

MASTER'S THESIS ACEX30

Removal of Metals from Stormwater in Rain Gardens with Different Sorption Materials

Master's Thesis in Infrastructure and Environmental Engineering

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Department of Architecture and Civil Engineering

Division of Water Environment Technology

Urban water and contaminated materials

CHALMERS UNIVERSITY OF TECHNOLOGY

Göteborg, Sweden 2023

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ABSTRACT

Polluted stormwater is a common problem in urban environments. With large parts of cities made up of impermeable areas and an expected increase of heavy rain, this is an issue that likely will get more attention in the coming years. In order to relieve pollution effects on the aquatic ecosystems' stormwater treatment alternatives must be developed. One possible treatment system is rain gardens, where polluted stormwater can be treated by both infiltration and retention techniques. This project was carried through at a pilot rain garden located in Gårda where stormwater from the catchment is treated in bioretention filters with different sorption materials such as ash, biochar, peat, and a control filter with compost. Three types of sampling were done, one after that different amounts of water volumes were added and then the effluents were sampled, the second at different depths in the filter beds through soil-water samplers under vacuum, and the third through passive sampling from a pipe inserted in the filter bed at the corresponding depths as the soil-water samplers. Sampling was carried out during winter conditions when the plants in filter beds were believed to be inactive. Three sampling occasions were done of treated stormwater per sampling method and the untreated stormwater, i.e., the influent. The concentrations of eight metals, As, Pb, Cd, Co, Cu, Cr, Ni, and Zn, both dissolved and total metal concentrations, were analysed in all samples and additional analysis of TSS, VSS, IC, TOC, and TN were conducted in the laboratory. The results showed high removal capacities of the metals mostly occurring in high concentrations in urban stormwater such as Co, Cu, Ni, and total Pb and Cu. Arsenic in general and dissolved Pb, Cd, and Cr were not effectively removed, instead, these metals leached from all filters. Three metals, Cu, Cr, and Zn had concentrations exceeding the guideline values of the Gothenburg municipality for discharge of polluted water to recipients from all filters. The removal efficiency of total suspended solids (TSS) and volatile suspended solids (VSS) was high in all filters whereas the removal efficiency of cations, anions, dissolved organic carbon (DOC), and dissolved total nitrogen (TN) varied among the different filters. Of the four filters studied, biochar was the most efficient for the removal of metals and organic matter, and the remaining three filters performed quite similarly. The conclusion of this thesis work is that biochar filters are recommended for the removal of TSS, VSS, and metals emitted in urban environments. Further research is recommended to clarify why some metals, such as As leach from the soil and sorption materials. The results for the removal efficiency of anions, cations DOC, and TN were inconclusive and further studies are recommended.

Key words: Stormwater treatment, bioretention filters, toxic trace metals, winter conditions, biochar, peat and ash

Rening av dagvatten förorenat med metaller i regnbäddar med olika sorptionsmaterial

Examensarbete inom masterprogrammet Infrastruktur och miljöteknik

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Avdelningen för Vatten Miljö Teknik

Dagvatten och förorenade material i den urbana miljön

Chalmers tekniska högskola

SAMMANFATTNING

Förorenat dagvatten är idag ett problem i många städer eftersom de till stor del består av hårdgjorda ytor där vatten inte kan infiltrera ner i marken. Detta i kombination med att skyfall förväntas bli mer frekventa gör att det här är en fråga som kommer bli mer aktuell i framtiden. För att skydda de ekosystem som annars kan ta skada är det viktigt att utveckla lösningar som kan minska problemen med förorenat dagvatten. En sådan lösning är regnbäddar där dagvatten kan renas både genom infiltration och fördröjning. I en pilotanläggning för regnbäddar vid Gårda i Göteborg renas dagvatten från dess avrinningsområde i ett antal regnbäddar med olika filtermaterial som aska, biokol och torv samt ett kontrollfilter bestående av kompostmaterial. I projektet gjordes tre typer av provtagning, en efter att tre olika volymer dagvattenfylldts på jordbäddarna och sedan togs prover i utflödet, både med en markvattenprovtagare under vakuum på två djup och en passiv jordprovtagning på motsvarande djup. Provtagningen gjordes under vinterförhållanden när växterna i bäddarna var inaktiva. Dessutom provtogs det obehandlade inflödet. Åtta metaller, As, Pb, Cd, Co, Cu, Cr, Ni och Zn, analyserades både för lösta och totala koncentrationer. Analys gjordes även på totalt suspenderat material (TSS), flyktigt suspenderat material (VSS), anjoner, katjoner, löst organiskt kol (DOC) samt löst totalkväve (TN). Resultaten visade en god rening av metaller som förekommer i höga koncentrationer som Co, Cu, Ni samt total Pb och Cu. För As generellt samt löst Pb, Cd och Cr var reningen inte tillfredsställande utan läckage förekom. Trots god reningseffekt överskreds Göteborg Stads riktvärden för Cu, Cr och Zn och därmed krävs ytterligare rening av dagvattnet under vintertid när växterna är inaktiva. Resultatet för TSS och VSS var bra medan DOC, anjoner, katjoner och TN gav varierande resultat. Av de fyra undersökta filtren var det filtret med biokol som sorptionsmaterial som presterade bäst medan de resterande tre hade liknande resultat. Denna studies slutsats blir därmed att biokol är det rekommenderade filtret för rening med avseende på TSS, VSS samt metaller vanliga i urbana miljöer. Ytterligare studier krävs för att utröna varför vissa metaller, exempelvis As, lakas ut från filtren. Likaså bör ytterligare studier kring TOC, anjoner, katjoner och TN göras eftersom ingen slutsats kunde dras av resultaten i denna studie.

Nyckelord: Dagvattenrening, regnbäddar, giftiga metaller, vinterförhållanden, bioretentionsfilter, sorption, biokol, aska, torv

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Preface

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In this study, stormwater sampling and successive laboratory work have been done at a facility in Gårda, Gothenburg. The tests have been carried out from January 2023 to March 2023. The work is a part of a research project concerning treatment of pollutants in rain gardens at the Department of Architecture and Civil Engineering, Division of Water Environment Technology, Chalmers University of Technology, Sweden. The project is carried out as part of a doctorate at Chalmers University of Technology.

This part of the project has been carried out with Glenn Johansson as a researcher and Professor Ann-Margret Hvitt Strömvall as supervisors. All tests have been carried out in the laboratory of the Department of Architecture and Civil Engineering at Chalmers University of Technology. My lab instructor Amir Saeid Mohammadi is highly appreciated for his help with planning and conducting the tests.

Finally, it should be noted that the tests could never have been conducted without the sense of high quality and professionalism of the laboratory staff.

Göteborg June 2023

Anders Jardeby

1. Introduction

Beginning with the industrialisation pollutants, among them greenhouses gases such as carbon dioxide (CO₂), dinitrogen oxide (NO₂), and methane (CH₄), in the atmosphere and water have increased and are causing climate change (Smerdon, 2018). The effects include higher temperatures and changing weather patterns. It is of importance both to halt processes fuelling climate change and to find suitable mitigating adaptations. The repercussions of climate change, such as severe weather, smog, heat, and cold waves, will be most noteworthy in bigger cities where many buildings, infrastructure and people congregate. Buildings and infrastructure may escalate climate effects but could also, if used properly, be a part of the solution.

The seemingly ever-increasing rate of urbanisation in Europe is at 75 % as of 2021 and is expected to rise to 90 % at the end of this century (EU, n.d.; World Bank, n.d.). Urbanisation changes cities and nearby areas, especially regarding greens areas which tend to either be lost or increasingly difficult to maintain which impacts the stormwater management negatively (Tzoulas et al., 2007). Urbanisation has caused an increase in pollutants such as organic pollutants (OP), particulate matter (PM), nitrogen, phosphorus, microplastics, and toxic metals being transported by stormwater runoff into receiving waters without treatment (Järllskog et al., 2022; 2021). These pollutants can cause harm in receiving (water) environments due to persistence to degradation, an ability to travel long distances as they are transported from their sources, among them parking lots, roads, car washes, and fertilised lawns (Hermawan et al., 2021; Lange et al., 2022; Youngblood et al., 2017).

Therefore, to restore the levels of pollution in (water) environments the stormwater management must be improved. There are several possible alternatives such as biofilters, wetlands, retention ponds and/or bioretention systems. These implementations can mitigate the impact of climate change and/or urbanisation since they are designed to restore natural hydrological patterns and thereby targeting water quality and quantity by using sedimentation, filtration, decomposition, and sorption (Hermawan et al., 2021; Kaya et al., 2022; Lange et al., 2022; Tzoulas et al., 2007).

Improvements in stormwater management can be both physical and non-physical (Eckart et al., 2017). The former could be in the form of green roofs, rain gardens, bioretention cells, swales, permeable pavements, or ponds while the latter primarily comes in the form of community education. Low-impact development (LID) technologies are generally categorised into two categories, infiltration-based technologies, and retention-based technologies. Both give the desired effect of reduced effective impervious areas. A combination of the techniques is preferred to achieve the best results. The overall aim with LID is to try to restore the pre-development hydrological conditions which would lead to a more natural water cycle. The factors listed above, the increase of pollutants in stormwater, the fact that it is a major threat to aquatic ecosystems, and that climate change and urbanisation likely will bolster these issues make stormwater treatment an important research topic.

This master thesis is carried out within the frames of a research project which focuses on rain gardens' ability to treat urban stormwater. The research project is a collaboration between Chalmers University of Technology, Aquateam COWI, and COWI SE. The pollutants of interest to study in this project are metals, organic pollutants, microplastics, and nutrients.

In the research project, the focus is to study processes for the removal of metals and possibilities to recycle metals, absorb nutrients, degrade microplastics and OP in rain gardens. The bioretention filters in the rain gardens have different filter materials, either ash, biochar or peat, with or without plants (Johansson et al., 2023). In the research the treatment efficiency is investigated, and different soil properties and plants are combined to see which yields the best result. A pilot facility has been set up in *Gårda*, in central Gothenburg where untreated stormwater mainly from highway E6, but also from other nearby areas, in Gothenburg is used for irrigating the rain gardens (Björklund et al., 2009). The stormwater is flowing to a well situated by the side of the road, from where it is pumped to a tank.

Lastly, the stormwater from the tank can be pumped to a watering can and then the plants in the rain gardens are irrigated. Highway E6 through Gothenburg is one of the most trafficked roads in Sweden with over 86 000 vehicles per day whereof roughly 10 % is heavy vehicles (Trafikverket, n.d.). Therefore, the road runoff is highly polluted.

1.1 Aim

The aim of this project was to investigate the treatment efficiency of polluted stormwater in a rain garden that consisted of several filter columns with different sorption materials such as ash, biochar and peat. There was also a fourth type of filter, a control filter. All filters used in this experiment had vegetation. The aim was also to understand the removal processes of toxic metals and make correlations with other chemical parameters such as concentrations of nanoparticles pH, TSS, VSS, redox potential, conductivity, turbidity, and DOC/TOC. A new sampling technique was investigated to study the concentrations of dissolved metals in the soil water at different depths in some selected filters. The soil-water samplers were originally developed to study groundwater but the suitability for stormwater moving through soils was investigated in this project.

1.2 Research questions

- How effectively can rain gardens remove metals from stormwater in winter conditions?
- How does the different sorption filter material in the rain gardens affect the metals regarding transport, both how the process is functioning and the efficiency of the process?
- Are soil-water samplers a suitable technique for tracking the treatment of metals in stormwater at different depths?

1.3 Limitations

In this study only the processes for the removal of metals and general chemical parameters are researched. Due to time constraints, two sampling sessions per filter were made.

2. Background

As a consequence of fossil fuel burning that has increased since the start of the industrialisation the concentration of CO₂ and other greenhouse gases has increased (Smerdon, 2018). Other factors include land use change and there is overwhelming evidence that this is the cause for the global warming which in turn may be devastating for several areas such as biodiversity, ecosystems, agriculture, ocean life, the global economy, and thereby, the well-being of human societies. Yet it is not known exactly what will happen in these areas since there are still many insecurities and time for adjustments which may have a mitigating effect. However, what is very likely is that the weather patterns will change bringing both more droughts and floods, more unbalanced patterns of precipitation (Smerdon, 2018).

Urbanisation is resulting in increased areas of impervious land which has an impact on watershed hydrology and water quality. The urban development cause more urban runoff and negatively affects the possibilities for storage, infiltration and groundwater recharge of rainfall while simultaneously compiling pollutants from impervious areas which in turn leads to a lower water quality locally and regionally. Both human activities and natural processes are sources of pollutants of different kinds, such as physical, biological and chemical, and are emitted in the form of suspended solids, oil and grease, organic carbon, nutrients and toxic metals (Li & Davis, 2009).

As most of the cities in Europe, Gothenburg is highly urbanised and consequently, there are fewer green areas left. Consequently, many ecosystems in the cities are affected negatively, as are human health and well-being (Tzoulas et al., 2007). A major issue with stormwater management is that it is still common that polluted urban stormwater is released to receiving water even though no treatment has taken place. This may harm the receiving environment. This problem can be rectified by implementing stormwater bioretention systems which reduce the total metal concentrations in stormwater efficiently (Lange et al., 2022).

Regarding urban runoff, impervious surfaces are vital since they facilitate the accumulation and reduction of pollutants in build-up and wash-off processes (Markiewicz et al., 2017). On days with no precipitation, the pollutants are accumulated and when it rains the pollutants are washed away. Two other factors that affect the distribution of pollutants are wind and degradation which stalls the accumulation after a couple of days. A major problem with urbanisation is that it often contributes to an increased pollution load in stormwater. The pollutant load can affect the receiving waters negatively due to the toxicity of several substances which occur in stormwater. Among the pollutants which may cause the worst effects are metals, nutrients, organic pollutants, and microplastics. Pollutants can occur in different phases, dissolved form, attached to particles and colloids (Müller et al., 2020). Several parameters regarding metal pollutants are dependent on their distribution between solid and liquid phases (Hvitved-Jacobsen et al., 2010). Transport, transformation, and effects of pollutants are three examples of this. The distribution is also influencing how efficient the removal of pollutants is in urban drainage systems designed for filtration and retention.

2.1 Stormwater issues

Urbanisation and climate change cause more extensive runoff than the current stormwater management systems are designed for. Urbanisation changes the environment in a way that aggravates the natural processes, mainly infiltration, which increases stress on conventional stormwater systems. The conventional stormwater systems have consisted of curbs and gutters which lead the water to the sewers. From there the water is either released to a water body (river or lake) or to a treatment plant depending on design. Urbanisation removes the pervious areas which result in increased peak flows, reduces infiltration rates and groundwater recharge is affected negatively. An issue with large cities is that retrofitting existing infrastructure is required to solve the problems that the growing city has instead of installing new (Eckart et al., 2017).

2.1.1 Stormwater management practices

In order to reduce the negative effects of urbanising development, an initiative called the Green Infrastructure (GI) concept was created (Tzoulas et al., 2007). GI focuses both on the quality and quantity of green spaces in urban environments. The idea is that constructed, and correctly maintained GI can be a guide for urban development yielding a framework combining economic growth and nature conservation. By planning a city with the GI-concept in mind, opportunities for integration between dissimilar fields such as urban development, nature conservation and public health is possible. If green areas in urban and peri-urban are prioritised, then biological diversity can be conserved, and habitat fragmentation be reduced since isolation is less probable.

In addition to providing better stormwater management, green infrastructure can give secondary benefits such as reduced energy consumption, emissions and urban heat island effect and improved air quality (Eckart et al., 2017). Moreover, community liveability and public health improvements as well as providing habitats for wildlife are potential benefits of green infrastructure. By using green infrastructure, the aesthetics of a city is improved.

Ecosystem services are closely interlinked with the health of an ecosystem and therefore ecological stress will cause a decrease in both quality and quantity of ecosystems. However, a healthy ecosystem will provide several beneficial ecosystem services. The GI-concept is a way to improve ecosystem health and consequently public health, e.g., by alleviating air pollution and urban heat island effects (Tzoulas et al., 2007).

2.2 Low Impact Development (LID)

Managing urban stormwater is important from all three perspectives of sustainability, ecological, economic, and social and subsequently in all urban development (Eckart et al., 2017). In order to adapt to the changing conditions caused by urbanisation and climate change urban stormwater management must evolve to meet the new demands. The overall aim with LID is to try to restore the pre-development hydrological conditions which would lead to a more natural water cycle. However, for the most part that is not possible so instead adaptations are implemented. There are two improvements that the adaptations can be designed for, either reducing stress on the urban stormwater infrastructure or improving the overall resilience to climate change. In LID the focus is on quantity as well as quality, to decrease the risk of stormwater pollution (Fletcher et al., 2014). LID installations can bring several benefits to urban infrastructure, including but not restricted to, reduced runoff volume, improved infiltration, reduced peak flow, extended lag time, reduced pollutant loads, and increased base flow. Improvements of stormwater management can be both physical and non-physical. The former could be in the form of green roofs, rain gardens, bioretention cells, swales, permeable pavements, or ponds while the latter primarily comes in the form of community education.

One way of solving issues in urbanised cities is to retrofit the existing infrastructure as opposed to building brand new ones (Eckart et al., 2017). Infrastructure that is suitable for retrofitting is parking lots, pavements, and buildings for example. Moreover, increasing the use of existing pervious areas such as parks, lawns, and gardens. Additionally, another possibility is directing runoff from impervious area to pervious areas or retention facilities. Finally, a city planning decision that can improve stormwater management is to have higher density development in one place and then leave undeveloped land nearby the prerequisite being that the city is still developing.

Several measures for stormwater treatment have been developed to try to alleviate the negative effects of untreated stormwater (Dagenais et al., 2018). There are systems that are very complex and designed to increase evapotranspiration, infiltration and reduce runoff volumes with the goal of lowering the impact of pollutants in aquatic environments.

There are several important aspects to maximise the viability of LID measures, the previously mentioned location but also soil type and conditions, type of plants, sunlight, rainfall, land use and topography

(Fletcher et al., 2014). Shallow water tables and permeable soils limit the viability of LID strategies. However, research has shown that there are still some feasible solutions, wet ponds and constructed wetlands namely. Two significantly impactful attributes of infiltration and surface runoff processes are the physical properties and effective depth of soil. LID technologies are generally categorised in two categories, infiltration-based technologies and retention-based technologies. Both give the desired effect of reduced effective impervious areas. A combination of the techniques is preferred to achieve the best results.

2.2.1 Infiltration-based techniques

The main benefit of using infiltration techniques is the support for the restoration of baseflows provided by recharging subsurface flows and groundwater (Eckart et al., 2017). These techniques are dependent on on-site conditions and therefore there is a variety of ranges where they can be used. Examples of infiltration-based techniques are bioretention areas also known as rain gardens, swales, infiltration trenches and porous pavements.

Rain gardens are small areas in the city designed to reduce and treat stormwater runoff in close proximity to the source. Implementation is possible in both residential and commercial settings. As with many other LID strategies important factors taken into consideration in the design phase are soil type, site conditions, and land use. In a rain garden, there are several different components that perform different tasks depending on their attributes to lower the impact of stormwater (Fletcher et al., 2014). Common plants in rain gardens include trees, perennials, and shrubs. A well-designed rain garden is functioning much like natural landscapes and services that can be provided include encouraging infiltration, capturing runoff, promoting evapotranspiration, recharging groundwater, and reducing peak flow and pollutant loads. One of the main advantages with rain gardens is the fact they apart from pollutant removal also provide other desired benefits in cities, such as enhanced biodiversity and mitigated effect of the urban island (Dagenais et al., 2018). Furthermore, rain gardens can be incorporated in cities relatively easily.

Additionally, an interesting fact about rain gardens is that vegetation is not necessarily needed (Dagenais et al., 2018). Stormwater passing through the soil media are sufficient to ensure some treatment regardless of vegetation. The physical and chemical processes of filtration, adsorption and precipitation are ensuring that pollutant removal is taking place and on top of that biofilm and bacteria account for processes as well. However, most manuals on rain gardens recommend vegetation because of the uptake of nutrients, the support of biological growth, and the contribution to a reduced need of maintenance by preventing clogging and maintaining porosity (Eckart et al., 2017). Additionally, vegetation also removes pollutants such as TSS, metals and organics.

Furthermore, transpiration is another process that is impacting stormwater treatment (Muerdter et al., 2018). Water is taken up by the root of the plant, passing through the plant and subsequently evaporating from the leafs. The transpiration of the vegetation helps increase the volume of treated stormwater since less water is reaching the surrounding soil, which is another advantage of having vegetation in rain gardens. Moreover, permeable pavements can also be used to handle stormwater since they, contrary to regular pavements, allow for infiltration to the ground (Fletcher et al., 2014). Additionally, pollutants in stormwater will not accumulate because the water is infiltrating continually.

2.2.2 Retention-based techniques

The main idea with retention-based techniques is to reduce flow. Examples of these techniques include wetlands, ponds, green roofs, and rainwater harvesting (Eckart et al., 2017). Wetlands and ponds have been used for several years and are effective in pollutant removal, however, they have a limited impact on reducing overall runoff volumes. This is due to the fact the only losses are from evapotranspiration. Green roofs are an efficient and space-efficient solution to reduce urban stormwater runoff by increasing the pervious surfaces in urban settings. Since green roofs use the fact that roofs are generally not used to a large extent and therefore can be claimed for stormwater management activities. A green roof is

covered, either completely or partially, with vegetation and with waterproof membranes leaks are avoided. Moreover, green roofs yield several other benefits such as extending the time the roofs can be in use, reducing energy costs, and conserving valuable areas that would otherwise be hosting stormwater management infrastructure. On top of that, well-designed green roofs can be implemented without the need for altering the structural design of a building.

2.3 Pollutants

Several pollutants can be found in stormwater, among them metals, microplastics, nutrients and organic pollutants.

2.3.1 Metals

Different metals have different properties such as solubility and toxicity depending on their ability to form complexes with organic and inorganic substances (Alenius & Hermansson, 2016). Because of different charges of the metals, the removal processes are depending on their ability to sorb to the surface of a particle. Four important factors for the speciation of metals in water are acidity, alkalinity, substance concentration, and redox potential. To understand metal speciation and the formation of metal complexes better a method called MINTEQ-model has been developed. By using the chemical equilibriums for metals, their speciation and solubility in natural waters can be calculated. Two factors that increase the interest in stormwater runoff metals are their omnipresence and that they are chemical elements, i.e., chemical transformation or destruction is not possible. Because of these two factors, they pose a threat to the aquatic environments of the receiving waters.

There are two main emission sources of metal pollutants in urban runoff and those are building materials and road traffic (Müller et al, 2020). Car washes, building sliding, vehicle brake linings, tire wear, and atmospheric deposition contribute to emissions of common metals such Cd, Cu, Cr, Ni, Pb, and Zn.

An important factor when evaluating the removal of metals from stormwater in rain gardens is metal speciation (Alenius & Hermansson, 2016). However, to this day, it is most common to focus solely on total metal concentrations, and studies on dissolved metals (fractions $< 0.45 \mu\text{m}$) is less common (Lange et al., 2020). Among the factors that can impact metal fractionation is vegetation and salt in stormwater. A study on metal speciation found that rain gardens affect the metal speciation of Cu and Zn, treatment of total concentrations are better than for dissolved fractions. Concentrations of particulate, colloidal and truly dissolved fractions were compared in that study. Moreover, an influence of metal speciation is the behaviour of metals in water, such as transportability and bioavailability. Therefore, metal speciation is important to take note of when evaluating environmental impact. For example, free metal ions are thought to be available for bioaccumulation and not settleable. Whereas metals bound to colloids are generally less bioavailable but are more settleable. Finally, particulate metals have the opposite characteristics, not directly bioavailable and settleable. Furthermore, metals can change between these three forms depending on environmental conditions, such as vegetation and salt content (Lange et al., 2020).

By understanding the metal speciation in the rain gardens, there are some of the processes that can be understood and explained. As stated earlier, salt has an impact on metal speciation and can affect the solubility of metals in both water and soils. Moreover, salt can lead to leaching of Cd, Cu, and Zn and increase the concentrations of dissolved Pb and Cu in the effluent. In the removal of metals in rain gardens, the uptake of plants accounts for just a small part. Nevertheless, vegetation can impact metal speciation by virtue of plant metabolism products and the creation of preferred flow paths because of roots. The results of processes like this are increased concentrations of colloidal and dissolved metals in the effluent. However, there have not been a lot of studies in this area and therefore this is subject to further studies.

Salt reduces the overall metal removal, but the impact depends on metal and fraction. For example, studies have shown that Cu and Cd have substantial differences between salt and no salt for all three

fractions, whereas for Zn this was only true for unfiltered samples and the other two showed no indicative difference. The speciation of metals is controlled by both the properties of the metals and the properties of the environment in which they are present, such as the pH, redox potential and the presence and concentrations and properties of molecules that could form complexes with the metals (Lange et al., 2020).

In natural systems, the speciation of metals is of great complexity and determines their mobility in the environment and their bioavailability (i.e., how easily they are taken up by organisms). Metal speciation therefore plays a key role in determining the potential bioaccumulation and toxicity of metals and should therefore be considered when assessing their ecological risks. Metal bioavailability and transport are in particular strongly related to the distribution over solid and liquid phases of the environmental matrix (Lange et al., 2020).

2.3.2 Microplastics

Microplastics in stormwater originate from various sources such as tyre and road wear and degradation of littered plastics (Gilbreath et al., 2019). It is mainly during rainfall that the microplastics are entrained in the stormwater runoff. Further studies are needed to understand the treatment of microplastics in bioretention systems and their potential efficiency.

2.3.3 Nutrients

High concentrations of nutrients such as nitrogen and phosphorus can cause eutrophication which is a common problem in aquatic environments. Eutrophication leads to deteriorating water quality and loss of aquatic habitats and because of this, it is a priority for many municipalities to treat stormwater (Dibiasi et al., 2009).

Nitrogen present in urban runoff is predominantly, 75–85 %, in dissolved forms. Apart from adsorption, the removal of nitrogen is mostly through biological processes. The nitrogen and compounds containing nitrogen will be degraded to fewer compounds. That is one reason for the importance of vegetation in bioretention systems, the degraded nitrogen particles can be collected by plants. Otherwise, it could lead to the leaching of nitrogen (Dibiasi et al., 2009).

2.3.4 Organic pollutants

Organic pollutants found in the environment are generally a result of extensive global development. Specific sources are human activities such as the industrial production of chemicals or farming (Eckart et al., 2017). The issue is that organic pollutants often are persistent to degradation and can transport long distances in water-air-soil systems. These pollutants have an important sink in cities. By stormwater runoff these compounds can then be transported to receiving waters in both dissolved and colloidal forms (Markiewicz et al., 2019). Because of this the mobility and consequently the toxicity of organic pollutants is higher than previously believed. The quality of urban runoff is a substantial pressure on aquatic ecosystems. Therefore, it is important to identify sources of OP and develop stormwater treatment facilities that can retain these pollutants. Studies have shown that traffic is one of the main contributors to the contamination of urban runoff. There are vast possibilities regarding traffic-related pollutants, such as tyres, auto bodies, exhaust gases and particles, road surfaces, and road infrastructure.

2.4 Construction of filter beds

Several parameters must be factored regarding the efficiency of sorbents like physical-chemical characteristics such as aromaticity, polarity, pore size and volume and surface (Markiewicz et al., 2020b). In addition to this sorption selectivity, mechanical stability, environmental impact and toxicity, cost-effectiveness and reusability potential are other important aspects. Challenges for stormwater treatment is finding sorption filters for degradation of organic pollutants, recovery of sorbed metals and microplastic separation after use. Another major challenge is clogging which limits the stormwater treatment and is a cause of physical, biological or chemical processes, either a combination of the three, or two or just because of one of them. Examples of clogging are fine-grained material occupying the

pores of a coarser material, interference of the sorption material in the form of degradation or biological growth or finally chemical precipitation caused by metal salts.

2.4.1 Biochar

Biochar is a porous, carbonaceous material produced when thermochemical pressure, such as pyrolysis, is put on biomass feedstock (Kaya et al., 2022; Lebrun et al., 2018). Biochars generally have high pH (above seven at least), the potential for high cation exchange, a big surface area and is therefore a viable solution as a filter material in a bioretention system, because of the high half-life which averages over 100 years. In comparison with normal soils, several parameters are increased, like pH, DOC and conductivity. Biochars consist of a non-carbonised fraction that can react with pollutants in the soil, forming complexes and thereby decrease the concentrations of these undesired elements in the soil solution (Lebrun et al., 2018). The use of biochars in rain gardens influences greatly the physical, chemical and biological properties of the soil, two examples are increased microbial activity and water-holding capacity (Murtaza et al., 2021). The extent of the influence depends on local conditions. Two other positive effects of biochar filters are increased soil porosity and infiltration rates.

Examples of thermochemical pressures include pyrolysis, hydrothermal carbonisation or gasification (Kaya et al., 2022). Biochar feedstock affects various characteristics of the product, for example contaminant removal capacity, elemental content and contaminant leaching. Consequently, a conscious selection of biochar feedstock based on the needs for stormwater treatment is possible. A variety of materials can be used to produce biochar such as organic materials (wood waste), industrial by-products (crop residues), municipal waste (newspaper) and lastly from non-conventional materials (manure). Biochar-based materials have a high specific surface area, micropore volume, surface charge density, and relatively low bulk density which makes it a good choice for the removal of toxic contaminants of concern in stormwater. In addition, biochar-based media can remove dissolved pollutants and thereby decrease the number of pollutants. Moreover, biochar is also effective in removing nutrients, metals and even persistent organic pollutants from waters through physicochemical and/or biological interactions such as degradation or plant uptake. Biochar is an adequate choice for removing trace metals, especially Cu and Zn.

When deciding on which type of media to use in a stormwater treatment facility for removal of dissolved pollutants there are several factors that must be taken into consideration (Kaya et al., 2022). These factors include hydraulic conductivity, effective storage volume, enhanced capacity, minimal leaching and/or release of undesired substances, long-term stability of the design, low cost and sufficient availability of materials.

Moreover, biochar with high ash contents often has a high pH which reduces the solubility of metals and increases removal efficiency (Kaya et al., 2022). The pH levels are important not only for metals but also for other inorganic pollutants regarding adsorption and desorption. Furthermore, it is of uttermost significance that the media does not alter the pH of stormwater since a considerable change can cause a reduced immobilisation of pollutants which in turn decreases the treatment efficiency (Kaya et al., 2022). Stormwater pH generally varies between 5.5-9.3, but this is site-specific. Two major factors are precipitation levels and nearby land use. Lastly, an aspect that might get overlooked when choosing an adsorbent is cost and availability.

In addition to being a sink for contaminants, such as nutrients and heavy metals, some biochars can be sources due to leaching, depending on feedstock and pyrolysis temperature or pH (Kaya et al., 2022). Another important factor to take into consideration is dissolved organic matter (DOM) which is biochar-derived and may cause mobilisation of constituents like Cu due to the high affinity of DOM.

2.4.2 Ash filter

Fly ash is formed as the main industrial waste by-product after the combustion of solid fuels (Alterary & Marei, 2021). The makeup of fly ash is mainly unburned carbon, metal oxides (Si, Fe, Ca and Al) and

inorganic substances. The unburned carbon is a desirable source of activated carbon since the price is relatively low. Moreover, activated carbon is a vital component for fly ash adsorption capacity. Since the content is subject to variation, the characterisation of fly ash is difficult. In earlier studies, ash has shown a good ability to adsorb toxic metals (Youngblood et al., 2017). In a study from 2021 of stormwater pollutants, the results showed that most of the toxic metals ended up in the top 5 cm of the ash filter media (Hermawan et al., 2021). The ash filter had double the amount of retained toxic metals in the top layer, crust layer, compared to the sand layers that also were included in the study and the infiltration rate was significantly lower for the ash filter at 31 mm/h. Taking metals accumulation and clogging potential into consideration it was concluded that the periodic maintenance of the ash filter was around 1.5 years.

Depending on the source, fly-ash can be problematic, for example, if municipal solid waste is the source due to the soluble salts (chlorides for the most part), metals (Cd, Cu, Hg, Pb & Zn), and dioxins (Karlfeldt Fedje & Andersson, 2020). To avoid problems, it is more common to landfill these fly ashes, which then have been stabilised to decrease potential leaching. Because of the high concentrations of metals, especially Cu and Zn, in fly ash landfilling is not the best option. Bioretention cells with fly ash as media have shown good removal efficiency for P (Kandel et al., 2017). One study found 85 % removal of P with minimal leaching whereas conventional filters had almost 50 % desorption.

2.4.3 Peat filter

One advantage of peat materials is the high adsorption capacity for metals. Factors such as which type of peat and solution properties, pH, metal ion concentration, and ionic strength is decisive for how high the adsorption capacity is (Kalmykova et al., 2008). Moreover, peat (sphagnum peat) has several desired properties such as acidic pH, high moisture content, and a low decomposition level as well as low bulk density, and high hydraulic conductivity. The latter two are indicators of good permeability and therefore well suited as a filtering medium (Kalmykova et al., 2008).

Peat is the first step in coal formation and is a soil material that contains organic matter in different phases of decomposition (Brown, et al., 2000). In order to form peat, areas with high moisture content and a growth rate that is exceeding the rate of decomposition is required. The process of decomposition to peat contains the preservation of vegetation and a subsequent slow oxidation. Factors such as regional climate, acidity, and degree of degradation is affecting the turnout of the final material.

Peat consists of two main components, minerals, and partially decomposed organic matter (Markiewicz et al., 2020b). The latter is made up of lignin, cellulose, fulvic and humic acids. The fulvic acids are the main reason for the low pH between 3–6 of peat soils. In a column pilot study, peat filters effectively removed total suspended particles and total organic carbon from stormwater but leached dissolved organic carbon.

Peat moss has shown a good capacity for removing toxic metals, oil, detergents, dyes, pesticides, and nutrients from contaminated waters, and more than nine-tenths of Cd, Cu, Zn, Ni and Pb was removed (Kalmykova et al., 2010). However, the removal of metals was temporarily decreased due to the leaching of DOC. While the peat filters showed an effective removal of the five earlier mentioned metals the results were not satisfactory for the removal of As and Cr. It should be noted that this study used a pH range higher than expected (6.7–8.0) compared to the common range between pH 3–6. Therefore, peat moss is not a suitable filter material if the stormwater has high concentrations of these two metals and the pH is neutral to basic. Peat moss has several advantages compared to other sorbents when it comes to use in filters such as availability, price, and high sorption (Kalmykova et al., 2008).

One unknown factor when it comes to peat moss use in filters is the durability (Kalmykova et al., 2009). When found in nature it is decomposing slowly because of the conditions it is formed under, poor in nutrients and below the water table in anoxic conditions. These conditions are unfavourable for the microbial activity responsible for decomposition, however, since the conditions in raingardens are

different, the decomposition rate can increase. Two factors that can lead to increased decomposition are changing redox conditions or an increase in pH. The former can increase the air available for microbial decomposition while the latter stimulates microbial growth in peat. As stated earlier, peat moss contains both humic and fulvic acids that are soluble in water and could thereby be leached. Studies have shown that this leaching occurs in two separate steps. The first, where large quantities are released, appears during the first flush when residual DOC are leached and the second is more modest in quantity but instead occurs for a longer time. The latter leaching is likely due to the continual degradation of the peat surface.

Neither freezing nor pH rise, or higher metal concentrations had a substantial effect on the metal retention or leaching of DOC (Kalmykova et al., 2009). DOC production and leaching is a common occurrence in organic rich peat but there are indications that microbiological activity is affected negatively by metal presence in the used solution and hence also peat decomposition. There is a risk of trace metals being transported by organic material in organic complexes through peat filters because of the former's high affinity for the latter. However, this phenomenon is more common in the earlier stages of the experiment when DOC leaching is highest. Thereafter, both the concentrations of DOC and metals are significantly lower and therefore a solution could be to wash the peat beforehand as this removes residual and easily soluble organic material which in turn decreases the risk of metal transport.

An earlier assumption was that metals with high stability constants for humic acids was the metals that would stay in complexes of humic acids and move through the filters (Kalmykova et al., 2010). However, more recent studies have shown that up to 40 % of the complexes can be retained, which is a big difference. The opposite types of metals, with low stability constants for humic acids, are removed to a higher degree in peat filters since they bind to the peat surface.

2.5 Transportation

Adsorption is a process where a substance is accumulated on the surface of a solid (Chowdhury, 2011). The substance is called adsorbate and the solid is called adsorbent. There are three components needed for this process in water treatment to occur and those are a solid adsorbent, filter material and contaminant water. However, this process is sensitive to changes in water properties such as pH, concentration, and temperature which may cause the process to revert. A reversion would mean a release of the adsorbate from the adsorbent, and a desorption for short. Adsorption can be divided into two categories, physical adsorption or chemical adsorption. Even though there are differences between the bindings of the materials, the difference is arbitrary, and boundaries are fluid. Conversely, absorption is one substance moving from a bulk phase to another. In addition to sticking to the surface, the substance is connected by pores or spaces between molecules. Some materials can accumulate substances on surfaces and in the interior, providing there is a complex structure (Chowdhury, 2011).

Sorption of metals is dependent on adsorbates being attracted to the adsorbent because of an arising electric charge on the surface of the adsorbent (Worch, 2012). The degree of attraction is depending on chemical properties and the form of the metal is present in. For example, generally free metal ions are more positively charged compared to a metal complex which means the former will have a stronger sorption to the negatively charged surface compared to the latter. In addition, electronegativity and electron configurations of positively charged ions are two other factors influencing the rate of attraction. Regarding the sorption of organic pollutants molecular weight and water solubility is the main impacting factors. The molecular weight of a substance has an impact on the absorbability of that substance, if the former increases, so does the latter. If the compound has low water solubility, it is likely that the compounds and the adsorbent has stronger attractive forces compared to the adsorbent and water. Therefore, the probability of the compound and adsorbent attaching to one another. Finally, important factors regarding adsorption, both positively and negatively, are the polarity of the adsorbent, pH, and the temperature of water, and the constitution of the solute.

Adsorption is taking place on the surface of a material making the surface area an important quality parameter of adsorbents (Worch, 2012). High porosity is desirable because that gives a large internal surface. With a large pore system and fine pore, a high internal surface is enabled. Moreover, fraction of large pores can facilitate the transport of adsorbates to the adsorption site. When discussing chemical adsorption, the chemistry of the surface is important. In a bid for optimal design and efficiency, a preferred method is to study equilibrium data of amounts of contaminants in waters compared to filter media (Kalmykova et al., 2009). When a state of equilibrium is reached, sorption and desorption are occurring at the same rate.

2.6 Filtration

For stormwater treatment of polluted water, filtration is a common method to remove specific undesired constituents of the stormwater (Metcalf & Eddy, 2004). The main removal with filtration is residual organic and inorganic and suspended solid. The most commonly used filtration method is by gravity through a granular media, and this is a straightforward design where water simply flows through the filter media. Most designs consist of a layer of sand on top of a layer of gravel.

Conventional gravity filters regulate the rate of filtration by managing the amount of effluent (Chalmers WET, 2014). The influent can be managed both automatically and manually to maintain the desired water level. The most important factor when it comes to the operational behaviour of the granular filter is the grain size of the filter material (Metcalf & Eddy, 2004). There are two factors affecting the lifetime of a filter and those are head loss or if the levels of pollutants in the water have exceeded acceptable levels. Ideally, the limiting levels are reached at the same time, however, this seldom happens, due to changing conditions and the quality of influent water.

A material that is coarse enough to have pore openings of a size sufficient to contain particles and at the same time so fine that suspended solids cannot filter through constitutes an ideal filter material (Chalmers WET, 2014). A filter that is too fine likely suffers from issues like high frictional resistance and head loss. The depth of the filter is another important aspect for extending the lifetime of a filter before saturation. Finally, a filter bed must be graded adequately to allow for effective backwashing.

2.7 International frameworks for a sustainable planet

In the latter years, several international frameworks for a more sustainable planet have been developed, among them the United Nations (UN) Sustainable Development Goals (SDG) and the Water Framework Directive (WFD) of the European Union (EU) (UN, 2023; *2030 Climate & Energy Framework*, n.d.). The Swedish government has set a couple of environmental objectives as well. The SDGs were adopted by the UN in 2015 and is a proclamation that all countries must take on responsibility for global cooperation (UN, 2023). There are 17 SDGs and the most relevant for this project are goal 6, *clean water and sanitation*, goal 11, *sustainable cities and communities*, goal 14, *life below water* and goal 15, *life on land*. It could be argued that other goals are fitting as well, but those would be secondary.

In addition to the UN SDGs, the European Union has set up a climate and energy framework for 2030. The goal is to reduce greenhouse gas emissions, increase the share of renewable energy and improve the energy efficiency (*2030 Climate & Energy Framework*, n.d.). Furthermore, since 2000 there is a *Water Framework Directive* (WFD) in place set out to stop the deterioration of EU water bodies and increase the status of rivers, lakes and groundwater (*Water Framework Directive*, n.d.). Since the inception of the WFD, it has been the highest prioritised water protection law in Europe. The objective is to have both good chemical and good ecological status, however for this project the former is applicable.

Finally, there is the Swedish environmental objectives which are 16 in total (*Swedish Environmental Objectives*, n.d.). But the most relevant for this project are *natural acidification only*, *a non-toxic environment*, *zero eutrophication*, *flourishing lakes and streams* and *a good built environment*.

3. Method

The project consisted of several sampling occasions, field days and then subsequent laboratory sessions. This was to ensure that data from different weather conditions was investigated, and that enough samples and analysis data were received. Alongside this a literature study accompanied the sampling and lab work, especially during the initial part of the project. After the lab work was done the results could be added to the thesis and when all data from the chemical analysis was available a comparison could begin. In addition to measuring the effluent from the different filters in the rain gardens, influent, i.e., the untreated stormwater, was measured and the efficiency of the filters in the raingarden for various pollutants was investigated.

The following metals were studied in this project: As, Pb, Cd, Co, Cu, Cr, Ni, Zn as most of these metals are emitted from traffic (Muller et al, 2020) and have reference values for emissions of polluted water from the Department of Sustainable Waste and Water in the City of Gothenburg as is seen in Table 1 (Göteborgs Stad, 2020). In addition to this, TSS, VSS, DOC, and TN were measured as well and finally an IC was conducted as well. TSS and VSS were analysed to get information about organic matter content in the stormwater. TN to get information regarding nutrients, DOC to investigate correlations with metals and IC to gain information about which ions are predominant.

Table 1. Guideline values for emissions of polluted water to stormwater networks and recipients, from Gothenburg municipality (Göteborgs Stad, 2020).

Metal	Guideline value
Arsenic	16 µg/L
Lead	28 µg/L
Cadmium	0.9 µg/L
Copper	10 µg/L
Chromium	7 µg/L
Nickel	68 µg/L
Zink	30 µg/L

3.1 Pilot facility

The pilot facility where the sampling was taking place is situated in Gårda in central Gothenburg, along the E6/E20 highway, and is presented in Figure 1. The catchment area is 52 000 m² of which 20 000 m² is impermeable (Björklund et al., 2009). The Gårda rain garden is situated along the highway and there are 13 bioretention filter constructed, with biochar, ash and peat with and without plants and one control filter with normal soil and plants. The construction of the different filters is described more in detail in Johansson et al., 2023).

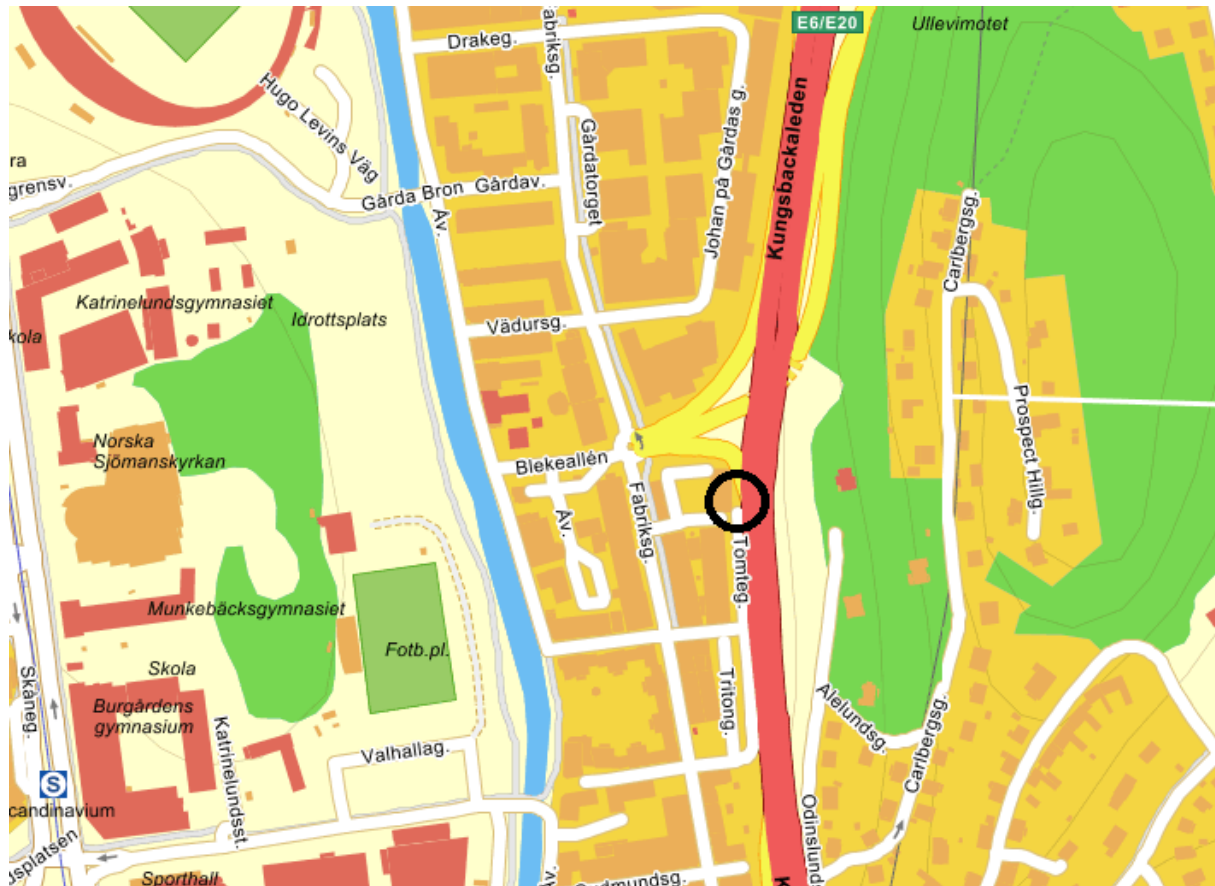


Figure 1. A map of central parts of Gothenburg with the location of the Gårda rain garden pilot facility marked by a red ring.

All the bioretention filters consist of six similar layers, a top layer, a main layer and four drainage layers, see Table 2. The main layer was the distinctive one and additionally peat, ash and biochar consisted of at least one sub layer. The top layer consists of *Hekla regnbädd* which is sandy loam with pumice stone. The main layer in the control filter consists of 85% *Hekla regnbädd* and 15% compost, while the peat filters have a main layer consisting of 60% *Hekla regnbädd* and 40% peat and under that a sub layer of 100% peat. Furthermore, the main layer of the biochar filters consists of 60% *Hekla regnbädd* and 40% biochar and under that a sub layer of 100% biochar and the main layer of the ash filters consists of 35% *Hekla regnbädd*, 50% ash and 15% compost and then two thin sub layers of biochar and peat respectively. The four drainage layers at the bottom of the filters consisted of two sand layers and two gravel layers.

Table 2. A schematic presentation of the different layers of the four different types of rain gardens.

Filter composition				
Layer/Filter media	Biochar	Ash	Peat	Control
Top soil	Hekla regnbädd	Hekla regnbädd	Hekla regnbädd	Hekla regnbädd
Main layer	60% Hekla regnbädd 40% Biochar	35% Hekla regnbädd 50% Ash 15% Compost	60% Hekla regnbädd 40% Peat	85% Hekla regnbädd 15% Compost
Specific layer	100 % Biochar	-	100 % Peat	-
Thin buffert layer	-	Biochar	-	-
Thin buffert layer	-	Peat	-	-
Drainage layer 1	0.4–0.8 mm filter sand	0.4–0.8 mm filter sand	0.4–0.8 mm filter sand	0.4–0.8 mm filter sand
Drainage layer 2	0.8–1.2 mm filter sand	0.8–1.2 mm filter sand	0.8–1.2 mm filter sand	0.8–1.2 mm filter sand
Drainage layer 3	3–5 mm gravel	3–5 mm gravel	3–5 mm gravel	3–5 mm gravel
Drainage layer 4	4–8 mm gravel	4–8 mm gravel	4–8 mm gravel	4–8 mm gravel

The rain gardens have several plants which were chosen because of a high tolerance for large amounts of water, drought, salinity, and their ability to break down organic pollutants (Johansson et al., 2023). Of the 13 bioretention filters in total ten have plants, as is presented in Figure 2. For this study, one rain garden with a main layer of ash, peat, and biochar, all with plants, and the control with plants were selected for sampling. The plants are the same in all filters which are *sea-buckthorn* (*Hippophae rhamnoides*), *red fescue* (*Festuca rubra*), *common rush* (*Juncus effusus*), and *sea thrift* (*Armeria maritima*) and these are presented in Figure 3. The bioretention filters are placed in cylinders made of polyethylene with a radius of 0.5 m and height of 1.2 m.

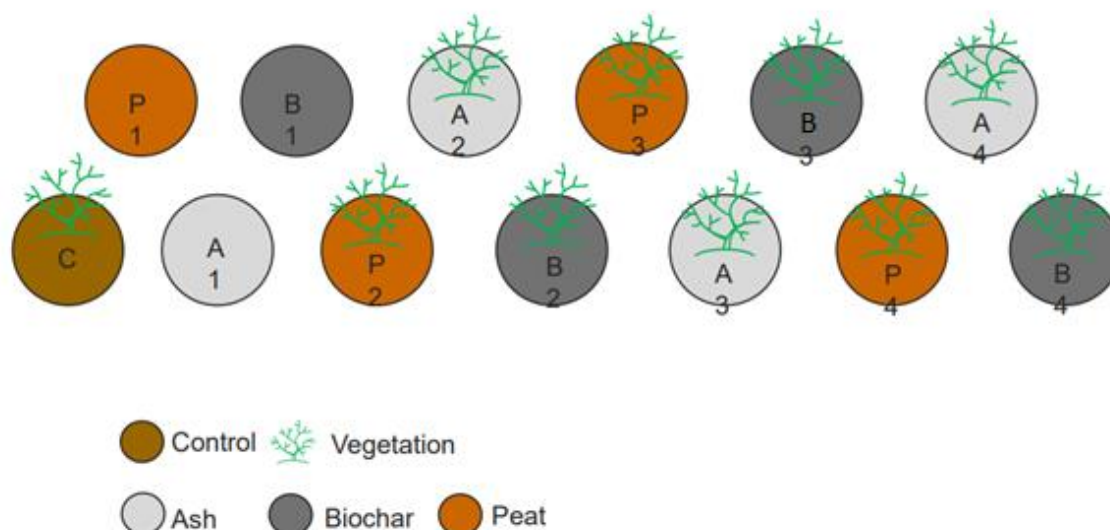


Figure 2. Location and occurrence of plants in the various bioretention filters in Gårda rain garden (from Johansson et al., 2023)



Figure 3. The plants used in the rain gardens, sea-buckthorn (*Hippophae rhamnoides*), red fescue (*Festuca rubra*), common rush (*Juncus effusus*), and sea thrift (*Armeria maritima*), respectively

Stormwater from the catchment area, mainly but not restricted to the highway, are collected in a sedimentation well and then pumped to two IBC tanks near the rain gardens. Thereafter stormwater was pumped from the tanks to the bioretention filters in the rain garden and distributed via watering cans as the experiments took place. Since the stormwater was taken from the catchment under varying weather conditions, the content of pollutants in the stormwater is supposed to vary.

In this project, the bottom ash was diluted due to the fact it is not known how the plants would react in an environment rich in metals. Therefore, a compost material was added to the non-ash, because there was a concern that the plants would not thrive otherwise. The reason for choosing ash was that ash is rich in metals that cannot be recycled otherwise. But here the prospect was that the plants can extract the metals (and other pollutants) from the ash. Then the plants can be combusted and from the fly ashes of that combustion, metals can hopefully be recycled as it is easier to recycle metals from fly ash than from the bottom ash used in this project. During the lifetime of the plants, they will treat pollutants such as OP, nutrients, and microplastics along with the metals (OVAM, 2019). The process responsible for these actions are called phytoremediation and can be further divided into separate processes, such as (phyto)-degradation, extraction or volatilisation. In the phytodegradation plants and microorganisms take up pollutants which then are rendered harmless through metabolic processes and/or enzymatic processes. Furthermore, in phytoextraction the pollutants are stored in the plant tissue after being taken up the plant itself or microorganisms. Finally, in phytovolatilization transpiration is the important process for handling pollutants.

An additional benefit with bottom ash is that the material has a large specific area where particles and dissolved ions can get adsorbed, and it is a draining material. Moreover, this ash is a rest product of combustion, yielding a low cost both in economic and environmental terms. No mining or production is required and on top of that the ash is binding CO₂. Finally, it is necessary to find new areas of use for bottom ash, as today it is mostly landfilled and therefore using it as a filter material is a potential future use.

3.2 Sampling and chemical analysis

When work was commenced in the pilot facility the collected stormwater from the catchment area was examined. Samples from all four types of filters were analysed. The measurements in the influents were done by using a Hanna multimeter instrument (HI 9829) as seen in Figure 4, which measured pH, redox potential, dissolved oxygen (DO), conductivity, turbidity and temperature. These parameters were measured to get an indication on the water quality which then could be used to explain the behaviour of the metals. Moreover, the colour of the sampled water was noted. After that the watering of the rain gardens began, first by adding 100 L, then 260 L and finally 140 L, an example of samples ready to be analysed is seen in Figure 5. The effluent was then measured for the same parameters as the influent. Consequently, the effluent was measured and sampled after 100 L, 360 L and 500 L, i.e., three times in total. The reason for this was to get information on the rain garden efficiency after some water, after more water and lastly, a lot of water infiltrating through the bed. The idea was to try and simulate

cloudburst similar events. These were the samples used for evaluating the TSS, VSS, TOC/DOC, IC and TN.



Figure 4. The Hanna multimeter instrument (HI 9829) used for the measurement of pH, redox potential, conductivity and turbidity.



Figure 5. Samples from the peat filter after addition of 100L, 360L and 500L of polluted stormwater compared with incoming stormwater in the bottle to the left.

Simultaneously, the sampling with soil-water samplers and the passive depth sampling were ongoing but they did not affect one another. Water samples were taken from three different depths: 0.45 m, 0.85 m and 1.14 m, with the third sampling spot being the same spot as the sampling took place for the volume sampling, i.e., the effluent from the filters. The two first ones are sampled by using soil-water samplers from Prenart that were inserted to the rain garden at the two first depths (0.45 m and 0.85 m) and gathered by using a vacuum pump. The samples at the bottom were sampled manually by pumping sample from the effluent storage tanks with a hose pump. The filter size in the soil-water samplers was 1.2 μm , meaning dissolved metals $<1.2 \mu\text{m}$ were analysed compared to receiving samples $<0.45 \mu\text{m}$ in the other samples sent for metal analysis after filtration at the laboratory. Finally, the third sampling was made where passive effluents at two heights/depths were sampled. The depths corresponded to the ones used for the soil-water samplers. In this project, sampling was commenced twice from each of the four bioretention filters in the rain gardens with different filter media. Therefore, the samples from the different filters will be called B I & B II for the first and second biochar samples and so on for the other filters as well.

The samples from these three types of sampling were sent to a commercial lab for chemical analysis of metal concentrations which were analysed by first shaking the sample well and then extracting 20 mL of the sample. Then 5 ml 7M HNO_3 (*DuoPur*) was added and then put in an autoclave for 30 min in 120° C heat. The concentrations are determined by ICP-MS SS-EN ISO 17294-2:2016. In some cases, the concentration of some metals was under the detection limit and then a concentration of one tenth of the detection limit (DL/10) was assumed for the calculation of the concentration. This assumption was made after consultation with the supervisor and examiner.

3.3 Soil-water samplers

The soil-water samplers are from a Danish company called PRENART. The samplers consist of porous PTFE (Teflon) which is saturated before installation with a solution of silica flour and in this way, the benefits of the two can be combined, with the inert character of the former and with the hydrological properties of the latter. The main advantage of this is that the water-repellent property of the PTFE is removed which then leads to a good capillary contact with the pore water in the soil. Moreover, the lifetime of the samplers is said to be long and can be used in different locations (if cleaned in between), easily installed in the soil, resistant to frost and made of sturdy materials. Finally, the result should not be affected by these samplers which do neither contaminate or retain elements or chemical compounds. In these experiments, *PRENART MINI*, OD 12mm, length 45mm, is most applicable for laboratory experiments or studies in thin soil horizons.

3.4 Description of laboratory methods

Total suspended solids (TSS) and volatile suspended solids (VSS) were done by following the ESS Method 340.2: Total Suspended Solids, Mass Balance (revised June 1993), which is described shortly in the next paragraphs (Valley, n.d.). TSS and VSS were measured to find the organic matter in the sampled water. The water sampled was filtered using a vacuum pump and a glass filter paper of 47 mm diameter and particle retention size of 0.7 μm , the setup is presented in Figure 5. When the filter was saturated, it was put in a pre-weighed aluminium cup and then first put in the oven at 105 °C for 18 hours and then weighed again before finally put in a furnace for two hours and weighed again. The weighing after the oven gave TSS while VSS were given the last time the sample was weighed.



Figure 6. The setup used for filtration when measuring TSS and VSS.

For DOC, IC and total dissolved N, the procedure began with filtering the water samples with a syringe filter of which material, 0.45 μm . DOC was measured to check correlation between DOC and metal concentrations. IC to get information regarding which ions are common in the sampled stormwater and total dissolved N to investigate nutrients. However, since all the samples were filtered TOC was not analysed. Furthermore, the conductivity was measured with a multimeter. The desired conductivity for the machine was $300 \pm 100 \mu\text{S}$ and if the measured conductivity exceeded this then the sample was diluted to approximately the desired level. The reason for diluting the samples was to be in the right interval for the machine used. DOC and total dissolved N were measured in the *Shimadzu TOC-V Series* while IC was analysed in the *Thermo-Fischer dionex ICS-900*. Several ions, both cations and anions were measured. Sodium, ammonium, potassium, magnesium, manganese, and calcium were among the former whilst acetate, chloride, bromide, nitrite, nitrate, phosphate and sulphate among the latter.

4. Results and discussion

By comparing the concentrations in the incoming stormwater with the concentration in the effluent samples, the removal efficiency could be calculated with equation 1. RE stands for Removal efficiency, $C_{Inc.Sw}$ for the concentration of the incoming stormwater and C_i for the concentration of the sample analysed. For the removal efficiency, it is then assumed that the volume of water in the inlet is equal to the volume of water in the effluent. This is a rough estimation as some water will get sorbed in the material in the columns. The plants were assumed to take up no water during winter, and evaporation was assumed to be zero. These calculations were done for selected metals and ion, TSS, VSS, DOC, and dissolved TN concentrations.

$$RE = ((C_{Inc.Sw} - C_i)/C_{Inc.Sw}) * 100 \quad (1)$$

4.1 Field study

In the field, several parameters were measured such as pH, redox, conductivity, total dissolved salts, salinity, turbidity and temperature, and results for all the samples are presented in Table 1 in Appendix. For the incoming stormwater, the pH ranged between 7.0–7.5, which is slightly basic. In the effluents a greater variety was found, 3.4–7.5, with the lowest values found in the effluent of the peat filters. The control filters also caused lowered pH, but only to levels around 6.5–6.8. The difference in pH was likely due to the difference of humic acid in the samples, with most humic acids found expected to be found in the peat samples (Kalmykova et al., 2010). In effluent from the ash and the biochar filters the pH had a smaller variation; the ash increased the pH slightly whereas the biochar increased slightly. Regarding the conductivity, it varied much between the samples and since all samples were measured only twice, it is more difficult to draw any conclusions.

4.2 TSS and VSS

For all filters, the removal efficiency (RE) for both TSS and VSS were higher in the first set of sampling compared to the second, with the removal of VSS in the ash filter being the only exception, see Table 3 and 4. The RE for TSS was slightly higher than for VSS for all filters. For TSS, see Table 3, the RE was above 95% for all four filters in the first experiment and above 70% for all the second the time, with peat still above 90% and both ash and the control filter above 80%. Biochar, with just over 70% had the least satisfactory result.

Table 3. Concentrations in influents and effluents, and the removal efficiencies of the two sets of samplings of TSS.

First sampling			Second sampling		
Sampling point	Concentration TSS [mg/L]	Removal efficiency [%]	Sampling point	Concentration TSS [mg/L]	Removal efficiency [%]
Inc. Sw	0.38	-	Inc. Sw	0.07	-
P4-100	0.002	99.5	P4-100	0.001	98.1
P4-360	0.005	98.6	P4-360	0.007	91.0
P4-500	0.002	99.38	P4-500	0.007	90.45
Inc. Sw	0.17	-	Inc. Sw	0.06	-
A4-100	0.002	98.85	A4-100	0.003	94.71
A4-360	0.0007	99.62	A4-360	0.01	81.60
A4-500	0.001	99.45	A4-500	0.01	82.88
Inc. Sw	0.283	-	Inc. Sw	0.05	-
B2-100	0.001	98.22	B2-100	0.007	86.67
B2-360	0.007	97.62	B2-360	0.01	72.84
B2-500	0.005	98.12	B2-500	0.01	74.66
Inc. Sw	0.27	-	Inc. Sw	0.18	-
C-100	0.0128	95.21	C-100	0.0176	90.11
C-360	0.0110	95.88	C-360	0.0208	88.30
C-500	0.0099	96.30	C-500	0.0268	84.94

Regarding VSS, see Table 4, the RE was above 90% for three of the four filters, with ash being the exception the first time. In fact, the ash had the three worst removal efficiencies of all examined filters for VSS. This result is contradictory compared to other results obtained on TSS in the other filter beds of ash in this study. Another interesting result is that for some filters the removal efficiency increased after more water was poured on, RE was higher for 500 L than for 360 L, which indicates good resilience in the rain gardens. There is also a possibility that the initial pouring of water led to leaching of pollutants already trapped in the filter material. However, compared to the first measuring point (after 100 L), there is no increase in TSS or VSS apart from the ash filter. Apart from the ash filter of the second sampling, which gave a negative RE, the lowest RE was found in the second sampling of biochar. Contrary to other results the RE for TSS and VSS was worst in biochar samples.

Table 4. Concentrations and removal efficiencies of the two sets of samplings of VSS.

First sampling			Second sampling		
Sampling point	Concentration VSS [mg/L]	Removal efficiency [%]	Sampling point	Concentration VSS [mg/L]	Removal efficiency [%]
Inc. Sw	0.07		Inc. Sw	0.06	
P4-100	0.001	98.60	P4-100	0.0008	98.05
P4-360	0.007	96.74	P4-360	0.002	90.95
P4-500	0.007	98.60	P4-500	0.0008	90.45
Inc. Sw	0.06		Inc. Sw	0.001	
A4-100	0.003	-11.42	A4-100	0.001	94.71
A4-360	0.01	42.60	A4-360	0.0006	81.60
A4-500	0.01	51.77	A4-500	0.0005	82.88
Inc. Sw	0.05		Inc. Sw	0.04	
B2-100	0.007	96.91	B2-100	0.001	86.67
B2-360	0.01	95.84	B2-360	0.001	72.84
B2-500	0.01	96.36	B2-500	0.002	74.66
Inc. Sw	0.18		Inc. Sw	0.04	
C-100	0.02	91.44	C-100	0.004	90.11
C-360	0.02	92.10	C-360	0.003	88.30
C-500	0.03	92.43	C-500	0.003	84.94

4.3 Anions and cations

In total concentrations of 13 ions were analysed with ion chromatography, and of these five were selected to be examined more closely: two cations, sodium and ammonium and three anions, chloride, nitrate and phosphate and the results are presented in Table 5 and 6. In some samples the concentrations were below the detection limit and then an approximative value was chosen. This occurred for ammonium, nitrate, and phosphate.

The removal efficiency for sodium (Na^+) was generally positive in the ash and peat filters while the biochar and control filters showed positive results one time and negative one time and indicate a lower cation exchange capacity of the biochar (Munera-Echeverri, et al., 2018). However, the removal efficiency was consistently better at the first sampling occasion compared the second. Moreover, for ammonium (NH_4^+) the concentration in several samples was under the detection limit and was therefore assumed to be one tenth of the detection limit in the calculations of RE. Because of this the removal efficiency for the filters were changing a lot and it is difficult to interpret these results. However, peat showed negative RE in all samples while the other three was difficult to draw any conclusions from. A study found that using peat as a growing media can lead to leaching of NH_4^+ in the initial stages (Messiga et al., 2020). However, leaching was only found in the second set of sampling which means that it is more likely to be correlated to treatment capacity in this case. That same study mentioned that biochar is well-suited to treat NH_4^+ due to factors such as surface area and cation exchange capacity.

RE for chloride (Cl^-) was consistently higher at the first sampling occasion compared to the second. Ash had positive RE in both samples and the contrary was found for peat. Biochar had a high RE at the first sampling but leached significantly the second time, and finally the control filter had a RE close to zero at both sampling occasions. Chloride has a high-water solubility which means that treatment of chloride is not easy (Li Y et al., 2020). However, chloride should not be leaching either, as is the case from the second set of sampling, so this is likely a case of insufficient capacity and that the filters need longer time to recover.

The RE of nitrate (NO_3^-) was highest for peat but compared with other ions the RE was quite low. The biochar had one positive and one negative RE, all in the interval between -10 and 10. Ash showed a negative removal efficiency as did the control filter; however, the removal efficiency was worse for the latter. To conclude, the nitrate was not that affected by the treatment, which the results of ± 10 RE shows. Due to the high-water solubility of nitrate, treatment is difficult but often adsorption is the most widely used treatment technique for removal of nitrate (Yong et al., 2022). This difficulty can be seen in the results which are not great.

Finally, phosphate (PO_4^{3-}) which for ash filters mixed had both positive and negative RE, but overall positive. For the biochar, control and peat filters the RE was negative with one or two exceptions.

Table 5. Removal efficiencies for the evaluated ions in the first set of sampling.

First sampling						
Sampling point	Removal efficiency [%]	Na^+	Removal efficiency NH_4^+ [%]	Removal efficiency Cl^- [%]	Removal efficiency NO_3^- [%]	Removal efficiency PO_4^{3-} [%]
P4-100	19.5		0	-6.40	-4.55	-36.0
P4-360	24.9		-94700	-1.54	0	-51.8
P4-500	26.7		-52000	-6.08	0	7.3
A4-100	50.3		99.8	73.9	-4.35	15.8
A4-360	44.1		99.8	26.8	-13.0	-3.24
A4-500	33.1		99.8	20.8	-8.70	-13.1
B2-100	71.0		-700	79.2	9.09	99.9
B2-360	50.6		96.4	57.1	4.55	-25.6
B2-500	34.5		96.4	33.4	4.55	-15.2
C-100	21.4		0	9.21	-145	-23.9
C-360	14.2		-33200	-3.09	99.7	99.9
C-500	2.19		0	-7.68	-27.3	-45.7

Table 6. Removal efficiencies for the evaluated ions in the second set of sampling.

Second sampling						
Sampling point	Removal efficiency [%]	Na^+	Removal efficiency NH_4^+ [%]	Removal efficiency Cl^- [%]	Removal efficiency NO_3^- [%]	Removal efficiency PO_4^{3-} [%]
P4-100	7.34		-65300	-45.7	99.8	0.42
P4-360	5.21		-81900	-18.2	14.8	-8.11
P4-500	-20.2		-2120	-55.0	7.41	-14.6
A4-100	29.2		-30400	8.28	11.5	99.9
A4-360	10.9		-109000	12.1	-23.1	15.5
A4-500	31.2		-34800	47.8	0	22.1
B2-100	-70.9		-2670	-108.5	-4.76	-89100
B2-360	-70.0		-1560	-40.9	-4.76	-91300
B2-500	-90.2		-454	-45.5	-4.76	-105000
C-100	21.4		95.5	-2.16	-44.4	99.9
C-360	-17.28		95.49	-35.77	-59.26	-29.46
C-500	-6.12		95.49	-14.97	-37.04	-29.96

4.4 Metals

All the results for the analysis of metals are presented in Table I in Appendix. Here the results for the eight selected metals will be presented. The first graph illustrates how the influent volume of added water affected the dissolved and total metal concentrations of metals, respectively. The second and third graph show the metals that were affected at different depths, and the graphs are divided into dissolved and total metal. Note that the soil-water samplers had a larger filter of 1.2 μm (collecting all metals less than 1.2 μm) than compared to all the other samples that were filtrated at the laboratory with much smaller 0.45 μm filters. The results for the dissolved particles (< 0.45 μm) are anyway comparable, since the laboratory filtrate the samples, but for the total metal concentrations the filter sizes are different since the first and third sampling does not have a upper limit which the sampling with the soil-water samplers have. In the third type of graphs there will only be results from six filters as AI and BI did not produce sample amounts large enough to conduct analysis on. The order in which the metals will be presented is: As, Pb, Cd, Co, Cu, Cr, Ni and Zn. For BII, the sample amount from the passive sampling was too small to analyse and that is why the BII in those graphs are discontinuous. In the graphs presenting the soil-water sampling and the passive sampling the fourth and last measuring point is an average of the corresponding metal concentration of 100 L, 360 L and 500 L. This to get a complete overview of the impact of depth to the treatment of the metals. Finally, the guidelines values for discharge of polluted water that are presented in some graphs are from the Gothenburg municipality. If no guideline value is found in the graph, the concentration was well below the guideline.

4.4.1 Arsenic

Figure 6a below shows how the concentration of As changes depending on the volume of water added in the influents and assumed to be equal to the volume of water in the effluents. The concentration of dissolved As is increasing from the influent to the effluent in all filters with the highest effluent concentration in AII. In Figure 6b the concentration of total As was increasing in all the filters but B I, which had the same influent and effluent concentration. However, the leaching (removal efficiency was worse) was significantly higher for dissolved As than total As.

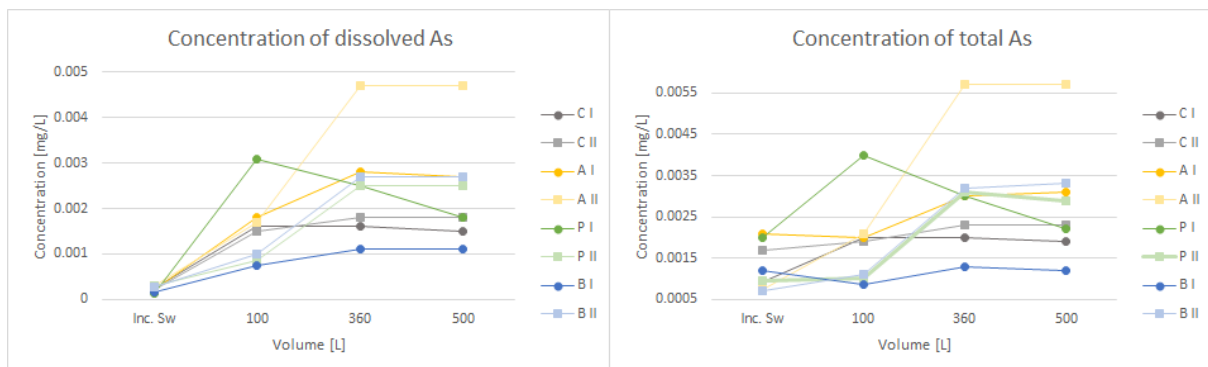


Figure 7ab. The variation in concentrations of dissolved and total As depending on the added volume of water to the influent. Note that all concentrations are far under the Gothenburg guideline value of 0.016 mg/L for polluted water.

The result in Figure 7 is reminiscent of that in Figure 6, i.e., and the concentration of dissolved As in the effluent increased in all eight filters compared to the influent. Note that both graphs have a logarithmic scale. The highest increase was again in AII. Moreover, the concentration of total As increased in all filters but BI, which only had a minor positive RE. Furthermore, the leaching was again, as in the first type of sampling, worse for the dissolved than the total As. However, the concentrations were higher in the latter. The concentration of total As is above the guideline values for several samples, but the concentrations in the effluent is below the guideline values.

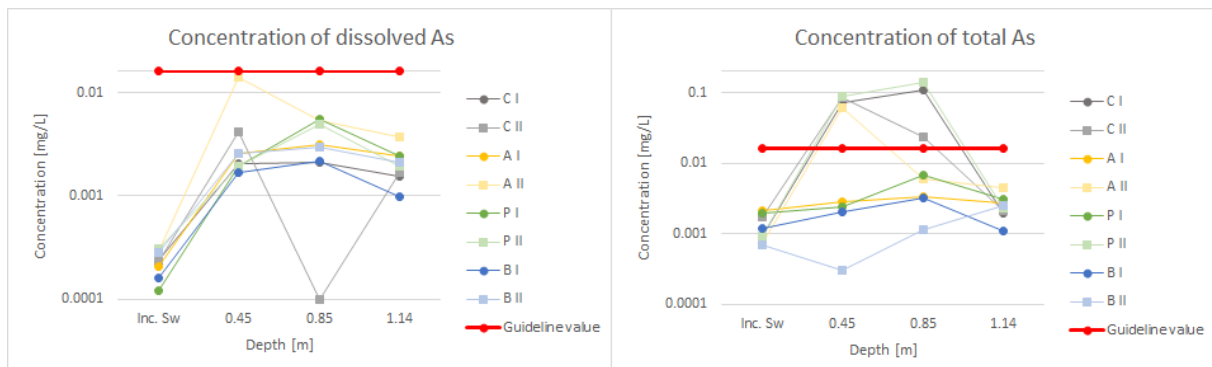


Figure 8ab. The variation of the concentrations of dissolved and total As by depth in the filter columns, and sampled with the soil-water samplers.

As is evident in Figure 8a, the pattern for the dissolved As continued, i.e., As leached from all filters and mostly from AII. Furthermore, total As leached in all filters even though some removal took place in the upper parts of the peat and control filters, as is presented in Figure 8b. Finally, once more the removal efficiency was worse for dissolved As than for total As and a reason for that could be the higher concentration of total As. Both graphs below use a logarithmic scale.



Figure 9ab. The concentrations of dissolved and total As variation by depth in the filters and sampled passively.

The removal efficiency of the dissolved As was negative in almost all samples from all filters, i.e., As was leaching from all filters. Because of the leaching the treatment of stormwater polluted with As did not give a desired outcome. However, the concentrations of As in the effluent were never above the guideline values from the Gothenburg municipality for emissions of polluted water even though some of the samples at intermediate depths were above. Therefore, the leaching does not contribute to additional hazards for the environment. All graphs of As depicts low concentrations of As in the incoming stormwater and higher in latter stages which indicates leaches in the rain gardens. Previous studies on As have found that removal of these metals is better in acidic environments, pH 3-6, since sorption is better in such conditions (Kalmykova et al., 2009). Instead, As is leaching in environments that are suitable for other metals which was the case in this study where the pH seldom was below 6. In order to have a satisfactory level of removal of As a lower pH is required.

4.4.2 Lead

The effect of volume on dissolved and total Pb concentrations are shown in Figure 9a and 9b respectively. For dissolved Pb only A II had a positive net removal efficiency and the rest had a negative net removal efficiency. The opposite was true for total Pb, only P II had a negative net removal efficiency. Interestingly, in the first set of sampling the removal efficiency was at least 85 % for all four filters.

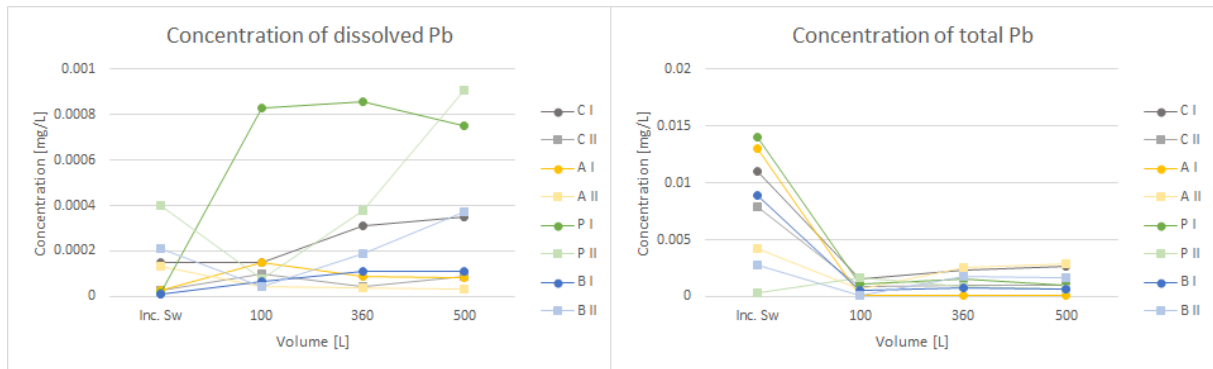


Figure 10ab. The variation in concentrations of dissolved and total Pb depending on the added volume of water to the influent. Note that all concentrations are far under the Gothenburg guideline value of 0.028 mg/L for polluted water.

In figure 10a the concentration of dissolved Pb is significantly higher in B II where leaching is taking place. Because of this substantial difference between filters, the graph uses a logarithmic scale. But from the Appendix it is possible to see that leaching is occurring in all filters but A II. However, for the total Pb, seen in figure 10b all filters had a positive removal efficiency and once more the first sampling gave a better result than the second sampling when comparing removal efficiencies. Additionally, in the first set of sampling the removal efficiency was again over 85 %. The graphs are presented with logarithmic scales.

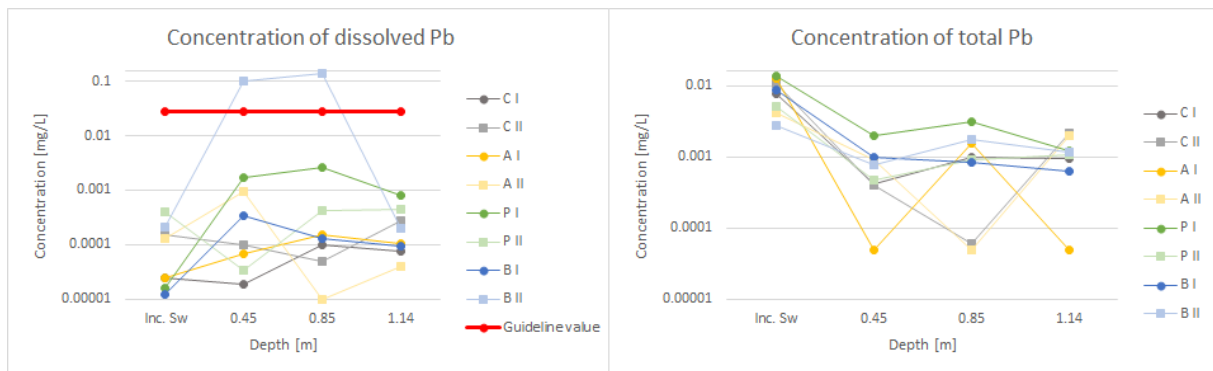


Figure 11ab. The concentrations of dissolved and total Pb variation by depth in the filters and sampled passively.

As with the last graph, the lead concentration in B II is vastly higher than the others. Figure 11a shows the concentration of dissolved Pb, where the net removal efficiency in B II was (just) positive and A II as well, even though leaching occurred in the upper parts of the filter. For the remaining four, i.e., P I, P II and C I and C II the removal efficiency was negative. The concentration of total Pb is presented in figure 11b and shows that the removal efficiency in all filters was net positive. The graphs are presented with logarithmic scales.

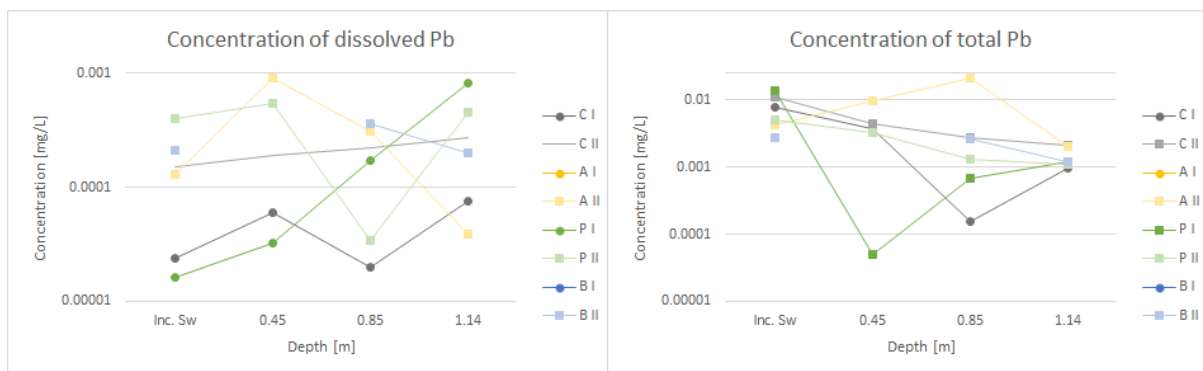


Figure 12ab. How the concentration of dissolved and total Pb, respectively, changed during the passive sampling.

To conclude, the removal of dissolved Pb for was negative for most samples while for total Pb it was positive. The filters that gave a net positive removal efficiency for dissolved Pb were ash and biochar. Interestingly, in the second set of sampling, the removal efficiency was consistently higher than in the first set of sampling, which indicates that the removal efficiency of Pb increased after usage.

4.4.3 Cadmium

The treatment of dissolved and total Cd is presented in Figure 12a and 12b, respectively. The removal efficiency for all filters but A I and B I were negative. For the total metal concentrations, it was only in the peat samples that total Cd was above the detection limit and in both leaching occurred but in P II the end concentration was below the detection limit again. This indicates some treatment in the lower parts of the filter; however, P I indicate the opposite, i.e., continuous leaching, which makes conclusions hard to draw.

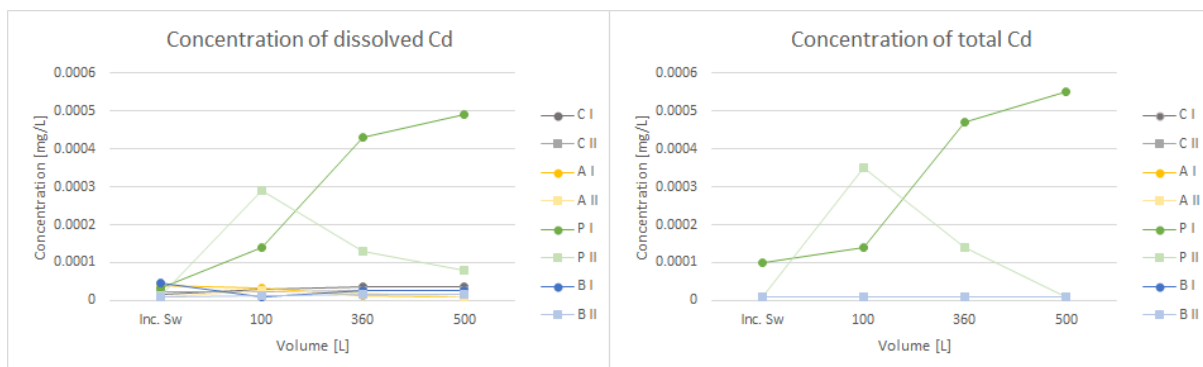


Figure 13ab. The variation in concentrations of dissolved and total Cd depending on the added volume of water to the influent. Note that all concentrations are far under the Gothenburg guideline value of 0.0009 mg/L for polluted water.

The concentrations of dissolved and total Cd are shown in Figure 13a and 13b (the graphs are presented with logarithmic scales). Regarding the dissolved Cd, A I and B I samples leached in the initial parts but recovered and ultimately had a positive removal efficiency. The rest of the six samples leached at all three sampling spots. For total Cd, where the detection limit again would prove decisive, the only sample with an effluent above the detection limit was P I, the remaining seven was below. However, for the influent, not a single one was above the detection limit. Nevertheless, in many of the intermediate steps leaching occurred, but the interesting fact is that in the effluent the concentrations was again below detection limit.

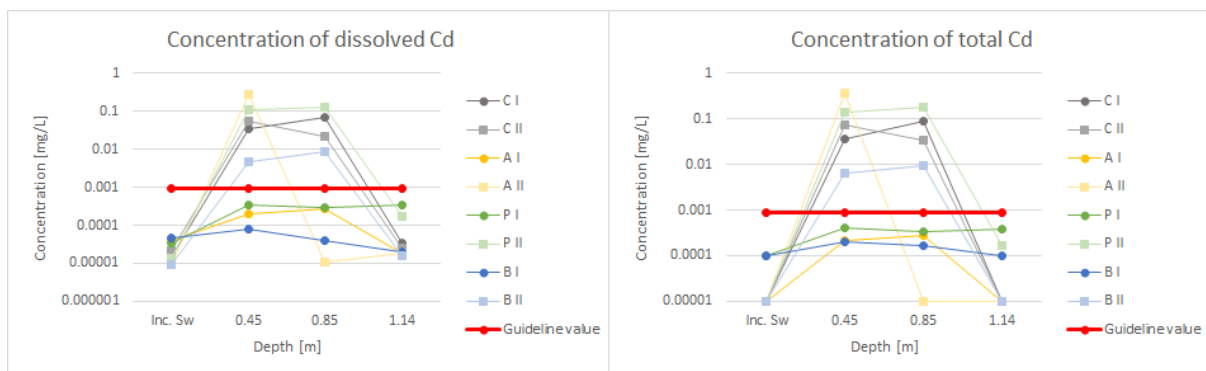


Figure 14ab. The variation of the concentrations of dissolved and total Cd by depth in the filter columns and sampled with the soil-water samplers.

In addition, the concentration of dissolved and total Cd is presented in Figure 14a and 14b, respectively (the graphs are presented with logarithmic scales). Initially, for dissolved Cd the removal efficiency was negative for all samples, which also can be seen in the Appendix. For the total Cd both the influent and effluent for all samples were below the detection limit and also for many intermediate sampling spots. Nonetheless, in P I, P II, A II and C I leaching was found in some of these sampling spots. Furthermore, it was then treated before it left the filter as it was below the detection limit in the effluent.

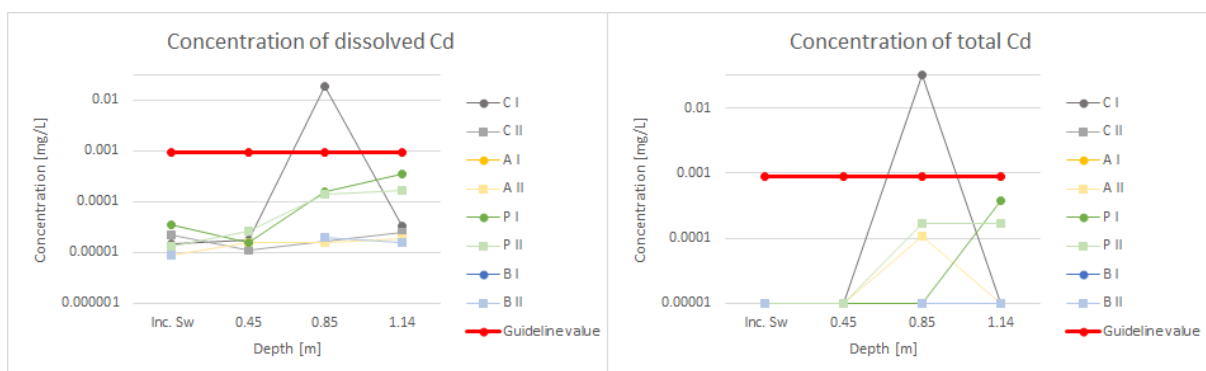


Figure 15ab. The concentrations of dissolved and total Cd variation by depth in the filters and sampled passively.

The removal efficiency for Cd was generally low and on top of that the total Cd was below the detection limit which makes a proper evaluation of the result difficult. For the dissolved Cd the removal efficiency was negative for the control and peat filters in all instances, while the removal efficiency for ash and biochar filters was net positive in the first set of sampling for both volume and soil-water samplers. According to the results Cd leached quite a lot, but any quantitative analysis on the leaching is impossible due to the concentrations being below the detection limit in many samples. However, since the levels of Cd in many samples were below the detection limit, they were well below the guideline value. Therefore, the concentrations of Cd can be seen as satisfactory even though the evaluation of removal efficiency requires further investigation.

4.4.4 Cobalt

The removal of Co is presented in Figure 15a and the removal is positive for all filters but P II and A II. High removal efficiency is found in A I, B I and C II. Continuing with the total Co, presented in Figure 15b, it showed a good net removal efficiency all in samples, with the lowest net removal efficiency at 74 %. Consistently better in the first set of sampling compared with the second set of sampling.

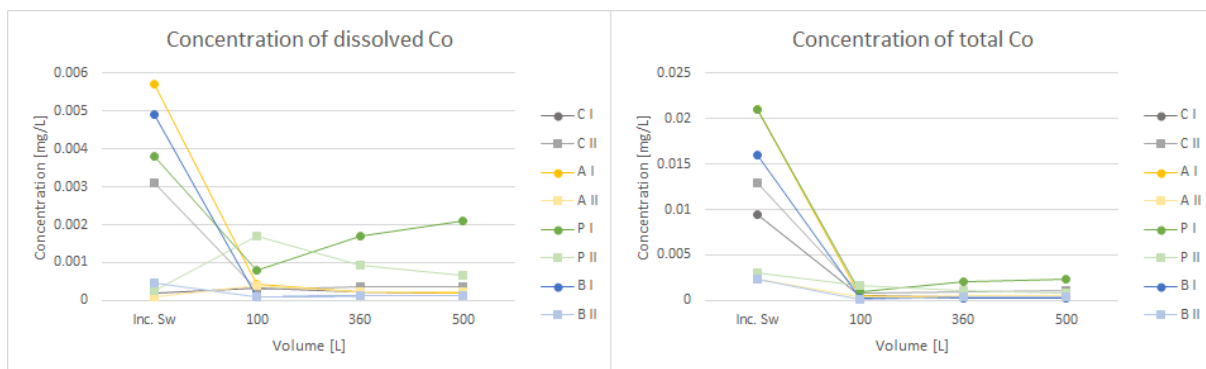


Figure 16ab. The variation in concentrations of dissolved and total Co depending on the added volume of water to the influent.

For the dissolved Co presented in Figure 16a the removal efficiency is positive for A I, P I, B I, B II and C II and negative for remaining three. On top of that, significant leaching occurred in C I, A I and B II as well. Additionally, the concentration of total Co is presented in Figure 16b. All samples had a net positive removal efficiency, however leaching was found in A II, P II, C I and C II at sampling spot two and three. The net removal efficiency was high, especially considering the initial leaching, at least 80 % in all but one, which had 61 %.

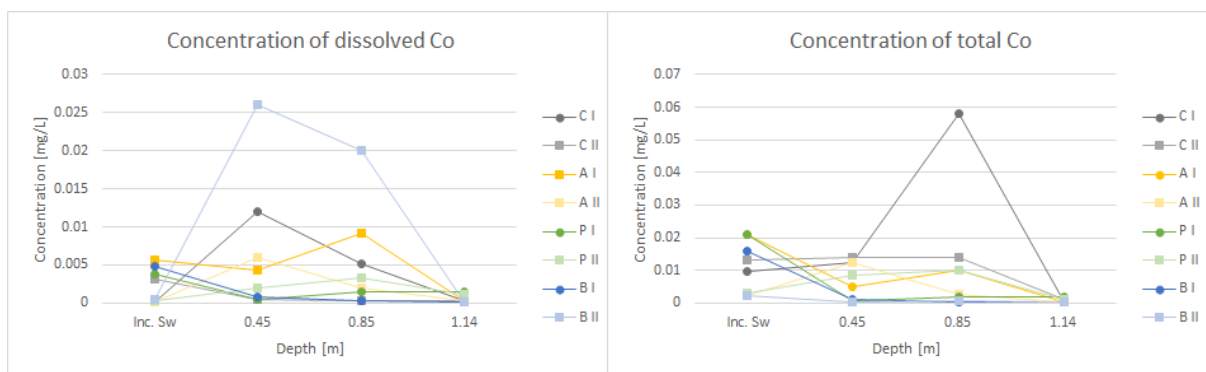


Figure 17ab. The variation of the concentrations of dissolved and total Co by depth in the filter columns and sampled with the soil-water samplers.

For the dissolved concentrations of Co presented in Figure 17a the net removal efficiency was positive for three filters P I, B II and C I, while P II, A II and C II had net negative removal efficiency. However, the significant factor in this graph is the monumental leaching in C II. For the total Co particles, the net removal efficiency was positive for all samples, with higher removal efficiency in the first set of sampling, seen in figure 17b. The graphs are presented with logarithmic scales.

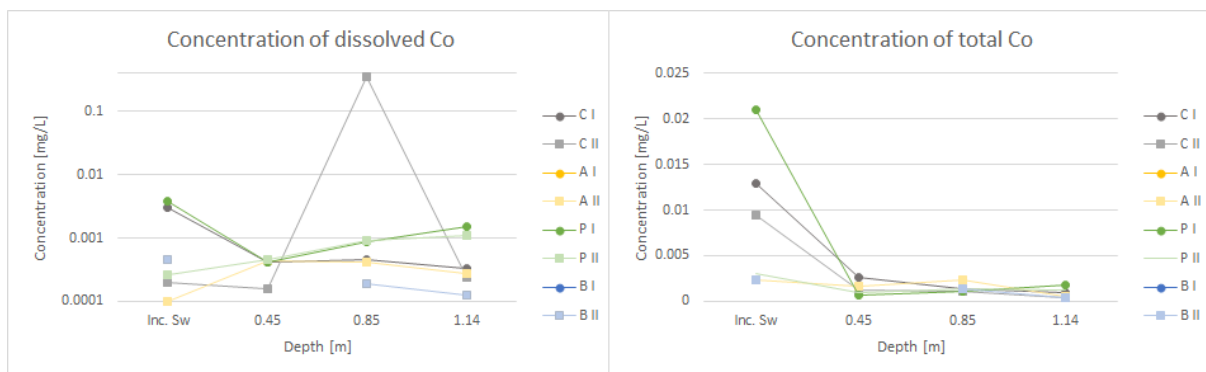


Figure 18ab. The concentrations of dissolved and total Co variation by depth in the filters and sampled passively.

Moreover, the removal efficiency for Co was good, especially for the samples from biochar. However, the second sampling of the dissolved Co in ash, control, and peat filters gave negative removal efficiency. In the sampling with the soil-water samplers, some leaching occurred in the second sampling of the ash, control, and peat filters.

4.4.5 Copper

The impact of volume on the dissolved and total Cu is presented in Figure 18a and 18b, respectively. The removal efficiency was net positive for all dissolved samples of Cu, although some leaching occurred, mainly in P I, and the removal efficiency not very high. For the concentration of total the removal efficiency was positive in all samples, with only minor leaching, never at levels above the concentration in the influent. Additionally, the removal efficiencies from the first set of sampling were higher compared with the second set. When comparing the concentrations of Cu in the effluents with the guideline values, it can be found that several samples exceed the recommended values, both for dissolved and total Cu. Guideline values are exceeded for all four types of filters but the A II exceeded guidelines for all three effluents for both dissolved and total Cu.

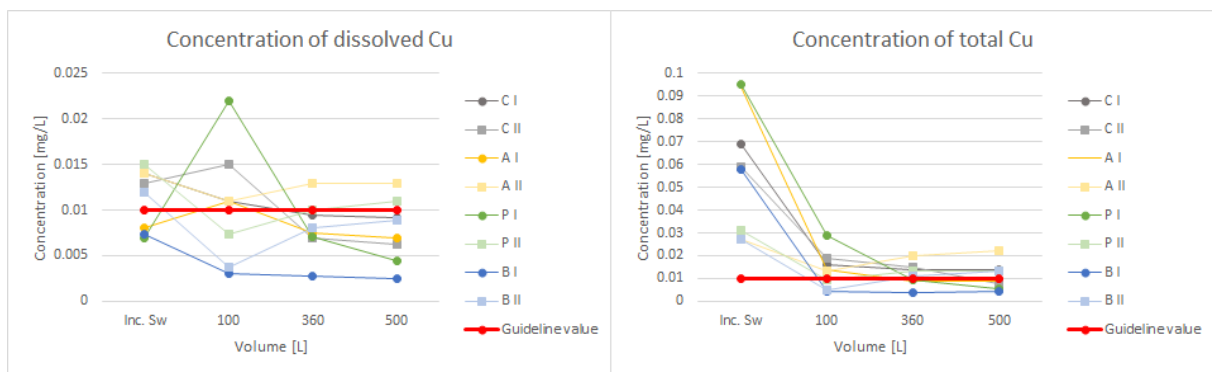


Figure 19ab. The variation in concentrations of dissolved and total Cu depending on the added volume of water to the influent.

Moreover, the impact of depth for the concentrations of dissolved and total Cu is shown in Figure 19a and 19b, respectively. Leaching of dissolved Cu occurred in all samples, however the net removal efficiency was negative only for A I and P I, but significant leaching also took place in B I, B II and A II. Furthermore, for the total Cu the net removal efficiency was positive but there was leaching in other parts of the filters, most notable in B II and A II. Moreover, the guideline values for Cu were again exceeded for both dissolved and total Cu, despite a substantial decrease in concentration before reaching the effluent. One ash and one peat samples exceed guideline values for both dissolved and total Cu while

the control samples exceed only for total Cu, however, regarding total Cu both samples are exceeding guideline values.

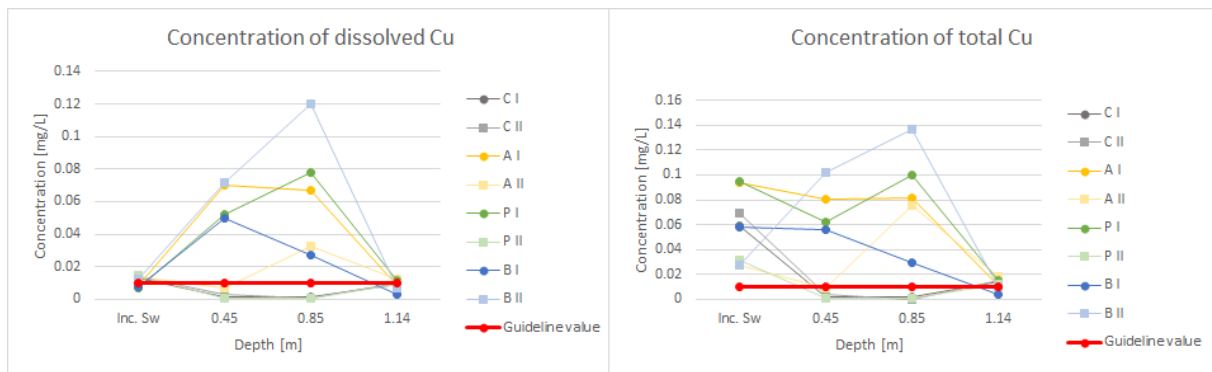


Figure 20ab. The variation of the concentrations of dissolved and total Cu by depth in the filter columns and sampled with the soil-water samplers.

The result of the passive sampling is presented in the figures below, 20a for dissolved Cu and 20b for total Cu. Despite substantial leaching of dissolved Cu at depths 0.45 m and 0.85 m for A II the net removal efficiency was positive and in fact only P I had a net negative removal efficiency. However, quite low removal efficiencies overall. Moreover, for the total Cu the removal efficiency was positive, despite pivotal leaching in A II, for all samples. Once again, the guideline values are exceeded in one ash and one peat sample for dissolved Cu while both samples of peat and control filter and the one ash sample exceeds the guideline values for total Cu.

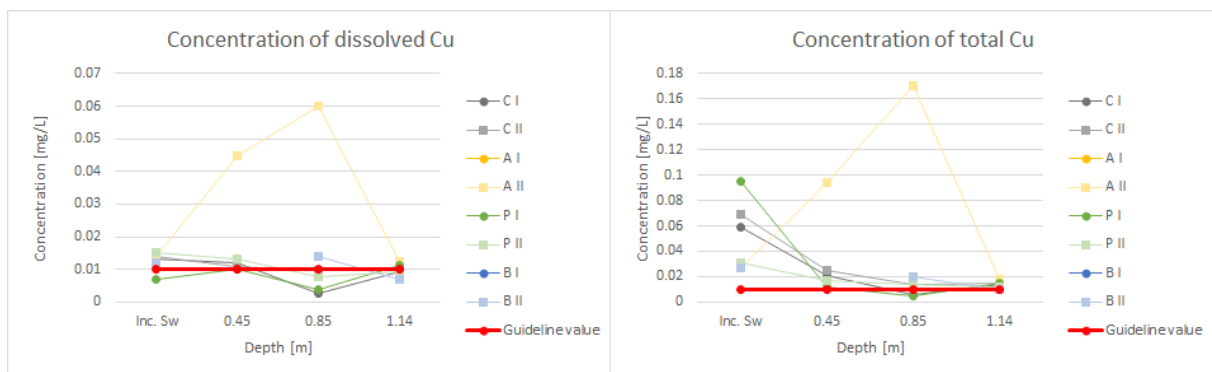


Figure 21ab. The concentrations of dissolved and total Cu variation by depth in the filters and sampled passively.

Cu responded desirable to the treatment of the rain gardens since both the biochar and control filters consistently showed positive removal efficiency. The ash and peat filters had both positive and negative net removal efficiencies and generally total Cu had better removal than dissolved Cu. Generally, the sampling with induced volume gave a good removal efficiency whereas the soil-water samplers showed some leaching.

4.4.6 Chromium

For the dissolved Cr the impact on concentrations by volume is presented in Figure 21a. In all eight samples the removal efficiency was net negative. However, some treatment took place but was reversed by the time the last sampling was made. Continuing with the impact of volume on the total Cr particles, which is presented in figure 21b. P II had a negative removal efficiency but in the other filters the removal efficiency was positive. Additionally, the removal efficiency of the first set of sampling was better compared with the second time. Several samples exceed the guideline value for Cr, regarding dissolved particles two samples of ash and one biochar. For total Cr, three ash and control samples and two peat and biochar samples.

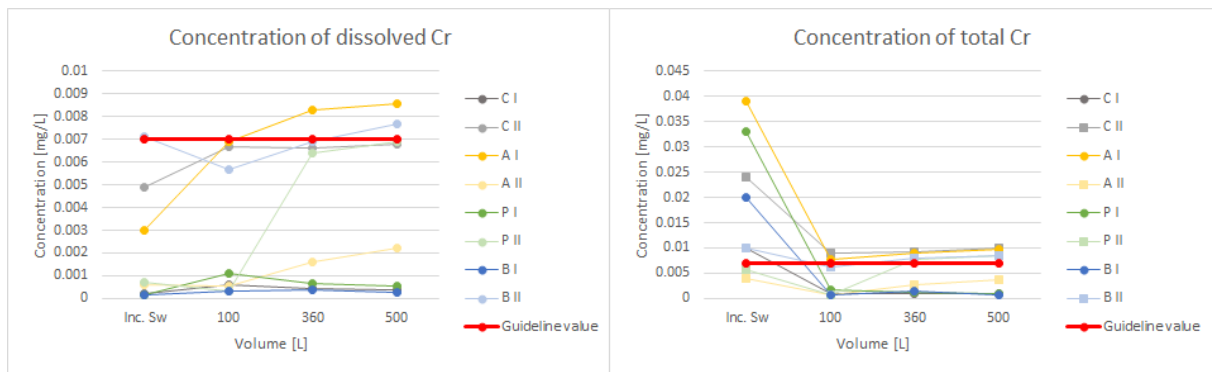


Figure 22ab. The variation in concentrations of dissolved and total Cr depending on the added volume of water to the influent.

The impact of depth for the concentration of Cr sampled by the soil-water samplers is presented in Figure 22a, for dissolved and 22b, for total Cr. For the former the removal was only (just) positive for B II and negative for the remaining seven. Noticeable is the leaching in P II. On to the latter where all samples showed a net positive removal efficiency and moreover the removal efficiency was higher in the first set of sampling. Only one sample of dissolved Cr is eclipsed the guideline value and that was an ash sample while one sample each from ash, biochar and control filters exceeded guideline value for total Cr. The graphs are presented with logarithmic scales.

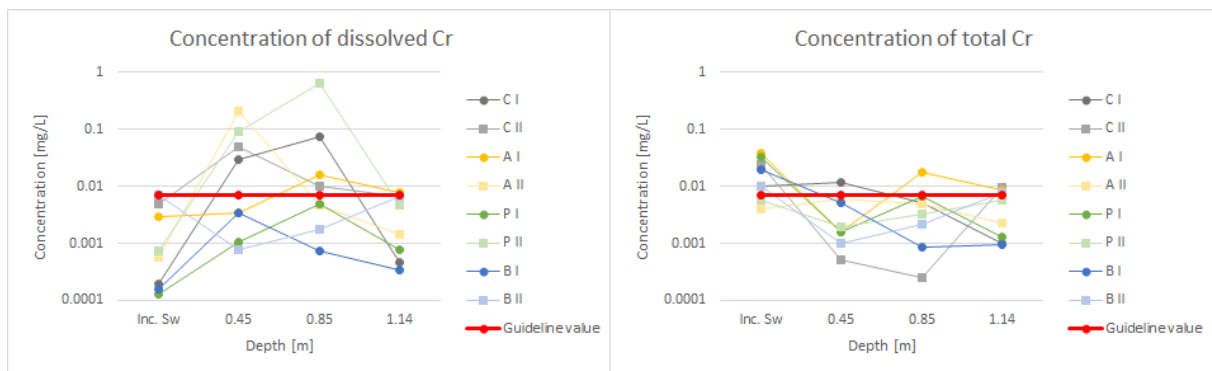


Figure 23ab. The variation of the concentrations of dissolved and total Cr by depth in the filter columns and sampled with the soil-water samplers.

Finally, the Cr concentrations for dissolved and total particles is presented in Figure 23a and 23b, respectively. For the former the net removal efficiency was negative for all filters but B II. Continuing with the total particles, the net removal efficiency positive for all filters but there are leaching occurring in all filters initially. No sample of dissolved Cr eclipsed the guideline value while for total Cr one sample from both biochar and control filters exceeded the guideline value. The graphs are presented with logarithmic scales.

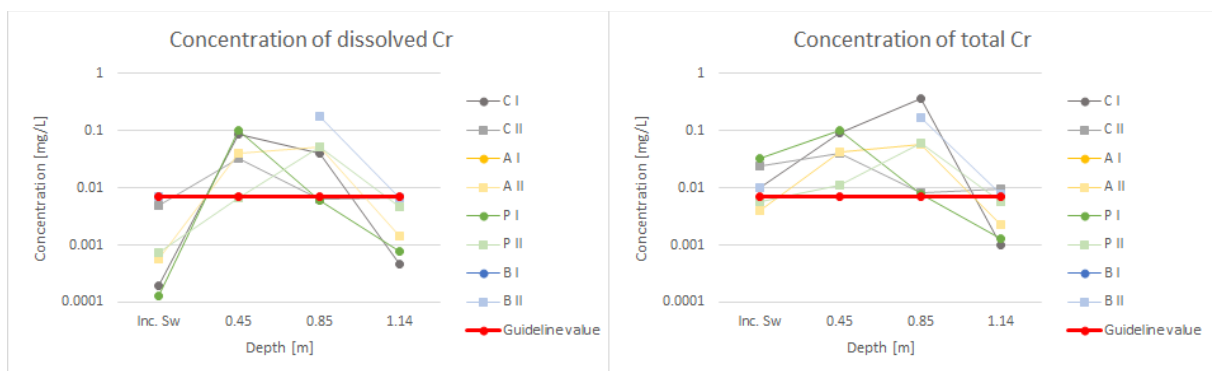


Figure 24ab. The concentrations of dissolved and total Cr variation by depth in the filters and sampled passively.

The removal efficiency of Cr proved to be good for the total metal but not the dissolved which leached a lot; hence the removal efficiency was negative. However, there were some exceptions, mainly positive removal efficiency in biochar samples, but generally this holds true. Finally, total Cr leached in the initial stages of both the soil-water sampling and the passive sampling but did recover in the latter stages, this could be because of the filter sizes, more on that later. As was the case with As removal, removal of Cr is better in acidic environments (Kalmykova et al., 2009). This is the reason why there are leaching of Cr occurring. A lowered pH would be beneficial for Cr but not for the majority of the other metals.

4.4.7 Nickel

The second to last metal presented is Ni, where the dissolved Ni is in figure 24a and total Ni is in 24b. In the case of the former, only two filters have net negative removal efficiency and that was A I and A II. While leaching occurs in all filters but the biochar and C II, the net removal efficiency was positive for all filters that are not ash. For the total particles the treatment of stormwater was better, and the removal efficiency is positive in all filters, even if A II is only in single digits.

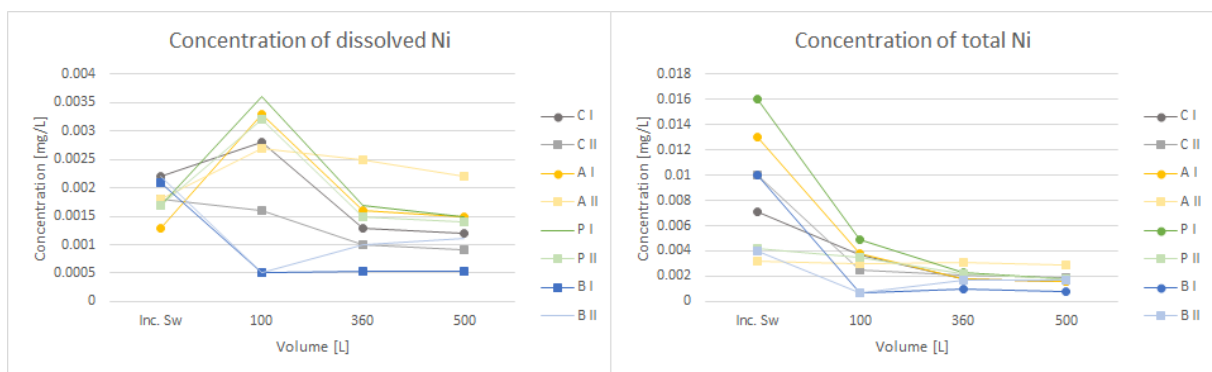


Figure 25ab. The variation in concentrations of dissolved and total Ni depending on the added volume of water to the influent. Note that all concentrations are far under the Gothenburg guideline value of 0.068 mg/L for polluted water.

The soil-water samplers' impact on the concentration of Ni over the depth of the filters is presented in figure 25a, for dissolved Ni, and figure 25b, for the total Ni. For dissolved Ni leaching was occurring in all filters but it was only half of them, A I, A II, P I and P II, that had a net negative removal efficiency. Consequently, B I, B II, C I and C II had a net positive removal efficiency. Moreover, the pattern was similar for the total Ni, leaching initially and then recovering for a net positive removal efficiency for all samples, not just the biochar and control samples. Remarkable leaching of both dissolved and total Ni in A I. No guideline value for Ni was transcended in the effluent, although some samples at higher depths did so. The graphs are presented with logarithmic scales.

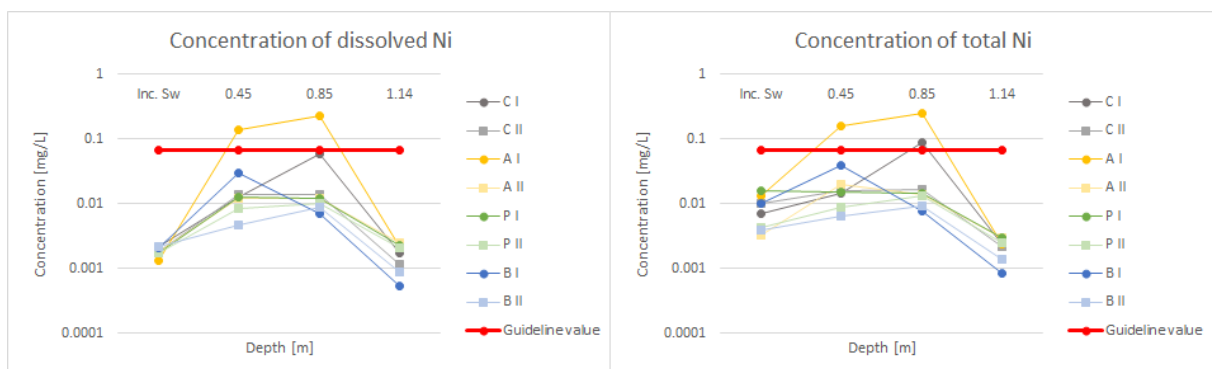


Figure 26ab. The variation of the concentrations of dissolved and total Ni by depth in the filter columns and sampled with the soil-water samplers.

Continuing with the third graphs, presenting dissolved Ni in figure 26a and total Ni in figure 26b. The pattern for the dissolved Ni was similar to the soil-water samplers, with A II, P I and P II having net

negative removal efficiency and consequently B II, C I and C II had positive removal efficiency. Furthermore, the net removal efficiency for total Ni was positive for all filters, even though the A II leached significantly. The graphs are presented with logarithmic scales.

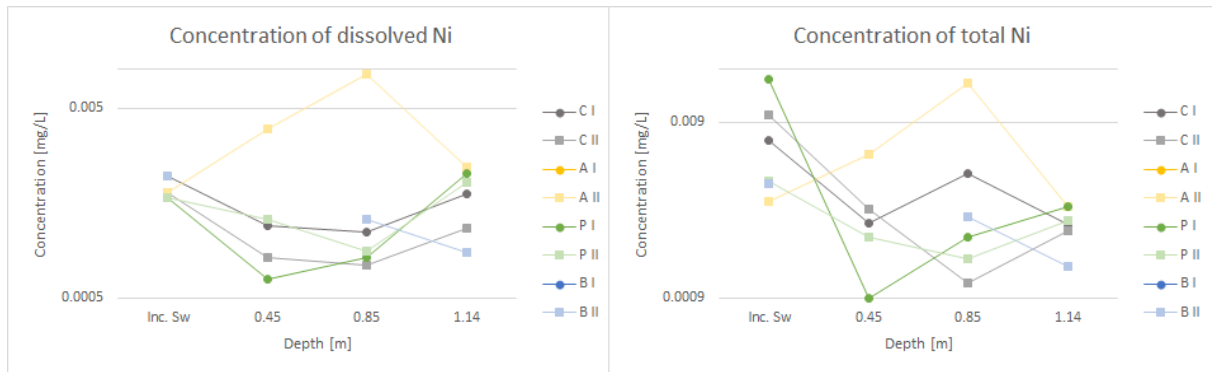


Figure 27ab. The concentrations of dissolved and total Ni variation by depth in the filters and sampled passively.

Furthermore, Ni showed good net removal efficiency overall for the total metal, even though there was some leaching, mainly in the samples from the soil-water samplers. The removal efficiency for biochar and control samples was consistently positive for dissolved Ni, whereas the peat samples were both positive and negative. Finally, the removal efficiency for the ash samples was consistently negative.

4.4.8 Zinc

Finally, the dissolved and total concentrations of Zn is presented in figure 27a and figure 27b, respectively. For the dissolved Zn P I and P II have a net negative removal efficiency while the other filters have positive removal efficiency of at least 73 %, which is quite high. Furthermore, the removal efficiency for the total Zn is positive in all filters. This is also evident in the graph. Only one sample, P I had concentrations higher than the guideline value but for two added volumes 360 L and 500 L for both the dissolved and total Zn.

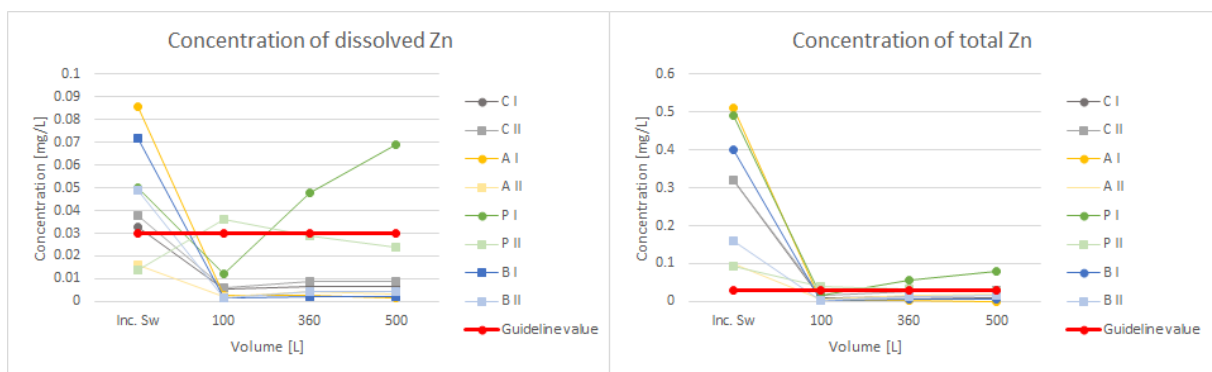


Figure 28ab. The variation in concentrations of dissolved and total Zn depending on the added volume of water to the influent.

The concentrations for the Zn that was dissolved in the soil-water samplers are shown in figure 28a and figure 28b. The former presents the dissolved Zn, which has a net negative removal efficiency in P II and some leaching in P I, A II and C I but apart from P II the net removal efficiency is positive. For the latter the A I, P I and P II has some leaching, but they are recovering by the time the water is reaching the effluent. To summarise, the net removal efficiency is positive for all filters. Better removal efficiency in the first set of samples compared to the second. The guideline value for dissolved Zn was exceeded

in the P I sample and for total Zn both peat samples exceeded the guideline value. The graphs are presented with logarithmic scales.

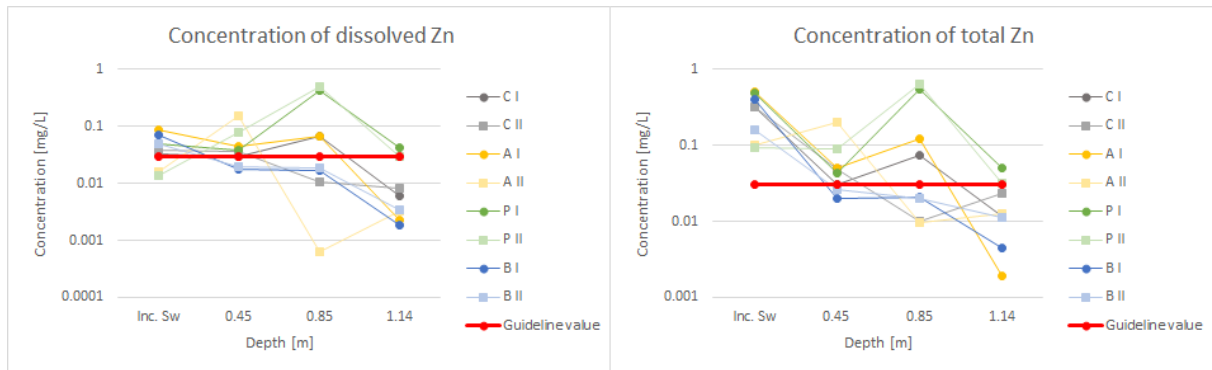


Figure 29ab. The variation of the concentrations of dissolved and total Zn by depth in the filter columns and sampled with the soil-water samplers.

For the second depth graph showing the dissolved and total concentrations of Zn in figure 29a and 29b, respectively. The dissolved Zn has some leaching P I, P II and A II but only P II has net negative removal efficiency. Consequently, P I, B II, A II, C I and C II had a net positive removal efficiency. Finally, the removal efficiency for total Zn was net positive for all filters but A II leached from a depth of 0.45 m to 0.85 m. Even though the sample size was halved, it should be noted that the first set of sampling had a higher removal efficiency than the second. The samples exceeding the guideline value was P I for dissolved Zn and both peat filters for total Zn.

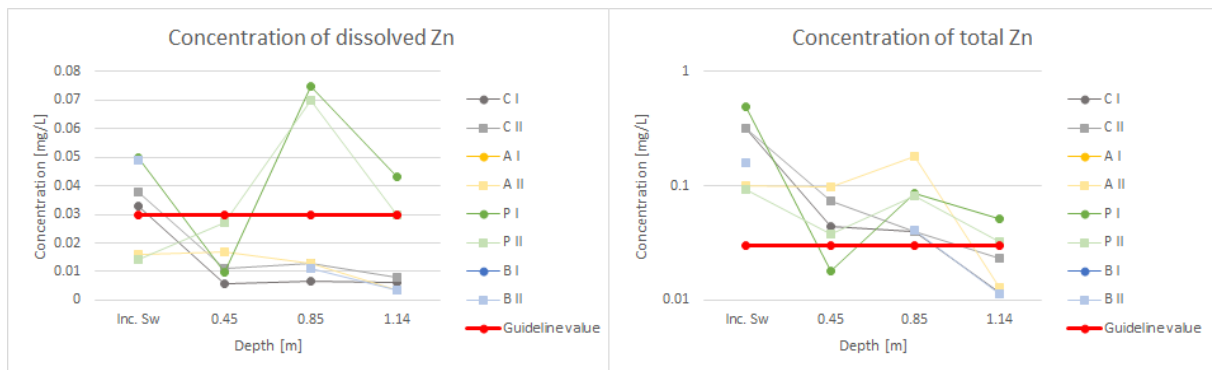


Figure 30ab. The concentrations of dissolved and total Zn variation by depth in the filters and sampled passively.

A short summary of the treatment metals gave the following result. Overall, the metal that had the highest removal efficiency was Zn. Zn was removed in all filters, except for the peat filters, for all three sampling methods. Some leaching of Zn was detected in the ash and control filters, however, the net removal efficiency for all samples of the ash and control filters was positive at all occasions, and was quite high (>70%).

4.4.9 Difference in removal efficiency between dissolved and total metal fraction concentrations

In general, the result for dissolved metals had both positive and negative RE whereas the total metal concentrations had a positive RE. There were two exceptions, total As and total Cd, but for the latter the concentrations were so low, around the quantification limit of the analysis method, that it was not possible to draw any conclusions for the removal efficiency. The remaining six metals, i.e., Pb, Co, Cu, Cr, Ni, and Zn, had a positive removal efficiency. For dissolved metals four metals, As, Pb, Cd, and Cr, had a negative RE while the remaining four had a positive removal efficiency. This pattern was consistent for the three types of sampling. A similar conclusion was found in another study, that it

is easier to treat total metals than dissolved metals (Lange et al., 2020). For example, larger particles can get stuck as they are passing through the filter which rarely happens for dissolved particles.

4.5 Correlation

Selected chemical analysis parameters were paired, and their correlation was investigated. All the eight metals were compared with pH, redox potential and DOC. The full result is presented in Table x in the Appendix. Only a few correlations could be found. For example, Zn, both dissolved and total Zn, had a correlation of 0.5 to the redox, -0.5 to pH and -0.4 to DOC. Dissolved As had a correlation of -0.5 to pH and -0.4 to Co and other than that no significant correlations were found.

Additionally, pH and TOC, pH and conductivity and redox and conductivity were compared and nitrate and ammonium. However, the correlation between the different parameters was between -0.25 – 0.26 which corresponds only to a weak correlation, either positively or negatively.

4.6 General discussion

The recommended guideline values from the Gothenburg municipality are exceeded for several metals in the effluent. These metals are Cu, Cr, and Zn and this was found for multiple samples. This means that the treatment of these metals is not sufficient, even though Zn for example has a high removal efficiency. Additional treatment is therefore required in order to fulfill the guideline values. However, the pollutants are not originating from activities this project is responsible for and therefore there is no requirement to fulfill these requirements as a part of this project. The guideline values from the Gothenburg municipality is simply to have something to compare with. Exceeding values can be found in effluents from all four filters but especially peat has several accounts of exceeding the guideline values. Effluents exceeding the guideline values come from ash and the control filters as well, while the biochar filters only exceed in two occasions.

Previous studies have shown a high removal of Cd, Cu, Ni, pb and Zn in peat filters but not satisfactory results for As and Cr (Kalmykova et al., 2009). A high RE for Pb, Cu and Zn have been found in other studies as well, showing that certain metals are well-suited for removal in bioretention (Davis et al. 2003).

The fact that Cu, Cr, and Zn, exceeded the guideline values that the Gothenburg municipality have set means that to oblige these demands a further treatment step must be implemented before the water can be released into nearby water bodies. However, this is not a responsibility for this project, but it is what is needed to reach the goals set by the Gothenburg municipality. These three metals had a very high removal efficiency, but the concentrations in the incoming stormwater were too high. This result is desired and should be the main takeaway from this project, not that fact that the metals are exceeding guideline values since some of them did that already before treatment. Effluents from all four filters had concentrations exceeding guideline values. On top of that, it was not only the total metal concentrations that were above the guideline values, but this was also found for dissolved metals which is more of a concern. This is because dissolved particles can easier but bioaccumulated and thereby cause harm to both individual animals and the ecosystem in large. A previous study from (Johansson et al., 2023) showed that Cu exceeded the guideline values but not Zn, which was the case in this study. Overall, the studies conducted shows a high RE of several metals from the start. Those studies were done during summer months when the plants are active and in this project, sampling was done during winter, so it is likely that the efficiency is a bit lower.

A factor increasing the uncertainties of the result was the fact that some metals were under the limit of quantification, both for influent and effluents and at different depths. The metals were Pb and especially Cd, and in both cases, it was the total metal concentration that was under the detection limit. For other metals, there were some samples where the concentration was under the limit of quantification but since it was only a few occasions it did not impact the result as much. When a sample had a concentration below the detection limit an assumption of one tenth of the detection limit was made. The removal

efficiency of some of these samples was therefore zero because of this and in other cases the removal efficiency of other samples was very low, indicating considerable leaching. Since the concentration was not known, this was deemed to be the best solution, but it does not remove any of the uncertainties that the low concentrations cause. An interesting fact is that it was the total Cd that was below detection limit, but the dissolved Cd was, even though the dissolved should be a part of the total concentration of Cd. However, the reason for this was that the detection limit was lower for the dissolved Cd compared with the total Cd. The guideline value for Cd was nine times higher than the detection limit and therefore concentrations below the guideline value is no cause for harm.

To rectify this and remove the uncertainties more sampling is recommended and/or finer instruments during measuring. For the project as whole, this was a minor problem but for total Pb and Cd this affected the result. However, the result of the total Cd showed that leaching occurred in many filters, but quantifying the leaching exactly was not possible.

A pattern that was found for many metals were that leaching occurred at 0.45 or 0.85 m for both dissolved and total metals and both in the soil-water sampling and the passive depth sampling. This was most common for the dissolved soil-water samples but also the dissolved samples of the passive sampling showed leaching. However, since the former had more samples (8 compared to 6) than the latter it is possible that the result is more similar. The results of the dissolved soil-water samples had higher peaks in concentration. Even though leaching occurred the treatment worked satisfactory in the lower parts of the rain gardens since the concentrations in the effluent was significantly lower than the peak concentrations. Leaching occurred in all four filters, but ash was the filter with most leaching.

When comparing the total metal concentrations of the latter two sampling types, a surprising result occurs. The concentrations of several metals were higher in the soil-water samples than in the passive depth sampling even though the latter had a larger filter size. A more expected result would have been the opposite and then the difference would have been explained by the difference in filter size.

When discussing passive sampling, i.e., the third type of sampling it is important to take into consideration that only the control and peat filters were sampled twice. Due to insufficient volumes of the samples from the ash and biochar filters in the first round of sampling, it was not possible to obtain results. Because of this the result from the biochar and ash filters was weighed higher and impacted more than it otherwise would. Those two samplings would have been nice to have to compare the result of the ash and biochar samples.

Overall, the fact that there were only two sampling occasions for the four filters adds some uncertainty to the result. Take the first sampling in the ash filter where the removal of VSS was significantly lower than for comparable samples, even a negative removal efficiency. In fact, the removal efficiency of VSS for the three samples was the three lowest obtained for both TSS and VSS removal. Therefore, it is likely that a mistake was made, either in sampling or during measuring. Since a decision was made to sample twice per filter the impact of this was larger than it would otherwise be if more sampling sessions had been carried out. In other cases, this could have been discarded as an outlier. However, due to time constraints it was deemed not possible to have another set of sampling sessions. This was the compromise that had to be made. However, since this project is a part of a bigger project more data on the treatment is available in this larger study by Johansson.

On a couple of occasions, the removal efficiency was higher in the first set of samples compared to the second set of sampling for all the filters. This was the case for total Pb, Co, TSS, Na⁺ and for both dissolved and total Zn. Regarding the metals, this was the case for all three types of sampling, volume dependent, soil-water sampling and passive depth sampling. In addition to this, total and dissolved Cu and total Cr also showed this pattern for the volume sampling and finally total Cr for the passive sampling as well. Finally, if the result from the first ash sampling of VSS is disregarded then it could be argued that VSS shows this pattern as well. The tendency found here was that the filters need some time

to recover before treatment can be as efficient as it was the first time. It should also be noted that the difference in removal efficiency differed between the various measured parameters, in some cases the difference was very small but in other cases the difference was significant. A possible explanation for this pattern is that the processes that handle pollutants in the rain gardens need more time to convert/bind them to less harmful compounds or transport to other parts of the plants (OVAM, 2019). For example, up take by plants or decomposition by bacteria. During the colder climate of winter, there are less activity in the plants and by bacteria and therefore the treatment process is probably more time consuming than it would be during other times of the year, when for example the energy obtained from sun light is more accessible.

However, the opposite was found in seven instances, dissolved Pb in all three different types of sampling and dissolved Cu and Ni in the passive sampling, again with the caveat that only two of four filters could be compared. Finally, Mg^{2+} and Ca^{2+} also had better removal in the second set of sampling. This was significantly fewer than the opposite. However, it is interesting that dissolved Pb consistently had better removal efficiency in the second set of sampling, since it was the only metal that had this result.

The best treatment of stormwater occurred in the first sampling of total metal particles where concentrations were measured after 100, 360 & 500 L, respectively. Except for As, which was not treated in a satisfactory fraction, and Cd, which was below the detection limit, the remaining six metals (Pb, Co, Cu, Cr, Ni & Zn) had a high removal efficiency and the concentrations were significantly lower in the effluent compared to the influent. In comparison, for dissolved particles, four metals were found to be leaching, As, Pb, Cd, Cu, Cr, while Co and Ni had a positive removal efficiency. Finally, Ni showed satisfactory removal for all samples but the ones that had passed through the ash filters.

It is easier to treat total metals compared to dissolved particles since the latter cannot be treated by physical infiltration (Lange et al., 2022). This was found in this study as well; the removal efficiency of total metals was consistently better than for dissolved. For example, both dissolved Pb and Cr leached but the total metal concentration for the corresponding samples had a positive removal efficiency. The removal activities responsible for removal of dissolved did not have a high enough capacity which could be because the study was done during winter. Lange also mentions that salt can have an impact on removal of dissolved metals and since this was during winter the highway where the collected stormwater flowed was likely treated with de-icing salts. This could be another reason for the leaching of dissolved metals.

Looking at the removal efficiency of metals in the four different filters, the result found was that the biochar filter was best, the control filter second best, followed by the ash filter and last the peat filter. Even though intermittent leaching, in large due to filter size, took place in the second and third sampling method the result in the end was the same for all three sampling methods. It was a bit surprising that the ash filters performed better than the peat filters since the latter consists of relatively high amounts of metals. The biochar filter had the best removal efficiency, quantitatively, in the soil-water sampling, while the other three had similar results, quantitatively, for all three sampling methods.

Regarding using soil-water samplers as an approach to measure metal concentrations from rain gardens, the fact that the filter size, 1.2 μm , is larger than the conventional filter size, 0.45 μm is not optimal. Because of this, when evaluating the graphs illustrating the changes in concentration, it is very likely that a concentration spike will appear. This could give the illusion that significant leaching is taking place in the upper half of the filter when it rather is two different filter sizes that are being compared and causing the irregular result. If it were possible to have soil-water samplers with a smaller filter size, interpretations of the result would be easier. It is difficult to draw any conclusions regarding leaching/treatment since only the passive sampling had the same filter size at all sampling spots, but the sampling was not complete, with only the peat and control filter having two samples. Both ash and biochar had one insufficient sample and then biochar had one sampling with only three out of four sampling spots being accounted for. If these problems could be amended, then soil-water samplers could

be used. However, having a filter for the total metal concentrations is not ideal since usually no filter is used and the result for total metals then a skewed a bit as well.

Regarding the soil-water samplers' suitability as a device for tracking treatment the filter size must be taken into consideration. Since the filter size is 1.2 μm it is not consistent with either dissolved (0.45 μm) or total metals the result will always be skewed in some way. In this instance the sampling with soil-water samplers caused what appeared to be a significant leaching for the dissolved metals. However, this could also be because of a nearly tripled filter size. The total metal concentrations are limited since not all particles cannot be accounted for with a filter. This is a problem whichever filter is chosen so therefore a decision must be made, is it more important to have precise sampling for either dissolved or total metals or is it better to have samples that can provide some information about both. It depends on what the objectives of the conducted study.

5. Conclusion

For the sampling done after the rain gardens had been irrigated with large amounts of water, they showed a good removal efficiency for most metals, especially when the metals were in total form, with As and Cd being the exceptions. As leached from all filters, whilst Cd either leached in the intermediate steps or was below the limit of quantification. Dissolved Pb, Cd, and Cr leached substantially, however, for the rest of the metals the removal efficiency was high.

Furthermore, the sampling with soil-water samplers indicated leaching in the intermediate steps for all metals but the overall removal efficiency of the metals was still more positive than negative. Leaching was found for As, dissolved Pb, Cd, dissolved Cr. Additionally, some leaching was found in dissolved Co, dissolved Cu, and dissolved Ni but these also had several samples where the removal efficiency was high. Pb, Co, Cu, Cr, and Ni had a positive removal efficiency for the total metal concentration and finally, Zn had a high for both forms. Depending on the leaching of the metals not occurring in the stormwater, it is recommended to analyse the original concentrations of these metals in the materials used in the bioretention filters.

Moreover, for the passive sampling, the following metals was found to leach, As, dissolved Pb, Cd, dissolved Cr. Dissolved Co and dissolved Ni had mixed results and total Pb, total Co, total Cu, total Cr, total Ni and Zn had a high removal efficiency.

A final note that should be mentioned is the fact that for the volume and passive sampling Ni leached consistently in the ash filters. Ni responded desirably to the treatment in the other filters which makes this a bit surprising.

Regarding the performance of the filters, the biochar filters had the best result for all three sampling methods. The second best was the control filter, third ash and the peat filter had the worst result. This was also consistent for the different sampling methods, although it should be noted that for the ash and biochar, only one sampling was made in the third, passive sampling.

Finally, regarding the guideline values of the Gothenburg municipality, the guideline values were only exceeded for, Cu, Cr, and Zn. However, they were still exceeded in several samples, especially in the peat filter effluent. The biochar filters had the best result concerning guideline values and the ash and control filters had similar results. To summarise, for both removal efficiency and guideline values, the biochar filter was best, followed by ash, and the control filter performing similarly, and peat performed worst.

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Appendix

Concentration of total and dissolved metals

Table I Concentrations of total and dissolved metals, obtained from a commercial lab.

Provtagningsdag	Provets märkning	Arsenik As (filtrerat)	Arsenik As (uppslutet)	Bly Pb (filtrerat)	Bly Pb (uppslutet)	Kadmium Cd (filtrerat)	Kadmium Cd (uppslutet)	Kobolt Co (filtrerat)	Kobolt Co (uppslutet)	Koppar Cu (filtrerat)	Koppar Cu (uppslutet)	Krom Cr (filtrerat)	Krom Cr (uppslutet)	Nickel Ni (filtrerat)	Nickel Ni (uppslutet)	Zink Zn (filtrerat)	Zink Zn (uppslutet)
		mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
2023-02-08	P4-P1	0,00055	0,00059	0,000032	< 0,00050	0,000016	< 0,00010	0,00041	0,00064	0,01	0,012	0,1	0,1	0,00063	0,0009	0,0096	0,018
2023-02-08	P4-P2	0,00055	0,00065	0,00017	0,00068	0,00016	0,00016	0,00087	0,0011	0,0038	0,005	0,006	0,0078	0,00082	0,002	0,075	0,087
2023-02-08	P4-L1	0,00078	0,00096	0,00068	0,00081	0,00014	0,00016	0,00018	0,00027	0,021	0,025	0,00041	0,00062	0,0051	0,0062	0,015	0,017
2023-02-08	P4-L2	0,0028	0,0034	0,0013	0,0016	0,00015	0,00017	0,00072	0,00094	0,039	0,05	0,0024	0,0033	0,006	0,0074	0,21	0,27
2023-02-08	P4-500	0,0018	0,0022	0,00075	0,001	0,00049	0,00055	0,0021	0,0024	0,0045	0,0056	0,00055	0,00088	0,0015	0,0018	0,069	0,079
2023-02-08	P4-360	0,0025	0,003	0,00086	0,0015	0,00043	0,00047	0,0017	0,0021	0,0071	0,0094	0,00068	0,0013	0,0017	0,0023	0,048	0,057
2023-02-08	P4-100	0,0031	0,004	0,00083	0,0011	0,00014	0,00014	0,0008	0,001	0,022	0,029	0,0011	0,0016	0,0036	0,0049	0,012	0,017
2023-02-08	Inc.SW	0,00012	0,002	0,000016	0,014	0,000034	< 0,00010	0,0038	0,021	0,0069	0,095	0,00013	0,033	0,0017	0,016	0,05	0,49
2023-01-23	A4-L1	0,0026	0,0028	0,00007	< 0,00050	0,0002	0,00022	0,0043	0,0049	0,07	0,081	0,0034	0,0016	0,14	0,16	0,044	0,051
2023-01-23	A4-L2	0,0018	0,0019	0,000086	0,0009	0,00015	0,00016	0,0052	0,0056	0,038	0,046	0,009	0,0099	0,13	0,14	0,039	0,069
2023-01-23	A4-500	0,0027	0,0031	0,000082	< 0,00050	0,00001	< 0,00010	0,00021	0,00023	0,007	0,0086	0,0086	0,0098	0,0015	0,0016	0,0018	< 0,0020
2023-01-23	A4-360	0,0028	0,003	0,000086	< 0,00050	0,000012	< 0,00010	0,00023	0,00026	0,0075	0,009	0,0083	0,0089	0,0016	0,0018	0,0025	0,0022
2023-01-23	A4-100	0,0018	0,002	0,00015	< 0,00050	0,000032	< 0,00010	0,00043	0,00051	0,011	0,014	0,0069	0,0077	0,0033	0,0038	0,0025	0,0033
2023-01-23	Inc.SW	0,00021	0,0021	0,000024	0,013	0,000039	< 0,00010	0,0057	0,021	0,0081	0,094	0,003	0,039	0,0013	0,013	0,086	0,51
2023-02-13	B2-L2	0,00099	0,0012	0,000075	< 0,00050	0,000023	< 0,00010	0,00016	0,00017	0,016	0,017	0,00042	< 0,00050	0,0041	0,0046	0,0097	0,012
2023-02-13	B2-L1	0,0011	0,0016	0,00017	< 0,00050	0,000041	< 0,00010	0,00042	0,00061	0,025	0,028	0,0017	0,0026	0,015	0,02	0,009	0,01
2023-02-13	B2-500	0,0011	0,0012	0,00011	0,00063	0,000027	< 0,00010	0,00012	0,00025	0,0025	0,0045	0,00028	0,00072	0,00052	0,00079	0,002	0,0047
2023-02-13	B2-360	0,0011	0,0013	0,00011	0,00072	0,000026	< 0,00010	0,00014	0,00029	0,0027	0,0039	0,0004	0,0015	0,00054	0,001	0,002	0,0049
2023-02-13	B2-100	0,00074	0,00086	0,000067	0,00054	0,000008	< 0,00010	0,000098	0,0002	0,0031	0,0043	0,00035	0,00066	0,00051	0,00071	0,0016	0,0036
2023-02-13	Inc.SW	0,00016	0,0012	0,000012	0,0089	0,000045	< 0,00010	0,0049	0,016	0,0073	0,058	0,00016	0,02	0,0021	0,01	0,072	0,4
2023-02-28	A4-L2	0,0054	0,0059	<0,000010	< 0,00050	0,000011	< 0,00010	0,0019	0,028	0,033	0,075	0,0046	0,005	0,012	0,014	0,00063	0,0097
2023-02-28	A4-L1	0,003	0,0042	0,0018	0,0027	0,0002	0,00028	0,00027	0,00046	0,082	0,11	0,0012	0,0018	0,0037	0,006	0,045	0,061
2023-02-28	P4-L1	0,00053	0,00058	0,00028	0,00054	0,000076	< 0,00010	0,00014	0,0002	0,034	0,043	0,00043	0,00058	0,0025	0,0027	0,023	0,027
2023-03-14	C-L2	0,00017	< 0,00020	0,000039	< 0,00050	0,000012	< 0,00010	0,000012	0,00014	0,0044	0,0069	0,0002	< 0,00050	0,0028	0,0034	0,0021	0,002
2023-03-14	C-L1	0,00034	0,00042	0,00002	< 0,00050	0,000011	< 0,00010	0,00004	0,00011	0,0057	0,0073	0,00015	< 0,00050	0,0014	0,0016	0,0037	0,0049

Provtagningsdag	Provets märkning	Arsenik As	Arsenik As	Bly Pb	Bly Pb	Kadmium Cd	Kadmium Cd	Kobolt Co	Kobolt Co	Koppar Cu	Koppar Cu	Krom Cr	Krom Cr	Nickel Ni	Nickel Ni	Zink Zn	Zink Zn
		(filtrerat)	(uppslutet)	(filtrerat)	(uppslutet)	(filtrerat)	(uppslutet)	(filtrerat)	(uppslutet)	(filtrerat)	(uppslutet)	(filtrerat)	(uppslutet)	(filtrerat)	(uppslutet)	(filtrerat)	(uppslutet)
		mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
2023-02-21	C-L1	0,0011	0,0011	0,000093	< 0,00050	0,00003	< 0,00010	0,00022	0,00027	0,018	0,02	0,00056	0,0064	0,0067	0,0076	0,016	0,016
2023-02-21	C-L2	0,0002	0,00021	0,000067	< 0,00050	0,000015	< 0,00010	0,000098	0,00018	0,007	0,0087	0,00034	0,00051	0,0058	0,0089	0,0066	0,0074
2023-03-14	C-P1	0,00058	0,0011	0,00019	0,0045	0,000011	< 0,00010	0,00042	0,0026	0,011	0,025	0,033	0,04	0,00082	0,0029	0,011	0,074
2023-02-21	C-P1	0,00051	0,00098	0,00006	0,0037	0,000018	< 0,00010	0,00016	0,0012	0,012	0,021	0,087	0,091	0,0012	0,0024	0,0057	0,044
2023-02-21	C-P2	0,00087	0,0011	0,000042	0,0016	0,000009	< 0,00010	0,000077	0,00041	0,0091	0,016	0,17	0,18	0,00056	0,0023	0,0033	0,02
2023-03-14	C-P2	0,0012	0,0012	0,00022	0,0028	0,000017	< 0,00010	0,00046	0,0013	0,0095	0,014	0,064	0,0084	0,00074	0,0011	0,013	0,04
2023-02-28	P4-P1	0,0014	0,0018	0,00054	0,0033	0,000026	< 0,00010	0,00045	0,00089	0,013	0,017	0,068	0,011	0,0013	0,002	0,027	0,038
2023-03-14	B2-L2	0,00083	0,0009	0,00034	< 0,00050	0,000065	< 0,00010	0,00016	0,0002	0,036	0,041	0,00053	0,00064	0,0026	0,0028	0,0057	0,006
2023-03-14	B2-L1	0,00082	0,0013	0,00015	< 0,00050	0,000029	< 0,00010	0,00011	0,00018	0,036	0,051	0,00039	< 0,00050	0,0023	0,0032	0,01	0,013
2023-02-28	P4-L2	0,0011	0,0015	0,0002	0,001	0,000097	0,00013	0,00027	0,00037	0,038	0,055	0,00048	0,001	0,003	0,004	0,15	0,19
2023-02-21	Inc.SW	0,00024	0,00093	0,000024	0,0079	0,000015	< 0,00010	0,0002	0,0095	0,013	0,059	0,00019	0,01	0,0022	0,0071	0,033	0,32
2023-02-21	C-100	0,0016	0,002	0,000098	0,00087	0,00003	< 0,00010	0,00032	0,00052	0,015	0,019	0,00059	0,001	0,0028	0,0037	0,0052	0,0097
2023-02-21	C-360	0,0016	0,002	0,000043	0,001	0,000036	< 0,00010	0,00022	0,00041	0,007	0,015	0,00044	0,00095	0,0013	0,0018	0,0063	0,014
2023-02-21	C-500	0,0015	0,0019	0,000088	0,001	0,000035	< 0,00010	0,00018	0,00038	0,0062	0,0079	0,00036	0,00098	0,0012	0,0016	0,0063	0,011
2023-02-28	P4-P2	0,00068	0,00081	0,000034	0,0013	0,00014	0,00017	0,00091	0,0013	0,0076	0,014	0,053	0,062	0,00088	0,0015	0,07	0,082
2023-02-28	A4-P2	0,009	0,011	0,00031	0,021	0,000016	0,00011	0,00042	0,0023	0,06	0,17	0,052	0,058	0,0076	0,015	0,013	0,18
2023-02-28	A4-P1	0,0046	0,0058	0,00092	0,0096	0,000016	< 0,00010	0,00043	0,0016	0,045	0,094	0,04	0,043	0,0039	0,0059	0,017	0,099
2023-03-14	B2-P2	0,0014	0,0018	0,00036	0,0026	0,00002	< 0,00010	0,00019	0,0013	0,014	0,02	0,18	0,17	0,0013	0,0026	0,011	0,041
2023-02-28	A4-360	0,0047	0,0057	0,000039	0,0025	0,000018	< 0,00010	0,00024	0,00047	0,013	0,02	0,0016	0,0026	0,0025	0,0031	0,0043	0,016
2023-02-28	A4-100	0,0017	0,0021	0,000044	0,00063	0,000027	< 0,00010	0,00038	0,00044	0,011	0,013	0,00053	0,0006	0,0027	0,003	0,0022	0,0045
2023-02-28	A4-500	0,0047	0,0057	0,000034	0,0029	0,000011	< 0,00010	0,00022	0,0005	0,013	0,022	0,0022	0,0036	0,0022	0,0029	0,0033	0,018
2023-02-28	Inc. SW	0,00029	0,00074	0,00013	0,0042	0,000009	< 0,00010	0,0001	0,0023	0,014	0,027	0,00058	0,0039	0,0018	0,0032	0,016	0,1
2023-02-28	Inc. SW-2	0,00031	0,00094	0,0004	0,0051	0,000013	< 0,00010	0,00026	0,0031	0,015	0,031	0,00074	0,0058	0,0017	0,0042	0,014	0,093
2023-02-28	P4-100	0,00086	0,001	0,000078	< 0,00050	0,00029	0,00035	0,0017	0,0017	0,0074	0,0093	0,00034	0,00058	0,0032	0,0035	0,036	0,039
2023-02-28	P4-360	0,0025	0,0031	0,00038	0,0017	0,00013	0,00014	0,00092	0,0011	0,01	0,013	0,0064	0,0078	0,0015	0,0022	0,029	0,032
2023-03-14	P4-500	0,0025	0,0029	0,00091	0,0019	0,000078	< 0,00010	0,00067	0,0008	0,011	0,013	0,0069	0,0085	0,0014	0,0018	0,024	0,026
2023-03-14	Inc. SW	0,00028	0,00071	0,00021	0,0028	0,000009	< 0,00010	0,00046	0,0024	0,012	0,027	0,0071	0,01	0,0022	0,004	0,049	0,16
2023-03-14	B2-360	0,0027	0,0032	0,00019	0,0018	0,000016	< 0,00010	0,00014	0,00041	0,008	0,011	0,0069	0,0081	0,001	0,0017	0,0043	0,014
2023-03-14	B2-500	0,0027	0,0033	0,00037	0,0017	0,000017	< 0,00010	0,00014	0,00044	0,0089	0,013	0,0077	0,0085	0,0011	0,0017	0,0046	0,016
2023-03-14	B2-100	0,001	0,0011	0,000046	< 0,00050	0,000013	< 0,00010	0,000093	0,00018	0,0038	0,0049	0,0057	0,0063	0,00051	0,00071	0,0015	0,0035
2023-03-14	Inc. SW-2	0,00023	0,0017	0,00015	0,011	0,000022	< 0,00010	0,0031	0,013	0,014	0,069	0,0049	0,024	0,0018	0,01	0,038	0,32
2023-03-14	C-100	0,0015	0,0019	0,00015	0,0015	0,000023	< 0,00010	0,00029	0,00081	0,011	0,016	0,0067	0,009	0,0016	0,0025	0,0062	0,015
2023-03-14	C-360	0,0018	0,0023	0,00031	0,0023	0,000027	< 0,00010	0,00036	0,00097	0,0094	0,014	0,0066	0,0093	0,001	0,0021	0,009	0,025
2023-03-14	C-500	0,0018	0,0023	0,00035	0,0026	0,000026	< 0,00010	0,00035	0,0011	0,0092	0,014	0,0068	0,01	0,00091	0,0019	0,0089	0,029