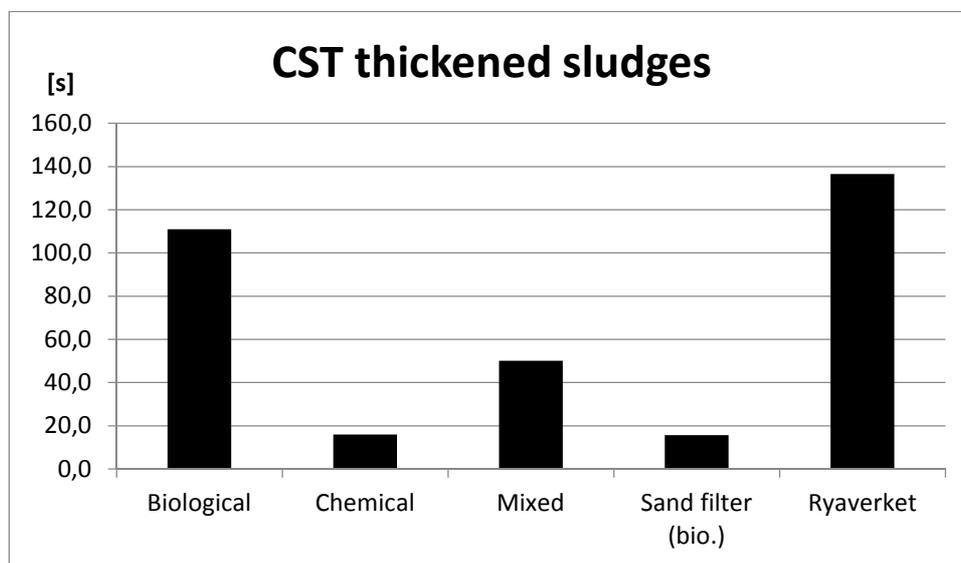


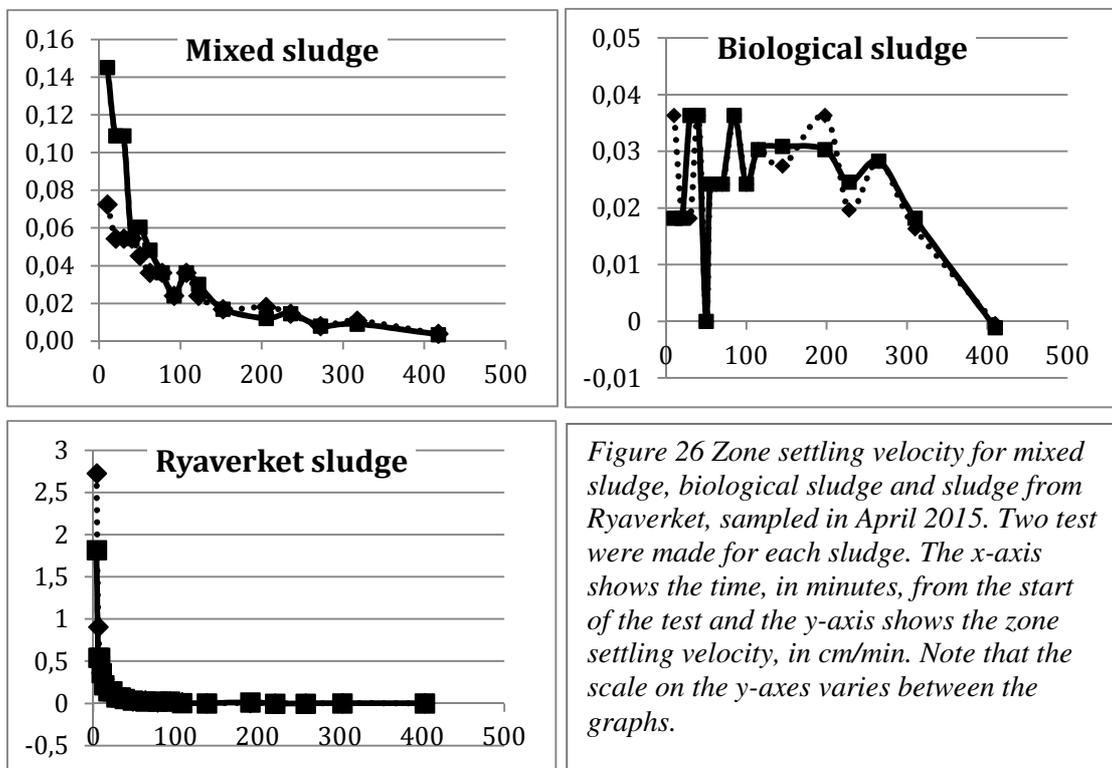
Figure 25 Average capillary suction times for thickened sludge, from the samples obtained in April 2015. Capillary suction time is a measurement of dewaterability in sludge.

As can be seen in Figure 25, the CST increases when the sludge is thickened. The most significant difference is for the Ryaverket sludge, where the CST is more than ten folded for the thickened sludge. The biological and the mixed sludge has CSTs closer to the unthickened sludge. This implies that the effect of thickening was small for these sludge. For both tests the CST of the mixed sludge is closer to the chemical sludge than the biological sludge. The amount thickened sludge from sand filter (chemical line) was not sufficient to perform a CST test.

Jin, et al. (2004) made CST measurements on activated sludge and obtained values between 12 – 20 s. The authors showed a correlation between the CST and the bound water in the sludge. In this study, the sludge with high CST had a high bound water content as well. Sawalha (2011) made measurements on several types of sludge and obtained in average 8.0 s for activated sludge, 83 s for waste sludge from an activated sludge process and above 1000 s for digested sludge. The value for activated sludge corresponds well to the one obtained for Ryaverket sludge. The biological sludge at Brudaremissen has a CST corresponding to the literature value for waste sludge. The mixed sludge consists to the larger part of chemical sludge. This is also seen when studying the results.

The zone settling velocities were only possible to evaluate from a few of the tested sludge. For the chemical sludge and both sludges from sand filters it was not possible to plot a zone settling velocity curve. Raw data for calculation of the zone settling velocities are presented in Appendix 10. The zone settling velocities for sludge at the second sampling occasion are shown in Figure 26. Two parallel tests were performed in identical cylinders at the second sampling.

The appearances of the curves are different between the sludges. It can however be seen that the parallel tests of the sludge match well. The Ryaverket sludge (see Figure 26) has a high zone settling velocity initially before thickening starts. A significant change in sludge volume can be expected after a couple of hours of settling and thickening. For this sludge, settling would be a useful alternative for dewatering. However, the Ryaverket sludge is obtained from a well-functioning wastewater treatment plant, therefore these results should be expected. The pilot plant at Brudaremassen has not been running for a long time, and the sludge show a different zone settling velocity curve. The mixed sludge, displayed in Figure 26, has a curve that has a similar appearance as the Ryaverket sludge, even though the magnitude is lower. This implies that settling could be used to reduce the volume of the sludge, however it would be less efficient. The biological sludge has a curve that diverges from the others. The magnitude of the zone settling velocity is lower, and it does not have a parabolic appearance. It is possible that the zone settling velocity would be more similar to the Ryaverket sludge if the pilot plant was more adjusted.



When comparing the CST and the zone settling velocity curves, the results are similar. The sludge with the most increased CST, Ryaverket sludge, is also the sludge that has the best zone settling. The sludge with the lowest increase in CST, biological sludge, is also the one with the worst zone settling. Overall, the implication is that the dewatering characteristics of the biological sludge likely would not be improved by settling. Furthermore, its characteristics are not very good to begin with, as shown by the large CST. According to the results the dewaterability is better for the mixed sludge than the biological sludge. Still it is worse than the sludge from Ryaverket.

7.2.5 Chemical composition of the sludge

The concentrations measured in the sludge from the biological unit and from the chemical precipitation are shown in Table 11. Additional results are found in Appendix 11. When evaluating the quality of the sludge these concentrations are compared to Revaq guidelines, reference values for polluted soil from the Swedish EPA and concentrations measured in sludge from other treatment plants. See Swedish EPA (2009) for more details.

Table 11 Concentrations of different compounds in sludges from Brudaremossen.

Parameter	Unit	Biological sludge	Chemical sludge
Aliphatic >C16-C35	mg/kg TS	381	43
PAH		Not detected	Not detected
PCB		Not detected	Not detected
Ba	mg/kg TS	2050	3330
Cd	mg/kg TS	0.239	0.705
Cr	mg/kg TS	30.8	35.4
Cu	mg/kg TS	35.3	31.6
Ni	mg/kg TS	12.6	7.17
Pb	mg/kg TS	6.81	18.3
Zn	mg/kg TS	112	176
Fe	g/kg TS	102	281
Mn	mg/kg TS	1820	398
Fixed solids	% of TS	46.6	74.2

As seen in Table 11, the organic contaminants PAHs and PCBs were not detected in the sludge. A few of the Revaq parameters were detected in low concentrations. When comparing to reference values from the Swedish EPA it is seen that only a few compounds exceed the limitations. Aliphatic hydrocarbons >C16-C35 in biological sludge and Cd in the chemical sludge exceeds the lower limit value, for sensitive land usage. The concentrations of the metal barium greatly exceeds guideline values for both sensitive and non-sensitive land usage.

The low concentration of organic contaminants in the sludge was not predicted. The compounds are partly removed in the chemical precipitation step, and therefore it was expected that they would be found in the sludge. The question is where they actually end up. It is possible that organic contaminants accumulate in the sand and activated carbon filters. Therefore samples should be taken from the filters and sent to analysis. If the concentrations are high the filters must be taken care of at the end of their lifecycle. There might be necessary to handle the backflush differently if the concentrations of contaminants are high.

The concentration of toxic metals is similar to concentrations found in other sludges from water and wastewater treatment (Blomberg, 1997; Mattsson 2015). Some compounds, such as Cu, are found in significantly lower concentrations in the Brudaremossen sludge. The concentration of iron in the Brudaremossen sludge is very high, especially in the chemical sludge. This is expected since the concentration in the incoming leachate is elevated. The chemical sludge contains more fixed material and less organic content, compared to the biological sludge.

When the sludge samples for chemical analysis were obtained they were allowed to settle. The supernatant was then removed before the samples were sent to analysis. Due to the settling it would be expected that the total solids value would be higher than in the test for physical properties. This is not the case for the biological sludge. One reason for this could be that the sludge was in fact thicker when studying the physical properties. Another possibility is that the biological sludge did not settle long enough. Therefore some particles could have been missed in the chemical test.

7.2.6 Handling of sludge

The concentrations of contaminants in the sludges do not motivate separated handling of them. The physical properties of the chemical and the biological sludge was different according to the tests. The chemical sludge had very good settling and thickening properties. This was not the case for the biological sludge. If the sludges would be mixed, it is possible that the chemical sludge improves the settlement of the biological sludge. A simple gravitational settling tank could be efficient in reducing the volume of the combination of the sludges.

Thereafter mechanical dewatering can be applied, which can greatly decrease the sludge volume (Werther and Ogada, 1999). Examples of applicable methods are belt presses and centrifuges. The location of the Brudaremossen landfill, in a recreational area, makes drying beds an unsuitable treatment option. Further studies are needed to determine a suitable handling.

A more detailed study is needed to assess the feasibility of the sludge treatment and handling. The treatment of the sludge is partly based upon how it should be disposed, as some alternatives have higher requirements of its properties (Svenskt Vatten, 2014). The concentration of compounds in the sludge does not alarmingly exceed any guideline values, and it is far below Revaq limits. Apart from the iron concentration, there is nothing remarkable in the sludge. Literature claims that sludge is not suited for incineration if the concentrations of metals are high (Gray, 2010).

The reasonable alternatives for sludge disposal are landfilling, incineration and reuse in materials. The results from the single chemical test of the sludges shows that the concentrations of contaminants are low. Drawing conclusions from these results, all three options are suitable. However, if the sludge would be reused in materials, the effect of the high concentrations of iron and barium should be studied. The leachate from Brudaremossen contains many different contaminants, often in low concentrations, some known to be carcinogenic or harmful to animals and the environment. It is possible that the analysed sludge sample is not representative, since the concentrations of contaminants are unexpectedly low. It is expected that the concentrations in the sludge will increase when the plant has been in operation for a longer time. Also, the group of known organic contaminants and their negative consequences are still growing and developing. It is possible that the risk of the contaminants in the sludge is underrated. For these reasons, it is recommended to dispose the sludge cautiously. Therefore options to reuse or put the sludge in a non-hazardous landfill are not applicable. Hopefully, incineration can be a suitable option. No incineration plants solely for sludge exists in Sweden (Svenskt Vatten, 2013). However there are waste incinerations plants capable of handling sludge. The capacity

exists as well. If future sludge analyses show elevated concentrations of contaminants the sludge needs to be placed in a landfill for hazardous goods. In order to find the preferred alternative, incineration or hazardous landfilling, a life cycle analysis should be made, both for the environmental and the economic aspects.

7.3 General discussion

The samples taken during this project have been grab samples. The leachate and sludge most likely varies in contaminant concentration over time therefore cluster samples may give more correct result. The reason why cluster sampling have not been applied is because it makes the sampling more demanding and there is a risk that contaminants may oxidise or vaporise if the sample are stored longer. The grab samples give an understanding of the current situation but it is hard to see trends and the long-term performance of the treatment if only weekly/monthly samples are taken. Therefore seasonal changes in the leachate quality and treatment settings (flow, chemical dosage etc.) have not been included or evaluated in the study since the project took place during a limited time with limited data.

For the physical testing of the sludge some tests were performed on sludge older than 48 hours. There is a possibility that this have affected the result. For the second testing (13 April) visible differences in the biological sludge was seen. For the other sludges no clear differences was observed but it might still have affected the result.

For toxic metals and organic contaminants the environmental discharge criteria is reached by the chemical treatment line in the pilot plant. The criteria is fulfilled without treatment of toxic metals since the incoming leachate has a low concentration of these. For the organic contaminants the concentration was below or slightly over the environmental criteria in the incoming water. The chemical precipitation treatment step have a good removal rate for Fe, TSS and PCB 7. Also aliphatic hydrocarbons (>C16-C35) and PAH H are removed to some extent. The sand filter removes Fe, TSS, PAH L, PAH M and to some extent carbon, PCB 7 and aliphatic hydrocarbons. Neither chemical precipitation nor sand filter removes toxic metals. The final treatment step, activated carbon filter, removes carbon, Zn and Cu. The MBBR treatment does not seem to improve the removal further for organic contaminants and toxic metals.

The environmental discharge criteria that has been used comes from different sources. The most local guideline has been counseled. For some compounds, especially PAH L, M and H, no suitable guideline was found. Therefore the individual PAH compounds have been compared to mostly Canadian guidelines, see Appendix 1. The effluent measurements at Brudaremossen on PAHs are far below these values. The Canadian guidelines are strict compared to some European values. However there are some Canadian guideline values on specific PAH compounds that are of higher magnitude, these have been disregarded.

Based on measured discharge concentrations from the pilot plant it would be practical to continue discharging the treated leachate to Ryaverket WWTP. The contaminant concentrations are below the environmental limit and the effect on the sludge at Ryaverket is assumed to be very low. The landfill Sofielund outside Huddinge pre-treats the leachate before discharge to Henriksdals WWTP, which still has a Revaq

certification. However, continued discharge to Ryaverket needs to be thoroughly investigated to ensure that no negative effects occur. Discharging the leachate to a different recipient requires new pipes and a better treatment of nutrients. This is estimated to be more expensive (Lundh, 2015a) and possible no improvement in environmental impact will follow.

7.4 Comparison with other treatment plants

Compared to the other studied leachate treatment plants Brudaremossen reaches a similar result. The following discussion will compare the pilot treatment plant at Brudaremossen with leachate treatment at Sofielund, Vankiva and Japan M. A compilation of the removal efficiencies is seen in Table 12.

Table 12 Comparison of treatment efficiencies in four leachate treatment plants. All concentrations are given in µg/L and the removal rate is written as R [%]. The first compound group is the aliphatic hydrocarbons >C16-C35.

Compounds	Hässleholm		Sofielund		Japan		Brudaremossen chemical line	
	Before	R [%]	Before	R [%]	Before	R [%]	Before	R [%]
>C16-C35	39	97	-	-	-	-	10	52
PAH L	0.34	99	0,72	72	3,7	92	0,47	84
PAH M	0.54	100	0,51	45	0,68	98	0,58	98
PAH H	0.05	100	-	-	0,016	75	0,03	26
PCB 7	-	-	-	-	0,00023	93	0,01	96
Fe	12000	88	-	-	-	-	11	99
Cu	110	83	35	-150	-	-	0,88	59
Cr	69	23	35	21	-	-	3,3	22
Ni	89	44	62	3,7	-	-	6,9	13
Zn	360	89	150	64	-	-	9,1	77

For all organic contaminants the outgoing leachate had concentrations below the detection limit, except for one measurement occasion for PCB 7. When calculating the removal rate, half of the detection limit have been used as the effluent concentration. Therefore the removal rates are approximates for Brudaremossen. The removal rate of PAH L and PAH M (84 % and 98 % respectively) were comparable to the results at the reference treatment plants. For PAH H only a 26 % removal rate was achieved at Brudaremossen. The incoming water had a low concentration and it makes it more difficult to treat, nevertheless the other treatment plants did manage to have higher removal despite their low concentration of PAH H in the incoming leachate. Japan M had only 0.02 µg/L in incoming leachate and removed 75%. Brudaremossen reached a 96 % removal of PCB 7. Only Japan M measured PCB 7 out of the other plants and reached 93 % removal. For aliphatic hydrocarbons (>C16-C35) the removal was lower at Brudaremossen with 52 % removal. At Sofielund aliphatic hydrocarbons (>C16-C35) were below detection limit and at Vankiva the removal was 97 %.

As for organic contaminants some toxic metals were also measured below the detection limit. In the effluent from the treatment lines, Pb, Cd, Hg and in half of the

effluent samples Cu and Zn were below the detection limit. When comparing the plants performance in removal of metals it can be seen that Zn, Cr and Fe reached similar results. For Cu the removal was slightly lower than at Vankiva. The removal of Ni differs between the facilities (Vankiva 44 %, Sofielund 3.7 % and Brudaremissen 13 %). All influent and effluent samples of Pb and Cd were below detection at Brudaremissen and therefore not possible to compare. The removal of toxic metals at Brudaremissen is good and similar to the ones achieved at the reference treatment plants. Brudaremissen have much lower concentrations of metals in the influent than the other plants and it is more difficult to remove low concentrations.

The reference facilities are full scale leachate treatment plants and they have been operating for longer time. Therefore they have had time to adjust and optimise the treatment. Brudaremissen is a newly started up pilot treatment plant hence the removal is expected to improve by time. The leachate at these different plants are not the same and the discharge criteria may differ. Therefore the plants have individual goals with the treatment. Overall, the efficiency obtained at the Brudaremissen pilot leachate treatment plant is fair and comparable.

7.5 Further studies

This study has been conducted on samples from a limited time period, February to May 2015. In order to fully evaluate the performance of the plant, leachate samples should also be obtained during summer, fall and winter. This will be done by Ramböll Sverige AB. A follow-up to this study should then be made to ensure that the conclusions are still valid.

The concentrations of different contaminants in the effluent were low. It would be interesting to know how much these contaminants affect the quality of the sludge at Ryaverket. It should be determined whether or not the leachate from Brudaremissen has a severe impact on the Ryaverket plant. This may require a lower detection limit when analysing the samples. When knowing the actual effluent concentration a better conclusion can be made and the impacts on Ryaverket sludge can be determined.

Several studies have already been conducted on the leachate at Brudaremissen. Removal of contaminants in filters was studied by Kalmykova et al. (2014) and the efficiency of biological treatment was evaluated by Porsgaard and Söderström (2015). One aspect that has not been the primary target in any study is the performance of the chemical treatment unit. It would be interesting to know how adjustments to pH, additives, retention time etc. affects its efficiency. A pre-investigation of the subject was performed before the construction of the pilot plant (Lundh, 2014). A more detailed study could evaluate if it also is possible to adjust the treatment to allow removal of toxic metals. Efficient removal of toxic metals was not achieved at Brudaremissen, however it is possible to do this according to literature (Fu and Wang, 2011). Another aspect could be to test other means of separation. The lamella sedimentation could perhaps be adjusted, or replaced with different techniques.

Further tests should be performed on the sludge to determine how its dewatering properties can be improved. No attempt was made in this study to evaluate how the properties changed if different treatment options are used. Possible future studies are

to evaluate how different chemical additives can be used to improve the dewatering of the sludge. Small scale tests of different mechanical dewatering equipment could be performed. This would however require additional resources and facilities not available in this study.

As a last proposal of further studies, the total environmental impact of the treatment facility could be investigated. From this, it might be possible to change to more sustainable treatment alternatives, such as less energy and chemical consuming solutions.

8 Conclusions

The pilot leachate treatment plant at Brudaremossen was evaluated regarding the removal of organic contaminants and toxic metals. The results show that the plant sufficiently removes organic contaminants, to below the detection limits. The toxic metals are not reduced significantly. However the concentrations in the incoming leachate, before treatment, were low and less than environmental discharge limits.

The organic contaminants are mainly removed in the chemical precipitation step and the sand filter. The MBBR does not seem to improve the removal of organic contaminants and toxic metals any further. A correlation between many different parameters were made. It was seen that the organic contaminants correlated better to TSS and total Fe than to TOC, DOC and COD.

The facility was compared to three similar leachate treatment plants and it was concluded that Brudaremossen reaches a comparable discharge quality. The removal of a few parameters (PAH H, Aliphatic hydrocarbons (>C16-C35) and Cu) were lower than in the reference plants. However, these plants are full scale facilities, which have been optimised and stabilised. By time, it is plausible to assume that Brudaremossen will have improved its efficiency.

The tests performed on the sludge showed that gravimetric settling could efficiently reduce the sludge volume. This would be an efficient and low cost method to handle the sludge, especially for the chemical sludge. The physical properties of the sludge will likely improve when the pilot plant is more adjusted. The results indicate that further treatment with mechanical dewatering will be possible and necessary. The sludge contained low concentrations of contaminants, which was unexpected. A further investigation of the sludge is required to ensure the low concentrations. If no contaminants are found, the sand and carbon filters should also be analysed for the content of contaminants.

The results from the chemical analysis in this study should be followed up when the pilot plant has been in operation for a longer time. This would make it possible to account for seasonal changes. With additional data from a longer time period a more reliable conclusion can be made. If possible, it would be of interest to analyse with lower detection limits for both sludge and leachate to ensure a satisfactory removal rate.

Based on the measurements, the contaminant concentrations are low in the treated leachate and would likely not affect the sludge at Ryaverket much. This should however be further investigated. The sludge produced at Brudaremossen should preferably be incinerated, otherwise placed in a landfill for hazardous goods. During this project, it seems as the treatment plant at Brudaremossen will fulfill its purpose, with regard to organic contaminants and toxic metals.

9 References

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Appendices

- Appendix 1 – Compilation of environmental limits
- Appendix 2 – Leachate analysis 2008 – 2012
- Appendix 3 – Correlation theory and MATLAB code
- Appendix 4 – Detailed results from leachate analysis
- Appendix 5 – Results from sludge sampling March 2015
- Appendix 6 – Detailed results from total solids in sludge
- Appendix 7 – Detailed results from total suspended solids in sludge
- Appendix 8 – Detailed results of water content in sludge
- Appendix 9 – Detailed results of capillary suction time for sludge
- Appendix 10 – Raw data for zone settling velocity
- Appendix 11 – Detailed chemical composition of sludge
- Appendix 12 – Sludge Volume Index

Appendix 1 – Compilation of environmental guidelines

The table below shows environmental guidelines for water. These values were used for evaluating the quality of the leachate and the effluent from the treatment process. Note that the theoretical background to the values varies between the sources and that not all values may be directly applied to the conditions at Brudaremossen landfill. For details, see the main report.

Compound	Type of compound	Subclass	Limit [µg/L]	Source
Aluminium	Metal	Metal	87	US EPA recommendations
Arsenic	Metal	Toxic Metal	15	City of Gothenburg
Cadmium	Metal	Toxic Metal	0,4	City of Gothenburg
Chromium	Metal	Toxic Metal	15	City of Gothenburg
Copper	Metal	Toxic Metal	10	City of Gothenburg
Iron	Metal	Metal	1000	US EPA recommendations
Lead	Metal	Toxic Metal	14	City of Gothenburg
Mercury	Metal	Toxic Metal	0,05	City of Gothenburg
Nickel	Metal	Toxic Metal	40	City of Gothenburg
Zinc	Metal	Toxic Metal	30	City of Gothenburg
Acenaphthene	Organic contaminant	PAH	5,8	Canadian guidelines
Anthracene	Organic contaminant	PAH	0,1	EU annual average
Benz (a) anthracene	Organic contaminant	PAH	0,018	Canadian guidelines
Benzo (a) pyrene	Organic contaminant	PAH	0,05	City of Gothenburg
Benzo (g,h,i) perylene	Organic contaminant	PAH	0,082	EU annual average
DEHP	Organic contaminant	Phthalates	1,3	EU annual average
Fluoranthene	Organic contaminant	PAH	0,0063	EU annual average
Fluorene	Organic contaminant	PAH	3	Canadian guidelines
Napthalene	Organic contaminant	PAH	2	EU annual average
Octylphenols	Organic contaminant	Other	0,1	EU annual average
PCB	Organic contaminant	PCB	0,014	City of Gothenburg
PFOS	Organic contaminant	Other	0,0063	EU annual average
Pyrene	Organic contaminant	PAH	0,025	Canadian guidelines
Cyanide	Other	Other	5,2	US EPA recommendations
Oil index	Other	Other	1000	City of Gothenburg
pH	Physical parameter	Other	6-9	City of Gothenburg
Suspended material	Physical parameter	Other	25000	City of Gothenburg
Aliphatic Hydrocarbons C16-C35	Sum parameter	Hydrocarbons		
TOC	Sum parameter	Organic material	12000	City of Gothenburg
Total nitrogen	Sum parameter	Nitrogen	1250	City of Gothenburg
Total phosphorus	Sum parameter	Phosphorus	50	City of Gothenburg

The guideline values from the City of Gothenburg are not necessarily applicable in the rest of Sweden. For such landfills the following table is used.

Compound	Swedish EPA Lakes and freshwater quality classes				
	1	2	3	4	5
Chromium	0,3	5	15	75	>75
Copper	0,5	3	9	45	>45
Nickel	0,7	15	45	225	>225
Zinc	5	20	60	300	>300
Lead	0,2	1	3	15	>15
Cadmium	0,01	0,1	0,3	1,5	>1.5
Arsenic	0,4	5	15	75	>75
	EU max. allowed conc.		Unit for all: µg/L		
Mercury	0,07	µg/L			

In order to become certified for Revaq, landfill leachate cannot contribute significantly to the contaminant concentration in sludge. In a pre-study made by Allmyr and Sternbeck (2014) the following concentrations was seen as a low risk level in sludge. The concentrations was used when evaluating the sludge quality at Brudaremassen.

Organic contaminants	Low risk level [mg/kg TS]
Tri(2-butoxyetyl) phosphate	650
DEHP	6 500
Diisobutyl phthalate	20
Diisononyl phthalate	23 000
Dibutyl phthalate	1 700
Perfluorooctanesulfonic acid (PFOS)	0.12
Anthracene	80
Benzo (a) pyrene	28
Nonylphenol	350
Polybrominated diethylether (BDE-47)	440
Polybrominated diethylether (BDE-99)	100
Aliphates >C16-C35	80 000

Appendix 2 – Leachate analyses 2008 – 2012

The results from the leachate analyses at Brudaremissen between 2008 and 2012 are shown in the table below. The leachate was sampled from the outlet of the pond at Brudaremissen.

Leachate Analyses 2008-2012					
		min	max	mean	median
Flow	L/s	4.0	7.5	5.1	5.0
Temperature	°C	3.6	14	8.3	8.3
Colour		26	350	214	198
pH		7.1	8.1	7.5	7.40
Conductivity	mS/m	150	287	217	210
Alkalinity	HCO ₃ mg/L	100	1700	1080	1100
Suspended solids	mg/L	10	69	30	25
COD _{Mn}	mg/L	17	40	25	21
TOC	mg/L	28	410	59	39
Oxygen	mg/L	4.6	9.5	7.6	8.2
Oxygen saturation	%	39	79	65	69
Total phosphorus	mg/L	0.06	0.28	0.14	0.11
Ammonium nitrogen	mg/L	39	114	74	75
Nitrate nitrogen	mg/L	0.06	0.50	0.28	0.19
Total nitrogen	mg/L	47	110	78	78.
Calcium	mg/L	84	112	98	99
Chloride	mg/L	119	283	194	192
Iron	mg/L	4.1	21	12	11.
Manganese	mg/L	0.49	0.97	0.62	0.63
Aluminium	mg/L	0.01	0.22	0.04	0.03
Arsenic	µg/L	0.05	1.70	1.07	1.03
Lead	µg/L	0.30	1.55	0.71	0.66
Mercury	µg/L	0.02	0.08	0.07	0.10
Cadmium	µg/L	0.04	0.10	0.08	0.10
Zinc	mg/L	0.00	0.15	0.02	0.01
Chromium	µg/L	4.3	19	6.5	6.0
Nickel	µg/L	1.2	16	7.2	6.8
Copper	µg/L	0.81	3.19	1.57	1.20
Cyanide	mg/L	0.00	0.02	0.01	0.01
Phenols	µg/L	5.0	80	19	17
Total PCB	µg/L	0.02	0.35	0.26	0.35
PCB 7	µg/L	0.00	0.07	0.05	0.07
PCB-28	µg/L	0.00	0.01	0.01	0.01
PCB-52	µg/L	0.00	0.03	0.01	0.01
PCB-101	µg/L	0.00	0.01	0.01	0.01
PCB-118	µg/L	0.00	0.01	0.01	0.01
PCB-138	µg/L	0.00	0.01	0.01	0.01
PCB-153	µg/L	0.00	0.01	0.01	0.01
PCB-180	µg/L	0.00	0.01	0.01	0.01
Benz(a)anthracene	µg/L	0.02	0.05	0.02	0.02
Chrysene	µg/L	0.01	0.05	0.03	0.02
Benzo(b,k)fluoranthene	µg/L	0.01	0.04	0.03	0.02
Benzo(a)pyrene	µg/L	0.02	0.04	0.02	0.02

Indeno(1,2,3-cd)pyrene	μg/L	0.01	0.02	0.02	0.02
Dibenzo(a,h)anthracene	μg/L	0.01	0.02	0.02	0.02
Sum carcinogenic PAH	μg/L	0.01	0.20	0.12	0.13
Napthalene	μg/L	0.01	1.40	0.22	0.07
Acenaftylen	μg/L	0.02	0.10	0.04	0.03
Fluorene	μg/L	0.03	1.25	0.60	0.53
Acenaphthene	μg/L	0.01	1.44	0.57	0.58
Phenanthrene	μg/L	0.02	0.96	0.19	0.05
Anthracene	μg/L	0.02	0.66	0.13	0.07
Fluoranthene	μg/L	0.03	0.39	0.17	0.16
Pyrene	μg/L	0.03	0.29	0.13	0.12
Benzo(g,h,i)perylene	μg/L	0.01	0.20	0.03	0.02
Sum other PAHs	μg/L	0.16	4.50	1.90	1.50
Oil index	mg/L	0.05	0.39	0.14	0.10

Appendix 3 – Correlation theory and MATLAB code

Correlation is measured by performing the following analysis for each parameter. This will give a value between - 1 and 1, where 1 is a perfect positive correlation and -1 is a perfect negative correlation. A value of 0 means that there is no correlation at all between the components. Correlation is calculated using the following formula.

$$C = \frac{\sum(x_i y_i)}{\sqrt{(\sum(x_i - \bar{x})^2 \times \sum(y - \bar{y})^2)}}$$

C = coefficient of correlation

x = parameter 1

\bar{x} = mean value of parameter 1

y = parameter 2

\bar{y} = mean value of parameter 2

The MATLAB code used to perform the analysis is written below. In order to be able to use it, an Excel file containing the measurement values are required.

```
% The following program creates a correlation matrix from the data in a
% given Excel file

clc, clf, close all % Clears all variables and closes figures

categories=char('SumPCB','Temperatur','TOC','DOC','COD',
'TSS','TotN','TotP','Fe','Fe Filt','Mn','Cu','Cr','Ni',
'Zn','naftalen','acenaftylen','acenaften','fluoren','fenantren','antr
acen','fluoranten','pyren','bens(a)antracen','krysen','PAH,summa
16','PAH,summa c','PAH,summa ö','PAH,summa L','PAH,summa M','PAH,
summa H','Alifater >C16-C35','Aromater >C8-C10','Aromater>C10-
C16','PCB 28','PCB 52','PCB 101','PCB 118','PCB 138','PCB
153','PCB180'); % Labels the categories, if further analysis is
desired

mdata=xlsread('Varden.xlsx','Matrix kapad','C19:AQ30'); % Reads the
excelfile containing the measurement data set

corrmdata=corr(mdata,'type','Pearson'); % Creates a correlation
matrix of the parameters in the data set

xlswrite('Varden.xlsx',corrmdata,'Resultat','C3:AQ43') % Writes the
result into an excel file

% The code rows below performs a principal component analysis of the
data
% and visualises it. Details of principal component theory is found
% elsewhere.

[wcoeff,score4,latent4,tsquared4,explained4,mu4]=pca(mdata,
'VariableWeights','variance'); % Rescales the variables
coeff4 = inv(diag(std(mdata)))* wcoeff; % Orthonormalizes the
principal components for the rescaled variables
```

```
figure()
pareto(explained4)
title('Data')
xlabel('Principal Component')
ylabel('Variance Explained %')
```

```
figure()
biplot(coeff4(:,1:2), 'scores', score4(:,1:2), 'varlabels',
categories)
axis([-0.4 0.4 -0.4 0.4]);
title('Datan, PCA 1 och 2')
```

```
figure()
biplot(coeff4(:,2:3), 'scores', score4(:,2:3), 'varlabels',
categories)
axis([-0.4 0.4 -0.4 0.4]);
title('Datan, PCA 2 och 3')
```

Appendix 4 – Detailed results from leachate analysis

Detailed results from the leachate analyses are shown in the tables below. All toxic metals and organic contaminants except Hg and PCB 7 are given in µg/L. Hg and PCB 7 are in ng/L. Fe, COD, TOC, DOC, TSS are given in mg/L.

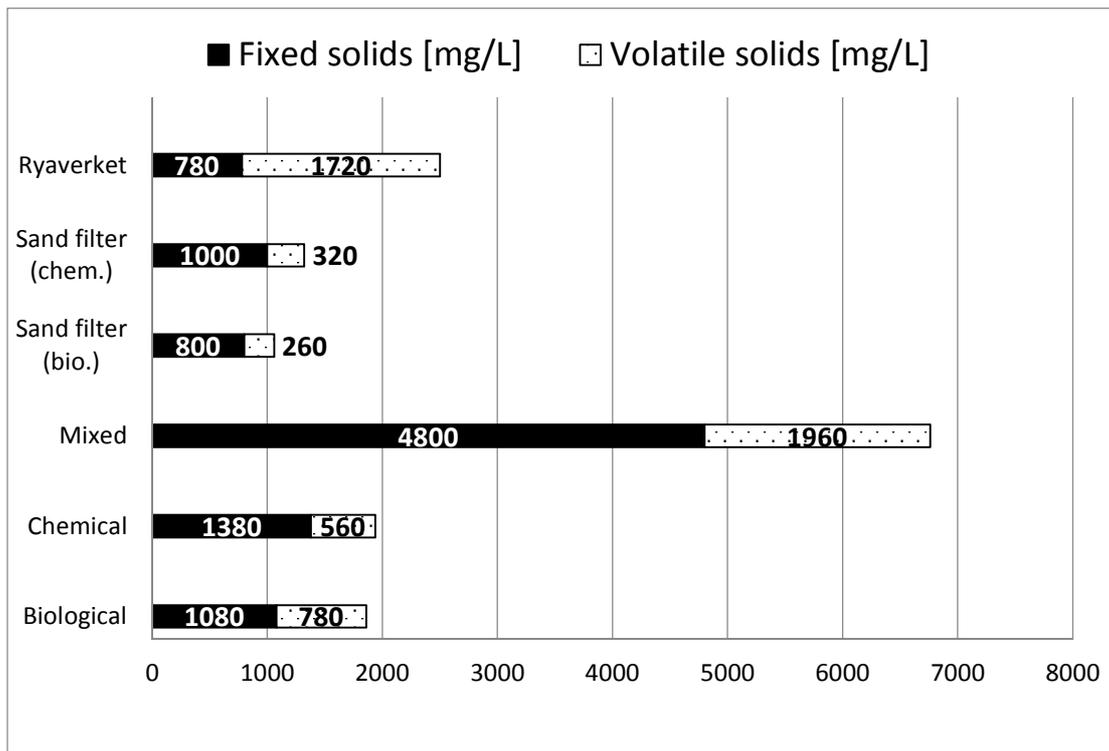
		11-feb	17-feb	26-feb	04-mar	10-mar	18-mar	26-mar	01-apr	09-apr	15-apr
A	Pb		<0,2								<0,2
	Cd		<0,02								<0,02
	Cu		<0,5								1.5
	Cr		3.3								3.2
	Hg		<5								<5
	Ni		7.9								5.9
	Zn		10								8.2
	PAH L		0.4				0.73				0.29
	PAH M		0.49				0.78				0.47
	PAH H		0.022				0.027				0.032
	>C16-35		10				10				11
	PCB 7		10				94				18
	COD		94				10				83
	Fe		13				35				11
	TOC		30				33				27
DOC		28				29				26	
TSS		32								32	
B	Pb		<0,2				<0,2				<0,2
	Cd		0.02				<0,02				<0,02
	Cu		6.2				3.6				2
	Cr		2.9				3.8				2.6
	Hg		<5				<5				<5
	Ni		8.5				7.4				6
	Zn		24				19				7.2
	PAH L		0.52		0.38		0.98		0.72		0.73
	PAH M		0.72		0.54		1.3		1.1		1.4
	PAH H		<0,04		<0,04		0.011		<0,04		0.033
	>C16-35		11		<10		<10		<10		<10
	PCB 7		7.2				3.8				4.7
	COD		90				86				80
	Fe	5.1	3.3			5.2	1	5.9	6.4		3.4
	TOC		29				33				27
DOC		25				33				27	
TSS	19	11	5.8	10	12	6.3	20	13		12	
F	COD		92				320				220
	Fe		0.12		0.15		0.33		0.33		0.51
	TOC	54	32	19	54	21	100	45	52	62	68
	DOC		31		55		110		52		62
	TSS						21				5.5
G	COD		60				53				230
	Fe		0.13		0.07		0.13		0.18		0.75
	TOC		16		44		24		49		66
	DOC	33	15	8.6	44	10	18	25	49		60

		11-feb	17-feb	26-feb	04-mar	10-mar	18-mar	26-mar	01-apr	09-apr	15-apr
F'	Pb		0.3				<0,2				<0,2
	Cd		0.024				<0,02				<0,02
	Cu		7.7				3.2				1.1
	Cr		3.3				3.3				2.9
	Hg		<5				15				<5
	Ni		8.2				7				5.8
	Zn		14				5.4				4
	PAH L		<0,15				<0,15		<0,15		<0,15
	PAH M		0.01				0.013		0.022		0.01
	PAH H		<0,04				<0,04		<0,04		<0,04
	>C16-35		<10				<10		<10		<10
	PCB 7		2.7				1.8				2.7
	COD		73				46				120
	Fe		0.11				0.3				0.57
	TOC	28	27	23	23	23	20	32	21	29	39
DOC		26			23	19		21		38	
G'	Pb		<0,2				<0,2				<0,2
	Cd		<0,02				<0,02				<0,02
	Cu		<0,5				<0,5				0.58
	Cr		1.7				2.7				3.2
	Hg		<5				<5				<5
	Ni		4.7				7.3				6
	Zn		<3				<3				3.3
	PAH L		<0,015		<0,015		<0,015		<0,015		<0,015
	PAH M		<0,025		<0,025		<0,025		<0,025		<0,025
	PAH H		<0,04		<0,04		<0,04		<0,04		<0,04
	>C16-35		<10		<10		<10		<10		<10
	PCB 7		1.5				<0,1				<0,1
	COD		30				38				61
	Fe		0.04		0.15		0.08		0.06		0.1
	TOC	2.5	9.2	12	17	13	16	24	17	25	23
DOC		9		17		16		17		23	

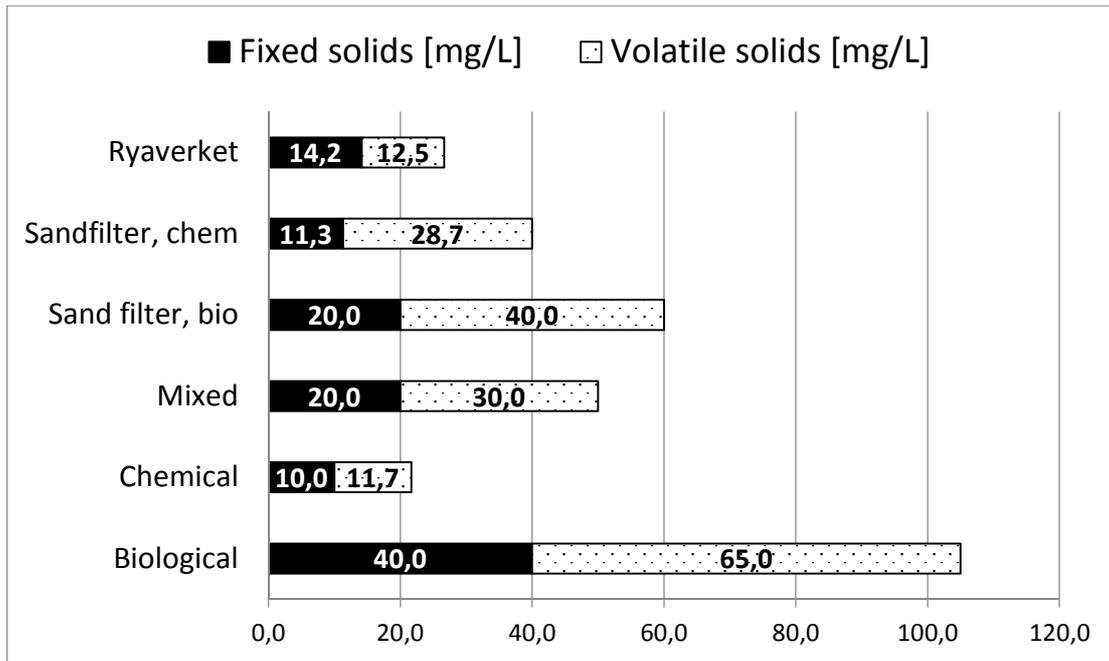
Appendix 5 – Results from sludge sampling March 2015

This appendix contains the results from the first test of physical properties of the sludge from Brudareossen. The sampling was performed 2015-03-16.

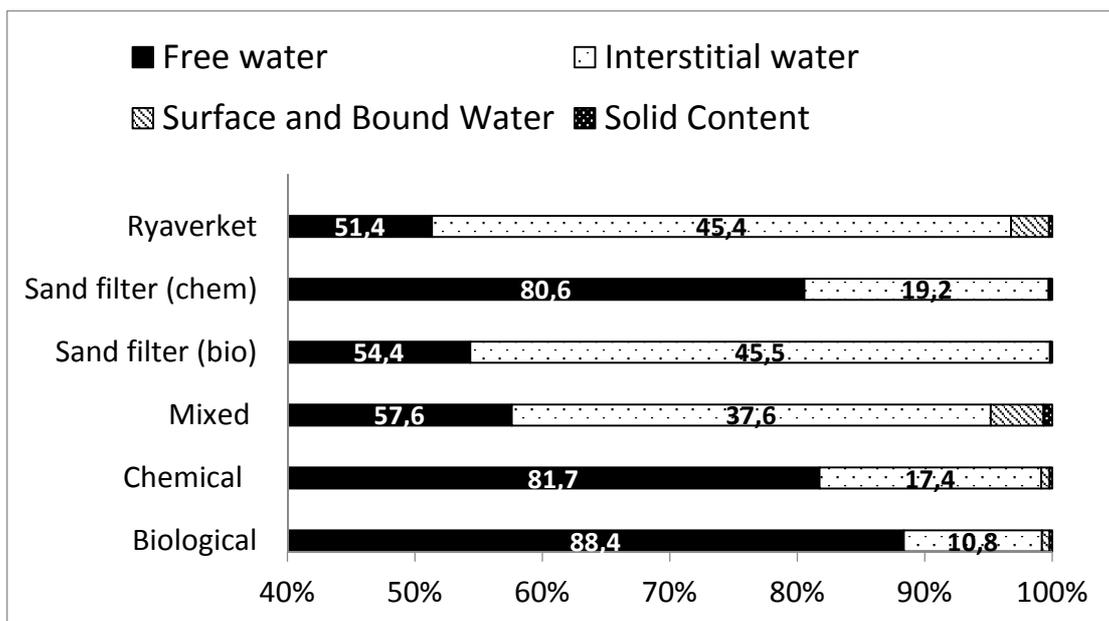
The solids content in the sludge from the samples obtained in March are shown in the figure below. The concentrations are given in mg/L. Fixed solids are red and volatile solids are blue.



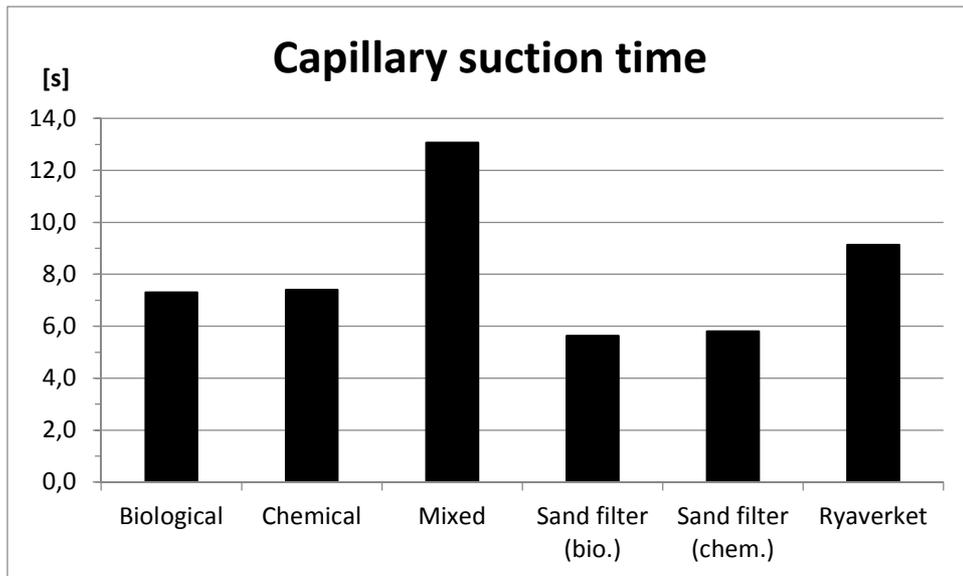
The total suspended solid measured in the supernatant from the settling test at the first sampling in March 2015 is shown in the figure below. The values shown are averages. The concentrations are given in mg/L. Fixed solids are red and volatile solids are blue.



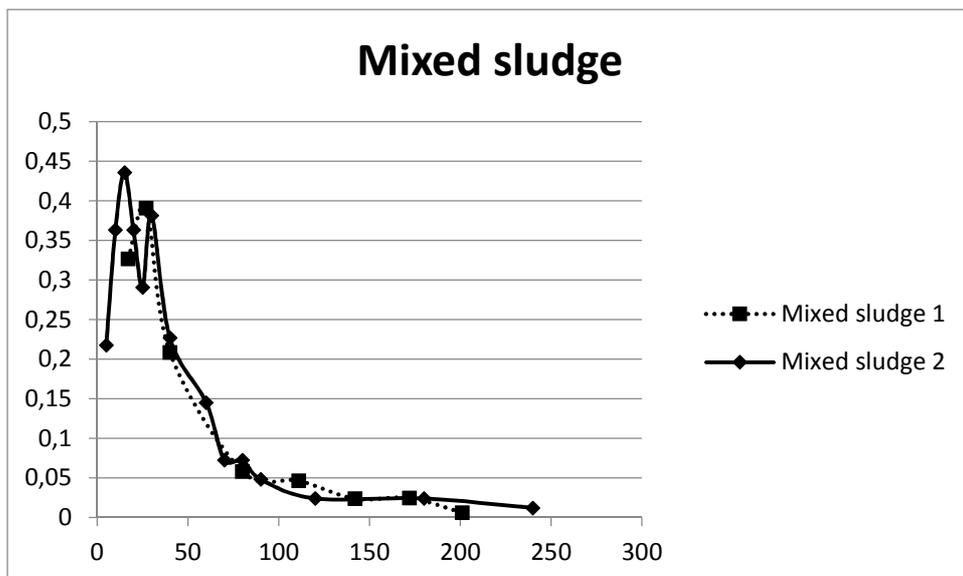
The figure below shows the percentages of different types of water, for the first sampling, in March 2015.



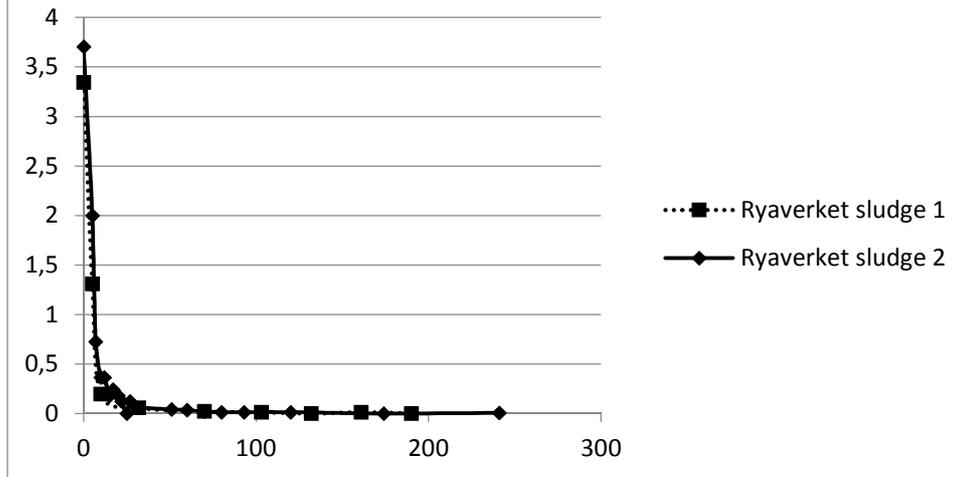
The figure below shows the average capillary suction time for the sludge sampled in March 2015. No CST was measured on thickened sludge at this occasion.



The zone settling velocities for the sludge from the first sampling occasion, in March 2015 is shown in the figures below. At this occasion the two tests were performed with 18 hours in between. In both figures the x-axis shows the time in minutes and the y-axis shows the zone settling velocity in cm/min.



ZSV Ryaverket sludge



Appendix 6 – Detailed results from total solids in sludge

The detailed results from the laboratory analyses of the sludge are presented in this chapter. They are presented in the same order as they are introduced in the report. Detailed results from the analyses of total, volatile and fixed solids are shown in the tables below. The test on thickened sludge and the test performed in March were made at room temperature (20°C) while the test on normal sludge in April was performed on sludge with a temperature of approximately 15.5°C.

April 2015	Biological			Chemical			Mixed		
Total solids [mg/L]	22460	22500	22320	1960	2140	2040	31740	32180	32400
Average total solids	22427			2047			32107		
Volatile solids [mg/L]	12660	12960	13300	380	440	400	8260	8440	8440
Average volatile solids	12973			407			8380		
Fixed solids [mg/L]	9800	9540	9020	1580	1700	1640	23480	23740	23960
Average fixed solids	9453			1640			23727		

April 2015	Sand filter (bio.)			Sand filter (chem.)			Ryaverket		
Total solids [mg/L]	1200	1320	1420	1320	1040	1260	2120	3480	3060
Average total solids	1313			1207			2887		
Volatile solids [mg/L]	220	180	340	300	580	140	1340	2360	1980
Average volatile solids	247			340			1893		
Fixed solids [mg/L]	980	1140	1080	1020	460	1120	780	1120	1080
Average fixed solids	1067			867			993		

Thickened sludge	Biological 1		Biological 2		Chem 1		Chem 2	
Total Solids [mg/L]	28060	27640	27620	27300	22540	22840	26960	27240
Volatile solids [mg/L]	15820	15860	15560	15620	6100	6100	6940	7140
Fixed solids [mg/L]	12240	11780	12060	11680	16440	16740	20020	20100

Thickened sludge	Mixed 1		Mixed 2		Sand filter (bio.)		
Total Solids [mg/L]	48300	49260	46820	54780	6880	7000	6120
Volatile solids [mg/L]	12180	12260	11580	18080	3780	3900	3300
Fixed solids [mg/L]	36120	37000	35240	36700	3100	3100	2820

Thickened sludge	Sand filter (chem.)	Rya 1		Rya 2	
Total Solids [mg/L]	7650	21540	22480	22460	23820
Volatile solids [mg/L]	1950	15120	16040	15900	16900
Fixed solids [mg/L]	5700	6420	6440	6560	6920

March 2015	Biological	Chemical	Mixed	Sand filter (bio.)	Sand filter (chem.)	Ryaverket
Total solids [mg/L]	1860	1940	6760	1060	1320	2500
Volatile [mg/L]	780	560	1960	260	320	1720
Fixed Solids [mg/l]	1080	1380	4800	800	1000	780

Appendix 7 – Detailed results from total suspended solids in sludge

The results from the analyses of total suspended solids in the supernatant from the zone settling tests. Some negative values were obtained for the tests performed in April 2015.

April 2015	Biological				Chemical			
Conc. of volatile solids [mg/L]	331	351	364	398	9	23	13	11
Conc. of fixed solids [mg/L]	204	198	151	131	3	-15	11	17
Percentage organic material	62	64	71	75	74	269	54	40

April 2015	Mixed				Sand filter (bio)		
Conc. of volatile solids [mg/L]	89	55	47	49	21	25	23
Conc. of fixed solids [mg/L]	-17	-57	31	29	15	7	27
Percentage organic material	123	-4150	60	63	58	78	46

April 2015	Sand filter, chem			Ryaverket 1		Ryaverket 2	
Conc. of volatile solids [mg/L]	17	15	11	31	43	45	41
Conc. of fixed solids [mg/L]	-1	5	1	21	-15	-1	11
Percentage organic material	104	74	89	59	151	101	78

March 2015	Chemical Sludge		Sand filter, bio		Sand filter, chem	
Concentration of volatile solids [mg/L]	10	13	50	30	33	24
Concentration of fixed solids [mg/L]	13	7	30	10	7	16
Percentage organic material	43	67	62	75	83	60

March 2015	Biological		Mixed Sludge		Ryaverket	
Concentration of volatile solids [mg/L]	60	70	25	35	20	5
Concentration of fixed solids [mg/L]	60	20	20	20	13	15
Percentage organic material	50	78	56	64	60	25

Appendix 8 – Detailed results of water content in sludge

The detailed results from the tests for free water are shown in the tables below. Three tests were performed for all sludge at the second sampling occasion. The filtration worked decently.

April 2015	Chemical			Sand filter, bio			Sand filter, chem		
Free Water [%]	91.6	82.3	95.9	91.8	91.7	95.9	81.3	91.7	95.8
Av. Free Water [%]	89.9			93.2			89.6		

April 2015	Biological			Mixed			Ryaverket		
Free Water [%]	48.3	52.9	49.2	80.1	79.4	74.5	96.2	85.5	91.2
Av. Free Water [%]	50.1			78.0			91.0		

At the first sampling occasion, there were performance problems with the vacuum filtration unit. A few tests had major issues with the equipment; their results are written in *italic* in the table.

March 2015	Biological		Chemical		Mixed	
Free water [%]	87.1	89.6	77.1	86.4	73.6	<i>41.6</i>
Av. Free water [%]	88.4		81.7		57.6	

March 2015	Sand filter (bio.)		Sand filter (chem.)		Ryaverket	
Free water [%]	45.7	63.0	92.7	68.4	<i>61.7</i>	41.0
Av. Free water [%]	54.4		80.6		51.4	

The tables below show the results from the test for free and interstitial water. The content of the different types of water and solids are given in percent.

April 2015	Biological	Chemical	Mixed
Free water	50.1	89.9	78.0
Interstitial water	28.9	9.3	0.0
Water Content	97.8	99.8	96.8
Surface and Bound Water	18.8	0.6	18.8
Solid Content	2.2	0.2	3.2

April 2015	Sand filter (bio)	Sand filter (chem)	Ryaverket
Free water	93.2	89.6	91.0
Interstitial water	6.3	10.0	5.7
Water Content	99.9	99.9	99.7
Surface and Bound Water	0.4	0.2	3.0
Solid Content	0.1	0.1	0.3

March 2015	Biological	Chemical	Mixed
Free water	88.4	81.7	57.6
Interstitial water	10.8	17.4	37.6
Water Content	99.8	99.8	99.3
Surface and Bound Water	0.6	0.7	4.1
Solid Content	0.2	0.2	0.7

March 2015	Sand filter (bio)	Sand filter (chem)	Ryaverket
Free water	54.4	80.6	51.4
Interstitial water	45.5	19.2	45.4
Water Content	99.9	99.9	99.8
Surface and Bound Water	0.1	0.1	3.0
Solid Content	0.1	0.1	0.2

Appendix 9 – Detailed results of capillary suction time for sludge

Capillary suction times are shown in the tables below. All samples were tested at room temperature in the lab (approximately 20 °C). The table below shows tests on the samples from April 2015.

Type of sludge	CST [s]			Average CST [s]
Biological	87.1	70.6	80.6	79.4
Chemical	5.9	5.3	5.7	5.6
Mixed	36.3	33	30	33.1
Sand filter (bio.)	5	4.6	5.1	4.9
Sand filter (chem.)	4.9	4.6	4.2	4.6
Ryaverket	9.4	8.7	9	9.0

The table below shows CST tests made on thickened samples from April 2015. The 1 and 2 written after some of the samples refer to the settling cylinder from which the sample was obtained. Note that the amount of thickened sludge in the cylinder containing sludge from the sand filter from the chemical treatment line was too small to perform CST tests.

Type of sludge	CST [s]		Average CST [s]
Biological 1	98.1	116.1	111.1
Biological 2	117	113	
Chemical 1	15.1	14.2	16.0
Chemical 2	17.1	17.5	
Mixed 1	46	46.9	50.1
Mixed 2	53.2	54.1	
Sand filter (bio.)	19.5	11.8	15.7
Ryaverket 1	144.4	120.3	136.6
Ryaverket 2	141.9	139.9	

The table below shows results from CST tests on samples from March 2015.

Type of sludge	CST [s]			Average CST [s]
Biological	7.7	7.1	7.1	7.3
Chemical	7.5	6.4	8.3	7.4
Mixed	13.5	13.1	12.6	13.1
Sand filter (bio.)	5.9	5	6	5.6
Sand filter (chem.)	6.3	5.9	5.2	5.8
Ryaverket	9.5	9.1	8.8	9.1

Appendix 10 – Raw data for zone settling velocity

Mixed	Time [min]	10	20	30	40	50	62	77	92	107	122	152	205	235	272	317	417	1382
1	Height [cm]	36.2	35.5	35.0	34.4	33.9	33.3	32.8	32.2	31.9	31.3	30.6	29.7	29.2	28.6	28.2	27.2	23.3
	Velocity [cm/min]	0.07	0.05	0.05	0.05	0.05	0.04	0.04	0.02	0.04	0.02	0.02	0.02	0.02	0.01	0.01	0.01	0.00
2	Height [cm]	35.9	34.4	33.3	32.2	31.7	31.0	30.2	29.7	29.3	28.8	27.9	27.0	26.6	26.1	25.7	24.8	21.3
	Velocity [cm/min]	0.15	0.11	0.11	0.05	0.06	0.05	0.04	0.02	0.04	0.03	0.02	0.01	0.01	0.01	0.01	0.01	0.00

Biological	Time [min]	10	20	30	40	50	55	70	85	100	115	145	198	228	265	310	410	1375
1	Height [cm]	36.2	35.9	35.7	35.5	35.1	35.1	34.8	34.4	33.9	33.5	32.6	31.2	30.1	29.3	28.1	26.4	26.8
	Velocity [cm/min]	0.04	0.02	0.02	0.04	0.00	0.02	0.02	0.04	0.02	0.03	0.03	0.04	0.02	0.03	0.02	0.00	0.00
2	Height [cm]	36.2	36.1	35.9	35.5	35.1	35.1	34.8	34.4	33.9	33.5	32.6	31.0	30.1	29.2	27.9	26.1	27.2
	Velocity [cm/min]	0.02	0.02	0.04	0.04	0.00	0.02	0.02	0.04	0.02	0.03	0.03	0.03	0.02	0.03	0.02	0.00	0.00

Ryaverket	Time [min]	4	6	8	10	13	18	23	28	33	38	48	63	78	93	108	138	191	303	403	1368
1	Height [cm]	20.6	15.2	13.3	12.3	11.2	9.7	9.0	8.3	7.9	7.5	6.8	6.4	5.9	5.7	5.4	5.0	4.8	4.6	4.3	3.9
	Velocity [cm/min]	2.73	0.91	0.55	0.36	0.29	0.15	0.15	0.07	0.07	0.07	0.02	0.04	0.01	0.02	0.01	0.00	0.01	0.00	0.00	0.00
2	Height [cm]	17.7	14.1	13.0	11.9	10.8	9.7	9.0	8.3	7.9	7.5	7.0	6.4	6.1	5.7	5.4	5.2	5.0	4.6	4.3	3.9
	Velocity [cm/min]	1.82	0.55	0.55	0.36	0.22	0.15	0.15	0.07	0.07	0.05	0.04	0.02	0.02	0.02	0.01	0.00	0.01	0.00	0.00	0.00

Chemical	Time [min]	0	5	15	25	30	43	58	73	88	103	133	186	216	253	298	398	1363
1	Height [cm]	-	-	1.45	2.05	2.15	2.15	2.15	2.15	2.15	2.15	2.15	2.15	2.15	2.15	2.25	2.25	2.25
2	Height [cm]	-	-	1.15	1.25	1.25	1.25	1.25	1.25	1.25	1.25	1.25	1.25	1.25	1.25	1.25	1.25	1.15

Appendix 11 – Detailed chemical composition of sludge

This table shows the result from the chemical measurement of the sludge.

Parameter	Unit	Biological sludge	Chemical sludge
Aliphatic >C8-C10	mg/kg TS	<10	<10,0
Aliphatic >C10-C12	mg/kg TS	<20	<20
Aliphatic >C12-C16	mg/kg TS	20	<20
Aliphatic >C16-C35	mg/kg TS	381	43
Aromatic >C8-C10	mg/kg TS	<3	<3,00
Aromatic >C10-C16	mg/kg TS	<7,75	<7,75
Aromatic >C16-C35	mg/kg TS	<1	<1,0
PAH:		Not detected	Not detected
PCB:		Not detected	Not detected
Chlorinated benzenes		Not detected	Not detected
4-tert-octylphenol	mg/kg TS	0.419	0.056
4-nonylphenols	mg/kg TS	2.4	0.59
DEHP	mg/kg TS	6.3	5.5
PFOS	µg/kg TS	<10	<10
PFOA	µg/kg TS	<10	<10
BDE 47	µg/kg TS	0.17	<0,10
BDE 99	µg/kg TS	0.17	<0,10
As	mg/kg TS	3.07	6.89
Ba	mg/kg TS	2050	3330
Cd	mg/kg TS	0.239	0.705
Co	mg/kg TS	3.23	4.35
Cr	mg/kg TS	30.8	35.4
Cu	mg/kg TS	35.3	31.6
Hg	mg/kg TS	<0,2	<0,2
Ni	mg/kg TS	12.6	7.17
Pb	mg/kg TS	6.81	18.3
V	mg/kg TS	7.12	30.7
Zn	mg/kg TS	112	176
Mo	mg/kg TS	6.57	2.3
Sb	mg/kg TS	1.12	2.2
Sn	mg/kg TS	18.7	61.8
Ag	mg/kg TS	0.15	0.259
Fe	mg/kg TS	102 000	281 000
Mn	mg/kg TS	1820	398
TOC	% of TS	22.9	7.48
Fixed solids	% of TS	46.6	74.2

Appendix 12 – Sludge Volume Index

The sludge volume index is presented in the table below. It is calculated from the results from the zone settling velocity tests and the tests for total solids.

	Biological	Chemical	Mixed	Sand, bio	Sand, chem	Ryaverket
March	16.6	46.3	102.9	-	14.0	81.4
April	39.3	20.5	26.2	-	-	70.5