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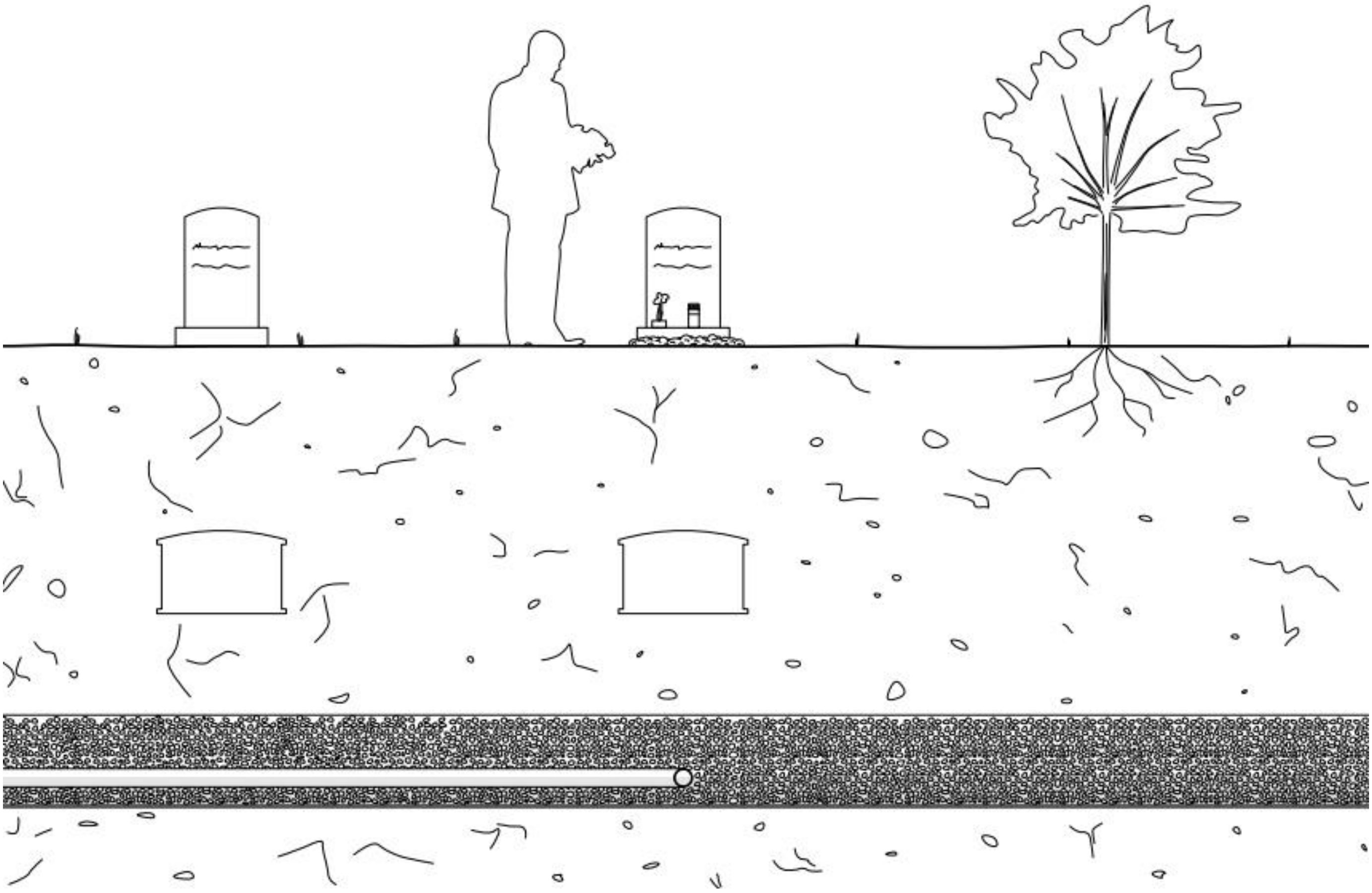

Mechanisms behind grave soil leachate and remediation options for drainage water

Master's thesis in Infrastructure and Environmental Engineering

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samhällsbyggnadsteknik,
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Schematic of drained cemetery soil. © ADAM DAHLIN, 2018
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ABSTRACT

Cemeteries have recently received more attention regarding drainage water in burial areas and environmental and human health risks related to potential contamination of receiving waters. The aim of this thesis is to supply the reader with an understanding of the processes behind grave soil leachate and what contaminants may exceed environmental quality guidelines. A suitable approach on how burial fields and stormwater systems can be governed in accordance with environmental law has been developed by demonstrating a case study of Fridhem cemetery in Gothenburg.

The most commonly found pollutants in grave soils are nitrogen, phosphorus, metals and microbes. All of them are included in a complex and dynamic web of reactions which is governed by for example contaminant loading, humidity, oxygen and pH. Several laboratory experiments and field investigations have proven that clay minerals, organic material, Ca, Al and Fe strongly influence retention of most contaminants of interest in grave soils. Beneficial environmental condition may therefore result in a low pollution risk.

Sand filters have, since a few pilot constructions were implemented in Sweden, become a somewhat generalised solution for grave soil leachate in many cemeteries and different contexts. An evaluation of the drainage water and performance of a constructed sand filter at Fridhem cemetery in Gothenburg, showed that the water was exposed to relatively low pollution, with only nitrogen slightly exceeding the regional guideline value. Although the burial field was not at full capacity at the time of sampling, it seems unlikely that high concentrations, posing an environmental risk, will prevail eventually. The grave soil is dominated by clayey textures with moderate organic material, providing a high surface area and good retention potential from organic material and clay minerals. The sand filter showed satisfying performance against both nutrients (N and P). However, metal concentrations increased at the outlet, which mainly was due to the distinctively low concentration and may have been natural variations or leaching from the filter media.

The case study of Fridhem cemetery clearly demonstrates the importance of conducting a comprehensive investigation of both geology and hydrogeology, which will provide an understanding of the risk of pollutant leaching from graves.

Key words: Cemetery, Decomposition, Grave soil, Drainage water, Constituent dynamic, Remediation

Mekanismer bakom lakvatten i gravjordar och reningsmetoder för dräneringsvatten

Examensarbete inom masterprogrammet Infrastruktur och miljöteknik

Institutionen för Arkitektur och samhällsbyggnadsteknik
Avdelningen för Vatten Miljö Teknik
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SAMMANFATTNING

Kyrkogårdar och dräneringsvattnet från begravningsområden har nyligen fått mer uppmärksamhet på grund av föroreningsrisker kopplade till recipienter och människors välmående. Syftet med detta arbete är att få en fördjupad förståelse bakom processerna till föroreningsspridning i jord och lakvatten, samt vilka föroreningar som utgör en risk. Arbetet har utformat ett tillvägagångssätt för att utvärdera hur stor risk olika begravningsplatser utgör genom en att använda Fridhem kyrkogård i Göteborg som studieobjekt.

De vanligaste föroreningarna i gravjordar är kväve, fosfor, metaller och mikroorganismer. Samtliga föroreningar påverkar varandra genom komplexa kemiska reaktioner och biologiska processer, och bidrar till olika grader av bindnings- och transportpotential i jord. Andra faktorer som föroreningsmassa, fuktighet, syrehalt och pH påverkar dessa processer. Flera laboratoriestudier och fältmätningar har identifierat att lermineraler, organiskt material, Ca, Al och Fe starkt påverkar bindningen i gravjordar. Om fördelaktiga miljöer är berörda kan gravområden utgöra väldigt små risker.

Markbäddar har efter test i ett fåtal pilot-konstruktioner blivit en generaliserad lösning för hantering av dräneringsvatten på kyrkogårdar i Sverige. Efter flertalet mätningar av dräneringsvattnet på Fridhem kyrkogård visade det sig att två gravområden utgjorde en väldigt liten risk för miljön. Kväve var den enda parametern som översteg miljöförvaltningens riktlinjer för utsläpp till recipienter, dock med liten marginal. Trots att kyrkogården inte utnyttjade sin fulla gravkapacitet vid provtagningstillfället är det osannolikt att höga föroreningshalter kommer att uppstå i framtiden och utgöra en stor risk. Jorden består huvudsakligen av leriga jordmassor med måttliga halter av organiskt material, vilket bidrar till en stor specifik yta för adsorption. Sandfiltret visade bra resultat och vid utloppet överstegs aldrig miljöförvaltningens krav under de tre provtagningstillfällena. Bättre reningskapacitet uppstod vid högre föroreningshalter. Metallhalterna var genomgående väldigt låga under undersökningsperioden, och visade ofta högre värden vid utloppet än vid inloppet. Detta kan bero på normala koncentrationsvariationer eller att filtermaterialet urlakas.

Arbetet vid Fridhem tydliggör betydelsen för en omfattande förundersökning av både geologi och hydrogeologi, vilket lägger grunden till hur allvarliga risker för recipienter och människors hälsa som kan uppstå.

Nyckelord: Kyrkogård, Nedbrytning, Gravjord, Dräneringsvatten,
Konstituentdynamik, Reningsteknik

Contents

Preface

Mechanisms behind grave soil leachate and remediation options for drainage water is the final project and concludes my studies at Chalmers' Infrastructure and Environmental engineering program. I would like to thank Therese Örnmark for the thesis suggestion. It has been a very interesting and educational project that I will have use for in the future. Therese, Jessica Karlsson and Henning Sandström from the regional cemetery administration helped me throughout my project, provided me with information and documents and answered all my questions, for which I am grateful.

I would also like to thank several people from Chalmers for their help with my project. My supervisor Karin Björklund and examiner Ann-Margret Hvitt Strömvall for their support with my sampling, laboratory analysis and report; Måna Pålsson and PhD student Masoumeh Heibati for their help in the laboratory; and my opponent Desirée Persson for providing some general thoughts and insight of how I could improve my project.

Finally, I want to thank everyone involved with the project at Björlanda cemetery, Norconsult and Gothenburg cemetery administration, for allowing me to participate. It has been interesting and challenging to use everything I have learned in an ongoing project.

Gothenburg May 2018

Adam Dahlin

1 Introduction

1.1 Background

Drainage water from cemeteries is becoming a more pressing issue that needs management. In 2015, the city of Gothenburg and the environmental agency demanded investigations regarding drainage water for Fridhem, Östra and Billdal cemeteries. These areas were mainly chosen due to their connection to the existing wastewater network, or the effluent being discharged to a local creek. Drainage water from cemeteries is classified as wastewater and must therefore fulfil pollution requirements before reaching the recipient, in accordance with environmental law (1998:808) and public safety. The ambition of the city of Gothenburg and Kretslopp och Vatten (KoV) is to separate the combined sewer system, and instead, manage stormwater and wastewater in different networks. This will mitigate incoming flows to Ryaverket wastewater treatment plant and provide more stable conditions for operation, while less polluted stormwater can be discharged to receiving waters before or after small scale treatment depending on the water quality. Privately owned operations, such as cemeteries, will therefore acquire sole responsibility for the wastewater they produce, and implementation of remediation if necessary and financially reasonable.

After implementation of cemetery regulation changes and inquiries towards the church management in Gothenburg, Fridhem cemetery has constructed a remediation station for its drainage water. The water is collected in drainage conduits under the coffins and led to an aerated sand filter. However, the sand filter also manages the remediation of leachate water from a nearby compost. The influent conduit is therefore a mixture of both leachate and drainage water. Prior to implementation of the drainage system, the drainage water quality was unknown, and only one set of influent and effluent samples has been analysed since construction. Therefore, the performance efficiency of the sand filter is unknown. The concentrations of contaminants in the influent are assumed to have increased due to an increase of burials in the graveyard. It is therefore of interest to investigate how the sand filter is performing today and evaluate how a future scenario might look like, with a denser cemetery.

Information and past experiences regarding drainage water in cemeteries and related risks are limited and few documents lay the foundation from which planning and implementation of mitigating measures are based upon in Sweden. Without further understanding of the processes behind grave soil leachate and which contaminants may pose a threat, cemeteries are likely to generalise solutions that has worked in the past, but which might not be necessary.

1.2 Aim and objectives

The aim of this master thesis is to provide a comprehensive literary study on mechanisms behind grave soil leachate and how different remediation methods may perform and reduce pollution levels to fulfil set water quality requirements. The investigation includes crucial factors of how the relationship between soil and water quality influence release of contaminants from grave soil. The purpose of the study is to supply the reader with sufficient information so that decisions can be made regarding mitigating measures for cemeteries with different environmental conditions.

The objectives are to

- provide a comprehensive and scientific literary study on drainage water in cemeteries, its content and past experiences
- evaluate the treatment efficiency of the sand filter through field sampling at Fridhem cemetery and laboratory analyses
- through support from the literary study and analysis of the results from Fridhem cemetery, find suitable approaches for mitigating measures

1.3 Limitations

The literary study includes the most important aspects related to drainage water in cemeteries; laws and regulations, the decomposition process, constituent behaviour in soils, examples of past cemetery investigations and remediation methods. However, not all parts will be investigated to the same extent, for example, all potentially applicable remediation methods will not be researched. The literature will mainly be based upon scientific articles and is therefore assumed to be reliable.

The project initially set out to also investigate if the cemetery soils are exposed to “first flush”, meaning higher concentrations of pollutants are leached in the early stages of precipitation events. This has been excluded from the study due to difficulties in collecting samples at the right time, and the sampling and analyses performed by an external laboratory had to be ordered in advance.

The first sampling occurred in 2017, and the parameters of interest were therefore decided upon prior to this project. The analysis is based and restricted to measurements of Zn, Cu, Pb Hg, total nitrogen, total phosphorus, total coliform and E. *coli*. Furthermore, general parameters which could be analysed at Chalmers laboratory were limited to pH, conductivity, dissolved oxygen, biological oxygen demand (BOD7), total suspended solids, total nitrogen, total organic carbon and dissolved organic carbon. However, due to a malfunction of the analyser, the concentrations of total organic carbon, dissolved organic carbon and nitrogen in the water samples showed unreliable results in two analysing attempts at Chalmers. These parameters were therefore excluded from the results. To strengthen the conclusions made from identified trends of the results, more than three samples for each sampling location would be preferred.

2 Literary study

2.1 Laws, regulations and guidelines

There are several laws, regulations and guidelines that provide a foundation for preventing and mitigating environmental impact and securing a sustainable future. Many can be applied directly to managing water resources in and near cemeteries. Different regions generally set their own goals and regulations, and the responsibility lies therefore on different governing levels. The regional guidelines for effluent water to receiving waters and stormwater in Gothenburg have been developed with regard to the Swedish environmental code (1998:808) and the European Water framework directive (WFD). Further support in the development of the regional guidelines was provided by the Swedish agency for marine and water management and the Swedish environmental protection agency (Mossdal, 2013)

2.1.1 The EU water framework directive

Sweden has been a member state of the EU since 1995 (European Union, 2017) and should therefore keep up with established rules and treaties (European Commission, 2016). The European commission has developed a framework – the Water Framework Directive (WFD) – for management and protection of ecosystems and water resources and was put into action the year 2000, after which all member states had a timeframe in which they could set their own guidelines and goals. However, on a national level, the guidelines differ depending on the countries' financial resources and environmental status (European Community, 2000). Drainage water from cemeteries are not mentioned specifically in WFD. However, several articles should be applied in the management of cemetery operations.

Article 4 in the directive state that, (i) member states shall implement measures necessary to prevent or limit the input of pollutants into groundwater and (iii) implement the measures necessary to reverse any significant and sustained upward trend in the concentration of any pollutant resulting from the impact of human activity in order to progressively reduce pollution of groundwater.

Article 9 in concurrence with *Annex III* state that member states shall apply the polluter pays principle, including all costs regarding recovery of water services and environmental and resource costs. An economic analysis is to be conducted and cover necessary calculations both short and long term, including forecasts of supply and demand for water with correlating costs.

Article 17 state that preventative measures in accordance with *Article 4* shall be carried out with criteria for assessing good groundwater chemical status, which should be defined by environmental characterisation guidelines in *Annex II.2.2* and *Annex V 2.3.2* and *2.4.5*.

Article 24 state that member states shall bring into force the laws, regulations and administrative provisions necessary to comply with this Directive.

2.1.2 The Swedish environmental code

Alongside the WFD is the Swedish environmental code which provides fundamental and comprehensive laws and guidelines regarding environmental responsibility and sustainable development. Drainage water in cemeteries is governed by several of these regulations.

Chapter 2 of the Swedish environmental code states that *2§* the accountable for a specific activity shall possess sufficient knowledge regarding risks related to that activity, and *3§* all operations which may cause inconvenience or harm must provide sufficient precautions and use the best available technology and equipment to preserve environmental quality and human health. *7§* Previous paragraphs in chapter 2 shall be applied if not unreasonable to fulfil such demands. An assessment, weighing environmental impact, costs and improvement of mitigating measures, will be considered.

Chapter 9 states that *2§* drainage water from cemetery soils is classified as wastewater, and *1§* wastewater effluents to recipients is considered an environmental hazard. *4§* The government may provide regulations and prohibit effluent of such wastewater in certain regions if necessary to preserve environmental quality and protect human health. *6§* The government may similarly prohibit effluent of wastewater to the environment if the accountable does not have a permit or before a notice has been processed. *7§* Wastewater shall be remediated before reaching the recipient to prevent any consequences that could jeopardize human health or environmental quality. Treatment facilities or an equivalent solution shall be implemented as a countermeasure. The government may prohibit establishment of such construction without a permit or notice.

Chapter 28 states that *10§* the responsible party for countermeasures against wastewater pollution may be given permission by the environmental court to claim owned land or property to implement such plans.

Chapter 29 states that *8§* the accountable for operation of wastewater management may be assigned a fine or sentenced to a maximum of two years in prison if negligently disregards prohibitions set out by the government in chapter 9 4§.

2.1.3 Regional guidelines for Gothenburg

The target values presented in Table 1, established by the city of Gothenburg, are generalised values for effluent water quality to all environmental and municipal stormwater systems (Mossdal, 2013). The values were updated in 2013 and decided upon with help from various environmental-governing agencies, reports and regulations, and wastewater managing enterprises and associations. The parameters include the most common contaminants found in wastewater and were therefore prioritised in the guideline. If other harmful substances, absent from the list, are found in effluent waters a case-specific evaluation should be made. Due to the difficulty and financial investment of full investigation, it is recommended that small-scale operations conduct an evaluation of expected contaminants, concentrations and sensitivity of the recipient. These guidelines are not legally valid without a notice from the environmental administration, due to the vast sensitivity range of different recipients. The administration also decides how intensively monitoring of effluent water quality should be conducted. If the target values cannot be fulfilled, a site-specific investigation should be made, and if environmental conditions allow it, higher concentrations may be allowed. However, mitigating measures should always be applied within reasonable technical and financial limits. If certain remediation methods have proved successful, sampling and control of effluent water for similar operations can in some instances be adjusted and generalised for the same practitioner.

Table 1. *Gothenburg target values of effluent water to recipients (Mossdal, 2013).*

Parameter	Target value	
Arsenic	15	µg/l
Chromium	15	µg/l
Cadmium	0.4	µg/l
Lead	14	µg/l
Copper	10	µg/l
Zinc	30	µg/l
Nickel	40	µg/l
Mercury	0.05	µg/l
PCB[1]	0.014	µg/l
TBT[2]	0.001	µg/l
Oil index	1000	µg/l
Benzo(a)pyrene	0.05	µg/l
MTBE[3]	500	µg/l
Benzene	10	µg/l
pH	6–9	
Total Phosphorus	50	µg/l
Total Nitrogen	1250	µg/l
TOC	12	mg/l
Suspended solids	25	mg/l

[1] Polychlorinated biphenyls, [2] Tributyltin, [3] Methyl tert-butyl ether

Gothenburg guideline values cover many important pollutants; however, no information is included regarding bacteria or other potentially harmful organisms. To investigate if there is a risk towards the environment or human health, indicator species are often measured. The most common indicators are Escherichia *coli* (E. *coli*), which is the most prominent specie of the bacteria family Enterobacteriacene, and Enterococci (Ohlsson, Karlsson and Gustafsson, 2011). Both are found in human intestines and warm-blooded animals. These bacteria may be used to evaluate the risk of existing pathogenic species. A list of other bacteria in human intestines which most likely is found in grave soils can be seen in Appendix A.

According to the Swedish water and wastewater association and the Swedish agency for marine and water management, water quality is likewise deemed as excellent or good with concentrations of E. *coli* and Enterococci in the ranges of 0-1000 and 0-400 cfu/100 ml, respectively (Table 2 and 3). However, transition and coastal waters have higher requirements due to the salinity reducing the half life time of most indicator bacteria, while other pathogenic microorganisms may prove to be more persistent, such as viruses (Who, 2003). Even though Gothenburg lacks guidelines for microorganisms for effluents to receiving waters, water quality assessments values according to Table 2 and 3 should still be applied. An investigation of five coastal/transition waters and one inland water source in Halmstad municipality showed that the inland water source had lowest E. *coli* and Enterococci concentrations of 150 and 40 cfu/100 ml, respectively, which were average values over five samples and analyses (Ohlsson, Karlsson and Gustafsson, 2011). The coastal/transition waters generally had average concentrations varying between 570-1060 for E. *coli* and 120-270 cfu/100 ml for Enterococci; However, one location showed elevated concentrations of 4780 and 2750 cfu/100 ml for E. *coli* and Enterococci, respectively.

Table 2. E. *coli water quality assessment in cfu/100 ml by Swedish organisations.*

E. *coli*	Excellent	Good	Satisfying YES/NO
Swedish assessment for recreational waters and swimming in a single sample [1]	<100	100-1000	>1000 NO
EU assessment for recreational waters and swimming in 95 percentile samples [1]	<500	500-1000	>1000 YES
Quality for inland waters [2]	<500	500-1000	
Quality for transition and coastal waters [2]	<250	250-500	

[1] Ohlsson, Karlsson and Gustafsson (2011)
[2] Havs- och Vatten myndigheten (2016)

Table 3. *Enterococci water quality assessment in cfu/100 ml by Swedish organisations.*

Enterococci	Excellent	Good	Satisfying YES/NO
Swedish assessment for recreational waters and swimming in a single sample [1]	<100	100-300	>300 NO
EU assessment for recreational waters and swimming in 95 percentile samples [1]	<200	200-400	>400 YES
Quality for inland waters [2]	<200	200-400	
Quality for transition and coastal waters [2]	<100	100-200	

Furthermore, according to Grebel et al. (2013), effluents to surface waters should not exceed 10000, 400, 400 and 104 cfu/100 ml for total coliform, faecal coliform, E. *coli* and Enterococci, respectively.

2.1.4 Burial regulations

The Swedish burial regulation (1990:1147) states that 10§ a grave must be dug deep enough to cover the coffin with a soil layer of minimum 1 m thickness. Graves are therefore commonly dug to a depth ranging from 1.6 – 2.8 m: a coffin is 60 cm high and it is preferred to fit more coffins in the same grave for ethical and density purposes (Arvidsson, 2008). The depth is also decided depending on groundwater conditions, which can force more shallow graves to be dug if it is not possible to drain and lower the groundwater table. There is no mention in the burial regulation in what clothing the deceased can or cannot be buried, and instances allow for burial with meaningful personal belongings. Therefore, it may be possible to find decomposed products that potentially could risk environmental damage or human health. The regulation is however much stricter if the deceased wishes to be cremated.

2.2 Decomposition processes and products

Burial areas in cemeteries can in some respects be regarded as landfills, due to the drainage water's tendency of transporting different contaminants and by-products from coffins, human remains and their decomposition (Dent, Forbes and Stuart, 2004). There are however differences in the decomposition timeframe between coffin burials and cases where the remains are in direct contact with the surrounding soil, mainly as a result from different oxygen supply. The location of the cemetery will also influence the degradation, since the soil composition of different compounds will vary, and microorganisms will in some instances be more active. The decay process of a human body begins only moments after death. However, it is not uncommon for bodies to be covered in protective substances, such as formaldehyde, to delay any outward initial decomposition, for the purpose of burial ceremonies or rituals (Kim *et al.*, 2008). Cemeteries can also pose risks from past practices, when embalming fluids contained arsenic and mercury, and hazardous chemicals from preservative agents on coffins were used (Spongberg and Becks, 1999). If the right conditions arise insufficient degradation of fatty acids can preserve bodies for decades or even centuries, and hinder the reuse of graves after a certain burial time (Fiedler *et al.*, 2015)

2.2.1 Composition of a human body

Most of the human body consists of water, 50–64 % according to studies done by Dent (2002) and Schoenen & Schoenen (2013). The remaining constituents consists of approximately 5–15 % bones and 30–35 % organic substances. The bones consist of minerals and the organic substances are mainly proteins, fats and a small amount of carbohydrates. There are also inorganic constituents present, such as nitrogen, phosphorus, calcium, natrium, iron and copper. In general, there are significant variations of the compostion depending on lifestyle and size. Obesity can for example lead to much higher fat percentages, in contrast to an active lifestyle and exercise.

2.2.2 Decomposition process

The human body goes through immediate change after death and is continuously degraded for years to come (Zhou and Byard, 2011). The interaction of autolysis and putrefaction are the initial decomposition processes and are defined by self-digestion, breakdown of tissue and modification of constituents. The autolytic process arises when blood flow stops, causing waste products to build up and damage the membrane of cells. When the membranes are ruptured, hydrolytic enzymes are released and utilised as a catalyst for splitting fats, carbohydrates and proteins. The autolysis can therefore progress at a faster rate in organs that have high water content and are dense with lysosomes, vast enzyme containing membranes, for example, the lungs and brain. Enzymes can also be derived from microbes and fungi in the surrounding soil. Simultaneously with the destruction of membranes, liquids, rich with amino acids, sugars, glycerol and fatty acids, are released and further utilised by microbes in the putrefaction process.

Both aerobic and anaerobic microorganisms contribute to the decomposition of a human corpse. The aerobic bacteria resorts to respiration and oxidises glucose for extraction of energy. Since the body contains a limited amount of oxygen this provides exemplary conditions for anaerobic decomposition (Dent, 2002; Schoenen and Schoenen, 2013) and fermentation, and degradation of organic nutrients. Decomposition by respiration produces more energy compared to fermentation and can degrade organic components fully to produce carbon dioxide and water. Fermentation cannot degrade all by-products that are produced, and soluble organic substances are commonly left in the soil (Schoenen and Schoenen, 2013). There are several different species of microorganisms responsible for the decomposition of human tissue. The most common organisms in the putrefactive decomposition stage are anaerobic spore bearing bacilli, coliforms, diphtheroids, micrococci, proteus species and *clostridium perfringens*. They mainly originate from intestines and the respiratory tract (Zhou and Byard, 2011).

The availability of oxygen will greatly influence the rate of the decomposition process. When all oxygen in the body is consumed, full decomposition through respiration, by aerobic microorganisms and aerobic fungi, will be restricted to the surface of the corpse. However, the available oxygen in the body and in the coffin is not enough to support full decomposition. Some additional oxygen is provided when the coffin begins to disintegrate and decompose. Coffins made from more easily degradable wood are therefore preferred for faster decomposition. The contribution of oxygen from external soil is however strictly dependent on soil properties, mainly degree of compaction and gas diffusion potential (Dent, Forbes and Stuart, 2004; Schoenen and Schoenen, 2013). The soil around the coffin generally contains more air than surrounding soil due to backfilling and has higher potential for diffusion. The soil above the grave will attain various degrees of compaction depending on the backfilling procedure. Commonly, excavators are used to dig, fill and manage earth retaining structures if the soil is unstable (Arvidsson, 2008), which causes a continuous increase in compaction towards the surface and air diffusion from above ground becomes restricted. However, gas diffusion is generally significantly decreased beyond 1 m depth even if the soil is porous. Also, decomposed organic products such as carbon dioxide and methane might prohibit air to reach the corpse as they diffuse towards the ground surface. Percolating groundwater and rainfall can provide oxygen for respiration, although, at greater depths, dissolved oxygen is more likely to stay in a hydrated state rather than as free oxygen (Dent, Forbes and Stuart, 2004). The large reduction of oxygen diffusion in water compared to air can act as a barrier and neglect the flow of oxygen to the surface of a corpse if water is present. Due to the lack of oxygen from the environment, when the available oxygen has been utilised, decomposition by respiration will proceed very slowly and most of the human remains will be degraded by putrefaction and anaerobic fermentation (Schoenen and Schoenen, 2013).

2.2.3 Decomposition products

The constituents included in human tissue undergoes different processes and produces by-products, mainly fat, protein and carbohydrates, that have various significance and impact on the environment.

2.2.3.1 Fat

The most dominant fat in human adipose (fat) tissue is triglyceride (neutral fat), which consists mainly of saturated and unsaturated fatty acid molecules (Dent, Forbes and Stuart, 2004; Forbes, Dent and Stuart, 2005). The bound fatty acids in triglyceride will together with water and enzymes, released during the decomposition process, transform during hydrolysis[1], hydrogenation[2] or oxidation reactions and form fatty acids through lipolysis[3]. Since the body contains enough water to satisfy for example hydrolysis, this process can occur even in dry soils. Microbial activity is also a key element in production of fatty acids, polyhydroxy fatty acids specifically, and fatty acids can contribute significantly to the cell mass of microbes (Schoenen and Schoenen, 2013). However, the microbial fermentation process cannot break down fatty acids further. In anaerobic environments the fatty acids can therefore be preserved due to their macromolecular[4] insolubility in water and absent decomposition through respiration. Contradictory to the soluble by-products from the putrefaction process, the produced fatty acids cannot be transported to the environment through percolating water or groundwater flow. Instead, they tend to stay put in the grave soil until full decomposition.

The mixture of different fatty acids can provide a protective coating on the body known as adipocere, which consists of mainly palmitic acid, stearic acid and myristic acid, and to a lesser extent hydroxy and oxy fatty acids (Forbes, Dent and Stuart, 2005; Fiedler *et al.*, 2015). Though the production of adipocere is inevitable, regardless of soil type, humidity or aeration, it can be microbially degraded if the environmental conditions allow it (Fiedler *et al.*, 2015). It can be degraded through respiration to carbon dioxide and water if oxygen is available due to fat substances being an energy source for bacteria. Although, roughly three times as much oxygen is necessary to utilise fatty acids as an energy source compared to a similarly sized carbohydrate molecule. Degradation of adipocere is therefore commonly a slow process and can only progress on the surface since there is no oxygen inside of the adipocere mass. The microorganisms grow in a biofilm on the adipocere, preferably in the most oxygenated environment (Schoenen and Schoenen, 2013).

Schoenen & Schoenen (2013) describes how a cemetery relocation in Zürich excavated over 600 human remains in 1913. Müller, who was involved in the project, investigated the vast coverage of adipocere that kept one third of the corpses intact.

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[1] Degradation through reaction with water

[2] Reaction of a compound with hydrogen

[3] Breakdown of fat tissue

[4] Component of a larger molecule

Amongst the findings were that no adipocere was found in graves that had available oxygen and diffusion between the corpse and the atmosphere. Some graves proved to have exemplary conditions for preservation of adipocere, however, the degradation of wooden structures in the soil gave way for air channels which provided enough oxygen to bacteria and respiration of the adipocere. Furthermore, a project in the Netherlands during the 1950's reburied 110 adipocere covered remains close to the surface, for the purpose of studying the rate of decomposition in well aerated environments. After four and a half years 75 % of the corpses had been fully decomposed (Schoenen and Schoenen, 2013). Dent, Forbes and Stuart (2005) set up a laboratory experiment and investigated the significance of different soil types in the formation of adipocere. They included sand, silty sand, clay and sterilised loamy sand soil. The results showed that adipocere formed in all soils and that the soil itself has little impact on adipocere production when a humid and anaerobic environment is considered. Fiedler et al. (2015) suggests that fatty acids can be decomposed by alternate pathways and not only by aerobic microorganisms. The body contain compounds that are perfectly suitable as a catalyst in autoxidation[5] of fatty acids, for example haemoglobin, methaemoglobin and haemin. This process can produce by-products such as dimers, polymers, keto, hydroxy and epoxy compounds, furans, aldehydes and alcohols. If strict anaerobic conditions apply, for example *Enterobacter* can utilise nitrate NO_3^-, and at lower redox potential, *Desulfobacter* sp. can utilise sulphate SO_4^{2-} for oxidation of fatty acids. Environmental conditions such as water content and available oxygen may change over time and can make it difficult to estimate the time of death for adipocere covered corpses. A cadaver was found in Switzerland in 1996 and a meticulous investigation showed that the body had been preserved, buried in sediment under water, since the 17th century (Thali *et al.*, 2011).

2.2.3.2 Proteins

Post mortem, proteins are broken down through proteolysis, a process that is driven by enzymes at varying rates (Dent, Forbes and Stuart, 2004). Proteins have different resistance to these enzymes and are therefore broken down both in early and late stages of the decomposition process. Degradation of the stronger proteins, for example keratin (fibre-rich protein in skin and hair), is commonly supported by physical wearing or microbes such as Streptomyces spp., which facilitates further decomposition by proteolytic enzymes. The rate of decomposition is also strongly affected by temperature, moisture and, as previously stated, microbial activity. Bacteria that are known to assist proteolysis are *Pseudomonas, Bacillus* and *Micrococcus* species.

Several studies have found nitrogen and phosphorus to be two of the common elements present in higher concentrations near cemeteries. Nitrogen is a constituent of amino acids in proteins and are utilised by microorganisms during deamination[6] of

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[5] Oxidation when reacting with air

[6] Removal of an amino group

proteins. Through chemical reactions post deamination, nitrogen can be released from the amino acids in the form of ammonia (Dent, Forbes and Stuart, 2004). Phosphorus can be found in several components of the human body, for example, nucleic acid and phosphorylated co-enzyme containing proteins, sugar and nucleic esters and fat tissue (phospholipids) in the brain and spinal cord (Oehl *et al.*, 2001; Dent, Forbes and Stuart, 2004)

2.2.3.3 Carbohydrates

Microorganisms aid the decomposition of polymer carbohydrates (polysaccharides) and produce simple sugars such as glucose, which eventually are broken down further (Dent, Forbes and Stuart, 2004). The decomposition can continue by fungi, aerobes or anaerobes. Fungi tend to oxidise sugar monomers to organic acids while anaerobes such as Clostridia spp. may also produce alcohols. Aerobes may fully degrade sugars through fermentation and produce carbon dioxide and water, simultaneously as methane, hydrogen and hydrogen sulphide.

2.3 Soil dynamics and transport of constituents

Various constituents are decomposed and released to the soil during and after the degradation of human remains. Some may benefit the environment while others pose a risk. Chemical and microbial reactions that take place in the soil do not only produce compounds that may threaten water sources, but also the atmosphere as greenhouse gases are emitted. Some of the significant elements that are included in these reactions and derive from cemetery operations are nitrogen, phosphorus and heavy metals.

2.3.1 Nitrogen

Nitrogen is already present in grave soils prior to human decomposition in a variety of states and compositions. Cadavers contributes with organic nitrogen, which may disassemble in the presence of microorganisms and join the transformation cycle. These nitrogen species may accumulate in the soil, be transported in the environment with groundwater flow or be released to the atmosphere in gaseous form (Dent, Forbes and Stuart, 2004). The pathways that govern nitrogen are nitrification, denitrification, assimilatory reduction to biomass (will not be investigated further) and dissimilatory reduction to ammonium (DNRA) (Müller and Clough, 2013), see Figure 1.

The nitrification process is the transformation of NH_4^+ to NO_3^-, which is achieved through several oxidation steps (Henze *et al.*, 2002). Different autotrophic[7] species of bacteria are responsible for individual oxidation stages. Firstly, *Nitrosomonas* converts $NH_4^+$to nitrite NO_2^- and consequently, *Nitrobacter* produces NO_3^-. Other species such as *Nitrospira*, *Nitrococcus* and *Nitrosocystis* are less significant in this process. The optimal conditions for autotrophic nitrification are an oxygenated environment and a pH between 5–8 and 7–9 to benefit activity for *Nitrobacter* and *Nitrosomonas*, respectively (Dent, Forbes and Stuart, 2004).

Denitrifying bacteria include autotrophic species that can utilise nitrate as oxidising agent when oxygen is limited (Dent, Forbes and Stuart, 2004). However, oxygen is preferred amongst these bacteria. Denitrification commonly takes place in anaerobic environments with bacteria such as *Achromobactre*, *Bacillus*, *Micrococcus* and *Pseudomonas*. These bacteria convert nitrate to gaseous products in several reduction steps and commonly use various carbon substrates as electron donors (Müller and Clough, 2013). Nitrate is firstly reduced to nitrogen dioxide NO_2^-, subsequently to nitric oxide NO, nitrous oxide N_2O and lastly nitrogen gas N_2 (Dent, Forbes and Stuart, 2004).

The DNRA and denitrification processes both reduce nitrate and the bacteria responsible for the various transformations compete for the utilization of the bound

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[7] Simultaneous production of organic compounds

oxygen (Van Den Berg *et al.*, 2015). Further studies are required regarding the relationship between these bacteria.

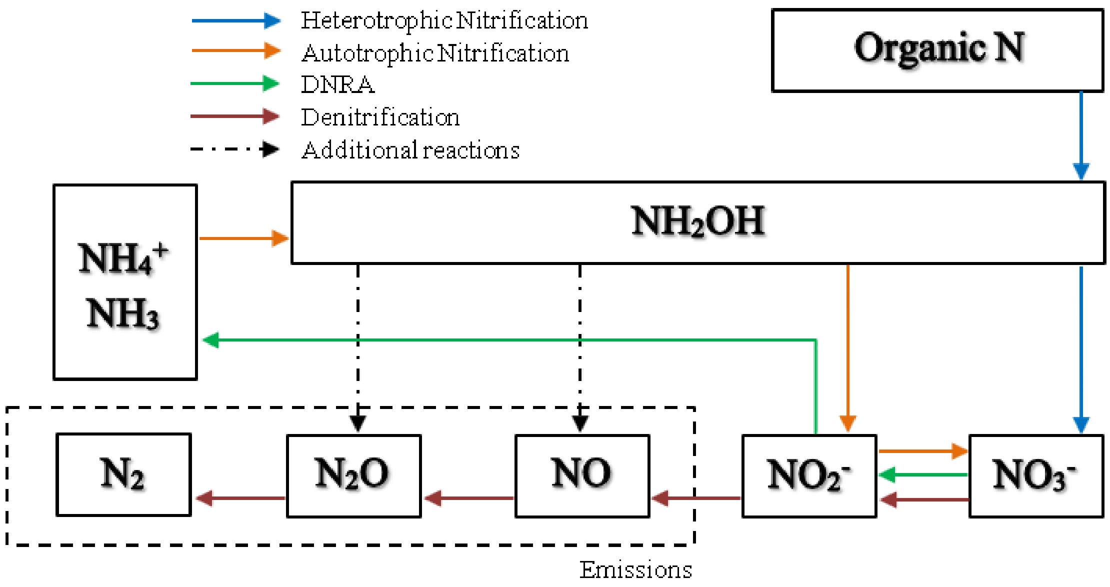

Figure 1. *Microbial oxidation and reduction pathways of nitrogen. Author's own copyright.*

Van Den Berg et al. (2015) attempted to cultivate DNRA bacteria to dominate the denitrification process and concluded that the optimal environmental conditions for this to occur are limited nitrate availability and a high ratio of electron donors in relation to nitrogen compounds as electron acceptors, i.e. high COD:N[8]. The study used a conventional denitrifying bacterial culture from activated sludge in which acetate and nitrate were the main substrates. The ratio was thereafter regularly increased until optimal COD:N conditions of approximately 9:1 prevailed. Previous studies of dissimilatory nitrate reduction to ammonium also indicated these environmental parameters to be of high significance. The thriving species of bacteria consisted of Deltaproteobacteria.

Nitrogen in soil dynamics is a complicated process and there are several factors affecting the produced products. Emissions of NO can lead to air pollution and ground level ozone, acidic rain and eutrophication, which all pose risks to both the environment and human health (Russow, Stange and Neue, 2009). N_2O is a severe greenhouse gas, contributing to global warming and depletion of the stratospheric ozone layer (Zhang, Müller and Cai, 2015). Approximately 37 % of the total N_2O sources is derived from microbial soil reactions. Russow, Stange and Neue (2009) tracked nitrogenous reactions in soil using the Kinetic ^{15}N method. This method plants radioactive nitrogen isotopes in the various nitrogen pools and by monitoring the abundance of ^{15}N over time, reactions and products can be calculated. The purpose of

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[8] Mass ratio of chemical oxygen demand to nitrogen

the study was to quantify the emissions of NO and N_2O and identify which microbial reactions that were responsible for the transformations. Generally, the environmental parameters that governed both nitrification and denitrification were the same as mentioned previously, namely, aerobic and anaerobic conditions respectively. However, the emissions of NO were considerably higher compared to N_2O in an anaerobic environment, which was explained by the laboratory set up and soil porosity. This enabled NO to be released to the atmosphere before it could metabolise further to N_2O. The small N_2O emissions however, derived solely from denitrification of NO in a strict anaerobic environment, and with a longer detention time in more humid and compacted soil, a higher quantity of N_2O can be expected. In a well aerated environment 12 % of NO_2 production was derived from denitrification, opposed to full production from nitrification, which proves that micro-sites of anaerobic soil aggregates exist in aerobic environments and may contribute to denitrification and emissions of toxic gases. Additionally, in an aerobic environment, the small N_2O emissions were not explained by either nitrified ammonia or denitrified nitrate. It is believed that these emissions instead derived from heterotrophic nitrification of organic nitrogen (Russow, Stange and Neue, 2009).

Most nitrified N_2O in soil is known to derive from autotrophic nitrification. However, recent work has identified heterotrophic nitrification of organic nitrogen to potentially be a significant contributor in acidic soils. Zhang, Müller and Cai (2015) investigated this issue by applying the ^{15}N method, but also considered organic nitrogen as a third pathway. The paper summarized data from several studies over a vast variety of soils that had used the ^{15}N method and the Ntrace model. The Ntrace model quantifies N_2O production but analyses transformation processes such as DNRA and oxidation of organic nitrogen, in contrast to the ^{15}N method, which analyses production and consumption rates of nitrogen compounds. The data were used to identify relationships between production pathways and environmental parameters. A linear relationship was clear between increasing autotrophic nitrification rate and increasing soil pH. Similarly, the heterotrophic nitrification rate increased linearly with a higher C:N ratio. However, pH also impacted the heterotrophic nitrification rate and increased in lower pH, with a low-production threshold when the soil pH exceeded 6 in forests. This relationship was not as strong since low rates were seen in low pH soils and vice versa. The denitrification process was influenced strongly by organic content, with increased activity in organic rich soils. This is mainly due to organic carbon, which is the electron donor, being one of the most important factors for denitrification. The data and paper lacked information about how the aeration influenced these relationships in general. Most data were retrieved from aerobic environments and it is not clear exactly how anaerobic environments would change the results and production of N_2O. Heterotrophic nitrification was concluded to be a crucial aspect of N_2O emissions, particularly in soils characterized by high organic content and low nitrogen input. The process contributed with up to 85 % of the total production between pH 4.2 and 8.4 (Zhang, Müller and Cai, 2015).

In addition to nitrogen ending up as emissions to the atmosphere or washed away with groundwater, the organic nitrogen, originating from for example amino acids in human remains, can be utilised by plants and microbes (Schmidt, Näsholm and Rentsch, 2014). Plants can utilise nitrogen more easily if low pH soils benefit the transformation from ammonia NH_3 to ammonium ions NH_4^+ (Dent, Forbes and Stuart, 2004) or if microbial mineralization of organic matter more easily provides available nitrogen (Müller and Clough, 2013). Studies on nitrogen uptake in roots have shown great diversity, including quaternary nitrogen compounds, proteins and even microorganisms which may integrate ammonium ions into their microbial tissue (Dent, Forbes and Stuart, 2004; Schmidt, Näsholm and Rentsch, 2014).

2.3.2 Phosphorus

Phosphorus exists as both organic and inorganic compounds in soils. The inorganic fraction exists mainly as precipitates with positively charged metal ions or adsorbed to the surface of soil mineral particles. The main sources of organic phosphorus are organic matter in soils and litter in forests. In cemeteries specifically, decomposed human remains may account for quite large quantities of phosphorus, from for example proteins, esters and fat tissue (Dent, Forbes and Stuart, 2004). After decomposition and release of organic phosphorus the constituents can be adsorbed or utilised in microbial biomass or, in association with microbial phosphorus turnover[9], eventually be utilised as orthophosphates[10] in plants metabolism (Richardson and Simpson, 2011). Organic and inorganic phosphorus undergo several very complex transformations and depends on relationships between anion cation exchanges, microorganisms and biota which are not yet fully understood (Shen *et al.*, 2011). In fact, there are generally so many influential factors that the availability of phosphorus in soils must be systematically evaluated due to exchanges between many phosphorus pools.

The formation of inorganic phosphorus and the resilience of precipitates is highly dependent on the availability of reactive substrates, but also soil pH, redox potential and organisms. Several studies have proven that soil particle surfaces including aluminium and iron ions and oxides with these metals, strongly dictate soil reactivity towards inorganic phosphate ions in both acidic and calcareous soils (Shen *et al.*, 2011; Bünemann, 2015; Achat *et al.*, 2016). However, the strength of surface charges depends on soil pH and benefits the formation of different phosphates, for example $H_2PO_4^-$, HPO_4^{2-} and PO_4^{3-}, as well variously oxidised metal ions or formed oxides, which all possess higher reactivity in acidic soils. Organic substances, clay minerals and carbonates may also provide reactive surfaces. Organic carbons and other sources of anions may increase the availability of phosphate ions in soils since these ions, like phosphate, adsorbs onto positively charged surfaces. Achat et al. (2016) found that soil texture influenced the reactivity and that a higher fraction of clay content led to a

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[9] Rate of which microorganisms incorporates and releases certain substances

[10] Reactive phosphate (salt or ester of phosphoric acid or molecule with a phosphate [PO_4^{3-}] group)

retention increase. This was explained by an addition of phosphate adsorption sites in clays with higher mineral content and thus leaching is more probable in sandy soils (Havlin, 2004). Although, the change in overall phosphorus adsorption was not as significant for an increase in clay content compared to other parameters. All other factors that influence phosphate availability in soil also affect potential surface reactivity (Achat *et al.*, 2016). For example, organic carbon in soils is also an energy source for microbial immobilization[11] of phosphate.

Microbial activity has a major impact on phosphorus retention in soils. Bacteria provide beneficial transformations of both organic and inorganic phosphorus through mineralization, solubilization, immobilization and mobilization (Bergkemper *et al.*, 2016). Communities made up of microorganisms are extremely diverse and may culture over 10 000 different species per gram of soil. Microbes responsible for solubilization of inorganic phosphorus are for example *Actinomycetes*, *Pseudomonas* and *Bacillus* spp. Environmental parameters strongly dictate thriving organisms. In acidic forests Rhizobiales bacteria maintained phosphorus transformations predominantly under regular concentration in the upper soil layers. When the phosphorus concentration is close to depletion, oligotrophic[12] bacteria are more likely responsible for phosphate availability (Bergkemper *et al.*, 2016). The trait of such solubilizing bacteria is their ability to acidify surrounding soil and media, mostly in association with calcium phosphates. Other bacterial solubilization methods include dissolution of phosphate precipitates by utilising protons, various anions from organic carbons or secretion siderophores[13] (Richardson and Simpson, 2011). Organic phosphorus may undergo microbial mineralization and increase available phosphates in soil. Bacteria release hydrolytic phosphatase and nuclease enzymes which cleave phosphate from esters and acids, respectively (Richardson and Simpson, 2011; Stigter and Plaxton, 2015; Bergkemper *et al.*, 2016).

In addition to transforming and increasing availability of phosphate, bacteria can incorporate phosphorus in their microbial biomass and thereby immobilise it. Energy is gained by degrading organic carbon compounds while enzymes produce phosphate from organic and inorganic forms. The phosphate is thereafter stored in the bacterial cells and polyphosphates are composed which eventually are utilised as stored energy when phosphates are released (Jurtshuk Jr., 1996). Wang et al. (2016) showed that microorganisms' secretion of enzymes and immobilization of phosphate can be correlated with available carbon. This metabolic process, especially for plant growth-promoting bacteria, may benefit plants in the rhizosphere[14] as biota can only utilise orthophosphates (Wang *et al.*, 2016). If bacteria have a higher turnover rate, commonly in carbon-abundant environments, phosphate may be available for plants to a higher extent, however, microbes tend to compete for phosphorus uptake

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[11] Incorporation of phosphorus in microbial biomass

[12] Organisms that can survive in nutrient restricted environments

[13] Strong binding substrate used to mobilize iron ions

[14] Soil that is influenced by root secretion and metabolism in plants

(Bergkemper *et al.*, 2016) and can therefore act both as a sink and source for plant-available phosphorus (Oehl *et al.*, 2001). During phosphate-deficient conditions, roots can secrete similar enzymes to microbes to promote orthophosphate availability (Bünemann, 2015).

Due to all these retention factors, phosphorus can prove to be quite immobile through soils. However, when phosphorus is available to a larger extent and saturate processes such as chemical reactions, microbial turnover and plant uptake, more severe leaching can occur. Older studies have suggested that phosphorus leachate may be up to eight times higher in sandy soils compared to clayey soils (Hartshorn *et al.*, 2016). Inhomogeneous soils and bacterial clouds must also be considered, and phosphorus may be abundant in areas distant from retention processes. The lowest mobility of phosphorus is commonly in the rhizosphere if soils are rich in biota and roots (Havlin, 2004). Soils characterized by low or non-existent root metabolism can therefore be prone to leaching. However, other soil parameters such as organic minerals, hydraulic conductivity, profile and groundwater table are also affecting the process. Salazar et al. (2011) concluded in their investigation on phosphorus mobility in soils, that hydrogeological conditions and precipitation patterns vastly dictated the concentrations of phosphorus in drainage water. The first rainfall after a dry period flushed available phosphorus from the soil, after which the concentration gradually decreased and stabilised. An increase in rainfall intensity could however leach out further phosphorus even after moderate rains, proving that both dry periods and precipitation intensity are important factors of leachate.

2.3.3 Heavy Metals

Heavy metals are some of the most toxic inorganic contaminants, even in low concentrations. They exist in ionized form or as compounds with other reactive substances in the soil. Some of the common heavy metals that are of interest include Cd, Cu, Ni, Hg, Pb and Zn. All these metals possess a positive charge in their ionized form, and are adsorbed to different surfaces in soils, which are often negatively charged. Several factors in soils decide how efficiently heavy metals are adsorbed and to accurately estimate the effective adsorption rates of different soils, specific concentrations of all reactive substrates and environmental parameters must be included. Soils with a high clay content have proven to be more efficient in adsorbing heavy metals, mainly due to the high fraction of clay minerals and colloids with negative charges, while smaller particles simultaneously provide a higher specific surface area (Uddin, 2017). Soil parameters such as hydraulic conductivity, molecular diffusion potential and dispersion are also influential on metal adsorption (Peng *et al.*, 2018). Furthermore, soil pH is one of the most important factors of adsorption, together with negatively charged anions and presence of metal oxides, hydroxides, sulphides, carbonates and phosphates. A high pH generally results in a higher retention ratio; however, due to different mechanisms in the adsorption process, soils with low pH may also retain significant amounts of heavy metals.

The adsorption of heavy metals can be characterized in different ways. For example, as two different mechanisms, specific and non-specific adsorption (Bradl, 2004). The first is identified as inner sphere-complexes, or covalent bonds, and is the stronger mechanisms. Non-specific adsorption however, form more easily reversible outer sphere complexes, such as electrostatic bonds. Heavy metals may also be absorbed into the pore space of other particles, for example clay minerals, and total dissolution of these compounds might be necessary to regain a dissolved ionized form. Specific adsorption is to a large extent governed by the attraction of functional groups on soil surfaces. Hydroxyl [-OH], carboxyl [-COOH] and sulfanyl [-SH] are examples of functional groups which provide strong bonds with heavy metals. These functional groups act as an electron donor after deprotonation[15], and the heavy metal ion is the electron acceptor due to its positive charge. These reactions may also occur with ligands[16] as ternary complexes. The ligands can be many different compounds such as organic acids, but they can also be inorganic. The ligand is similarly deprotonated to form these complexes. Ternary formations like this possess stronger bonds than normal adsorption reactions and are the result of a surface functional group, a metal and a ligand, and occur as metal-like or ligand-like reactions. A metal-like reaction is defined by the ligand being adsorbed and bound by a complex that consists of a surface functional group and a metal. A ligand-like reaction is defined by the ligand being bound in between the same complex chain, see equations 1 and 2, where S denotes the soil particle surface, Me is a metal, Lig is a ligand and H_m denotes the deprotonation of the ligand. Metal-like reactions are more likely to take place in high pH soils and ligand-like reactions in low pH soils.

$$\mathrm{S{-}OH} + \mathrm{Me}^{n+} + \mathrm{H_mLig} \leftrightarrow \mathrm{S{-}OMe{-}Lig}^{(n-m-1)+} + (m+1)\mathrm{H}^+ \qquad (1)$$

$$\mathrm{S{-}OH} + \mathrm{Me}^{n+} + \mathrm{H_mLig} \leftrightarrow \mathrm{S{-}Lig{-}Me}^{(n-m-1)+} + (m+1)\mathrm{H}^+ + \mathrm{H_2O} \qquad (2)$$

Precipitation of metals is not to be confused with adsorption onto a surface. Precipitation is mainly a function of pH and anion-cation concentrations, which forms a solid phase. However, the reactions occur between similar constituents such as oxides and hydroxides and can be characterized as an additional reaction succeeding adsorption, see equation 3. A relationship can often be identified between adsorption and precipitation depending on the availability of metal cations and functional groups (Bradl, 2004).

$$\mathrm{S{-}O{-}MeOH_2^+} + \mathrm{Me}^{2+} + 2\mathrm{H_2O} \leftrightarrow \mathrm{S{-}O{-}MeOH_2^+} + \mathrm{Me(OH)_2(s)} + 2\mathrm{H}^+ \qquad (3)$$

Peng et al. (2018) monitored the adsorption and precipitation of various heavy metals in different soils, mainly by organic matter and mineral phases such as iron and aluminium hydroxides and oxides. The soil pH varied in most scenarios and

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[15] Release of a proton [H^+]

[16] An ion or molecule that binds to a central atom.

relationships were drawn between adsorptions rates, pH and constituent concentrations. The results showed that soil organic matter and minerals like the ones studied in this project significantly influenced metals adsorption rates, and that various sites and metal species may possess different kinetic behaviour in its environment. In general, a higher pH value and more organic matter resulted in a higher retention of metals. Although, the adsorption rates were quite similar for Cd, Ni and Zn with a pH ranging between 5.5 and 6.5. Copper and lead seemed to have a much stronger correlation with pH alterations, even for a small increase of 0.5, and retained more metals at higher pH. Nickel showed a strong correlation with organic matter concentration during a constant pH in different soils, and a high concentration resulted in more retention. Both copper and lead had stronger adsorption compared to Cd, Ni and Zn (Peng *et al.*, 2018).

Acosta et al. (2011) proved in their study on heavy metal retention, with further support from previous studies, that adsorption of the investigated metals, Pb, Cu, Cd and Zn, were influenced by soil salinity. The salts were $CaCl_2$, $MgCl_2$, NaCl and Na_2SO_4 and the metals were affected differently by the various salts. The salt cations function as competitors with the metal ions, causing less metals to be adsorbed and precipitated, however, the contribution of corresponding anions in the salt causes other adsorption complexes to be formed. Generally, a higher salinity resulted in more metal leachate through the soil material. The soils had a high pH of over 7.5 and high concentrations of calcium carbonate. Of all cations in the salts, Ca^{2+} had the most significant impact on metal mobility, the reason being a stronger positive charge. Among the anions, sulphate was most efficient in mobilizing Cu and Zn, which partially was due to lower stability of complexes formed with chloride and the soils having a higher sulphate content. However, chloride caused higher mobility of Pb and Cd. The study also concluded that a higher clay and silt fraction in the soil caused a higher metal retention, and the high pH of the soil caused large amounts of Pb and Zn to be adsorbed by carbonate crystals (Acosta *et al.*, 2011).

The existence of heavy metals will also affect the properties of microorganisms and plants in the soil. Several processes are known to be impaired by metal occurrence, such as carbon mineralization, nitrogen transformations and production and utilization of microbial and biotic enzymes. Furthermore, bacterial populations are commonly decreased, or provide more sustainable conditions for heavy metal resistance bacteria (Kamal, Prasad and Varma, 2010).

2.3.4 Formaldehyde

Formaldehyde CH_2O is a metabolic intermediate of organic material and is produced naturally during oxidation of organic substances (WHO, 2002, 2005). It is therefore present in the environment and in most organisms and is emitted in soils by for example bacteria and vegetation. Formaldehyde is a carcinogenic substance (Persson, 2016) with toxic properties in higher concentrations when in contact with skin or digested. The World Health Organisation have not established any recommended limits in drinking water production due to the insignificant occurrence in regular drinking water compared to the tolerable limit of 2.6 mg/l (WHO, 2011). Cemeteries however, which may contribute as large anthropogenic source of formaldehyde in grave soils, can contribute to a much higher concentration depending on the use of embalming fluids. Formaldehyde is not readily adsorbed to particles in soils and highly soluble in contact with water, therefore, it's considered as a mobile contaminant (WHO, 2002). In terms of persistence in soil and groundwater, the high mobility is compensated by rapid hydration and transformation to glycol and various heterotrophic microorganisms may utilise formaldehyde as an organic energy source. Depending on the biodegradation efficiency, the half-life has been estimated to vary between two days and two weeks in groundwater, and one day to one week in soil.

Allemann et al. (2018) investigated the mobility of formaldehyde in South African soil samples with a depth of 190 mm. The purpose was to evaluate if soil type, rainfall intensity, temperature and acidic rain influenced leaching. The soils were clay, silt and sand, rainfall intensities consisted of prolonged rain and heavy rain, soil temperatures were 20 or 30 °C and the artificial rain had a continuous pH of 4 or 6. After the 24-week test period, the results showed that the highest leachate concentrations occurred 6 weeks into the simulation and decreased after approximately 10 to 14 weeks. Soil type was the only parameter that seemed to influence the leachate, of which sand provided the most critical conditions towards formaldehyde leachate, most likely due to its more permeable characteristics and open pore space compared to silt and clay. However, only 2.6 % of the total formalin[17] content buried in the samples ended up as leachate, the remaining solution was most likely decomposed (van Allemann, Olivier and Dippenaar, 2018).

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[17] Formaldehyde diluted solution

2.4 Previous cemetery investigations

To get a wide understanding for which contaminants may be commonly found in different environments and soils, several studies had to be reviewed due to each study being limited in terms of investigated contaminants and soil conditions. Therefore, not all sub-chapters are necessary to understand the conclusions drawn in relation to the case study of Fridhem cemetery. However, each investigation provides some insight to different risks in specific environmental conditions. To understand some of the main factors and trends, read chapters 2.4.4, 2.4.6 and 2.4.7.

2.4.1 Microorganisms in Vila Nova Cachoerinha, Vila Formosa and Areia Branca, Brazil

Pacheco et al. (1991) investigated the microbiology of three cemetery soils with different environmental conditions and soil contents. The microbial analysis covered amongst other microorganisms, total coliforms, proteolytic bacteria and heterotrophic aerobic and anaerobic bacteria, see Appendix B.

Among the tested cemeteries, Vila Nova Cachoeirinha cemetery was characterized as sandy sediments with clayey layers and had the highest count of total coliforms and anaerobic bacteria and contained a significant number of aerobic bacteria. This soil had a water table depth of 4–9 meters and the clay layers may have functioned as impermeable barriers, which could have resulted in limited aeration and reduced conditions compared to the other tested soils, and anaerobic bacteria could thrive. Additionally, the clay-rich layers may contain more organic material, which benefits microbial growth, compared to the other two cemeteries.

Vila Formosa cemetery was governed by similar water table fluctuations and the profile consisted mainly of alternating layers of different sediment thicknesses and grain size, however, in contrary, dominated by aerobic bacteria. This soil may supply the bacterial biomass with oxygen at this depth due to varying particle size and higher gas diffusion potential from the atmosphere, or availability of reducible substrates. Even though aerobic bacteria dominated, the results showed the lowest microbial concentrations. This can be explained by the low water table and distant sampling from the burial depth, which causes the unsaturated sediment to act as a natural filter against both microorganisms and organic material.

Areia Branca was characterized by marine sediments, high porosity and high hydraulic conductivity. The water table was strongly governed by sea level due to its coastal location and varied between 0.6 and 2.2 meters. This provided a well oxygenated environment for bacteria and was the main reason behind the dominating aerobic fraction. Anaerobic bacteria were moderately high in comparison, which could be due to the shifts in water table depths and opportunity for these bacteria to grow in the saturated zone with lower oxygen supply. Although, it should be noted that facultative anaerobic bacteria may grow in aerobic environments, while oxygen can be toxic or restrict growth for obligate anaerobic bacteria (Holland, Knapp and Shoesmith, 1987).

In general, all cemeteries were highly contaminated with microbes and constituted a health risk. It was clear that the health risks were a consequence of cemetery operations since both proteolytic and lipolytic bacteria, which both are involved in the decomposition of protein and adipose tissue, was found in high concentrations (Pacheco *et al.*, 1991).

2.4.2 Microorganisms in Itaquera and Santo Amaro, Brazil

The cemeteries in Itaquera and Santo Amaro consists mainly of loamy soils with high sand content (Zychowski and Bryndal, 2015).

The Itaquera cemetery has a steep slope and an aquifer, covered by an impermeable layer with high concentrations of Al and Fe, indicating that even though high permeability contributes to transport of contaminants, significant retention can occur (Peng *et al.*, 2018). The samples showed high concentrations of total coliforms, *Shigella* and *Klebsiella* spp.. The cemetery was concluded to be the microbial source in the investigation. However, it was inconclusive if these microorganisms were a direct cause of cadaver decomposition in grave soils, or from faulty construction of sewage system and graves, resulting in additional leaching.

The cemetery in Santo Amaro is located on a gentle slope with a descent of approximately 12 m. High concentrations of both heterotrophic and proteolytic bacteria were found in two of the investigated wells. One well specifically showed higher counts of *E. faecalis* and E. *coli*, 91 and 36 CFU[18]/100 ml, respectively. Some cemeteries in Brazil are more prone to leaching contaminants than in other regions of the world. The tropical climate can induce more precipitation, and combined with sandy soils transportation is more likely, depending on the height of the groundwater table. For full references of these two investigations, see Silva et al. (2008) and Abrão (2007) in Zychowski and Bryndal (2015).

2.4.3 Microorganisms in Luz de Tavira and Seixas, Portugal

Rodrigues and Pacheco (2003) conducted a groundwater analysis on cemeteries in Portugal, however, no comprehensive information was given on the environmental parameters for the separate sites. The cemeteries are relatively new, and the first graves were dug in the end of the 20th and early in the 21th century. The cemetery in Luz de Tavira is located in a porous aquifer in southern Portugal and Seixas is a region in the northwest, which is governed by coastal tides. No information was given about the lithology at Seixas cemetery. Two sample locations were included from

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[18] Colony Forming Unit

both cemeteries, inside the burial area and 300 and 290 m away for Lux de Tavira and Seixas, respectively.

Both cemeteries found high counts of total coliforms, faecal coliforms, sulfite reducing bacteria and heterotrophic bacteria, indicating a potential hazard for water resources, see Appendix B. Outside the cemetery peripheries could a clear trend of reduced concentrations could be detected, especially for Seixas cemetery. However, for Luz de Tavira, similar maximum values were found outside the periphery for faecal streptococci, proteolytic bacteria and heterotrophic bacteria (Rodrigues and Pacheco, 2003), which could indicate high mobility of microorganisms, especially considering that proteolytic and heterotrophic bacteria are generally abundant in grave soils during decomposition, and that the cemetery was recently constructed. This can be explained by beneficial climate for microbial growth, as the south regions of Portugal are governed by Mediterranean conditions (Zychowski and Bryndal, 2015), but mainly the porous aquifer as a means of transport.

2.4.4 Heavy metals in Northwest Ohio, USA

Spongberg and Becks (1999) identified metal contamination in a large cemetery, dating back to the 19th century. The study measured adsorbed metals and samples were collected in two similar soil types, characterized as silty clay loam and fine sandy loam. The main difference is a stream that often floods parts of the cemetery, which most likely has resulted in a slightly larger particle fraction distribution in one of the soils. Both had moderate permeability and organic matter concentrations. The soil was exposed to moist conditions and a water table above burial depth during wet periods. Most soil samples had a pH ranging from 7 to 9 and was taken near subsidence surfaces, indicating decomposition of coffins. Areas inside the cemetery and outside the periphery were investigated, with several sampling depths at each location. A river, to which the stream is connected, was located very close to the cemetery.

Reference samples were taken for both soil types outside the cemetery periphery. For the silty clay, the only anomaly in relation to the background samples was the As concentration. The other soil resulted in elevated concentrations of all measured metals, Cu, Pb, Zn and Fe. It is believed that the As originated from past cemetery practices when arsenic-containing embalming fluids and wood preservatives were used, especially since all As sample locations were near old graves from before the 19th century. The elevated metal concentrations in the fine sandy loam are believed to be a consequence of material choice of caskets. In general, for most metals, the concentrations decrease with distance from the grave, both vertically and horizontally. However, all measurements indicated quite low environmental risk, including As, due to low concentrations in all sample locations. It is possible that higher concentrations prevailed here once, although, due to the local stream and nearby river, much of the content may have been flushed out of the cemetery area and accumulated elsewhere, possibly in sediments further down the river.

2.4.5 Comprehensive constituent analysis in southwest Germany

Fiedler et al. (2012) investigated a cemetery in southwest Germany which had been active since 1974 and consisted of 74 graves. The purpose was to comprehensively analyse the content of the grave soil, including a total of 22 metals and other elements, see Table 4. The soil derived from Triassic red sandstone and was of sandy clay loam, with an approximate ratio of 7:1:2 between sand, silt and clay, respectively. However, there were some variations and the soil beneath the coffins generally contained a larger fraction of clay. The cemetery was mainly governed by lateral water flow which partially resulted in saturated conditions. Adipocere could thereafter be formed due to a lack of sufficient aerobic decomposition of fatty acids. To reuse graves after 25 years, lingering remains had to be moved to a mass grave on the premises in 2006. The samples included five replicates from the soil slightly above the coffin and directly underneath, and a total of 40 graves were investigated. An analysis was also conducted on the soil within the coffins and the adipocere-covered remains. Soil reference values were retrieved from samples 40 m outside the burial field periphery.

Table 4. *Elements identified in grave soil in Germany (Fiedler et al. 2012).*

		Elements			
As	Cd	K	N	Pb	Zn
Ba	Co	Mg	Na	S	Fe
C	Cr	Mn	Ni	Sb	
Ca	Cu	Mo	P	Sr	

The results indicated that the element concentrations in the soil above and underneath the coffin were similar for all samples. In relation to the reference soil samples on similar depth, however, significantly elevated concentrations of Ba, Ca, Na and Zn were identified below the coffins; the same for Ba, Ca, Cu, Mo, Sr and Zn above the coffins. Concentrations of N and C were much higher above and below the coffins compared to the reference values. P and S however, were below the detection limit in all sample locations except for inside the coffins, and the highest concentration of S was found in black humus material. This may be a result of either sulphate oxidation of fatty acids in the adipocere tissue or sulphur-containing amino acids from protein, which results in precipitated iron sulphide in the presence of iron (Dent, Forbes and Stuart, 2004; Fiedler *et al.*, 2015). The soil inside the coffins, governed by decomposed material, showed a further significant increase of Ba, Ca, Cr, Cu, Na, Pb, Sr and Zn when compared with the surrounding soil; As, Fe, K and Mg showed an opposite behaviour.

Graves began to be reused in 2006, and consequently caused mixed soil conditions above and below the coffin, which most likely is the reason why similar concentration has arisen. Fidler et al. (2012) reasoned that adipocere was formed in most graves.

However, during exhumation, some instances indicated partially decomposed adipocere and as a result, less remains. A hypothesis was that the roots from trees and other vegetation, which had stretched across the burial area and through graves, absorbed significant amounts of water and lead to a more aerated environment and hence aerobic degradation. Dewatering and nutrient uptake in the rhizosphere could also explain the low abundance of phosphorus. As explained by Wang et al. (2016) and Richardson and Simpson (2011), carbon rich soils may benefit microbial phosphorus mineralization, precipitate dissolution and turnover rate, which increase the production of bioavailable phosphorus for plant uptake. The soil in this cemetery is governed by high C:N ratio, approximately 10-13:1, and might have resulted in bioavailable production and depletion of phosphorus in other locations but inside the coffins.

2.4.6 Nitrogen, phosphorus, heavy metals and microorganisms in Nikaia cemetery, Greece

A recent investigation by Massas, Kefalogianni and Chatzipavlidis (2018) evaluated the state of Nikaia cemetery, abandoned in 2003 after approximately 80 years of operation. The purpose was to see if the old grave soil in the area was in good enough condition for establishment of an urban park. In total, 58 samples were taken and analysed. The sampling points were analysed in different layers to retrieve results that represent top soil, above grave soil, grave soil and below grave soil at the depths 0–30, 30–80, 80–150 and 150–250 cm respectively. All samples were tested for nitrogen, phosphorus and the heavy metals Pb, Zn, Cu, As, Ni, Mn and Cr, while 48 samples were tested for microbial load. *Clostridium perfringens*, coliforms and Enterococci were detected in the majority of all samples, additionally, *Salmonella* spp., *Listeria monocytogenes* and E. *coli* were analysed but no samples indicated any presence of these bacteria. Vegetation such as grass and bushes have grown to cover the entire area due to neglected maintenance. The soil has mainly been exposed to temperatures between 25–35 °C, an unsaturated layer below burial depth, aerated conditions and some surface runoff opposed to full drainage as a result of a gentle slope (1:10). A study by Argyraki & Kelepertzis (2014) showed that the soil naturally contains very high concentrations of Ca (13–16 %), Fe (2.3–2.7 %), and Al (4.1–4.9 %) specifically. However, other metals may also exist in naturally high concentrations. This is due to limestone and alluvium being the parent materials, which are naturally rich in metals and vastly influences the soil composition (Argyraki and Kelepertzis, 2014). Limestone furthermore contributes to alkaline conditions and the pH buffers around 8 in all soil samples in the cemetery. Finally, the soil contains quite little organic material and is characterized as medium to fine grained with a clay content of approximately 30-35 % (Massas, Kefalogianni and Chatzipavlidis, 2018).

The results showed in general small variations with depth, which was reasonable since the cemetery regulations required reburial of the same grave after 3–5 years, producing mixed soil conditions. The ammonium concentration proved to be constant throughout all four layers, however, the nitrate seemed to accumulate and increase with depth linearly, meaning that total nitrogen also increases with depth. This is most likely a result of additional aeration when consistently mixing the soil every 3–5 years, which induces significantly higher nitrification rather than denitrification. Also, the main species involved in the nitrification process, *nitrosomonas* and *nitrobacter*, benefits from the alkaline pH governed in the soil (Dent, Forbes and Stuart, 2004). Zhang, Müller and Cai (2015) showed that there is a linear relationship between increased autotrophic nitrification rate and increased pH, and that the heterotrophic nitrification rate increase linearly with a higher C:N ratio. The results show that the highest ratio exists at the second and fourth depths, 0-30 and 80-150 cm, respectively. Additionally, since the cemetery has not had any further significant anthropogenic nitrogen sources, it can be assumed that both nitrate and ammonium is leaching downwards and is utilised by the biota that is now disseminated across the entire ground surface. Ammonium ions may also be less likely to adsorb to colloids in the soil due to the abundance of different metals and low concentration of organic carbon. The ammonium concentration has probably not been constant throughout the different layers in the past, but rather nitrified and accumulated due to these processes. This indicates that it is possible that the aquifer underlying the karstic geological layer has been exposed to nitrate leaching for quite some time.

Phosphorus concentration were abundant in all layers of the soil, but the highest concentrations were found in the top three layers and resulted similarly to approximately 300 mg/kg of soil. However, of the total phosphorus, roughly 10 mg was available for utilisation, while the rest consisted of mineralized compounds and precipitates. The separation of total and bioavailable phosphorus is logical, considering the naturally high Ca, Fe and Al additions to the soil. Peng et al. (2018) showed that aluminium and iron oxides and hydroxides strongly dictates phosphorus adsorption and precipitation in high pH soils, such as the soil in this cemetery, and can be assumed to have a very high impact in combination with the mixed aerated conditions and oxidation potential. The high immobilization of phosphorus can also be explained by formed solids, calcium phosphates, aluminium phosphates and iron phosphates. Furthermore, the bioavailable fraction has the highest value in the third layer, the grave soil, and declines thereafter in both vertical directions. This indicates that the cemetery must have had an anthropogenic contribution to the phosphorus concentrations in the soil, but probably does not provide any significant leaching to the surrounding environment but is rather retained in the cemetery soil. The declining trend towards the surface can be explained by developing biota and uptake by the root system over the past 15 years.

Heavy metals were found in various concentrations, although, it was difficult to evaluate if they have an anthropogenic or natural origin due to the underlying metal-rich rock material (Argyraki and Kelepertzis, 2014). Other metals than Al, Fe, and Ca can be assumed to have a geogenic origin, namely, Ni, Cr and Mn derived from the ophiolitic parent rock. However, elevated concentrations of Pb, Zn, Cu and As are regarded as being anthropogenic. No samples showed any protrusive signs that leaching may be an issue in the cemetery. The concentrations were quite similar throughout all four sample depths, except for Pb which showed a slight decrease with depth below the second layer. This was discussed as being the result of mixed soil. When comparing these two studies, Argyraki and Kelepertzis (2014) and Massas, Kefalogianni and Chatzipavlidis (2018), see Table 5, some differences can be identified.

Table 5. *Measured metal concentration in top soil. Values are presented as mg/kg soil and the interval represents mean values ± standard deviations for Massas, Kefalogianni and Chatzipavlidis (2018). Bold font is used for highlighting metals with increased concentration. The last row is the bioavailable fraction of the metals.*

	Anthropogenic				Geogenic			
Depth	Pb	Zn	Cu	As	Ni	Mn	Cr	Fe
0-10 cm [1]	3-47	18-91	11-36	22-32	117-174	455-628	**113-173**	23-27*10^3
0-30 cm [2]	43-50	**275-285**	**71-83**	16-19	120-165	640-690	20-23	25-27*10^3
0-30 cm [2]	2.2-2.7	1.5-3.2	1.2-1.5	NA[3]	1.6-1.7	3.2-4.5	NA[3]	1.6-2.1

[1]Argyraki & Kelepertzis (2014), [2]Massas, Kefalogianni and Chatzipavlidis (2018), [3]Not Analysed

Lead, As, Ni, Mn and Fe showed similar concentrations in both investigations. Zinc and Cu however, had elevated values when comparing the two investigations, see the bold values in Table 5. This might be a direct cause of the cemetery and coffins in the soil, since Spongberg and Becks (1999) also found higher concentrations of these two metals. If that is the case, coffins have possibly leached additional Zn and Cu between the investigations by Argyraki & Kelepertzis (2014) and Massas, Kefalogianni and Chatzipavlidis (2018). Chromium occurred in lower abundance in comparison with Argyraki and Keleperziz (2014). It is not certain that these soil samples were taken from the cemetery due to the concentration distribution map in Argyraki and Keleperziz (2014) being large scale and only show soil composition in the vicinity of the cemetery. Although, it seems likely, considering the similarity between most of the metals. The total metal content was generally not considered as hazardous and soluble metal ions were found in significantly lower concentrations. This can be explained by the alkaline pH, and probably, abundance of aluminium and iron oxides and hydroxides that result in adsorbed and precipitated complexes and compounds (Peng *et al.*, 2018).

The microbial analysis showed that the highest concentrations of *Clostridium perfingens*, Enterococci and coliforms were generally found in the top soil, however, a few samples indicated maximum concentrations in the grave soil layers as well. The count of microorganisms was higher than the reference safety values used in the study, which represented non-existent or insignificant risk, set by the US

Environmental Protection Agency and European Community. These standards are applied towards composted bio-waste and some marginal may be accepted. Additionally, Enterococci and coliforms were only detected in approximately 30 and 50 % of the samples, respectively, while *clostridium perfringens* were more abundant. Even though the safety standards were exceeded for some bacteria, the site was considered as an adequate location for an urban park since no samples contained *Salmonella* spp., *Listeria monocytogenes* or E. *coli*.

2.4.7 Pharmaceuticals, nitrogen, phosphorus and DOC in South Germany

Fiedler, Dame and Graw (2018) conducted a study of 12 cemeteries in southern Germany. The cemeteries were selected with the criterion of having been recommended installation of a drainage system, which was implemented at a depth ranging from 2.6 to 3 m prior to the investigation for all burial areas. The study compared the concentrations of pharmaceuticals in the potentially contaminated drainage water, with surface water upstream from the cemetery, which was unaffected by the cemetery operations. Furthermore, the study investigated the concentration of nitrogen, phosphorus and DOC[19]. The last burial occurred the same year or previous year to the time of sampling in six of the cemeteries and is unknown for the remaining cemeteries.

Only four and seven drugs were detected in the drainage water and surface water samples, respectively (Table 6). The Carbamazepine and Propranolol concentrations were always significantly lower in the cemetery drainage water in comparison with the respective upstream surface water. Additionally, previous studies and analyses of water sources have identified these pharmaceuticals in higher concentrations, compared to the results by (Fiedler, Dame and Graw, 2018).

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[19] Dissolved Organic Carbon

Table 6. *Pharmaceutical findings in the drainage and surface water from 12 cemeteries in Germany. All values are retrieved from Fiedler, Dame and Graw (2018). Concentrations are presented in ng/l, and number of detections is presented within the parenthesis. N/D – Not detected, *Below quantification limit.*

Pharmaceutical	Drainage water	Surface water
Atenolol	N/D	57-301 (2)
Metoprolol	23 (1)	450-2230 (2)
Propranolol	N/D	N/D
Carbamazepine	10-225 (7)	43-418 (5)
Chlorthalidone	N/D	N/D
Hydrochlorothiazide	(4*)	(7*)
Indomethacin	N/D	67 (1)
Naproxen	N/D	41-81 (2)
Diclofenac	N/D	129-574 (3)
Furosemid	N/D	N/D
Ibuprofen	(1*)	N/D
Trimethoprim	N/D	N/D

In contrast to the pharmaceutical findings, nitrogen phosphorus and DOC concentrations were consistently higher in the cemetery drainage water samples than in the surface water samples. The concentrations ranged between 0.2–68, <0.05–2.8, <0.1–1.9 and 1.3–11 mg/l for nitrate, ammonium, phosphate and DOC, respectively (Fiedler, Dame and Graw, 2018).

2.4.8 Pharmaceuticals in Portugal

Five cemeteries were investigated in 2015 by Paíga and Delerue-Matos. They collected two groundwater samples in each cemetery and analysed them for pharmaceutical contamination. A list of all investigated drugs can be seen in Table 7. All cemeteries initiated operation during the 20th century, however, due to discretion within the municipally and privately owned cemeteries, no information was released regarding the geological or hydrogeological conditions.

Table 7. *Investigated pharmaceuticals by Paíga and Delerue-Matos (2016). Number of detections (10 samples were analysed) is presented within the parenthesis. N/D – Not detected, *Below quantification limit.*

NSAIDs/Analgesics	**Psychiatric drugs**	**Antibiotics** (N/D)
Salicylic acid (10)	Venlafaxine (N/D)	Trimethoprim
Acetylsalicylic acid (N/D)	Trazodone (N/D)	Ofloxacin
Acetaminophen (4*)	Citalopram (N/D)	Ciprofloxacin
Hydroxyibuprofen (N/D)	Paroxetine (N/D)	Enrofloxacin
Carboxyibuprofen (N/D)	Norfluoxetine (N/D)	Sulfadiazine
Ketoprofen (10*)	Norsertraline (N/D)	Sulfapyridine
Naproxen (N/D)	Fluoxetine (8)	Sulfamethazine
Nimesulide (1)	Sertraline (3*)	Sulfamethoxypyridazine
Diclofenac (N/D)	Carbamazepine (10)	Azithromycin
Ibuprofen (10*)	Diazepam (N/D)	Clarithromycin
		Sulfamethoxazole
		Sulfadimethoxine

Eight of all the investigated pharmaceuticals were detected in the samples (Table 7). Nimesulide and Fluoxetine resulted in the lowest measurable concentrations of the investigated pharmaceuticals. The most abundant pharmaceuticals, Salicylic acid, Carbamazepine and Fluoxetine, were discussed to be present to a larger extent due to the two psychiatric drugs being more persistent, and although Salicylic acid is an easily degradable analgesic, it is commonly found since this drug is widely used (Paíga and Delerue-Matos, 2016). The concentration of Carbamazepine was within the same range as the investigation performed by Fiedler, Dame and Graw (2018) and should therefore not be hazardous to either human health or the local ecosystem.

2.5 Remediation methods for drainage water

Drainage water in cemeteries can be compared with leachate found in landfills (Fiedler *et al.*, 2012) or urban stormwater runoff, which generally have similar content. Water remediation from these origins focus mainly on nutrients, metals and microorganisms, and additional contaminants may be of interest depending on specific situations. The Center for Watershed Protection has developed a database over statistic of removal efficiencies for different contaminants, the National Pollutant Removal Performance Database. This database contains a total of 166 cases of stormwater remediation studies and includes dry ponds, wet ponds, constructed wetlands, filtering, infiltration, bioretention and open channels. It should be noted that these results are examples of performances and some fluctuations are to be expected since the analysed studies deal with varying flows and concentrations of pollutants. Other factors influencing efficiency are for example, if the study regarded mass contaminant load or concentration, removal computing method, design and environmental parameters in the catchment area (Center for Watershed Protection, 2007).

Not all compiled remediation methods and results in Table 8 achieve desirable reductions of all contaminants encountered in cemetery drainage water, namely, nutrients, heavy metals and microorganisms.

Suspended solids and particles is of much higher concern for stormwater runoff remediation as they are more likely to be flushed from impervious surfaces (Shammaa *et al.*, 2002). The values obtained by the Center for Watershed Protection for suspended solids removal should be considered as partially larger particles than what is expected to be found in strictly drainage water and are therefore more easily removed. The removal efficiency of total suspended solids in Table 8 is consequently assumed to be overestimated for drainage water. Furthermore, the treatment efficiencies for metals would be more reliable if more elements than copper and zinc were included. However, these two metals have proven to be of highest abundance from anthropogenic sources in cemetery soils according to Spongberg and Becks (1999), Fiedler et al. (2012) and Massas, Kefalogianni and Chatzipavlidis (2018) and may be two of the most common materials related to coffin leachate in drainage water.

Dry ponds have proven to have poor performance against nutrients and metals over several investigations, and bioretention systems has a poor removal of phosphorus. These two systems will therefore not be investigated further. Open channels had a negative impact on bacteria removal and even prompted growth in all three investigations. Microorganisms have shown, in previous investigations, to always be a risk in active cemeteries, see chapter 4.1, 4.2 and 4.3; open channels are therefore also excluded from further investigation. The remaining remediation methods seem to have decent to very good performance against all the concerned contaminants, with regards to potential fluctuations in the performance results, and will therefore be included in an in-depth investigation.

Table 8. *Comparison of removal efficiencies. All values are retrieved from Center for Watershed Protection (2007) and are presented as percentages.*

	TSS[1]	TP([2])	TN	Cu	Zn	Bacteria
Dry Ponds						
Median	49	20 (-3)	24	29	29	88
25th Percentile	18	15 (-8)	5	22	1	83
75th Percentile	71	25 (8)	31	42	59	92
No. Studies	10	10 (6)	7	4	8	2
Wet Ponds						
Median	80	52 (64)	31	57	64	70
25th Percentile	60	39 (41)	16	45	40	52
75th Percentile	88	76 (74)	41	74	72	94
No. Studies	44	45 (28)	22	23	34	11
Wetlands						
Median	72	48 (25)	24	47	42	78
25th Percentile	46	16 (6)	0	18	31	67
75th Percentile	86	76 (53)	55	63	68	88
No. Studies	37	37 (26)	24	12	19	3
Filtering						
Median	86	59 (3)	32	37	87	37
25th Percentile	80	41 (-11)	30	33	71	36
75th Percentile	92	66 (63)	47	67	91	70
No. Studies	18	17 (7)	9	13	18	6
Infiltration						
Median	89	65 (85)	42	86	66	-
25th Percentile	62	50 (55)	2	62	63	-
75th Percentile	96	96 (100)	65	89	83	-
No. Studies	4	8 (4)	7	4	6	0
Bioretention						
Median	59	5 (-9)	46	81	79	-
25th Percentile	15	-76 (-9)	40	37	37	-
75th Percentile	74	30 (49)	55	97	95	-
No. Studies	4	10 (5)	8	5	5	0
Open Channels						
Median	81	24 (-38)	56	65	71	-25
25th Percentile	69	-15 (-94)	40	45	58	-63
75th Percentile	87	46 (26)	76	79	77	-25
No. Studies	17	16 (14)	9	16	16	3

[1]Total suspended solids, [2]Bioavailable phosphorus

Table 9. *Efficiency order for specific contaminants. Based on the median values of Table 8*

	Median removal efficiency
TP	Infiltration > Filtering > Wet Ponds > Wetlands > Open Channels > Dry Ponds > Bioretention
TN	Open Channels > Bioretention > Infiltration > Filtering > Wet Ponds > Wetlands and Dry Ponds
Metals	Bioretention > Infiltration > Open Channels > Filtration > Wet Ponds > Wetlands > Dry Ponds
Bacteria	Dry Ponds > Wetlands > Wet Ponds > Filtering > Open Channels (Infiltration, Bioretention: No Data)

2.5.1 Retention systems

Retention systems can focus on retaining water to mitigate intense stormwater events or lower pollution levels in the water. Wet ponds and constructed wetlands are two systems that have proven to be successful in both categories and may be very useful for larger catchment areas which produce large flows and water volumes. The main difference is the design parameters and geometry, which causes them to be specifically useful in different situations (Wong, Breen and Somes, 1999).

2.5.1.1 Wet Ponds

Wet ponds are generally considered to be one of the most aesthetically pleasing stormwater practices. It is a retention method that aims to remove contaminants through settling of particles (Youn and Pandit, 2012). As suspended solids settle towards the bottom of the pond, biological and physical adsorption mechanisms ensure that particulate nutrients and metals simultaneously are reduced. The design relies on an effluent, often an orifice in combination with overflow, which generally increase with higher influent volumes, and depend on the storage capacity of the retention pond. The pond itself consists of a permanent pond volume and a storage capacity volume. When the height of the retention volume increases, the effluent increases due to higher water pressure or higher water level. The hydraulic behaviour in the detention pond is therefore largely dependent on design choice: surface area and depth. To limit the effluent volume and velocity, any permeable material or perforated film may be used in the outlet (Chrétien *et al.*, 2016).

A common problem with wet ponds is resuspension of settled particles. Inflow or wind may cause turbulence in the water body, which results in elevated effluent concentrations (Chrétien *et al.*, 2016). Another issue that can arise is erosion of the soil, at either the inlet, outlet or receiving water, for example a stream. This occurs when the flow is too big or concentrated, and the soil cannot withstand the accompanied force. Ponds should therefore be designed according to suitable drawdown velocities and erosion and resuspension risk control guidelines for the likeliness of certain rain events (Wong, Breen and Somes, 1999). To ease this issue, vegetation can be used on the periphery of the pond which protect the soil from erosion. These aquatic plants may also partially utilise nutrients from runoff (Youn and Pandit, 2012). High organic content in influent water, high BOD or COD, may lead to reducing conditions and settled phosphates and metals may again become dissolved into the pond (Wong, Breen and Somes, 1999). An impermeable layer, for example a clay layer, may be used in to avoid infiltration and contamination in combination with groundwater recharge and keep a permanent minimum water table in the wet pond (Sønderup *et al.*, 2015).

2.5.1.1.1 Performance of an agricultural wet pond

Chrétien et al. (2016) investigated the performance of a wet pond, connected to an agricultural area of 23.1 ha. The purpose was to determine if the pond was efficient in reducing stormwater peak flows and reduce nutrients. The pond's total storage capacity was under-dimensioned in relation to criteria discussed by the authors, equal to 146 m^3/ha. The outlet was made of a perforated vertical pipe; however, the perforated film was not designed to reduce any outflowing contaminants, rather control flow.

The results showed that the pond's hydraulic retention and peak flow reduction performed differently depending on the stormwater-volume to total-storage-capacity ratio:

- Effective for ratios <0.5
- Almost insignificant for ratios >3; resulting in <13 % reduction in peak flow

Furthermore, the results showed, for TSS;

- -100% to 93% removal for 20 events
- Increase occurred at 5 instances and for the lowest influent concentrations
- In 85% of all samples the effluent concentration was above 25 mg/l (Table 1)
- For effluent concentrations below 25 mg/l, the influent was below or near the guideline value initially

for TN;

- The concentration was below 1.25 mg/l (Table 1) in 7 and 9 out of 11 samples for the influent and effluent, respectively
- Negative performance occurred when the influent concentration was low
- One instance of increased concentration when the influent was above the guideline value (42%)
- The two samples with highest influent concentration resulted in high removal efficiencies of 90 and 82 %, respectively, and satisfying effluent concentration

for P;

- Negative performance in the 6 samples with lowest influent concentration (<0.14 mg/l)
- All effluent concentrations were >0.10 mg/l and above the guideline value

In conclusion, the pond was quite efficient in reducing all three contaminants, and mean removal ranged between 50–56 %, 52–52 % and 48–59% for total suspended solids, total nitrogen and total phosphorus, respectively.

2.5.1.1.2 Floating treatment wetlands

Hartshorn et al. (2016) investigated the effect of floating treatment wetlands (FTWs) in three wet ponds to see if they could further increase nutrient removal. FTWs utilises aquatic plants (macrophytes) and are kept at the surface of the wet pond by floating devices. They provide nutrient uptake, but also, their roots may act as biofilms for microbial populations, which can reduce the potential of bacterial contamination.

According to Hartshorn et al (2016), previous studies have shown that nutrient uptake were optimal at 5–15 °C. At higher or lower temperatures, the efficiency reduced with approximately 30 % and 15–20% for nitrogen and phosphorus, respectively. Furthermore, with the addition of soil to the floating device and aquatic plants, the reduction rate seemed to increase significantly. For full references of these two investigations, see Van de Moortel et al. (2010) and Tanner and Tom (2011) in Hartshorn et al. (2016).

This study investigated the performance of FTWs in three ponds by comparing values before and after implementation. Two ponds had an area greater than 2000 m^2, and one with an area greater than 1000 m^2. The largest pond stood out by having a long and rectangular shape, while the other two were strictly rectangular and govern by algal growth. The FTWs covered approximately 5–7 % of the total surface areas. All ponds were exposed to runoff from traffic and fertilizers with additional flow from a stormwater conduit. Samples were gathered at the inlet and outlet of each pond (Hartshorn *et al.*, 2016).

Generally, no large differences were identified between the pond’s performance before and after implementation of the FTWs, except for the larger rectangular pond whose nutrient source came from a nearby tomato field. This pond showed significant further reduction of phosphorus, from -3 % to 70 % reduction post FTW implementation, and was most effective during storm events compared to no precipitation. However, many results showed an increase of both phosphorus and nitrogen, which may be explained by surface runoff entering the pond between the inlet and outlet. In relation to the findings by Van de Moortel (2010), this study showed no notable correlation between temperature and reduction potential. The performance of FTWs could be more impressive if they were to cover a larger percentage of the pond’s surface area (Hartshorn *et al.*, 2016).

2.5.1.1.3 Wet pond in combination with permeable filters

A wet pond in Denmark proved to have insufficient phosphorus removal, 0.45 mg/l in the influent and 0.15 in the effluent, which was significantly higher than the set limit. Therefore, the treatment facility was retrofitted to a chain treatment with a sand filter and a crushed concrete filter following the wet pond. The two filters were separate and connected in parallel, which enabled Sønderup et al (2015) to perform individual reviews. The experiment investigated removal efficiency of a wide range of contaminants, including phosphorus, nitrogen, TSS, organic matter and the heavy metals Ni, Pb, Cu, Cr and Cd. The outlet of the pond is controlled by a water break which, at maximum flow, provides a minimum retention time of 0.9 h in the 1 m thick filters. The hydraulic properties of the systems were designed to provide constant saturation which minimized the remobilization risk of contaminants (Grebel *et al.*, 2013) and emergence of channels through the filter media (Sønderup *et al.*, 2015). Both filters had the same pore volume. However, the crushed concrete consisted of a larger grain fraction, and the retention time was generally lower for the crushed concrete due to a protection fleece covering the sand filter. The performance of these two filters are presented in chapter 2.5.2. In addition to the inlet pipe, the pond was exposed to large volumes of groundwater seepage and was by recommendation significantly smaller in relation to the catchment area.

Ponds with low volume to catchment ratio have shown to have significantly lower reduction of suspended solids (Sønderup *et al.*, 2015). However, this pond resulted in decent suspended solids and organic matter retention performance of 60 and 65 % as an annual mean, respectively. During one recorded intense rain event, concentrated runoff entered the pond due to flushing. Since it was a significant rain event, a large stormwater volume entered the pond subsequently, and the initial contaminant load did not have enough time to settle. Therefore, the effluent resulted in high concentrations. Although, the retention was generally quite constant through intense rain events and did not vary too much from the annual mean efficiency for either suspended solids or organic matter. The best performance occurred during smaller rain events which impelled longer retention times in the wet pond.

The phosphorus removal proved to be quite poor with an annual mean of 10 % of the dissolved fraction and 39 % of the particulate fraction. However, the removal of particulate phosphorus was approximately 3.6 times more effective during the summer months in relation to winter. The removal of dissolved phosphorus was partially believed to be biological uptake during the summer, reaching a maximum of 62 %, when macrophytes grew on the clay surface.

Nitrogen proved to have quite expected performance and similar to the values presented in Table 8, 39 % reduction as an annual average. However, the pond proved to have better than expected removal of metals (Table 10), and ranged between 40 and 91 % reduction. Copper, which is one of the most common metals related to disintegration of coffin materials (Spongberg and Becks, 1999; Fiedler *et al.*, 2012; Massas, Kefalogianni and Chatzipavlidis, 2018), had the highest removal efficiency. However, the dissolved fraction increased significantly, which most likely is due to transformation of solid state copper. Nickel, lead and cadmium, which all had concentrations lower or near the guideline values set by Gothenburg (Table 1), were also significantly reduced.

Table 10. *Pond removal efficiency for heavy metals. Values are retrieved from Sønderup et al. (2015) and are the results from the first month of operation.*

		Ni	Pb	Cu	Cr	Cd
Inlet µg/l	Particulate	6.4	7.3	72.3	26.1	0.5
	Dissolved	2.5	0.1	4.6	0	1
Outlet µg/l	Particulate	1.1	2.4	6.2	4.2	0.3
	Dissolved	2.8	0.1	15.4	0	1
Removal %	Particulate	83	67	91	84	40
	Dissolved	-12	0	-235	0	0

2.5.1.1.4 Indicator organisms in two costal wet ponds

Hathaway and Hunt (2012) investigated the performance of microbial reduction in two wet ponds. For more information about the wet pond properties, see Hathaway and Hunt (2012). The investigated parameters were E. *coli* and Enterococci and samples were gather at the inlet and outlet surfaces with a grab sampler. One sample was obtained for each stormwater event when there was an ongoing flow in the system. Fifteen and 18 storm events were analysed in total for pond 1 and pond 2, respectively.

The removal efficiency was impressive in both wet ponds with slightly higher performance in the first pond (Table 11). The input concentrations of both E. *coli* (included in total coliforms) and Enterococci was high in comparison to organisms found in the groundwater of cemeteries, for example, the investigations by Pacheco et al. (1991) and Rodrigues and Pacheco (2003), see chapter 4.1 and 4.3. Total coliforms in the three cemeteries in Brazil never exceeded a maximum concentration of 1600 MPN[20]/100 ml, and from the two cemeteries in Portugal the maximum number of coliforms was measured to 3900 MPN/100 ml; The average concentration of total coliforms compared to E. *coli* in these two ponds was also significantly lower.

Table 11. *Removal efficiencies of indicator organisms. Values are retrieved from Hathaway and Hunt (2012).*

		E. coli MPN/100 ml			Enterococci MPN/100 ml		
		In	Out	Reduction %	In	Out	Reduction %
Pond 1	Mean	2483	62	98	2356	237	90
	Maximum	24196	19863		24196	24196	
	Minimum	255	2		278	2	
Pond 2	Mean	1273	60	95	274	37	87
	Maximum	81640	3466		24196	1633	
	Minimum	10	2		2	2	

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[20] Most probable number

2.5.1.2 Constructed Wetlands

In contrast to the permanent water volume of wet ponds, wetlands remove pollutants while the storage volume periodically fills and drains, depending on the inflow (Wong, Breen and Somes, 1999). They are shallower in comparison and usually cover a larger surface area. In addition to the removal mechanisms in wet ponds, wetlands practices phytoremediation through dense vegetation consisting of mainly aquatic macrophytes, but also epiphytic biofilms, which grow on the surface of other vegetation without leaching nutrients. Phytoremediation in wetlands have been proven to have good removal of nutrients and metals. The flow resistance of vegetation in the wetlands contributes to a lower water velocity which is a key factor in retaining contaminants (Wong, Breen and Somes, 1999; Malaviya and Singh, 2012). Nitrogen is commonly removed by transformation through nitrification and denitrification processes due to microsites of aerobic and anaerobic conditions, which typically is vastly provided by the rhizosphere in wetlands (Wong, Breen and Somes, 1999). Another positive aspect of these microsites and drying and wetting cycles is rapid decomposition of organic material, and the organic material found in sediments are therefore usually derived from plants.

Wetlands have been reported to perform better than ponds under high microbial loading, however, the opposite applies for low loadings (Beutel, Whritenour and Brouillard, 2013). This is generally the result of a higher background concentration of bacteria in wetlands, due to an active rhizosphere, and when low concentrated influents are treated, the wetland will act as a source. Additionally, bacteria are more resilient in wetland environments (Kadlec, 2009). For example, attached particles and vegetation may block solar radiation, and there is often an abundance of biofilms.

Wetlands may function in several different ways, for example with surface flow or subsurface flow which may be horizontal or vertical (Malaviya and Singh, 2012). Subsurface flow occurs through a permeable material, in which the macrophytes grow, while surface flow takes place above substrate level. There are generally some performance differences between these two systems. According to an investigation and central tendency analysis by Kadlec (2009), based on 134 surface flow and 143 subsurface flow studies, the following can be said about surface flow in comparison to subsurface flow wetlands:

- may risk higher background concentration due to open waters tendency to attract wildlife and provide habitats
- more light may promote phytoplankton and consequently lead to higher concentration of suspended particles and phosphorus
- reduces organic nitrogen, ammonia and ammonium slightly better under low loading, and behave very similar to subsurface flow when the loading increase
- generally better performance against total nitrogen and more efficient nitrification process due to anaerobic conditions in subsurface flow wetlands

- more commonly applied when inlet concentrations are lower and are meant to polish the water quality
- slightly better phosphorus removal
- lower ability to insulate during cold winters

Smaller stormwater areas, for example drainage areas in cemeteries, may prove to have difficulties in providing a sufficient baseflow and govern preferable water levels. The term pocket wetland is usually applied in such situations, and are small wetlands that periodically, during dry periods, lose macrophytes and phytoremediation efficiency (Watershed Protection Techniques, 1996; Malaviya and Singh, 2012). Additionally, phytoremediation is dependent on climate and the efficiency differ with seasons. The highest performance is usually during summer and associated optimal growth conditions. However, macrophyte species thrive under different conditions, and for optimal and a more constant performance, it may prove efficient to use a variety of species in a wetland, especially considering that certain species benefits in various nutrient concentrated waters (Rai, 2008; Vincent *et al.*, 2018). *Phragmites australis*, *Typha* spp., *Scirpus* spp. and *Phalaris arundinacea* are some of the most common wetland macrophytes used in Europe (Ebrahimi *et al.*, 2013); furthermore, a list of naturally growing macrophytes that may be applied for wetland operations in Sweden is presented in Table 12. To obtain optimal phytoremediation performance, it is important to consider macrophyte water consumption, and consequently, the accompanied evapotranspiration for different species, especially for practices and climates that risks prolonged dry periods (Leto *et al.*, 2013).

Table 12. *Examples of wetland macrophytes found in Sweden (Möller, 2016).*

Family	**Species**
Typhaceae	Typha angustifolia
Typhaceae	Typha latifolia
Poaceae	Phragmites australis
Poaceae	Glyceria maxima
Poaceae	Phalaris arundinacea
Cyperaceae	Scirpus lacustris
Cyperaceae	Carex Rostrata
Ceratophylláceae	Ceratophyllum demersum
Potamogetonáceae	Potamogeton lucens
Potamogetonáceae	Potamogeton crispus
Iridáceae	Iris pseudocarus
Alismatáceae	Alisma plantago
Equisetáceae	Equisetum fluviatile
Polygonaceae	Persicaria amphibia

2.5.1.2.1 Subsurface flow

Leto et al. (2013) compared the performance of two macrophytes in constructed wetlands, namely, *Typha latifolia* and *Cyperus alternifolius*. The wetlands had a narrow and long shape and practiced horizontal subsurface flow. In total, nine wetland channels were investigated, of which one third were *T. latifolia*, *C. alternifolius* and unvegetated, respectively. The substrate consisted of silica quartz gravel in which the main elements were silica > iron > aluminium. The temperature ranged between -2.9 °C and 38.1°C, with mean temperature of 23.1 °C.

The average inflow contaminant concentrations and the reduction by the two macrophyte constructed wetlands can be seen in Table 13. In addition to a better performance by *T. latifolia* in all the presented parameters, during the summer months, this macrophyte proved to have a more rapid growth. Due to a higher above-surface biomass production, it can be expected that after harvest, *T. latifolia* will have contributed with a higher reduction of all absorbed contaminants, most likely nutrients and metals. This species was believed to outperform *C. alternifolius* mainly because of its ability to adapt to changing environmental conditions and tolerance high concentrations of contaminants. In general, the difference in water consumption was quite small, with a slightly lower water loss for *C. alternifolius*. This additional water loss should also be considered when reflecting on the outflow concentrations, since contaminants become more concentrated at high water losses (Leto *et al.*, 2013).

The results clearly demonstrated the effect vegetation in wetlands has on sedimentation of solids. The unvegetated wetland channel had approximately half the removal efficiency of *T. latifolia*. Logically, nutrients and microbes attached to settleable solids will result in lower outflow concentration in the vegetated channels. The difference in hydraulic influence in the channels was not investigated, and therefore, it is difficult to estimate how much of the reduction is contributed to macrophyte absorption, in contrast to higher addition of settleable solids due to proliferation, more rapid growth and additional friction and flow resistance, see TSS in Table 13. However, the difference in biomass production was lower than that of nitrogen concentration in the biomass between the two species, indicating *that T. latifolia* surely have a higher potential for nitrogen absorption within these environmental parameters.

Table 13. *Pollutant removal of the macrophytes Typha latifolia and Cyperus alternifolius in a constructed wetland. All values are retrieved from Leto et al. (2013), presented as mg/l and constitutes the average value over two years. The percentage removal efficiency is presented within the parentheses.*

	In	*Typha latifolia* (^{1})	*Cyperus alternifolius* (^{1})	Unvegetated (^{1})
TSS	28.7	10.2 (64.3)	15.2 (47)	19.2 (32.9)
TKN^6	17,8	8.5 (51.6)	11.2 (36.1)	14.2 (19.3)
TP	8	4.2 (47.9)	5.6 (31.7)	7 (14.2)
BOD$_5$	24.6	6.7 (72.4)	8.5 (64.8)	12.2 (49.6)
TC^{2,4}	23442	2511 (88.8)	4365 (80.5)	13804 (40.5)
SF^{3,4}	8318	1445 (81.2)	2089 (74.1)	5012 (39.2)
E. *coli*^{5}	1175	120 (89.5)	170 (85.5)	660 (43.5)

^{1}Percentage reduction, ^{2}Total coliforms, ^{3}Feacal streptococci, ^{4}MPN/100 ml ,^{5} CFU/100 ml, ^{6}Total Kjeldahl Nitrogen (organic material, ammonium and ammonia)

Vincent et al (2018) similarly investigated subsurface systems with different macrophyte species. Their study, however, focused on so called polishing treatment and investigated the efficiency of wetlands with lower influent concentrations. The climate was subtropical, and the experimental setup included parallel channels with the dimensions 13 * 4.5 m, and a substrate thickness of 0.6 m, which consisted of granite gravel. The investigated species were *Phragmites australis*, *Typha orientalis*, *Cyperus alternifolius*, *Thalia dealbata* and *Arundo donax*. The sampling occurred every two weeks between May and November. The average influent concentrations for TN, TP and BOD were 7.45, 0.22 and 5.3 mg/l, respectively, for samples gathered before August, and 9.84, 1.85 and 6.9 mg/l, respecitvely, during and after August. The increase in concentration was due to artificial increase, with the purpose to investigate the performance with a higher loading.

In contrast to the study by Leto et al. (2013), the dominating species that resulted in most rapid growth and biomass production was *C. alternifolius*, with an above-ground biomass of 80 kg. *T. dealbata*, *A. donax* and *P. australis* followed at similar biomasses, varying from 40 to 44 kg. *T. orientalis* had the lowest biomass production of 29 kg. The densest growth occurred near the inlet and consistently reduced towards the outlet for all macrophytes, which probably indicate nutrient deficiency along the length of the channel and proves nutrient uptake by the macrophytes. *C. alternifolius* showed the highest concentration of phosphorus in the entire channel and the highest total absorption of both phosphorus and nitrogen. The results should however be analysed with caution since only the above ground biomass was analysed, and some species may be more efficient in storing nutrients in their roots (Vincent *et al.*, 2018). *T. orientalis* for example, had almost equal biomass above and below surface near the inlet, and four times higher biomass below surface near the outlet, which were much larger ratios of below-surface biomass compared to other species. The outflow concentration curves throughout the study period was quite hard to analyse due to few sampling occasions and long timeframe between samples. However, in general, for lower concentrations of phosphorus before August, all macrophytes had quite similar performances, and the highest removal efficiencies occurred at the highest inflow concentrations. For nitrogen, the lowest outflow concentration varied between different macrophytes and sampling occasion. In general, the highest removal efficiencies occurred for *C. alternifolius* and *A. donax*, but the difference was not significant. *C. alternifolius* seemed to perform better against nitrogen after the significant increase of phosphorus loading, which may very well be due to more rapid growth and root network development, which allowed higher uptake of nitrogen as well.

Furthermore, it is important to emphasize that these results were obtained in a subtropical climate, and in colder climates, different macrophytes might be more dominant.

2.5.1.2.2 Surface flow

Beutel (2012) investigated three wetland channels in Moscow, each consisting of two cells and governed by phytoremediation from *Typha* spp.. The climate in this study was more applicable to Nordic conditions, according to weather stations located in Gothenburg (SMHI, 2018), with average temperature of -1.4 °C and 19.1 °C in January and August, respectively. The channels were narrow and long with a very large surface area due to high daily hydraulic loading from a wastewater treatment plant, and the wetlands' purpose was to polish the effluent. The very long channels made it possible to produce concentration profiles, which show the nutrient uptake. The pumped water from the treatment plant provided almost plug flow conditions with steady concentrations under the investigated events. The nitrogen concentration for example, had an average value of 2.95 mg/l and 5.8 mg/l for two consecutive years.

The wetland proved to be very efficient in reducing nitrate (Table 14), which to a large extent made up most of the total nitrogen. The ammonia concentration however, remained quite constant, was slightly reduced, or increased throughout the wetland channels. The phosphorus concentration showed a similar trend, although, increased more significantly. The influent phosphorus concentration was very low due to the treatment plant utilizing aluminium precipitation with filtration prior to the wetland. The phosphorus concentration might have increased due to higher background concentration. The surface flow was exposed to light and quite high solar radiation during the summer months, which may have resulted in higher concentrations of phytoplankton and corresponding suspended phosphorus (Beutel, 2012).

Table 14. *Inlet and outlet concentration of nutrients in constructed wetland during two consecutive years. All values are retrieved from Beutel (2012).*

	Year 1		Year 2	
Pollutant mg/l	Inlet	Outlet	Inlet	Outlet
Nitrate	3	0.2	5.8	0.5
Ammonia	0.11	<0.1	0.06	<0.1
Phosphorus	0.12	<0.25	0.05	<0.25

Maine et al. (2017) investigated the metal retention capability of two free surface flow wetlands located on the same site in Argentina (Appendix C). They received the effluent of metallurgical plants; however, influent concentrations were not always significantly higher than the regional guidelines for effluents to receiving waters in Gothenburg.

The measured concentrations showed that the inlet had a larger variance compared to the outlet concentrations, which proved to be quite stable even when the inlet concentration shifted. This indicated a good buffer potential of the wetlands, and most likely a reduction to background concentration. Initially, *E. crassipes* dominated the first wetland, however, after two years, *T. domingensis* started to take over and persisted. This was mainly due to the high pH in this wetland, and *T. domingensis* was more tolerable. *E. crassipes*, which is a floating macrophyte, stored most of the contaminants in the growing biomass, in contrast to *T. domingensis* which stored significantly more metals and slightly more phosphorus in the sediments, approximately 70–90 % and 62 %, respectively. Furthermore, of the stored contaminants in the *T. domingensis* biomass, significantly more metals were stored in the root system compared to the leaves. An opposite trend was identified for phosphorus. There were some concerns regarding decomposed plant litter with accumulated metal concentrations, and if they would leave the system. However, this was not the case, and instead, detritus became mineralized and added to the sediments. The behaviour of the wetland channels was similar to the investigation by Vincent et al. (2018), and the highest accumulation of contaminants occurred near the inlet. As demonstrated by Table 15, both wetlands have had success in reducing metals, even though their concentration was consistently quite low, although, not all effluent concentrations were in accordance to the regional guidelines in Gothenburg (Table 1).

Table 15. *Inlet and outlet concentration of metals in constructed wetlands. All values are retrieved from Maine et al. (2017) and presented as µg/l.*

	Wetland 1			Wetland 2		
Parameter	Inlet	Outlet	Reduction %	Inlet	Outlet	Reduction %
pH	10.8	8.3		7,9	8	
Fe	824	87	89.4	350	110	70.4
Cr	92	14	84.7	310	22	92.9
Zn	41	20	51.2	72	31	51.7
Ni	48	23	69.5	18	4	77.5

The system also experienced unexpected events, such as rodents and very high and temporary concentration of Cr. However, after the macrophytes were reduced, by wildlife ransacking and high accumulation harvest, the system managed to balance its performance and return to normal operation within a few months, proving the resilience of wetlands (Maine *et al.*, 2017).

The result in this study also demonstrates the drawbacks of the investigation by Vincent et al. (2018) who only analysed the above-ground biomass, and that *T. domingensis* accumulates most contaminants in the root network (metals) and surrounding soil or sediment (metals and phosphorus).

Beutel, Whritenour and Brouillard (2013) investigated faecal coliform reduction efficiency of two parallel surface flow wetlands with a pre-treatment sedimentation basin in Washington, USA. See Beutel, Whritenour and Brouillard (2013) for information on the design parameters. The inflow to the sedimentation basin varied between 30 and 500 cfu/100 ml over seven years of monitoring. The sedimentation basin proved to be quite successful considering the low inflow concentration and resulted in outflow concentrations between 6 and 80 cfu/100 ml. The wetlands however, proved to have poor performances and minor reduction or increase of faecal coliforms. The outflow concentration had minimum and maximum values of 17 and 1585 cfu/100 ml, with average concentrations of 142 and 188 cfu/100 ml. The reason of the increase in concentration was most likely due to higher background concentrations of faecal coliforms in the wetlands compared to the inflow concentration. Furthermore, relationships were identified in the pre-treatment sedimentation basin between inflow turbidity and faecal coliforms, and higher temperature in the wetlands were coupled with poorer performance, which probably is linked to more rapid microbial growth in the warmer seasons.

2.5.2 Filtration and infiltration systems

Permeable systems aim to remove contaminants through filtration, sorption and chemical and biological transformations that takes place in the media. Filtration and infiltration methods rely on similar processes. The main difference is that infiltration methods recharge the groundwater while filtration methods commonly includes an outlet to a water network (Davis and Birch, 2009). The filtration process in both methods generally removes particles effectively through size elimination, and subsequently the corresponding elements or microbes attached to that particle (Grebel *et al.*, 2013). However, if precipitated forms of contaminants, for example metals, are retained in the media, chemical or biological transformation processes resulting from microbial presence and redox potential can produce a higher bioavailable fraction which is more susceptible to transportation through the pore system. Therefore, systems must be designed with understanding of all coexisting retention processes for optimal removal efficiency. The infiltration or filtration media particle size dictates hydraulic conditions, including flow rate and water contact time, but also specific surface area. The choice of material governs sorption capacity and the media should therefore preferably contain surfaces that promotes complexation and precipitation, for example oxides, see chapters 2.3.2 and 2.3.3. Media that alters pH may also be an effective method for increasing the sorption potential of filtration and infiltration systems. Sand is one of the most commonly used materials, however, the success primarily relies on good particle removal and hydraulic behaviour, as dissolved phases may be more difficult to retain. A mix of different materials is therefore usually recommended for best performance, providing good filtration and sorption capacity. Another effective approach is to utilise the sorption potential of oxides in media coatings of filter and infiltration beds, for example aluminium and iron oxides and hydroxides, which Peng et al. (2018) proved to increase retention capacity. Other materials that have shown to have good sorption capacity are ash, shale, crushed cement or concrete and limestone (Grebel *et al.*, 2013). An adsorbent's capacity to retain contaminants depend on the relative surface charge strength of the media. The net surface charge can be determined by investigating the point zero charge (PZC) which indicates the pH of a neutral media (no surface charge) (Liu, Sansalone and Cartledge, 2005; Miyittah *et al.*, 2016). Examples of PZC values of different adsorbents can be seen in Table 16. The values consider circumstantial variations but may attain a slightly different pH. Activated carbon for instance, commonly possess several different surface functional groups, see chapter 3.3 for examples, and may therefore have PZC values that vary significantly (Miyittah *et al.*, 2016). The same can be established for soil species in common infiltration methods (Appel *et al.*, 2003). Thus, particle surface functional groups may behave amphoteric[21], and a pH below PZC results in positive surface charges, while a pH above PZC gives the surface charge a negative value. However, other factors such as roughness, specific surface area and potential for double layer adsorption on particles play an important role (Ahammed and Meera, 2010).

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[21] Negative or positive charge

Table 16. *PZC values for different adsorption media based on previous studies.*

Adsorbent media	Liu et al. (2005)	Miyittaha et al. (2016)	Naeema et al. (2007)	Appel et al. (2003)
Silica Sand	6.8	-	-	-
Activated Carbon	10.3	8.4	-	-
IOCS*	8.6	-	-	-
Iron oxide	-	-	8.0-8.5	-
MOCS**	6.4	-	-	-
Manganese oxide	-	4.1-5.0	-	-
Aluminium oxide	-	7.5	-	-
Titanium oxide	-	5.4-5.7	5.7	-
Ultisol	-	-	-	2.3-3.7
Oxisol	-	-	-	3.9-4.4

*Iron oxide coated sand, ** Manganese oxide coated sand

Providing certain redox conditions may also be an effective method for reducing pollutant levels in drainage waters, specifically organic material, but also metals. Zero-valent iron (ZVI) can be used to provide exemplary conditions for reduction, due to the low redox potential, and greatly reduce organic material concentrations (Grebel *et al.*, 2013). Yusmartini and Faizal (2017) showed that nano zero-valent iron (NZVI) and activated carbon with a 1:3 ratio reduced COD and BOD with 95 and 93%, respectively. Furthermore, the composition managed to reduce 100 % of Pb and Cr when 0.4 g NZVI-Activated carbon per litre was mixed in the influent water to a sand column bed. The reduction of organic material and absorption of ammonia consistently increased with NZVI concentration and contact time (Yusmartini and Faizal, 2017).

The hydraulic behaviour and moisture content in the filter or infiltration media affect contaminant removal. When media beds are periodically completely dried, colloids and pollutants risk remobilization (Grebel *et al.*, 2013) and channels can develop through the media (Sønderup *et al.*, 2015). However, this can be prevented by elevating drains or outlet valves in filtration systems. This may likewise be applied for management of environmental conditions and adjust redox potential, microbial activity and biological transformation processes. It is of most interest for bioretention and bioinfiltration systems, which are similar to vertical subsurface flow wetlands and generally govern higher microbial populations due to an active rhizosphere. Managing redox potential can be difficult and tend to vary depending on the concentration of dissolved oxygen and other compounds in the influent. Furthermore, the atmospheric contribution towards aerated conditions is quite restricted during these short time intervals and depend significantly on soil saturation. During longer contact times and with saturated media, the dissolved oxygen can be consumed and promote anaerobic conditions, which in turn benefits for example denitrification of nitrate. Complementing substances such as mulch and woodchips can be used to benefit microbial transformation and denitrification of nitrate, however, these substances may provide excess organic material and lead to high BOD levels in the effluent (Grebel *et al.*, 2013).

Generally, the retention capacity is higher during long contact times with the media as it promotes sorption processes. However, it should be noted that increasing the contact time will simultaneously increase the need of storage and retention tanks during intense and long rain events (Grebel *et al.*, 2013).

The sand and crushed concrete filters investigated by Sønderup et al. (2015), see description in chapter 2.5.1.1.3, were set out to have equal hydraulic properties and pore volume. The particle sizes varied between 3 and 35 mm for the concrete filter and <1 and 4 mm for the sand filter. However, due to a protection fleece on the sand filter media, included as a measure against some irregular sand behaviour, the concrete filter received a higher flow, higher contaminant loading and consequently shorter residence time for adsorption, which is assumed to affect the results.

The pH from the pond (filter inlet) varied between 6 and 9, and the alkaline conditions of the concrete filter initially resulted in effluent pH above 10. Therefore, acid dosing had to be operated to regulate pH levels for the first 8 months, after which the leaching of the filter had stopped. The concrete filter was assumed to have higher adsorption capability due to the higher content of Ca, Fe and Al in cement. After the 1-year study, it was concluded that more Ca washed out of the sand filter, and the concrete filter retained more Fe and Al in comparison to the sand filter; however, the sand filter retained a higher percentage of Fe and Al. It is unknown how the Ca would continue to leach after the study period (Sønderup *et al.*, 2015).

Suspended solids and organic matter was reduced quite efficiently in both filters. The annual mean ranged between 75 and 85 % for both parameters and filters and the outlet concentrations were below the regional guidelines set by Gothenburg. However, maximum concentration periodically exceeded the guidelines.

The filters had similar performance, 75-85 %, against the particulate fraction of phosphorus. The dissolved fraction showed a different trend and the effluent concentration increased after approximately 2-4 months after the operation began. This decrease to 24 and 13 % retention for the concrete and sand filters, respectively, was most likely the result of mineralization of particulate phosphorus which eventually was transported through the filter media. The increase of dissolved phosphorus in the sand filter effluent happened more slowly in comparison to the concrete filter, which could be explained by the higher loading in the concrete filter.

Both filters had poor performance against total nitrogen, 8 % reduction and 34 % increase in the concrete and sand filter, respectively. This was the result of an increasing ammonium concentration of unknown origin. However, it was assumed to derive from mineralization of organic nitrogen, but DNRA of nitrate could also be a source. A clear trend was identified between nitrate and ammonium, and when the ammonium concentration increased, the nitrate concentration decreased. This can be explained by a combination of the above-mentioned processes and possibly anoxic conditions which restricted nitrification and promoted leaching of ammonium. The concrete filter was probably not exposed to as severe anoxic conditions as the sand filter due to faster flow rate and inability of full oxygen consumption in the low residence time, even though both filters were constantly saturated. This was reflected in the relationship between ammonium and nitrate, and when ammonium concentration decreased in the inflow to the filters, the nitrate concentration increased more significantly in the concrete filter effluent.

The concrete and sand filters showed good performances, especially considering that the prior wet pond treatment removed some of the influent heavy metal concentration to the filters below the standard set by Gothenburg. Furthermore, the wet pond removed most of the settleable particles, and most of the treatment rely on adsorption capability. Cd proved to persist above the Gothenburg guideline value due to the dissolved fraction not being reduced in neither of the filters, see Tables 1, 17 and 18. Cu showed decent removal of both the particulate and dissolved phase, however, the guideline was still exceeded in the concrete filter. The dissolved Ni concentration increased significantly in the sand filter. This is most likely a result of leaching from the filter media since the measurements are from the first month of operation. However, another reason may be periodical mineralization of retained particulate Ni. This would be more likely if the particulate fraction was reduced with similar significance as the increase of the dissolved phase, which is not the case.

The study proved that a concrete filter may be just as efficient as a sand filter, however, there were some ambiguity regarding oil spills that could have risked the integrity of for example phosphorus removal, and might be the cause of reduced phosphorus efficiency in the concrete filter (Sønderup *et al.*, 2015).

Table 17. *Concrete filter removal efficiency for heavy metals. Values are retrieved from Sønderup et al. (2015) and are the results from the first month of operation.*

Concrete filter		Ni	Pb	Cu	Cr	Cd
Inlet µg/l	Particulate	1.1	2.4	6.2	4.2	0.3
	Dissolved	2.8	0.1	15.4	0	1
Outlet µg/l	Particulate	0.3	0.6	1.3	0.8	0.1
	Dissolved	2.2	0	13.2	8.2	1
Removal %	Particulate	73	75	79	81	67
	Dissolved	21	100	14	-	0

Table 18. *Sand filter removal efficiency for heavy metals. Values are retrieved from Sønderup et al. (2015) and are the results from the first month of operation.*

Sand filter		Ni	Pb	Cu	Cr	Cd
Inlet µg/l	Particulate	1.1	2.4	6.2	4.2	0.3
	Dissolved	2.8	0.1	15.4	0	1
Outlet µg/l	Particulate	0.3	0.5	1.4	0,8	0.2
	Dissolved	15	0	6.7	0	1
Removal %	Particulate	73	79	77	81	33
	Dissolved	-436	100	56	0	0

Ahammeda and Meero (2010) compared three filtration beds regarding their ability to retain metals and bacteria, and its correlation with turbidity removal. The three beds consisted of sand, iron oxide coated sand, and a mix of iron oxide coated and manganese oxide coated sand, respectively. The laboratory test was conducted in cylindrical glass containers. Neither bed was ever fully dried out during the experiment. Two influent waters with different properties were tested and acquired a contact time of 30 minutes with the filters. The filtered water derived from roof runoff and was governed by quite high metal concentrations, and from a canal with high counts of bacteria in relation to the roof runoff.

The uncoated sand filter had quite poor performance in comparison with the other two filters (Tables 19 and 20). The mixed coated filter media and the iron oxide coated sand had similar reduction potential for total coliforms, faecal coliforms, heterotrophic bacteria and Pb. However, the mixed coated sand filter seemed to perform better against Zn. All three filters had similar results for turbidity removal. The coated filters showed very good performance and reduced coliforms and heterotrophic bacteria with > 99 % and approximately 87 %, respectively. The removal efficiency of both metals was also good; however, the Zn concentration could not achieve the regional guideline standard for Gothenburg, which is not unrealistic considering the high influent concentration. The reason behind better performance of Zn removal in the mixed coated filter is most likely due to the manganese top layer having a PZC value below the influent water pH, and thereby, attaining a negative surface charge suitable for adsorbing metals, followed by strong adsorption of negative colloids and correlating bacteria by the positively charged iron oxide coated sand particles. The filter only consisting of iron oxide coatings may therefore have smaller potential for adsorbing metals. Nevertheless, the iron coated sand managed to reduce 91 % of the Zn concentration.

Table 19. *Contaminant reduction in roof runoff using three filtration materials. All values are retrieved from* Ahammeda and Meero (2010).

Roof runoff	Influent	Mixed Coated	Iron Coated	Sand
pH	6.6	6.9	6.9	6.8
Turbidity NTU	2.8	1.0	1.0	1.2
TC[1] MPN/100 ml	310	3	4	60
FC[2] MPN/100 ml	17	0[4]	0[4]	0
HB[3] CFU/100 ml	1300	170	181	1060
Pb µg/l	140	10	10	100
Zn µg/l	2200	60	200	1610

[1]Total coliform, [2]Feacal coliform, [3]Heterotrophic bacteria, [4]Expected value

Table 20. *Contaminant reduction in canal water using three filtration materials. All values are retrieved from* Ahammeda and Meero (2010).

Canal water	Influent	Mixed Coated	Iron Coated	Sand
pH	8.0	7.9	7.8	7.8
Turbidity NTU	12.0	1.5	1.5	1.6
TC[1] MPN/100 ml	4300	14	25	1260
FC[2] MPN/100 ml	170	2	2	50

3 Case study

Fridhem cemetery is located north of Gothenburg city, see Figure 1 in Appendix D, and is rich in both deciduous and coniferous forest. The cemetery has been operational since 1968 and covers 87 ha.

3.1 Geology and hydrogeology

In 2004, a geotechnical investigation was performed on the Muslim burial field by GF Konsult AB. The upper 2 m of the soil mostly consisted of sandy and gravelly clay landfill masses, with moderate humus organic material and occasionally brick construction residue was found. At a few sample locations, sand and gravel made up the main soil fraction. The first meter consisted of more mull compared to lower depths. However, sand and gravel were found in the clay throughout the entire area. The high clay content together with gentle slopes caused the central parts of the burial area to be water saturated and runoff was limited. However, runoff was deemed good in the southern parts and no investigation or sampling were conducted in the northern parts. See Appendix G for more information on all sampling pits and the soil type at different depths. According to another investigation performed by Norconsult, the landfill masses are placed on a soil layer of the Gothenburg moraine, and consists of mainly moraine and esker sediments with layers of clay. A map constructed by The Geological Survey of Sweden (SGU) indicates that the total soil depth varies between 5 and 10 meters.

The areas that have clayey soil with higher influence of gravel and sand, and a thickness of 1.5–2 meters, give rise to an upper waterbed and a higher water table, approximately 1–1.5 meters below ground surface level. In areas dominated by a thin layer of porous landfill masses on top of the clayey soil, no upper waterbed is able to form and a lower groundwater table at approximately 2 meters depth is found. However, even these porous soils are influenced by clay, which consequently results in restricted natural water flows in the entire areas. Furthermore, the groundwater table in these areas is governed by seasonal variations.

The graves in this area will be dug to a depth of 1.6 meters, and therefore, water drainage is necessary to avoid wet soil and saturation of coffins.

3.2 Drainage and remediation system

Norconsult performed calculations to estimate and quantify the drainage water by using runoff data from SMHI and an analytical approach with a computing software model. Their recommendation was to construct drainage conduits in area M3, M5 and M6, see Figure 1 in Appendix G, located approximately 2 meters below the ground surface. However, no drainage was ever implemented on M5.

The pressurized drainage conduit from the Muslim burial field is connected to the gravity fall stormwater conduit and the water is mixed with compost leachate before reaching the sand filter, see Appendix E. There are also three wells in this junction point. All wells in connection to this stormwater system have a sand trap for larger particles to settle and avoid them from reaching the sand filter. The drainage system of area Kv41 and Kv43 (Figure 1 in Appendix E) are connected to the sand filter, however, the drainage water is most likely of good quality since very few graves have currently been dug here.

Before reaching the sand filter, the inflow is divided into three separate aerated influent conduits to evenly distribute the water over the filter media. The top and bottom layers consist of 40 cm macadam, in which the influent and effluent conduits are located. The middle layer consist of 90 cm washed sand, separated from the macadam layers by protection fleeces. See designs in Appendix F. The effluent water is thereafter discharged to a nearby creek.

3.3 Development of graves in the area

There are currently 5627 graves, 1118 of which are buried in the 2 ha Muslim burial field on the eastern part of the cemetery (2018-05-18), see Figure 2 in Appendix D. The capacity is set to approximately 1400 graves, meaning 80 % of the space is currently occupied. This burial field is strictly governed by non-cremated coffin burials and natural decomposition with one coffin in each grave. The only drained burial fields are M3 and M6 as previously stated. The drainage system at M3 was installed after 80 interments, and the drainage conduits were placed in between the unoccupied graves (Figure 2 in Appendix G). Today, 221 (83 % capacity) is used at M3. Area M6 is a burial field for children and 345 interments (67 % capacity) occupies the area.

A future burial field is planned for the areas Kv41 and Kv43 and will also consist of coffin burials. However, each of these graves will hold 2-3 overlaying coffins.

4 Method

A total of 16 soil and water samples have been taken and analysed at Fridhem cemetery since January 2017, see dates and number of analysed replicates in Table 21. The samples were collected at the green and blue marked wells and the influent and effluent wells in connection with the sand filter, see figures in Appendices K and L. However, the compost leachate-well was excluded in one of the sets from 2018-04-16. All samples were refrigerated immediately after sampling.

Table 21. *Samples taken on Fridhem cemetery*

Time	Type	Number	Replicates	Sampled and analysed by
2017-01-25	Water	4	2	ALcontrol
2018-01-22	Water	4	1	ALcontrol
2018-04-16	Water	4	1	ALcontrol
2018-04-16	Water	3	1	Adam Dahlin
2018-04-16	Soil	1	3	Adam Dahlin

The water analysis methods used by ALcontrol are listed in Table 22. The remaining samples and parameters were analysed by the procedures presented in chapter 4.1 and 4.2. Furthermore, the parameters in chapter 4.1 and 4.2 aims to provide support for why the main parameters by ALcontrol attained certain values, and what processes can be expected to take place in the soil and drainage water.

Table 22. *Methods used by ALcontrol for the samples taken in 2017 and 2018.*

Parameter	**2017**	**2018**
TN	SS-EN 12260:2004	SS-EN 12260:2004
TP	SS-EN ISO 15681-2:2005	SS-EN ISO 15681-2:2005
Zn	ISO 17294	ISO 17294
Cu	ISO 17294	ISO 17294
Pb	ISO 17294	ISO 17294
Hg	Merlin	ISO 15587-2, ISO 17852mod
TC1	SS028167-2 MF	SS028167-2 MF
E. *coli*	SS028167-2 MF	SS028167-2 MF

From the commercial laboratory results, removal efficiencies were calculated by dividing the reduction or increase with the concentration before treatment, see equation 4. If concentrations were below the quantification limit, 50 % of the limit value was used. To summarize the removal efficiencies (RE), and to present how they change in between each sampling, the percentages in Table 37 were calculated according to equation 5.

$$\text{Removal efficiency} = (C_{in} - C_{out}) / C_{in} \qquad (4)$$

$$\text{Change in RE} = RE_1 - RE_2 \qquad (5)$$

4.1 Water samples

All water samples were taken with a handpump and poured in polyethylene containers. After 24 hours of storage, conductivity, dissolved oxygen and pH were measured using the electrochemical probe Multi 3630 IDS.

TOC samples were prepared by pouring 50 ml of sample from the drainage water, before sand filter and after sand filter in plastic tubes and stored in freezer. Similar procedures were performed for DOC samples; however, the samples were firstly filtered through sterile syringes with 0.45 µm polyethersulfone membranes. The samples were thereafter stored in a freezer for 8 days before measured by a TOC, DOC and TN analyser.

BOD_7 samples were prepared in sealed glass bottles with no available air space post dissolved oxygen measurements. The samples were stored in a dark space with a temperature of 19–21 °C for seven days, and the concentration of dissolved oxygen was thereafter measured again.

To analyse TSS, 1 µm glass fibre filters were prepared by washing with 20 ml ultrapure deionized water three times before using vacuum suction. The filters were thereafter dried for one hour in 104 °C and weighed. The total suspended solids in 50, 50 and 150 ml of sample from the drainage water, before sand filter and after sand filter, respectively, were filtered through vacuum suction, followed by three successive washes with 10ml ultrapure deionized water. All samples were once more dried for one hour in 104 °C and weighed. After drying, the samples were cooled in a desiccator for approximately 5 minutes. TSS was calculated with equation 6.

$$TSS = (DW_{FS} – DW_F) * 1000 / (V_S / V_U) \qquad (6)$$

TSS - Total suspended solids mg/100 ml
DW_{FS}- Dry weight of filter and sample in grams
DW_F - Dry weight of filter in grams
V_S - Sample volume
V_U - Unit volume

4.2 Soil sample

The soil sample was taken with a shovel at a depth of approximately 1 m in a newly dug grave in the southern area of M3, see Appendix G, and placed in a polyethylene bottle. The only analysed parameter for this sample was TOC, performed in three replicates with the basic LOI method. After three days of storage, all replicates of the sample were dried in 104 °C for 17 hours and thereafter cooled in a desiccator for 5 minutes. The organic carbon fraction of the clay was combusted in 550 °C for 2 hours, followed by 10 minutes cooling in a desiccator. After the final weighing, the organic carbon fraction of the soil was calculated using equation 7.

$$LOI550 = ((DW_{105} – DW_{550}) / DW_{105}) \quad (7)$$

LOI550 - Loss on ignition, TOC
DW105 - Dry weight before combustion
DW505 - Dry weight after combustion

5 Results and discussion

Table 23–37 present all results on water and soil quality at Fridhem cemetery. The values marked in red indicate concentrations which exceed the guideline values set by the Environmental Administration of Gothenburg (Table 1). The tables are organized according to water flow through the stormwater system and the time when the samples were analysed.

The analysis conducted during the first year of operation (2017) showed that the drainage water had very low concentration of all contaminants of interest (Table 23). Total nitrogen was the only parameter slightly above the guideline set by Gothenburg, which is 1.25 mg/l. Similarly, the compost leachate was slightly above the guideline for total phosphorus (0.050 mg/l), and moderately above the guideline for Zn (Table 24). However, after mixing of the drainage and compost water in the conduit junction, nitrogen was the only parameter above the guideline (Table 25). Concentrations of total phosphorus and all investigated metals were higher before reaching the sand filter influent-well, indicating that the sand traps are settling particle-bound pollutants in the stormwater system (Table 23-25). This trend is consistent for the three sampling occasions, thereby strengthening this argument (Tables 23-25, 27-29 and 31-33). Therefore, the importance of cleaning the sand traps should be emphasised, since over time, accumulation of phosphorus and metals can be assumed in these locations. Additionally, such conditions may provide an exemplary environment for microbial growth since there is abundance of organic material and nutrients, and the sand traps could act as a source of bacteria in the stormwater system. If inadequate maintenance and cleaning, intense rain events and high flows in the stormwater system may lead to rapid resuspension of the settled pollutants in the sand traps resulting in severe peak concentrations. Many particles may then reach the sand filter and cause clogging. Regular cleaning of the sand traps will therefore result in an optimized sand filter and prolong its life-time. Furthermore, maintenance of the sand traps in relation to cleaning of the sand filter is a much simpler process due to the sand filter being covered by a 1.1 m thick layer of backfilling masses.

Table 23. *Concentrations found in Fridhem cemetery drainage water 2017-01-25. The sampling temperature was 4.4 °C.*

2017-01-25	**Drainage water**			
Parameter	Replicate 1	Replicate 2	MU[2]	Unit
TN	1.6	1.6	± 0.24	mg/l
TP	0.032	0.032	± 0.0050	mg/l
Zn	11	16	± 3.0	µg/l
Cu	1.5	1.5	± 0.23	µg/l
Pb	0.21	0.21	± 0.19	µg/l
Hg	<5	<5	± 1	ng/l
TC[1]	<10	<10	-	cfu/100 ml
E. *coli*	<10	<10	-	cfu/100 ml

[1]Total Coliform, [2]Measurement uncertainty

Table 24. *Concentrations found in Fridhem cemetery compost leachate 2017-01-25. The sampling temperature was 4.8 °C.*

2017-01-25		**Compost leachate**		
Parameter	Replicate 1	Replicate 2	MU[2]	Unit
TN	0.83	0.83	± 0.83	mg/l
TP	0.061	0.061	± 0.61	mg/l
Zn	98	70	± 15, 11	µg/l
Cu	3.3	2.7	± 0.45, 0.41	µg/l
Pb	13	13	± 2.0	µg/l
Hg	14	14	± 2	ng/l
TC[1]	45	45	-	cfu/100 ml
E. *coli*	<10	<10	-	cfu/100 ml

Table 25. *Concentrations found in Fridhem cemetery sand filter influent 2017-01-25. The sampling temperature was 4 °C.*

2017-01-25		**Sand filter influent**		
Parameter	Replicate 1	Replicate 2	MU[2]	Unit
TN	1.6	1.6	± 0.24	mg/l
TP	0.023	0.023	± 0.0050	mg/l
Zn	5	5	± 3.0	µg/l
Cu	1.2	1.2	± 0.18	µg/l
Pb	<0.2	<0.2	± 0.19	µg/l
Hg	<5	<5	± 1	ng/l
TC[1]	<10	<10	-	cfu/100 ml
E. *coli*	<10	<10	-	cfu/100 ml

Table 26. *Concentrations found in Fridhem cemetery sand filter effluent 2017-01-25. The sampling temperature was 5.7 °C.*

2017-01-25		**Sand filter effluent**		
Parameter	Result	MU[2]	Unit	Removal Efficiency
TN	1.2	± 0.18	mg/l	25%
TP	0.0054	± 0.0050	mg/l	77%
Zn	13	± 3.0	µg/l	-160%
Cu	3.4	± 0.51	µg/l	-183%
Pb	0.43	± 0.19	µg/l	-330%
Hg	<5	± 1	ng/l	0%
TC[1]	<10	-	cfu/100 ml	0%
E. *coli*	<10	-	cfu/100 ml	0%

In general, the performance of the sand filter during the first year of operation was poor. Total nitrogen, which was slightly above the guideline, decreased with 25 % (Table 26). When analysing this result with measurement uncertainty, it is possible that the effluent concentration occasionally even exceeded the guideline value set by Gothenburg. The sand filter proved to have better performance for total phosphorus, 77 % removal. Due to the influent concentration already being below the guideline, the removal performance is surprisingly good considering it is the first year of operation and sand filters are generally governed by a start-up period with lower efficiency. The higher removal of phosphorus in comparison to nitrogen can be explained by the higher reactivity of phosphate, and is therefore more likely to form complexes and precipitates. Phosphorus is therefore most likely abundant in a particulate form, which would correlate with the findings by Massas, Kefalogianni and Chatzipavlidis (2018) and approximately 97 % particulate phosphorus of the total phosphorus in their soil samples. Additionally, there is a clear trend that the effluent metal concentrations are higher compared with the influent, indicating that the sand filter is leaching metals. This occurs in all sampling instances, see the negative removal efficiencies (Tables 26, 30 and 34). The leaching of metals may further enhance the retention of phosphorus as there is an abundance of metals, and possibility for metal-phosphate precipitation. However, the leaching seems to decrease towards the last sampling occasion, performed in April 2018, and the reduction percentage changes to positive values for Pb and Cu (Table 34). It is difficult to conclude if these changes are due to leaching, or if they are normal variations since the water contains low concentrations of metals, and all metals are far below the regional guideline values at the sand filter inlet. However, leaching seems probable due to Pb and Cu cease to increase in April 2018. The removal of phosphorus also consistently increases between the sampling occasions (Table 37). This should not be interpreted as an increase in phosphorus removal potential of the sand filter, but rather an increase of removal due to a higher influent concentration. As stated by the Centre for Watershed Protection, it is more difficult to achieve efficient remediation of clean water. This increase of influent concentration mainly derives from the compost leachate, and the concentration of phosphorus in the drainage water decreases with 50 % from 2017 to the two samples analyses performed in 2018 (Tables 23, 27 and 31). This is unexpected since the burial area is expanding consistently between the sampling occasions, as more graves are being dug. However, total nitrogen in the drainage water increases throughout the two consecutive years, 2017 and 2018 (Tables 23 and 27). Although the concentration is lower in April 2018 compared to January 2018 (Tables 27 and 31). This supports the hypothesis of variations in the drainage water concentrations and may very well be due to irregular maintenance of the sand traps, varying water flows in the conduits and periodical flushing of the grave soil. To be able to understand the development of drainage water concentrations further, more samples would need to be analysed in correlation with further interments.

Table 27. *Concentrations found in Fridhem cemetery drainage water 2018-01-22. The sampling temperature was 4.1 °C.*

2018-01-22	**Drainage water**		
Parameter	Result	MU^2	Unit
TN	2.3	± 0.35	mg/l
TP	0.016	±0.0050	mg/l
Zn	<3	± 3.5	µg/l
Cu	2.2	± 0.33	µg/l
Pb	<0.2	± 0.19	µg/l
Hg	<5	± 1	ng/l
TC^1	18	-	cfu/100 ml
E. *coli*	<10	-	cfu/100 ml

Table 28. *Concentrations found in Fridhem cemetery compost leachate 2018-01-22. The sampling temperature was 3.5 °C.*

2018-01-22	**Compost leachate**		
Parameter	Result	MU²	Unit
TN	2.9	± 0.44	mg/l
TP	2.5	± 0.25	mg/l
Zn	18	± 3.5	µg/l
Cu	1.8	± 0.27	µg/l
Pb	0.8	± 0.19	µg/l
Hg	<5	± 1	ng/l
TC¹	420	-	cfu/100 ml
E. *coli*	18	-	cfu/100 ml

Table 29. *Concentrations found in Fridhem cemetery sand filter influent 2018-01-22. The sampling temperature was 4.3 °C.*

2018-01-22	**Sand filter influent**		
Parameter	Result	MU²	Unit
TN	2.5	± 0.38	mg/l
TP	0.15	± 0.015	mg/l
Zn	<3	± 3.5	µg/l
Cu	1.3	± 0.20	µg/l
Pb	<0.2	± 0.19	µg/l
Hg	<5	± 1	ng/l
TC¹	<10	-	cfu/100 ml
E. *coli*	<10	-	cfu/100 ml

Table 30. *Concentrations found in Fridhem cemetery sand filter effluent 2018-01-22. The sampling temperature was 6.1 °C.*

2018-01-22	**Sand filter effluent**			
Parameter	Result	MU²	Unit	Removal Efficiency
TN	0.97	± 0.15	mg/l	61%
TP	0.0088	± 0.0050	mg/l	94%
Zn	7.7	± 3.5	µg/l	-413%
Cu	3.7	± 0.56	µg/l	-185%
Pb	<0.2	± 0.19	µg/l	0%
Hg	<5	± 1	ng/l	0%
TC¹	<10	-	cfu/100 ml	0%
E. *coli*	<10	-	cfu/100 ml	0%

Throughout the three sampling occasions, the only parameter of interest in the drainage water, in terms of exceeded guideline values, is nitrogen (Tables 23, 27 and 31). Since nitrogen in the burial area mainly derives from organic nitrogen mineralization and NH_4^+ production (Dent, Forbes and Stuart, 2004), it can be assumed that NH_4^+ is the most dominant nitrogen pool in the early stages of decomposition in the burial soil. However, due to a lowered water table and aerated decomposition, it is likely that NH_4^+ is partially nitrified to nitrate before flushed from the soil and drained in the conduits. Furthermore, the drainage water is governed by almost saturated dissolved oxygen conditions (Table 36) and could prompt further nitrification. The removal efficiency may therefore be improved by introducing complete or partially anoxic conditions in the sand filter. This can be accomplished by adjusting the effluent height to a certain elevation in relation to the sand filter media. If the bottom half of the sand filter is at constant saturation, anoxic microsites will be more dominant in comparison to an aerated pore volume. This could benefit adsorption of NH_4^+ and NO_3^- through ion exchange in combination with nitrification before reaching the anoxic microsites in the bottom half of the filter. The anoxic microsites would thereafter enhance the denitrification and reduce nitrate. The BOD results (Table 36) also indicates that there is a moderate amount of organic material that could act as electron donors for this microbial process. Additionally, as shown by the investigation by Fiedler, Dame and Graw (2018), several studies have proven that nitrogen seem to mainly exist as nitrate in both cemetery drainage water and groundwater in burial areas. However, due to the very low abundance of indicator organisms in all samples (Tables 23-34), it is unknown how this process would behave, and it is possible that the cultivation of bacteria in the filter media is very low, which consequently would result in both restricted nitrification and denitrification in the drainage water and sand filter, since microbial activity is limited. Generally, throughout all sample analyses, the nitrogen reduction efficiency in the sand filter is only good when the compost is contributing to higher influent concentrations, which is demonstrated by the results from April 2018 and an 84 % reduction (Table 34). This clearly shows the difficulty in reducing nitrogen concentrations that initially are very low, and it is under more polluted conditions that effective remediation mostly occur (Center for Watershed Protection, 2007). This is mainly due to nitrogen species not forming strong bonds compared to for example phosphorus, where the primary adsorption mechanisms are ion exchange (Groeschke *et al.*, 2017), or electrostatic attraction, most likely with colloids and minerals. Additionally, NH_4^+ and NO_3^- possess a relatively weak charge, and for filter media that periodically saturates and dries out, resuspension and remobilization may be a crucial factor of nitrogen retention (Grebel *et al.*, 2013).

The burial areas M3 and M6 (Figure 1 in Appendix G) are not at full capacity, 83 and 67 %, respectively, and more graves will be dug in the near future. Therefore, the concentration of most contaminants can be assumed to face an increase. However, when analysing the reduction of phosphorus in the sampling performed in 2017 and 2018, the retention of phosphorus appears to be very strong in the grave soil in these areas. Especially considering the increase in graves between the sampling occasions. A logical conclusion of these results could be that the soil retention limits the phosphorus concentration in the drainage water to this approximate level, which is below the guideline value set by Gothenburg, and more interments will, due to stabilized conditions, not result in a higher phosphorus concentration. However, for nitrogen, dilution with uncontaminated water from pristine areas in these zones seems to be a factor since the concentration increases from 2017 to 2018. It should be noted that M6 is a burial area for children, which subsequently results in smaller cadavers and possibly less contaminated grave soil. The extent to which the nitrogen concentration may increase under full capacity of M3 and M6 is difficult to quantify. However, when applying the interment capacity as multiplying factors for the drainage water concentration, it seems unlikely that the total nitrogen concentration will ever exceed 3 mg/l. Furthermore, it is important to emphasize that the current operation of the stormwater system includes drainage water from Kv41 and Kv43 (Figure 1 in Appendix E). Very few graves have been dug in these two areas, and drainage water of good quality can be assumed to dilute the concentrations found in the drainage water from M3, M6 and the compost leachate, before reaching the sand filter media. Therefore, the influent concentrations should be interpreted with cation, and it is not certain if the influent concentration is a result of strictly drainage water and compost leachate. This is also assumed to significantly influence the concentration reduction for metals from the drainage water and compost leachate to the influent well and is not only governed by settling of particles in the sand traps as mentioned previously. An approximation of the dilution factor for the investigated parameters could have been calculated with estimated water flows from a hydrogeological investigation. Unfortunately, estimated flows are only available for M3and M6, and the additional flows from the compost and Kv41 and Kv43 are unknown. Even if a dilution factor could be estimated, it is uncertain if they apply for these three sets of samples. No samples were collected during a rain event, and M3, M6 and the compost is located further away from the sand filter in relation to Kv41 and Kv43. This means that peak flows from each area will reach the sand filter at different times, and without a geological investigation of Kv41 and Kv43, infiltration rates to the drainage system cannot be estimated. For example, if Kv41 and Kv43 are governed by more permeable soils, the entire stormwater volume from these areas can be assumed to reach the sand filter before M3 and M6 have been completely drained.

In a future scenario, when M3, M6, Kv41 and Kv43 are at full capacity, the drainage water will likely have higher concentrations of nitrogen, phosphorus, and possibly, a higher count of indicator organisms. The metal concentration will probably continue being far below the guideline due to the very low concentrations today, and the absence of microbes in the drainage water from M3 and M6 indicates that the concentration of indicator organisms will unlikely exceed any limits that should be applied by Gothenburg (see chapter 2.1.3), even though an increase is expected. The graves in Kv41 and Kv43 will have two to three coffins in each grave, and depending on what soil type is dominant in these two areas, higher concentrations of nitrogen and phosphorus are to be expected in comparison with M3 and M6. With similar soil conditions, it is possible that the concentration slightly exceeds the guideline value set by Gothenburg for phosphorus, while nitrogen risks a more significant increase.

Table 31. *Concentrations found in Fridhem cemetery drainage water 2018-04-16. The sampling temperature was 5.3 °C.*

2018-04-16	**Drainage water**		
Parameter	Result	MU[2]	Unit
TN	2	± 0.30	mg/l
TP	0.018	±0.0050	mg/l
Zn	7.3	± 3.5	µg/l
Cu	2.3	± 0.35	µg/l
Pb	0.28	± 0.19	µg/l
Hg	<5	± 1	ng/l
TC[1]	<10	-	cfu/100 ml
E. *coli*	<10	-	cfu/100 ml

Table 32. *Concentrations found in Fridhem cemetery compost leachate 2018-04-16. The sampling temperature was 5.5 °C.*

2018-04-16	**Compost leachate**		
Parameter	Result	MU[2]	Unit
TN	7.6	± 1.1	mg/l
TP	1.2	± 0.12	mg/l
Zn	43	± 6.5	µg/l
Cu	6.2	± 0.93	µg/l
Pb	0.58	± 0.19	µg/l
Hg	7	± 1	ng/l
TC[1]	270	-	cfu/100 ml
E. *coli*	<10	-	cfu/100 ml

Table 33. *Concentrations found in Fridhem cemetery sand filter influent 2018-04-16. The sampling temperature was 3.7 °C.*

2018-04-16	**Sand filter influent**		
Parameter	Result	MU[2]	Unit
TN	4.5	± 0.67	mg/l
TP	0.22	± 0.022	mg/l
Zn	5.8	± 3.5	µg/l
Cu	2.1	± 0.32	µg/l
Pb	0.44	± 0.19	µg/l
Hg	<5	± 1	ng/l
TC[1]	55	-	cfu/100 ml
E. *coli*	<10	-	cfu/100 ml

Table 34. *Concentrations found in Fridhem cemetery sand filter effluent 2018-04-16. The sampling temperature was 5.4 °C.*

2018-04-16		**Sand filter effluent**		
Parameter	Result	MU[2]	Unit	Removal Efficiency
TN	0.72	± 0.15	mg/l	84%
TP	0.0085	± 0.0050	mg/l	96%
Zn	9.5	± 3.5	µg/l	-64%
Cu	1.7	± 0.26	µg/l	19%
Pb	<0.2	± 0.19	µg/l	77%
Hg	<5	± 1	ng/l	0%
TC[1]	<10	-	cfu/100 ml	91%
E. *coli*	<10	-	cfu/100 ml	0%

The graves are dug to approximately 1.6 m depth in M3 and M6, due to only one coffin being placed in each grave. The soil between 1 m and the depth of the drainage system will therefore significantly influence the retention of contaminants in the burial area. Table 35 shows that the clay at approximately 1 m depth contain a moderate amount of organic material. The organic material, together with an abundance of small clay mineral particles and large surface area, is an important factor that contributes to significant retention (Ahammed and Meera, 2010; Uddin, 2017). Surprisingly, only one sample indicated any indicator organisms in the drainage water (Table 27), meaning that the bacteria must be retained in the soil and coffin. The measured low concentration of phosphorus in the drainage water may therefore be partially linked with microbial immobilization, which together with adsorption mechanisms are retaining factors in the soil (Achat *et al.*, 2016; Wang *et al.*, 2016).

The drainage water proved to contain a very low concentration of total suspended solids (Table 36) and is far below the regional guideline of 25 mg/l. Suspended solids are partially the reason behind clogging of filters, since the particles eventually will form a layer on top of the sand. Fortunately, the risk of this happening is probably low for the life-span of a normal sand filter, approximately 15–20 years under well-managed conditions according to the Gothenburg environmental administration, if sand traps are regularly cleaned and low influent concentration continue. Considering that Kv41 and Kv43 are yet to be occupied, it seems reasonable that the retention capacity is kept moderately high for a long time as well.

Table 35. *Total organic carbon in the clay layer at 1 m depth in area M3. Sampling occurred 2018-04-16.*

2018-04-16	**Soil**		
Replicate	1	2	3
TOC	4.67%	5.36%	4.71%

Table 36. *Water quality results at Fridhem cemetery 2018-04-16 (Chalmers laboratory).*

2018-04-16

Parameter	Drainage water	Influent	Effluent	Unit	Removal Efficiency
pH	7.6	7.9	7.4	-	-
Conductivity	762.0	535.0	743.0	µS/m	-
DO	10.5	10.2	10.9	mg/l	-
BOD_7	4.6	5.9	1.7	mg/l	71%
TSS	1.6	1.2	0.1	mg/100 ml	89%

Table 37. *Performance difference of the sand filter between the analyses in 2017 and January 2018, and January 2018 and April 2018.*

Parameter	2017 – 2018-01	2018-01 – 2018-04
TN	36%	23%
TP	18%	2%
Zn	-253%	350%
Cu	-1%	204%
Pb	330%	77%
Hg	0%	0%
TC1	0%	91%
E. *coli*	0%	0%

Treatment of the stormwater at Fridhem cemetery is necessary. However, the main contamination source is currently not the burial areas, rather the compost leachate which produces most of the nutrient load and occasionally zinc concentrations above the regional guideline. If the drainage and leachate were operated separately, it could be argued that the drainage water from M3 and M6 is not in need of remediation. According to the Swedish environmental code, an assessment should be conducted, weighing the environmental impact of the potential pollution, the cost of implementing a remediation system and the environmental or social alleviation it brings. If it is assumed that the nitrogen level at full interment capacity is increased to 3 mg/l, the drainage water will exceed the Gothenburg guideline with an annual loading of approximately 4.6 kg. This is based on the estimated annual average drainage flow from M3 and M6 according to the investigation performed by Norconsult, 5–6 l/min (Norconsult, 2015). If the concentration instead stabilizes at 2–2.3 mg/l (reason being a reduced nitrogen loading in the drainage water after additional interments between January and April 2018), the burial area will only produce an excess annual loading of approximately 2 kg. Such low annual loading may seem insignificant enough that construction of a remediation station is unreasonable, especially when considering that lower influent concentrations result in less efficient removal performance and that the sand filter in this scenario contributes to elevated metal concentrations. However, this depends on the sensitivity of the environment and receiving water. Construction of a remediation station is seemingly more important when there is a risk of leaching additional contaminants, such as phosphorus and possibly metals. Therefore, it is crucial that a comprehensive analysis investigates the retention capability of the grave soil in question, specifically, the depth between coffin elevation and the drainage conduits. Soils governed by high surface area, clay minerals, organic material and high Ca, Al and Fe content, may provide exemplary conditions for retaining phosphorus, metals and microbes, and to a lesser extent, nitrogen. Additionally, environmental and social risks play a vital role in whether remediation is deemed to be necessary. If for example drinking water wells, recreational waters, ground water protection areas or a creek governed by active wildlife is located nearby, risk should be emphasized additionally, and remediation may be necessary even if high retention potential applies. Even if the water table is below coffin elevation, and aerated decomposition is possible, drainage and remediation may be required, especially if poor soil conditions prevail and there is an environmental or social risk. This is mainly due to the groundwater's potential to transport contaminants, for example in porous soil conditions. In such a scenario, distance to sensitive areas should also be considered. If any uncertainties arise regarding retention of contaminants, it could prove to be successful to construct the drainage system at larger depths to increase the thickness of the natural filter. By doing so, significantly more contaminants can potentially be retained (depending on soil type), and the additional costs from managing a higher water flow, bigger conduits and pumping of larger water volumes due to increased groundwater recharge, will most likely be compensated by avoiding a construction investment of a remediation station. Stormwater systems that can utilise gravity fall might instead be forced to construct pressurized conduits due to lower elevation of the system, this should be considered. Furthermore, if soils with very low hydraulic conductivity are a concern, an evaluation of the risk of water-accumulation in the upper soil layers should be conducted. It may be necessary to construct surface drainage to complement the system.

It should also be mentioned that during sampling, it was difficult to retrieve clean water from the wells located before the sand filter, especially in the drainage water and compost leachate wells. The very low water flow, consequently, lead to low water levels in the wells, and over time, a high contaminant loading had accumulated in the sand traps. When pumping water to the polyethylene bottles, some resuspension occurred, and particles ended up in the samples. This may be a significant source of all the investigated contaminants. Therefore, it is possible that the measured values pre-treatment, are higher than the results would show if the sand traps were regularly cleaned.

Due to relatively low flow and concentrations of all investigated parameters in the sand filter influent, utilising other remediation methods such as wet ponds or wetlands can result in worse performance. This is due to for example phytoremediation in wetlands not performing efficiently all year due to the seasonal climate variations (Malaviya and Singh, 2012) and cold winters in Sweden, low concentration of suspended solids resulting in low settling potential of adsorbed contaminants and microbial background concentrations in wetlands may act as a source (Kadlec, 2009). However, since phosphorus likely exists mainly in a particulate form, settling of particles and phosphorus removal could prove to be moderate in these systems as well. The surrounding forest also provides a risk of interaction with wildlife and could jeopardize the integrity of these treatment system. Several studies showed that ponds and wetlands had a neutral or negative impact when nitrogen, phosphorus and bacteria concentrations were low in the influent (Beutel, 2012; Chrétien *et al.*, 2016; Hartshorn *et al.*, 2016). Furthermore, using for example iron oxide coatings on the filter media to increase retention potential is not necessary since all sampling results validate that all contaminants are below the regional guidelines. However, it should be emphasised that sand filters should not become a generalised solution for cemetery drainage water simply because it seems to work within most contexts and environmental parameters.

Instead, the area, environment and retention capability of the grave soil should be considered to roughly estimate the contamination risk and develop customized, innovative and forward-thinking solutions from this information

Finally, since there is a lack of information about drainage water quality in correlation with soil characteristics, it is recommended that the Gothenburg cemetery administration cooperate with other municipalities and develop a database for this specific purpose. With time, estimating water quality depending on soil type will be easier and can potentially save money for organisations in the future.

6 Conclusions and recommendations

A clear trend could be identified from the analysed samples: the drainage water is currently governed by low concentrations of all investigated contaminants (nitrogen, phosphorus, metals and microorganisms). However, the concentration of nitrogen slightly exceeds the guideline value in Gothenburg. The cemetery is currently not operated at full capacity, and after additional coffin burials, especially at the areas Kv41 and Kv43, higher concentrations can be assumed for nitrogen since this nutrient seems to correlate strongest with burial capacity in the investigated grave fields. The highest contaminant load originated from the compost leachate, including nutrients, Zn and microorganisms. However, uncertainties regarding sampling resuspension and drainage water dilution questions the integrity of the drainage water concentrations and the sand filter performance.

The sand filter proved to be appropriate for the stormwater system at Fridhem cemetery. Wet ponds and wetlands may have a fluctuating performance or a negative impact on the relatively low influent concentrations, for example nitrogen, phosphorus and microbes. Additional retention capability from for example iron oxide coating is not deemed necessary since the effluent never exceeds the regional water quality guidelines. The sand filter is expected to fulfil the 15-20 years life-time, mainly due to the low influent concentration, and maintenance of the sand filter is probably not a requirement in the near future if sand traps in the wells are cleaned regularly. The sand traps may currently be acting as a source of contaminants as resuspended particles may be flushed from the traps to the sand filter. Although the increase of metals at the filter outlet probably is a consequence of filter media washing, it could partially be normal variations in the stormwater quality as very low concentrations were detected in all samples. The filter proved to perform better at higher influent concentrations of pollutants.

If the drainage water from M3 and M6 is managed separately from the compost leachate, remediation might not be necessary. The annual nitrogen load is currently very low and is not expected to increase significantly in the future. For practices with similar conditions, arguments for low contaminant concentrations, high investment cost and an environmental and social risk analysis should be presented to the environmental administration regarding the Swedish environmental code, chapter 2, *7§*.

For future drainage water management practices, it is recommended to investigate retention capability and transportation risk of the grave soil, including soil texture, hydraulic conductivity, organic material, Ca, Al and Fe. Additionally, sand filters should not routinely be the only solution for remediation of drainage water practices, as other practices may be more suitable.

7 References

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8 Appendices

Appendix A

Table 1. *Bacteria commonly found in human intestines. The table was retrieved from WHO (1998).*

Families and genera	Prominent species	Species isolated from the intestine
Pseudomonadaceae		*Pseudomonas aeruginosa (pyocyanea)*
Pseudomonas		*Ps. (Alkaligenes) faecalis*
Enterobacteriaceae	*Escherichia coli*	
Klebsiella		*Klebsiella (Aerobacter) pneumoniae*
Enterobacteriaceae		*Enterobacter (Aerobacter) aerogenes*
Proteus		*Proteus mirabilis*
Bacteroidaceae		*Bacteroides capillosus. B. oralis*
Bacteroides	*Bacteroides fragilis*	*B. clostridiformis. B. putredinis*
		B. coagulans. B. ruminicola
Fusabacterium		*Fusobacterium mortiferum*
		F. necrogenes. F. fusiforme
		F. girans
Neisseriaceae		*Neisseria catarrhalis*
Neisseriaceae		*Veillonella parvula*
Veillonella		*V. alcalescens*
Micrococcacene		*Staphylococcus albus*
Staphylococcus		*Peptococcus asaccharolyticus*
Acidaminococcus		*Sarcina centriculi*
Sarcina		*Acidaminococcus fermentans*
Peptococcus		*Steptococcus salivarius*
Streptococcaceae		
Streptococcus	*Streptococcus faecalis*	*Strep. sangius*
		Strep. viridans (mitior)
		Strep. faecium
Lactobacillacene		*Lactobacillus brevis*
Lactobacillus	*Lactobacillus acidophilus*	*L. casei*
		L. catenaforme. L. fermentum
		L. leichmanii. L. plantarum
Leptotrichia		*Leptrotrichia buccalis*
Bifidobacterium	*Bifidobacterium adolecentis*	*Bifidobacterium (Actinomyces*
	Bifidobacterium longum	*lactobacillu) bifum (bifidus)*
		Bif. Breve. Bif. Cornatum
		Bif. Eriksonii. Bif. Infantis
Ruminoccus	*Ruminococcus bromii*	*Peptostreptoccus intermedius*
Peptostreptococcus		*P. productus*

Table 1. *(Continued) Bacteria commonly found in human intestines. The table was retrieved from WHO (1998).*

Families and genera	Prominent species	Species isolated from the intestine
Propionobacteriacene		
Propionobacterium		*Propionobacterium*
		(Corynebacterium) acnes
Eubacterium	*Eubacterium (Bacteroides)*	*Prop. Granulosum*
	aerofaciens (biforme)	*Eubacterium contortum*
		Eu. Cylinderoides. Eu. Lentum
		Eu.limpsum. Eu. Rectale
		Eu. tortuosum. Eu. ventriosum
Corynebacteriaceae		*Corynebacterium pseudo-*
Corynebacterium		*diphtheriticum (hojmanni)*
		C. xerosis. C. ulcerans
Bacillaceae		*Bacillus cereus. B. subtilis*
Bacillus		*Clostridium cadaveris*
		Cl. innocuum
Clostridium	*Clostridium perfringens*	*Cl. maienominatum. Cl. ramosum*
	(welchii)	*Cl. sordellii*
	Clostridium paraputrificum	*Cl. certium. Cl. bifermentans*
		Cl. sporogenes. Cl. indolis
		Cl. sphenoides. Cl. feisineum
		Cl. difficile. Cl. oroticum

Appendix B

Table 1. *Summary of microbial findings in six cemeteries. All values in Vila Nova, Vila Formosa and Areia Branca are retrieved from Pacheco et al. (1991) and the values in Luz de Tavira and Seixas are retrieved from Rodrigues and Pacheco (2003).*

Country	Cemetery		CT[1]	CF[2]	SF[3]	CSR[4]	PROT[5]	CPH[6]	CPH[7]	LIP[8]
Brazil	Vila Nova Cachoeirinha (Mean)		163	2	8	7	1018	10821	16383	3942
	Vila Formosa (Mean)		14	3	8	14	268	9018	675	2520
	Areia Branca (Mean)		58	5	55	21	431	14699	4049	6433
Portugal	Luz de Tavira	Inside (Min-Max)	3-1850	1- 121	0 -11	23- 1100	0 -90	27-365		-
		Outside (Min-Max)	0- 595	0 -60	0 - 7	0 -48	2 -90	1- 293		-
	Seixas	Inside (Min-Max)	193-3900	0-4400	0- 580	4-4600	-	5-4800		-
		Outside	49	4	0	4	-	53		-

[1] Total Coliform, [2] Feacal Coliform, [3] Feacal Streptococci, [4] Sulfite Reducer Clostridia, [5] Proteolytic Bacteria, [6] Heterotrophic Aerobic Bacteria, [7] Heterotrophic Anaerobic Bacteria, [8] Lipolytic Bacteria.

CT, CF, SF, CSR and PROT are presented as MPN/100 ml, CPH and LIP are presented as CFU/100 ml.

Appendix C

Table 1. *Characteristics of the two investigated wetlands by Maine et al. (2017).*

	Wetland 1	**Wetland 2**
Size (with central separator)	50 * 40 m	20 * 7 m
Soil thickness	1 m	1.5 m
Water depth	50 cm	50 cm
Retention time	>7 days	>7 days
Sampling	Monthly (2002-2015)	Monthly (2009-2015)
Species	Floating and emergent	Emergent
	Typha domingensis	*Typha domingensis*
	Pistia stratiotes	
	Eichhornia crassipes	
	Salvinia rotundifolia	
	Panicum elephantipes	
	Pontederia cordata	
	Hydrocotyle ranunculoides	

Appendix D

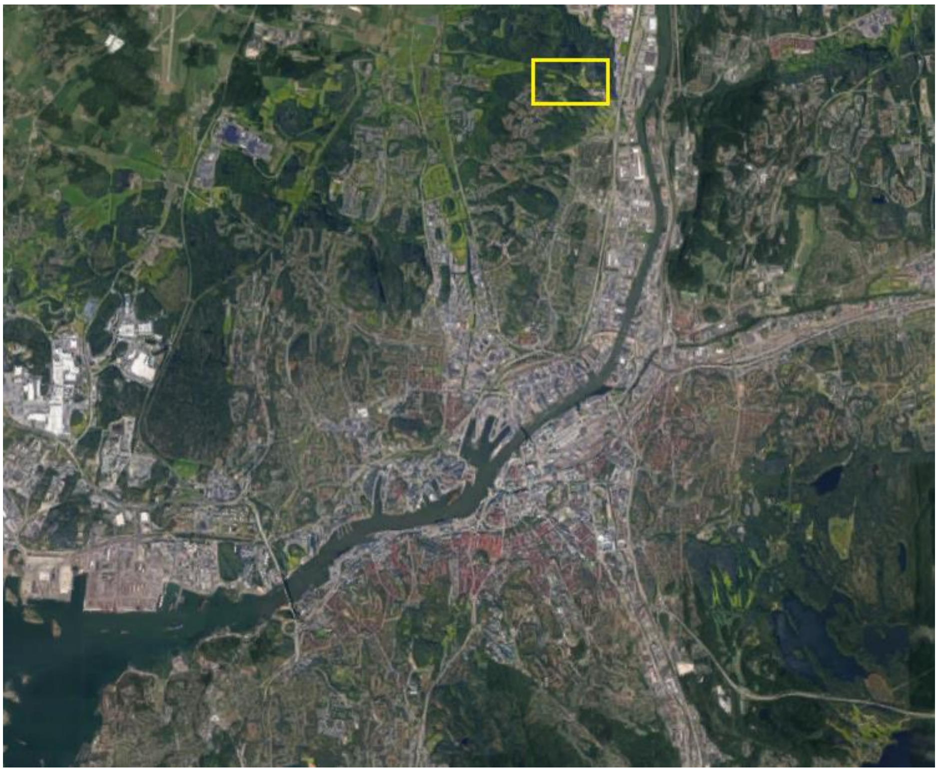

Figure 1. *Location of Fridhem cemetery.* ©2018 Google

Figure 2. *Location of the non-cremated burial area.* ©2018 Google

Appendix E

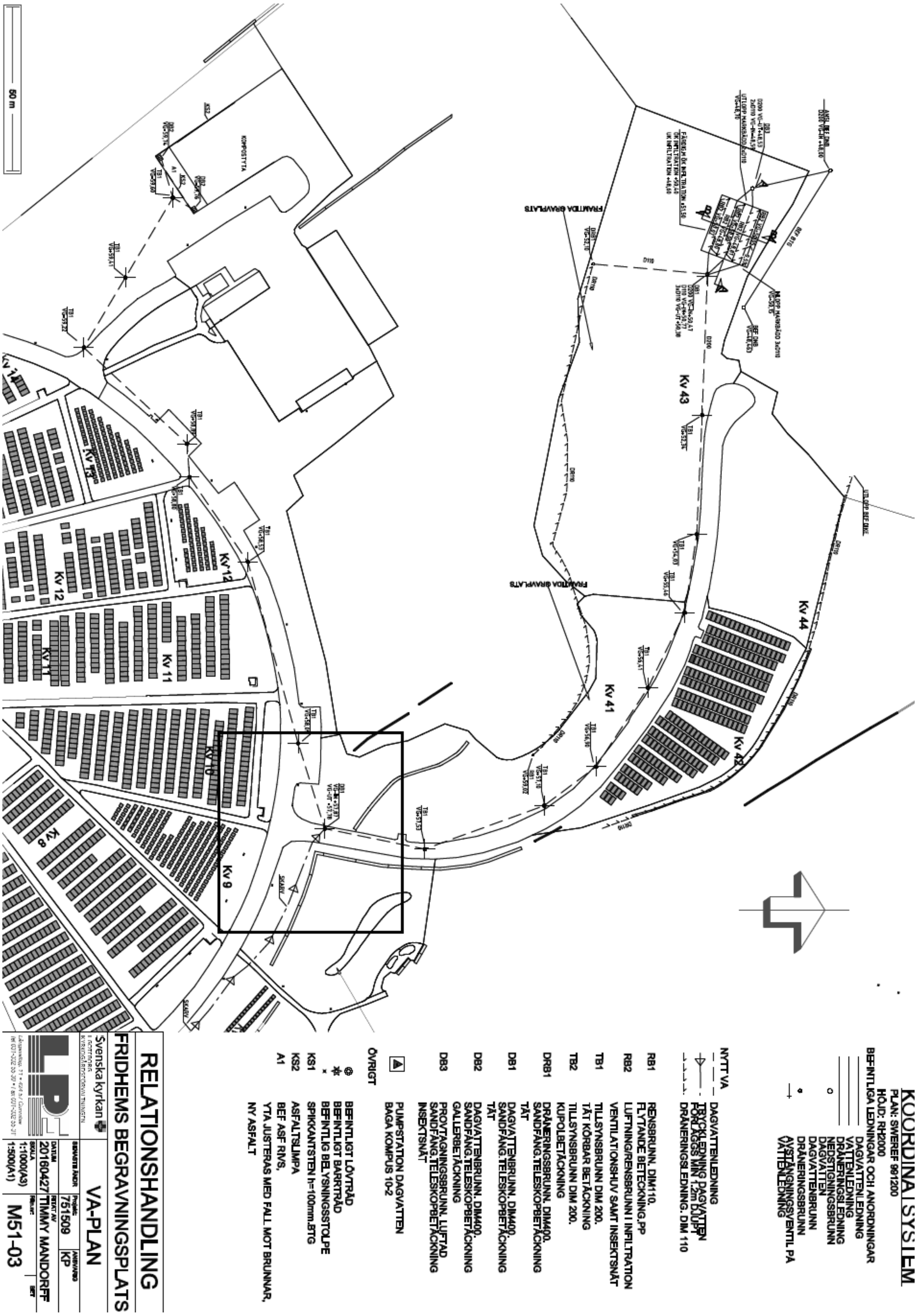

Figure 1. *Mapping of the grave soil and compost leachate conduits and sand filter. The sand filter can be seen in the north-west corner of the map. A detailed mapping of the drainage water and compost leachate intersection, area in the black box, can be seen in Figure 2.*

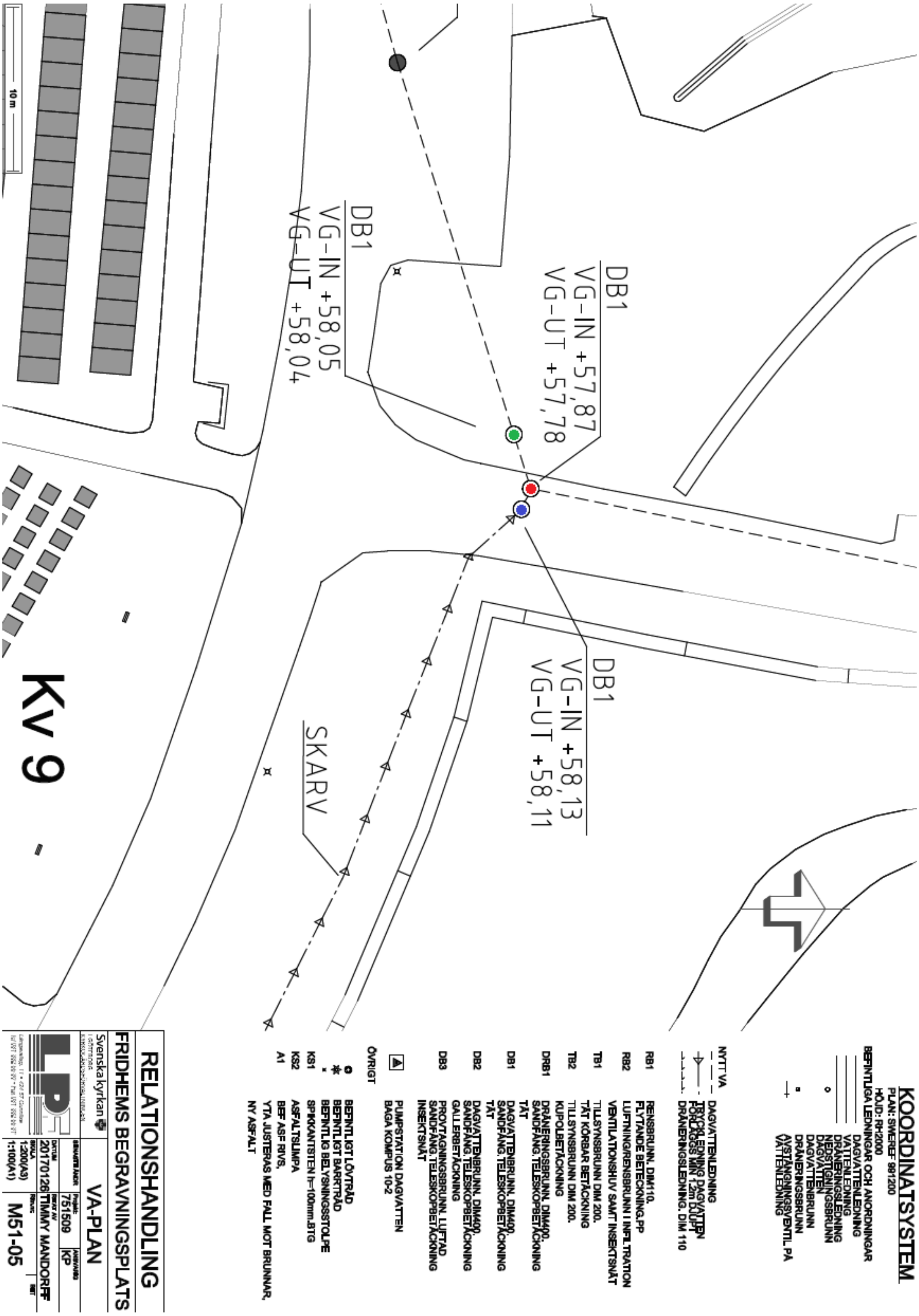

Figure 2. *Mapping of the drainage water and compost leachate intersection. The dashed line is a gravity stormwater conduit and the arrow dashed line is a pressurized conduit. The compost leachate attaches from the west and the drainage water from the east. The green, red and blue dots marks situated stormwater wells with sand traps for compost leachate, mixed water and drainage water, respectively.*

Appendix F

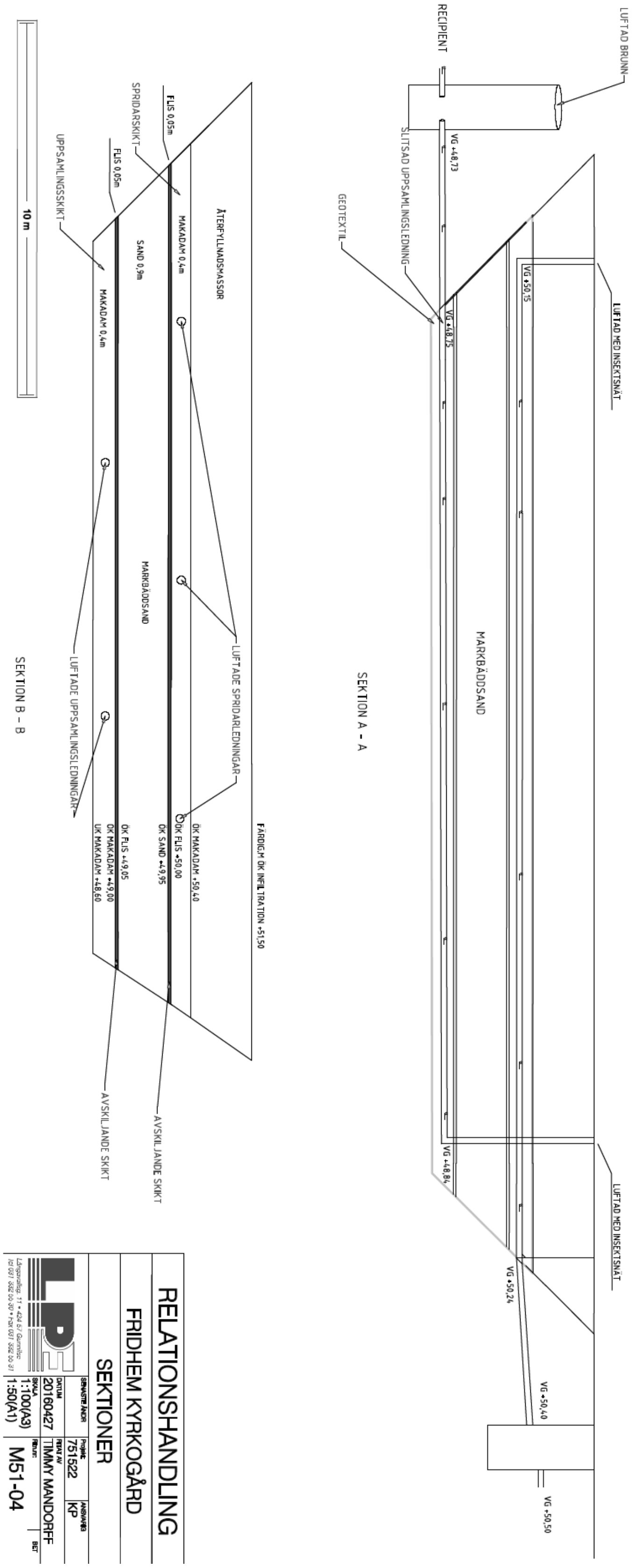

Figure 1. *Design of the sand filter at Fridhem cemetery. The angles of the cross-sections can be seen in Figure 3.*

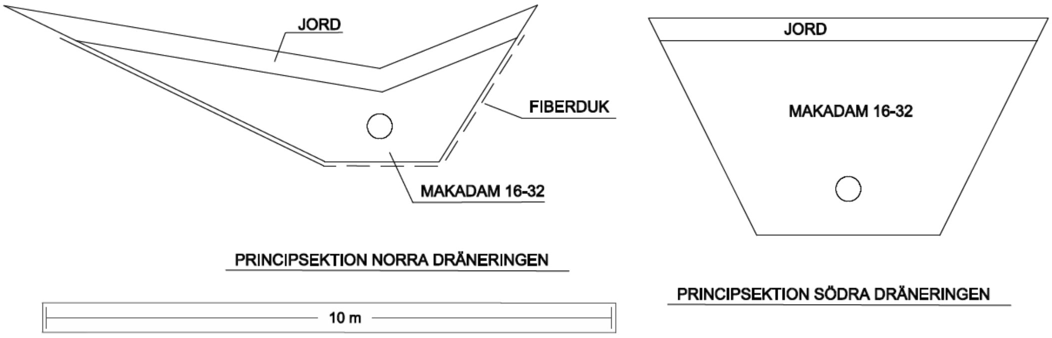

Figure 2. *Design of the drainage system at Fridhem cemetery. The cross-sections in this figure were originally drawn together with the cross-sections in Figure 1.*

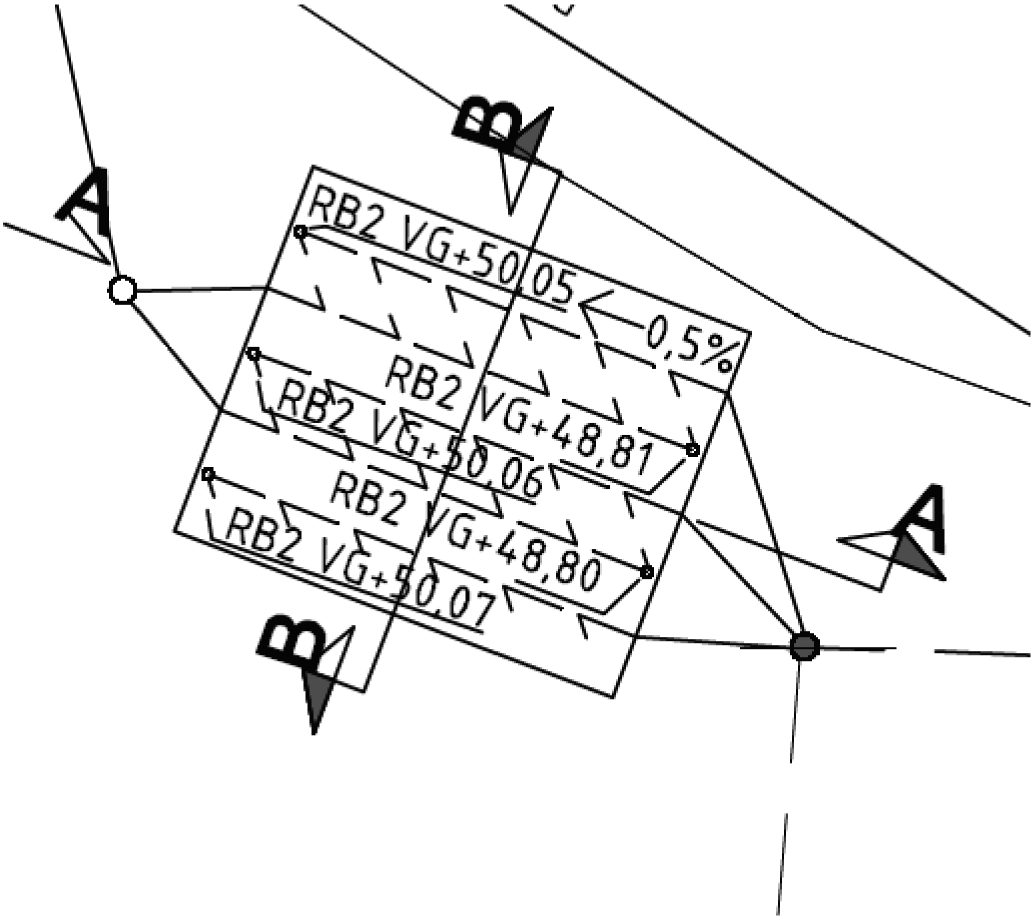

Figure 3. *Close-up of the sand filter from Appendix E Figure 1. Three conduits distribute the water evenly on the filter bed, and two conduits collected the water after filtration. A-A and B-B are the presented cross-sections in Figure 1.*

Appendix G

Table 1. *Summary of the geotechnical investigation by GF Konsult AB. The table includes 19 sampling pits (PG 1-19) spread over the burial area. The columns indicate at which depth the different soil types were found, in meters. PG 1-14 is located in the southern parts of the burial area, and the remainder in the central parts.*

PG1	0-0.3	0.3-1.5	1.5-1.8	Comment
	mu/gr sand with br	gr clay	mu/sa clay	-
PG2	0-0.4	0.4-1.0*	1.0-2.0	Comment
	mu/gr sand with residue	clay with wr		Moderate amount of rock
PG3	0-0.4	0.4-2.0*		Comment
	mu/gr sand with br	gr clay		Some rock
PG4	0-0.8	0.8-2.0		Comment
	gr sand with br	gr clay		Much brick
PG5	0-0.8	0.8-2.0		Comment
	gr sand with br/wr	gr clay with br		Much brick
PG6	0-1.0	1.0-2.0		Comment
	mu/sa clay	gr/sa clay with br		-
PG7	0-1.0	1.0-2.0		Comment
	mu/sa clay	clay with br/wr		Water table at 1.6 m depth
PG8	0-1.0	1.0-2.0		Comment
	mu/sa clay	gr/sa clay		-
PG9	0-1.2	1.2-2.0		Comment
	mu/sa clay	clay		-
PG10	0-1.7	1.7-2.0		Comment
	mu/sa clay	clay		-
PG11	0-0.4	0.4-0.8	0.8-2.0	Comment
	mu clay	sa clay with br	sa clay	-
PG12	0-1.1	1.1-1.5		Comment
	br	clay		-
PG13	0-1.0	1.0-1.3		Comment
	br	clay		wood and metal scraps
PG14	0-0.4	0.4-0.5	0.5-2.2	Comment
	sa gravel with br	sand	clay	-
PG15	0-1.3	1.3-1.8		Comment
	clay	mu/sa clay		-
PG16	0-0.8	0.8-2.2		Comment
	gr clay with br	clay		-
PG17	0-0.3	0.3-2.0		Comment
	mu/sa claywith br	clay		-
PG18	0-0.3	0.3-2.0		Comment
	mu/sa clay with br	clay		-
PG19	0-0.8	0.8-2.0		Comment
	mu/sa clay	clay		-

mu – mull (humus/organic material), gr – gravelly, sa – sandy, br – brick residue, wr – wood residue
*Blocks of rock

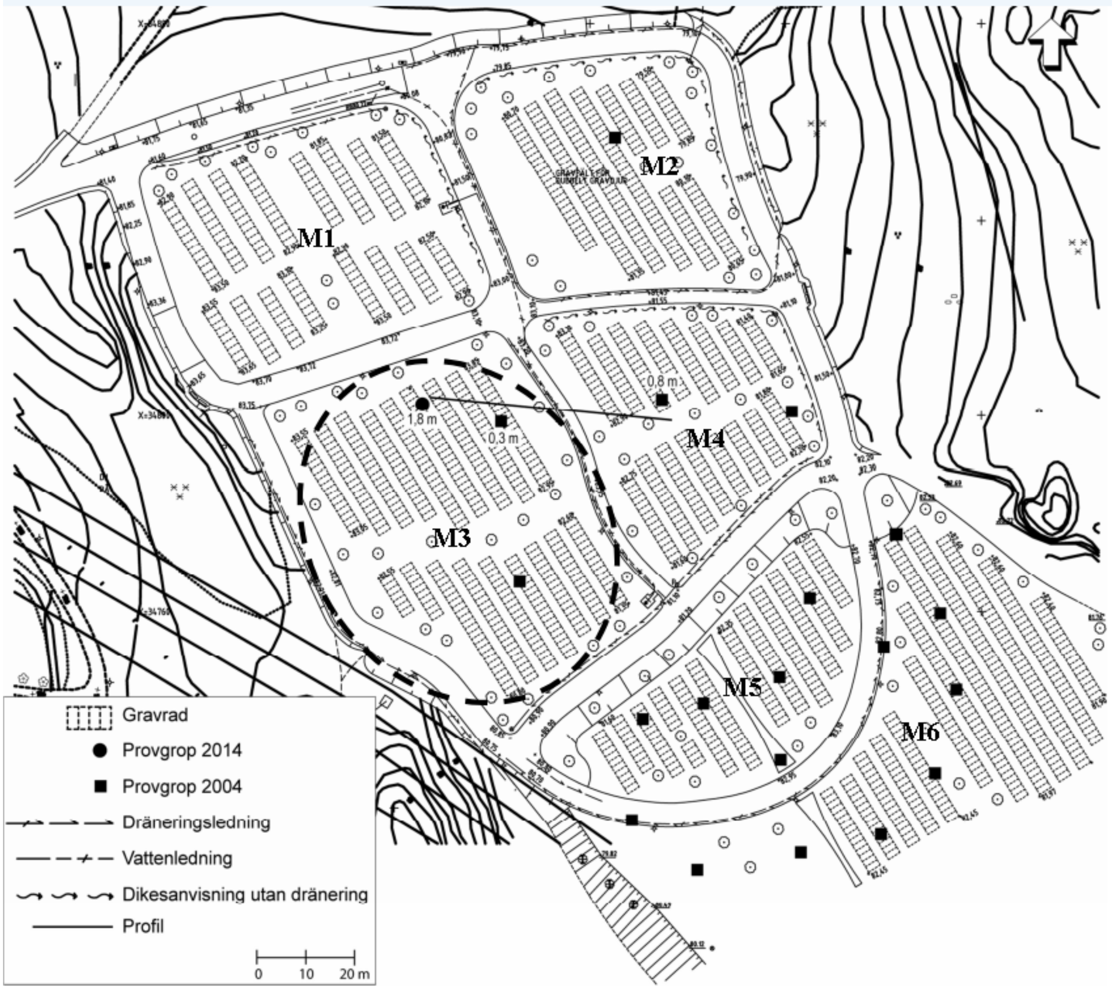

Figure 1. *Map of the burial area and investigation zones M1-M6. The area marked by the black ring is most exposed to water saturation. The black squares are the sampling pit locations listed in Table 1.*

Figure 2. *Location of the installed drainage conduits (pink lines) and display of occupied graves during installation, marked by dark grey coffins.*