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Effects of Drought on the Removal of Microplastics and other Environmental Pollutants from Urban Stormwater in Bioretention Filters

Master thesis

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Abstract

Various particles and pollutants, such as microplastics, organic pollutants and nutrients, accumulate in untreated urban stormwater. In order to minimise surface water pollution from stormwater discharges, this study investigated the effects of drought on the removal efficiency of pollutants in bioretention filters with different sorption materials (waste-to-energy bottom ash, biochar, or s*phagnum* peat). The study was carried out in a rain garden pilot plant with 13 bioretention filters, 10 of them planted (*armeria maritima, hippophae rhamnoides, juncus effusus* and *festuca rubra*) and 3 unplanted. One planted bioretention filter of each sorption medium and one control filter were exposed to a dry period of 13 weeks. The remaining filters were regularly irrigated with the untreated stormwater runoff from the adjacent highway and the surrounding sealed surfaces. At the end of the dry period, all filter columns were irrigated, inflow and outflow water samples were taken and subsequently analysed for the previously mentioned pollutants in the laboratory. In general, high removal rates for microplastics >10µm, ions and nutrients were achieved in all filters. However, the drought had an impact on the removal efficiency with regard to TSS, VSS, TOC and DOC for all sorption media during the first irrigation and with increased pollutant concentrations in the inflow. In the course of the study, regeneration of all filters was observed, regardless of the sorption medium.

Overall, it can be said that the different cleaning performance of the filters is primarily due to the different properties of the sorption materials and not the dry period. All sorption media contribute positively to the removal of pollutants. However, ash had the best cleaning performance in the overall comparison, therefore its implementation in rain gardens is recommended.

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List of Abbreviations

Al	aluminium
As	arsenic
BR	butadiene rubber
C	carbon
Ca	calcium
Cd	cadmium
Cl	chlorine
Cr	chromium
Cu	copper
DOC	dissolved organic carbon
DON	dissolved organic nitrogen
ET	evapotranspiration
Fe	iron
GSI	green stormwater infrastructure
K	potassium
KOH	potassium hydroxide
Mg	magnesium
MIBA	mineral fraction of incineration bottom ash
MP	microplastics
N	nitrogen
Na	sodium
NH_3-N	ammonia
NH_4-N	ammonium
Ni	nickel

NO_3-N	nitrate nitrogen
NO_x-N	nitrogen oxide
NR	natural rubber
O	oxygen
OP	organic pollutants
P	phosphorus
PA	polyamide
PAH	polycyclic aromatic hydrocarbons
Pb	lead
PB	polybutadiene
PC	polycarbonate
PE	polyethylene
PET	polyethylene terephthalate
PI	polyisoprene
PMMA	polymethyl methacrylate
PP	polypropylene
PS	polystyrene
PVC	polyvinyl chloride
RE	removal efficiency
SBR	styrene-butadiene rubber
SCM	stormwater management measures
SSF	stainless steel filter
TN	total nitrogen
TOC	total organic carbon
TP	total phosphorus

TRWP	tyre and road wear particles
TSS	total suspended solids
VSS	volatile suspended solids
wt%	weight percentage
Zn	zinc

1. Introduction

Climate change and growing urbanisation are central issues of the 21st century. Due to increasing global warming, precipitation patterns are changing, resulting not only in more and intense rainfall but also in longer periods of drought (Dai et al. 2018). Furthermore, the problem is exacerbated in urban environments as the reduction of green and permeable areas results in the loss of natural hydrological flow and infiltration processes, leading to increased surface runoff (Cook 2007). Urbanisation processes and associated land use changes are expected to result in a 75–100% growth in impervious surfaces. As a result, urban surface runoff is predicted to increase from the original 10% to 55%, and infiltration rates are estimated to decrease by 35% (Lee and Gil 2020; Macedo et al. 2017). In the context of changing precipitation patterns, it is necessary to adapt stormwater management.

Historically, the focus of stormwater management has been on preventing flooding (Fletcher et al. 2015). Today's approaches also aim to reduce pollutant concentrations in water discharges (Dagenais et al. 2018). In particular, road dust in heavily trafficked areas is considered a sink for pollutants. These include metals, nutrients, organic pollutants (OPs) such as aliphatic, aromatic, and polycyclic aromatic hydrocarbons (PAHs), microplastics (MPs), and nanoparticles (Björklund et al. 2009; Järlskog et al. 2022a; Järlskog et al. 2022b; Järlskog et al. 2021; Järlskog et al. 2020; Markiewicz et al. 2017; Müller et al. 2020; Pamuru et al. 2022; Polukarova et al. 2020). Contaminated road dust can then be discharged untreated into the sewer system during rain events, into water bodies, or other ecosystems via surface runoff (Anh et al. 2019). The discharge of contaminated stormwater runoff has serious consequences for receiving water bodies as it pollutes water, and alters the hydrography (Masoner et al. 2019). Appropriate mitigation strategies are required to reduce the impact on water bodies and groundwater. In this context, various stormwater treatment technologies and stormwater management measures (SCMs) have been developed in recent years (Dagenais et al. 2018). These combine the reduction of peak flows and flood protection to improve water quality (Sun et al. 2019).

Biofiltration systems are a common urban SCM. (Sun et al. 2019). They mimic natural systems by combining soil-based infiltration with filtration processes and pollutant retention (Cederkvist et al. 2017). Studies have already shown that biofiltration can capture and retain particles, metals, pollutants, and nutrients (Al-Ameri et al. 2018; Flanagan et al. 2019; Søberg et al. 2017; Valtanen et al. 2014; You et al. 2019), but there are few studies yet addressing the removal processes of MPs and OPs (Lange et al. 2023). In addition, the comparability of previous studies on biofiltration systems is limited due to the current lack of standardised analytical methods, variations in the parameters investigated, or small sample sizes (Österlund et al. 2023). For example, many studies have been carried out under laboratory conditions or with synthetic stormwater (Lim et al. 2015; Lucke and

Nichols 2015; Jiang et al. 2019; Wan et al. 2017), but there are only a few field studies (Shrestha et al. 2018). In general, bioretention filters are a promising method for stormwater treatment. However, there is still a need for further research, particularly at field scale and concerning MP removal processes. As treatment efficiency depends on various biological, chemical, and physical processes, as well as climatic and seasonal variations (Beral et al. 2023), the effects of different combinations of filter media and plants on pollutant and particle retention, as well as the effects of changing weather and climatic conditions, should be investigated.

1.1 Objective

The aim of this Master's thesis is to investigate the effects of drought on the treatment efficiency of pollutants in bioretention filters with different sorption materials to be used in rain gardens.

In this context, the retention and degradation capacities of nutrients, and pollutants such as MPs, and OPs in bioretention filters were analysed under field conditions in a pilot scale. Ash, a by-product of municipal waste incineration, biochar, and *Sphagnum* peat moss were used as sorbents. The different filter media were analysed both with and without the planting of thrift *(armeria maritima)*, sea buckthorn *(hippophae rhamnoides)*, common rush *(juncus effusus),* and red fescue *(festuca rubra)*. In total, one planted bioretention filter of each sorption medium and one control filter were subjected to a dry period of 13 weeks. The remaining filters were regularly watered with untreated stormwater. At the end of the drought period, all filter columns were irrigated, and water samples were taken on five field days.

1.2 Research Questions

The primary inquiry guiding this research is: What are the effects of prolonged drought on bioretention filters? Within this framework, additional questions can be formulated, including:

a) How do the properties of the filters develop over time after the resumption of irrigation? ***- Long-term implications of drought***
b) Do variations exist in the recuperation and operational dynamics among the sorption materials (ash, biochar, peat) post the dry spell? ***- Long-term implications of drought***
c) How does drought impact the capacity of bioretention filters to remove particles and pollutants such as MPs, TSS/VSS, DOC/TOC, cations, and anions from water? Are there notable alterations in filter efficiency compared to routinely irrigated filters? ***- Efficiency in pollutant removal***

d) Are there distinct advantages and disadvantages associated with the sorption materials (ash, biochar, peat) concerning their resilience during and after drought, alongside their efficacy in pollutant removal? ***- Comparative assessment of filter materials***

1.3 Limitations

The Master's thesis is part of a joint research project between Chalmers University of Technology, COWI SE and COWI Aquateam NO. It is a work done for the third campaign, which focuses on the effects and regeneration capabilities of the bioretention filters after a drought. The first two campaigns covered the start-up and stabilisation, as well as the effects on the bioretention filters of extremely heavy rainfall events. In addition, a Master's thesis has been written on the removal of metallic pollutants during the winter months. These issues are not considered in this thesis. The focus is solely on how the regenerative capacity, as well as the removal of pollutants and particles of nutrients, OP, MP, TSS/VSS, TOC/DOC, ions and TN, are affected by drought.

It is important to note that due to time constraints and high analysis costs, only 5 sampling days were conducted and each test per sample was performed only once. Therefore, no mean values can be calculated for each day and measurement errors cannot be completely excluded. Furthermore, no distinction was made whether the MP was present in fibres, spheres, or fragments.

2. Current State of the Art

As a result of urban development, stormwater contributes significantly to the pollution of surface waters. Stormwater is considered a diffuse source. Diffuse sources do not originate from a confined spring and are therefore much more difficult to regulate than point sources (Hsieh and Davis 2005; Wang et al. 2000). Depending on the increasing sealing of surfaces and the resulting higher volume of runoff, greater pollution of receiving waters is occurring. In addition, current management systems are not designed to handle the increased runoff (Dietz 2007; Lucke and Beecham 2011). The associated negative impacts of uncontrolled and untreated urban stormwater runoff are well documented (Al-Ameri et al. 2018; Paul and Meyer 2001; Walsh et al. 2005). Pollutants in stormwater, for example, not only result in general water quality degradation and hydrographic changes but also cause eutrophication, discolouration, and odour problems (Jiang et al. 2019).

Runoff from road surfaces has been identified as a major pathway for water pollution (Edwards et al. 2016; Galfi et al. 2016; Laña et al. 2016). Road runoff contains a combination of tyre and brake pad abrasion, oils, lubricants, winter maintenance residues, metals, sediments, nutrients, and the road material itself (Novotny et al. 2008; Trenouth and Gharabaghi 2016). Heavily trafficked areas in particular have high concentrations of MPs in road dust and are considered a major source of microplastic inputs in this regard (Järlskog et al. 2022a; Järlskog et al. 2022b; Järlskog et al. 2021; Järlskog et al. 2020; Kole et al. 2017). This was also shown in studies which identified road and tyre wear particles (TRWP) as the main source of MP in Sweden (Polukarova et al. 2020; Magnusson et al. 2016).

In order to minimise the adverse effects of urban stormwater discharges, the Green Stormwater Infrastructure (GSI) approach is gaining attention. It provides an alternative to traditional stormwater management by incorporating strategies that are integrated into the landscape and promote filtration, infiltration, soil storage, evapotranspiration, and stormwater reuse. By applying this approach, surface runoff can be reduced, and the effects of increasing land sealing can be counteracted. Examples of GSI include green roofs, infiltration trenches, ponds, and bioretention basins (Davis 2007; Roy et al. 2008; Shrestha et al. 2018). Bioretention systems are a common form of green infrastructure. These systems are being deployed progressively as an integrated urban drainage solution to reduce, control, and treat stormwater runoff in urban catchment areas (Beral et al. 2023).

2.1 Rain Garden Facilities

Rain gardens, also known as bioretention systems or biofilters, are a common form of green infrastructure. They are usually planted in basins filled with filter and sorption materials. By capturing urban stormwater runoff, they can reduce the flows, store the stormwater and minimise the pollutant content (Cook 2007). The functional principle is based on passive runoff treatment using the natural purification properties of plant and soil systems (Glaister et al. 2017). The purification efficiency depends on the combination of various chemical (sorption, precipitation, ion exchange), biological (phytoremediation, microbial transformation, transpiration), and physical (filtration, evaporation) processes (Beral et al. 2023; Liu et al. 2014). In this regard, not only the substrate, soil-based materials and plant composition are relevant, but also the age of the system, as well as seasonal and climatic conditions (Fowdar et al. 2022; Nie et al. 2013; Payne et al. 2014b; Walaszek et al. 2018). Not only does the composition of urban stormwater vary with the seasons, e.g. due to the use of winter and summer tyres or winter service residues in road dust, but also the temperatures of road runoff (Beral et al. 2023). Furthermore, seasonal variations can have an impact on treatment efficiency and runoff reduction. For example, the winter dormancy of many plant species in continental climates can temporarily reduce the performance of the system (Géhéniau et al. 2015; Paus et al. 2016). However, bioretention systems do not necessarily have to contain plants. Several studies have shown that pollutant removal can also occur in the substrate without the biological degradation of plants in the form of chemical and physical processes such as sorption, precipitation, and filtration (Dagenais et al. 2018). Reactions such as nitrification, oxidation, and fermentation take place through biofilms and various bacteria, whereby nutrients such as ammonium can be converted (Davis et al. 2012). However, plants contribute to increased runoff reduction and pollutant removal through transpiration, retention, absorption and filtration processes. In addition, the roots have a positive effect on biological degradation and can counteract clogging depending on the choice of plants. Therefore, they should be included in the design of rain gardens (Bratieres et al. 2008; Payne et al. 2014a).

In general, it should be emphasised that bioretention filters cannot be designed for every rainfall event. Systems are usually built for small events, often following long dry spells, as these are the events that carry the most polluted loads. Rainfall of a longer duration and greater intensity will exceed the infiltration capacity of the soil medium and result in surface runoff. In these cases, urban stormwater runoff cannot be fully treated, but the discharge of untreated runoff to water bodies can be reduced (Davis et al. 2012). In addition, the catchment itself is also relevant, as increased surface sealing leads to larger surface runoff for a given rainfall intensity (Sun et al. 2019). Conversely, if the rain garden is designed for excessive rainfall events, the area required will increase and the treatment efficiency per unit area will decrease. Furthermore, the quantity of water per unit area is reduced, which in turn can cause dryness and might result in plant death (Yang and Chui 2018). In this respect, the design of bioretention systems needs to be adapted to local conditions and catchment-specific management requirements.

Another aspect to consider is climate change. In the future, climate change is expected to cause not only higher rainfall intensities, but also more frequent and longer dry periods. According to the studies from Mangangka et al. (2015), the lower soil moisture content increases the retention capacity of the filter material and improves the purification performance. Lucke et al. (2015) on the other hand, found no influence of the preceding dry days and the associated low water content in the soil on the retention capacity and pollutant recovery efficiency. However, other studies have reported higher leaching of pollutants during the first flush followed by a rapid drop in concentration (Sansalone and Cristina 2004). This is probably because dry sediments and pollutants are more mobile and therefore more easily leached during the first irrigation (Österlund et al. 2023). Uncertainties remain in this respect and therefore more research is needed on the effects of dry periods on rain gardens.

Overall, rain gardens have a significant number of benefits. Not only do they remove pollutants and reduce peak flows as well as runoff, but they can also be flexibly integrated into the landscape, promote biodiversity and aesthetics, minimise the urban heat island effect, help to restore natural hydrological functions and provide a cost-effective way of reducing the burden on stormwater infrastructure (Shrestha et al. 2018; Bratieres et al. 2008; Coutts et al. 2013; Ellis 2013; Kazemi et al. 2009; Wadzuk et al. 2015). However, regular maintenance and inspection are required to maintain these attributes. Rain garden maintenance activities can vary from site to site. They may include inspection and cultivation of vegetation, weed prevention, ensuring surface infiltration with possible removal and replacement of top filter layers, erosion control as well as litter removal (Blecken et al. 2017; Davis et al. 2009; Lim and Lu 2016).

The following chapters outline the importance of different sorbents and the benefits of plants in bioretention systems. Furthermore, the most common pollutants and particles found in urban stormwater runoff are outlined.

2.1.1 Filter Sorption Material

There are many factors in the design and construction of bioretention systems that affect treatment efficiency and scale. One important aspect is the filter media. Depending on the design of the filter bed and the composition of the substrate, not only the filter depth but also the stratification, particle size, compaction, and clogging characteristics will vary (Kandra et al. 2014).

In the past, various media have been established for the treatment of urban stormwater, including (granulated) activated carbon, zeolites, sand, clay, diatomaceous earth, limestone, carbonates, sulphates, iron oxides, iron hydroxides, sawdust, pine bark, fibre sludge ash, shrimp shells, seaweed, *pinus sylvestris* bark and *sphagnum* peat (Bandura et al. 2015; Bortone et al. 2013; Fronczyk 2017; Fronczyk et al. 2016; Fronczyk and Garbulewski 2013; Wang et al. 2013; Kalmykova et al. 2008; Markiewicz et al. 2020). Sand, for example, helps to reduce runoff, minimise peak flows and particulate pollutants, but cannot remove dissolved substances (Barrett 2003; Shahrokh Hamedani et al. 2021; Zarezadeh et al. 2018). Mixing or layering with other materials, such as compost, can counteract the removal deficiencies of sand and maximise system performance. However, using compost can also promote leaching of nutrients and dissolved Cu (Ouyang et al. 2016). In this regard, it is important to analyse the synergies of the different materials and combine them accordingly in order to design the most efficient filter possible. This applies not only to sand but also to various filtration and sorption materials.

The composition of the soil has a particular influence on the infiltration processes, the infiltration capacity, and the sorption and desorption processes of chemical contaminants (Macedo et al. 2017). In this context, the hydraulic performance of the filter mixture is relevant, as the clogging properties limit the treatment of urban stormwater. One important aspect is the shape (angularity and surface texture) of the particles. For instance, non-spherical media such as granular sand promote clogging since they facilitate biofilm growth due to the increased surface area. Gravel or glass beads, on the other hand, are referred to as spherical media and have a more round and even shape (Kandra et al. 2014). In the context of biofiltration systems, clogging refers to the reduction of the infiltration rate. By reducing hydraulic permeability, overflows and periods of waterlogging can occur in the system, resulting in reduced system performance (Hatt et al. 2007). Further research is needed to understand the clogging processes in stormwater biofilter systems (Yong et al. 2013). It is also necessary to be aware of the hydraulic performance and long-term cleaning performance of the filter media to ensure sufficient treatment efficiency and system maintenance (Kandra et al. 2014).

Overall, filter and pollutant treatment performance varies depending on material composition and climate. Furthermore, the long-term behaviour of the media differs and can be additionally influenced by the inclusion of sorbents (Lim et al. 2015). In this respect, it is necessary to adapt the composition of the substrate to the specific conditions and objectives. It is important to distinguish whether the priority is the quantity of treated water, pollutant removal, longevity, maintenance requirements, cost, or a combination of several factors (Kandra et al. 2014; Kaya et al. 2022).

2.1.1.1 Ash

Every year almost one million tonnes of ash are produced from the incineration of municipal waste in Sweden (Karlfeldt Fedje et al. 2021). Untreated residual ash contains a variety of metals, including silicon, iron, calcium, and aluminium (Youngblood et al. 2017). The metals can be recovered through different techniques, such as mechanical separation and carbonisation. The material is also stabilised to prevent leaching. After the metals have been recovered, the ash is known as the mineral fraction of incineration bottom ash (MIBA) (Arickx et al. 2006; Blasenbauer et al. 2020; Freyssinet et al. 2002).

Using MIBA as a sorbent in bioretention systems has several advantages. For example, it is not only economical because it is a by-product of incineration, but it can also contribute to metal recovery via its metal sorption capacity. The active surface of the MIBA particles allows dissolved metals to be bound and subsequently removed by phytoextraction (Karlfeldt Fedje et al. 2021). It is also highly effective in the removal of OPs, which are very water soluble and have a low molecular weight (Markiewicz et al. 2020). Other positive properties are the high pH, low organic content, the presence of the macronutrients Ca, K, P, and Mg, which are essential for many plants, and the potential for use as a drainage agent (Karlfeldt Fedje et al. 2021; Johansson et al. 2024).

However, phytoextraction can only be utilised to a limited extent in the Nordic countries as the necessary plants, which act as hyperaccumulators, grow only to a limited extent (Odjegba and Fasidi 2007). In addition, MIBA contains potentially toxic metals such as Cr, Ni, and Pb, which can affect root growth. The metal complexes Cu, Zn, and Cl have a negative effect not only on plant growth but also on nutrient uptake (Karlfeldt Fedje et al. 2021). However, due to the stabilisation of the material during the treatment of residual ash to MIBA, the mobility of these metals in water is low (Karlfeldt Fedje et al. 2021). Another issue is the increase in soil salinity. Ash has an enriched content of soluble salts. Plants may suffer from salt stress due to the release of salts (Ferreira et al. 2003). In this respect, it is important to use plants with high salt tolerance in the rain garden when using MIBA as a sorbent. When implementing MIBA in a bioretention system, it is important not only to consider the site conditions for the particular application but also to analyse the potential environmental impact.

2.1.1.2 Biochar

In recent years, biochar has gained increasing attention as a decontaminant in urban stormwater treatment (Ulrich et al. 2017). Biochar is a porous carbonaceous material formed by carbonisation or pyrolysis under low-oxygen conditions (Ahmad et al. 2014). It can be synthesised from various types of biomass, industrial by-products, and municipal wastewater sludge (Akhil et al. 2021; Randolph et al. 2017; Tsang et al. 2019). The physicochemical properties of biochar vary depending on the raw material and production technology (Cao et al. 2017). For example, manure-based biochar has high levels of sodium, phosphorus, and potassium, while wood-based biochar has low concentrations (Lehmann and Joseph 2015). However, the production temperature also has a significant effect on the composition, pore volume, pore size, pH, and specific surface area (Mohanty et al. 2018). For instance, high pyrolysis temperatures result in increased carbon, phosphorus, potassium, and calcium contents, as well as high pH values and a large specific surface area (Weber and Quicker 2018). There is also a reduction in nitrogen, hydrogen, and oxygen content (Premarathna et al. 2023). For stormwater treatment, biochar with a high specific surface area and surface charge density, small micropores, and low bulk density is of particular interest (Kaya et al. 2022; Kookana et al. 2011).

In general, biochar is a cost-effective and sustainable material for removing pollutants from stormwater (Biswal et al. 2022). At around US$250/tonne, biochar is almost six times cheaper than activated carbon (US$1,500/tonne) (Inyang and Dickenson 2015). Despite the low price, biochar has a high retention capacity for various pollutants such as nutrients, metals, organic substances, MP, and pathogenic microorganisms (Tsang et al. 2019; Premarathna et al. 2023; Biswal et al. 2022; Kuoppamäki et al. 2021). In various studies reviewed by Biswal et al. (2022), filters mixed with biochar were able to reduce the total nitrogen content of urban runoff by 32–61%. Total phosphorus was reduced by 45–94%, metals by 27–100%, organic pollutants by 54–100%, and microbial matter by a log10 removal of 0.78–4.23. The range of variation in this respect is due to the different composition of the stormwater runoff, the temporal changes in the filter media, and the number of dissolved organic substances (Biswal et al. 2022). Similar results were also shown in the studies by Ashoori et al. (2019); Berger et al. (2019) and Xiong et al. (2019). The pollutant removal efficiency of the biofilter also depends on the type, amount and size of the biochar used as sorbent. Typically, less than 10% by weight of biochar is added to the filter medium (Xiong et al. 2019; Rahman et al. 2020). The removal efficiency can be further improved by increasing the biochar content (up to 30 wt%) (Biswal et al. 2022; Rahman et al. 2021). However, excessive use of biochar can temporarily inhibit nitrification processes due to the significant minimisation of ammonia (NH_4-N) (Premarathna et al. 2023). In this respect, the filter composition should be adapted to the respective objectives, local conditions, and stormwater composition. Other benefits of biochar as a sorbent in stormwater filtration systems include improved denitrification, the possibility of multiple regeneration of the filter material, low susceptibility to contamination, increased water and nutrient retention, and the promotion of microbial community and plant growth (Berger et al. 2019; Wei et al. 2019; Lu and Chen 2018). Additionally, the use of solid waste biochar contributes to the concept of green infrastructure (Premarathna et al. 2023).

At the same time, several studies have shown that biochar has a limited ability to remove dissolved organic nitrogen (DON) and that high concentrations of dissolved organic pollutants and phosphates inhibit metal adsorption (Liu et al. 2019; Nabiul Afrooz and Boehm 2017). Furthermore, biochar can act as both a source and a sink for pollutants. Depending on the feedstock and production technologies, leaching of metals and nutrients may occur, or contaminants may be transported via nanoparticles (Swaren et al. 2022). There are also several research gaps in the use of biochar in bioretention systems for urban stormwater treatment. For example, it is unclear to what extent the weathering processes of biochar filters affect the immobilisation of pollutants (Kaya et al. 2022). Also, most studies have been conducted at the laboratory scale and with synthetic stormwater (Biswal et al. 2022). In order to understand the behaviour of biochar as a sorption material, it is essential to investigate the removal efficiency under natural environmental conditions.

2.1.1.3 Peat

For more than 25 years, peat moss has been investigated as a way of treating pollutants in water (Brown et al. 2000). It is an organic material that contains both minerals and partially decomposed organic matter. Lignin, cellulose, fulvic, and humic acids, the minerals C, Ca, K, and O, and the metals Al, Mg, Fe, and Na are the main constituents (Markiewicz et al. 2020; Ahmaruzzaman 2008). Peat is formed under anoxic and nutrient-poor conditions, which inhibit microbial activity (Kalmykova et al. 2009). As a result, *sphagnum* peat in particular is considered to be one of the slowest decomposing organic materials (Johnson et al. 1990). Untreated peat has an acidic pH (Markiewicz et al. 2020). The growth of bacteria in peat filters can be stimulated by shifting the pH into the neutral to alkaline range, higher temperatures, or increased oxidation processes. The increased number of bacteria or changes in redox conditions can accelerate decomposition processes (Johnson et al. 1990). In addition to the low level of decomposition, peat has other advantages. Not only is peat readily available and inexpensive, as it covers 15–20% of Sweden's total land area, but it also has high hydraulic conductivity and a low bulk density (Johnson et al. 1990; Ågren et al. 2022). In addition, peat is a porous medium with a large active surface area, which favours sorption and adsorption processes (Markiewicz et al. 2020; Brown et al. 2000).

Several studies have already demonstrated the removal of OPs, metals, oil, and pesticides using peat filters (Björklund and Li 2015; Kalmykova et al. 2014; Kalmykova et al. 2010). Thus, the removal efficiency for metals (Cd, Cu, Zn, Ni, and Pb) is over 90%. Physical changes do not affect the treatment efficiency in this respect. However, As and Cr are exceptions. For example, the removal capacity for Cr was greatly reduced (30%) by increasing the pH. Arsenic, on the other hand, can hardly be removed (Kalmykova et al. 2009). In addition, the use of road salt during the winter months can temporarily reduce the sorption efficiency of metals, as the presence of Na^+ ions blocks the active structures so that fewer toxic trace metal ions can be bound to the organic material in a complex manner (Kalmykova et al. 2009).

On the other hand, no significant external factors have been found to influence the leaching of DOC on peat filters. However, there is an increased release of residual DOC in the peat at the start of peat filter operation, which temporarily reduces metal removal (Kononova 2013). To counteract this, peat should be washed before implementation as a filter medium. This removes excess and easily soluble organic matter (Kalmykova et al. 2009). Thereafter, a continuous reduced leaching of DOC from the filters is expected to occur as the peat decomposes (Kononova 2013).

However, there is an increased need for research into the removal and sorption processes of OPs. For example, it is still unclear whether the sorption processes are linear or non-linear. Since the concentration of pollutants is dependent on the degradation and sorting processes, further research is required (Markiewicz et al. 2020). However, it has been shown that peat filters are effective in sorbing high molecular weight and lipophilic OPs (Kalmykova et al. 2014).

Nanoparticles, on the other hand, were only slightly retained in the study of Markiewicz et al. (2020). The NPs present in the stormwater were reduced by only 22%. As pollutants can bind to NP, it is necessary to increase the removal efficiency of NP via biofiltration systems (Markiewicz et al. 2020).

2.1.2 Plants

Plants are an integral part of rain gardens (Dagenais et al. 2018). Therefore, an appropriate selection of plant species can not only improve hydraulic performance and water quality but also increase nutrient and nitrogen removal (Morash et al. 2019; Zhang et al. 2018). Nevertheless, research is scarce concerning the impacts of various species, particularly within regions characterized by substantial seasonal variations (Beral et al. 2023).

The contribution of plants to reduce runoff volume and peak flows has been well investigated (Skorobogatov et al. 2020). One method of mitigating runoff involves leveraging the transpiration processes occurring in the leaves of plants. Roots maintain the infiltration capacity of the soil (Shrestha et al. 2018; Szota et al. 2018). Thus, the growth, death, and subsequent decomposition of roots preserve soil porosity, which leads to a positive infiltration rate of the soil system (Le Coustumer et al. 2012). In particular, plant species whose roots have a high mass density and diameter contribute to maintaining hydraulic conductivity. They can also minimise clogging and retain stormwater in the system for prolonged periods (Goh et al. 2017). However, plant roots can also form macropores and thus favour the formation of flow paths. The presence of preferential flow paths can reduce the treatment efficiency of the biofiltration system and result in increased displacement of contaminants (Zhang et al. 2018).

In addition, plant roots contribute significantly to nutrient uptake and thus nutrient reduction by vegetation (Shrestha et al. 2018; Szota et al. 2018; Le Coustumer et al. 2012). For example, in the study conducted by Beral et al. (2023), an average removal of 55% TN, 81% TP, and 61% K took place. In the unplanted reference system, however, 6% of TN was released and 61% of TP as well as 22% of K were retained. Plants can also contribute to the removal of metals from stormwater. By the use of hyperaccumulators, metal particles are bound in the plant tissue so that they can be permanently removed by cutting off the leaves. Other processes of metal removal by plants include rizhofiltration and phytostabilization, but the contribution is small (Ali et al. 2013). To date, only a few studies have been carried out on the effect of plants on the removal efficiency of MP. Nevertheless, a positive correlation was identified between the establishment of bioretention plants and MPs within the size range of 20–100 µm. Conversely, there was no observed enhancement in efficiency for microplastics with a particle size exceeding 100 µm (Lange et al. 2021). In addition to existing research gaps of the pollutant retention of microplastics, there is currently a heightened demand for investigations into the effects of plants on the removal of organic pollutants in bioretention systems (Dagenais et al. 2018).

Another advantage of vegetated systems is higher evapotranspiration rates (Lucas and Greenway 2008). However, the percentage contribution to the water balance varies. For instance, in the research conducted by Hamel et al. (2011), evapotranspiration (ET) losses constituted approximately 10% of the total water balance, whereas in the study conducted by Li et al. (2009), they accounted for nearly 19% of the inflow. Nevertheless, it can be assumed that water losses due to ET in the growing season are up to 2.5 times higher than in unplanted systems. An increased ET not only minimises runoff but also promotes pollutant retention (Beral et al. 2023). Additionally, the lifespan of planted bioretention systems is almost twice as long (9 years) as that of unplanted systems (5 years) (Jiang et al. 2019).

In general, it is important to adapt the choice of plants to the conditions and distance objectives of the particular bioretention system. Depending on the location and stormwater composition, plants need to be able to withstand different weather and climate periods, flooding, pollutants, and temperatures (Johansson et al. 2024). In this respect, the use of native plant species is generally recommended. On the one hand, the risk of invasive neophytes should be minimised and, conversely, native plants are expected to have a higher survival rate and require less maintenance (Dagenais et al. 2018). However, since stormwater treatment facilities are man-made, the conditions in terms of hydrology, soils, adjacent vegetation, fauna, and biogeochemistry vary considerably from those found in natural ecosystems (Pickett et al. 2011). As a result, native plant species are not necessarily the better choice. In addition, the introduction of plant diversity in bioretention systems is beneficial. The presence of different plants can compensate for spatial and temporal variations and thus increase treatment efficiency (Dagenais et al. 2018). This is particularly relevant for retention systems in temperate climates with dry summers and cold winters. One aspect of this is the peak activity during the growing period of the plants. Some plants are particularly active at the beginning of the season, others at the end. The presence of a wide range of species and functional diversity can ensure the minimisation of runoff through ET and the removal of pollutants and nutrients throughout the growing cycle (Dagenais et al. 2018; Markiewicz et al. 2020). Other benefits of polycultures include multiple affinities for different pollutants and nutrients, seasonal growth patterns, water requirements, increased resilience, and aesthetics (Liang et al. 2011; Zhang et al. 2010).

Overall, plant selection is essential to maximise efficiency in terms of improving water quality, reducing runoff volume, and retaining pollutants in the bioretention system. In this respect, plants with high resistance to pollutants, drought, cold and waterlogging should be selected, especially in temperate climates. In addition, polycultures should have fast growth rates, high nitrogen concentrations, and a well-developed root system (Wang et al. 2017).

2.2 Pollutants and Particles

The current priority of stormwater management is not only to minimise runoff volume and peak flows but also to reduce pollutant concentrations (Dagenais et al. 2018). Road runoff in particular is a major contributor to increased pollution of receiving waters (Dietz 2007; Lucke and Beecham 2011). In this regard, untreated stormwater in heavily trafficked areas is considered a sink for pollutants. Contaminated dust consists of metals, tyre and brake pad abrasion, nutrients, oil, OPs, lubricants, hydrocarbons, sediments, and the road material itself (Müller et al. 2020; Pamuru et al. 2022; Polukarova et al. 2020; Novotny et al. 2008; Trenouth and Gharabaghi 2016). In addition, stormwater runoff from roads has been identified as a significant source of microplastic inputs (Järlskog et al. 2022a; Järlskog et al. 2022b; Järlskog et al. 2021; Järlskog et al. 2020; Kole et al. 2017).

In order to prevent the general deterioration of water quality and changes in hydrography, it is essential to reduce the pollutant content before the water is discharged. These pollutants are removed in bioretention systems by infiltration, filtration, sorption, ion exchange, plant uptake, and microbial transformation (Jiang et al. 2019). In addition, purification performance is strongly influenced by retention time, pollutant load, inflow volume, dry periods, and other factors such as media or plant selection (Jiang et al. 2019). However, the extent to which pollutants can be removed or retained using different filters and sorbent materials still needs to be analysed in various field experiments.

As several studies have already been carried out on metals in road runoff and their removal efficiency in bioretention systems (Al-Ameri et al. 2018; Søberg et al. 2017; You et al. 2019), these are not considered below.

2.2.1 Nutrients

Nutrients in stormwater are mainly present in the form of phosphorus or nitrogen. The input can originate from natural, anthropogenic, or biogenic sources such as atmospheric deposition, car washing, car exhaust, heavy vehicles, fertilisers, or plant residues (Yang and Chui 2018).

The effects of excess nutrients in stormwater are diverse and depend on several factors. For example, elevated levels of nitrogen and/or phosphorus can have negative impacts on ecosystems (Jiang et al. 2019). The altered nutrient content disturbs the natural balance, which in turn affects soil quality and plant growth (Razaq et al. 2017). When stormwater is discharged into watercourses, the high nutrient content also promotes eutrophication processes. The increased growth and subsequent death of algae cause oxygen depletion in the water, which can affect aquatic life (Yang and Toor 2018; Anderson et al. 2008; Correll 1998). In addition, the nitrogen compounds present in stormwater can contribute to the formation of nitrous oxide or ground-level ozone, which not only degrades air quality but also contributes to climate change (Massara et al. 2017). In order to minimise the impact on the environment, it is therefore necessary to monitor and understand the nutrient content of stormwater. Measures must then be developed and implemented to limit nutrient inputs to ecosystems.

The nitrogen content and the different forms of N in stormwater have already been analysed in several studies. For example, in the stormwater sample analysed by Jani et al. (2020), 47% of the N was present in dissolved organic form, 22% as particulate organic N, 17% as NO_x-N, and 14% as NH_4-N. (Jani et al. 2020) The study by Taylor et al. (2005) also reported 75–85% dissolved N. As nitrogen is predominantly present in dissolved form, it can be assumed that nitrogen decomposition takes place mainly through biological processes (Zinger et al. 2013). The nitrogen cycle is significant, involving the mineralisation of organic nitrogen, followed by nitrification of ammonia to nitrite and nitrate, denitrification, and subsequent assimilation by microorganisms and plants (Payne et al. 2014a). The degree of nitrogen retention in bioretention systems is highly dependent on the choice and diversity of plants as well as the retention time of stormwater in the system (Taylor et al. 2005). Plants with a fast growth rate are expected to be particularly effective in influencing nitrogen uptake (Dagenais et al. 2018). In addition, systems with a continuous saturated aerobic zone and sufficient carbon source contribute positively to nitrogen removal in bioretention systems (Shrestha et al. 2018). The presence of a saturated zone supports denitrification processes and can improve nitrate removal efficiency by up to 71% (Palmer et al. 2013). However, if the denitrification process is incomplete or NH_3N is nitrified, NO_3N can be released (Collins et al. 2010). Overall, the study of Wang et al. (2017) demonstrated good nitrogen removal rates. The reduction in this respect was 53–84% for TN and 43–92% for NO_3N.

Similar to nitrogen, phosphorus exists in stormwater and in soil in diverse forms, including dissolved, particulate, organic, or inorganic states (Roy-Poirier et al. 2010). The degradation processes primarily occur through passive sedimentation and adsorption, underscoring the significance of the filter medium in bioretention systems for the effective removal of particulate phosphorus. Utilizing media with a high phosphorus sorption capacity, such as calcite or iron-oxide-containing substances, is recommended (Dagenais et al. 2018). Dissolved phosphates can also be stored by plants. By cutting the above-ground plants, phosphorus removal of 1 and 5 g P/m^2 is possible (Dagenais et al. 2018). The phosphorus removal efficiency in bioretention systems therefore varies depending on the sorbent material, plant selection, and stormwater composition. However, an average removal rate of 65% can be assumed (Beral et al. 2023; Paus et al. 2014).

In general, N and P are dynamic nutrients that are found in varying forms during treatment. In this respect, it is still unclear to what extent the ageing processes of the bioretention system affect long-term nutrient retention (Dagenais et al. 2018). In addition, there is increased leaching of dissolved nutrients, particularly during long or intense rainfall events. The reduction in treatment efficiency is due to the reduced retention time in the media, which is insufficient for the bio- and physicochemical processes (Shrestha et al. 2018). Leaching of N and P results in variable removal rates (Poor et al. 2021). In order to mitigate this and to gain a better understanding of the removal efficiencies of different filter media concerning nutrient removal in bioretention systems, it is necessary to carry out further investigations under field conditions and with natural stormwater.

2.2.2 Organic Pollutants

Another component of urban stormwater runoff is organic pollutants. Up to 1100 different OPs have been identified to be emitted from road traffic (Markiewicz et al. 2017). PAHs, C20-C40 alkanes, alkylphenols, phthalates, aldehydes, phenolic antioxidants, bisphenol A, oxygenated PAHs, C5-C12 naphtha, amides, and amines should be removed from stormwater in descending order (Markiewicz et al. 2019). The treatment of these pollutants holds particular significance, as these substances pose risks not only to the environment but also to human health. Given their potential persistence, carcinogenicity, toxicity, teratogenicity, and mutagenicity, addressing these pollutants is crucial for safeguarding both environmental integrity and human well-being (Jiang et al. 2019; Markiewicz et al. 2019). Previous studies have reported PAH removal efficiencies of 31–99% per rainfall event (DiBlasi et al. 2009). In general, only a few studies have been conducted on the treatment of OP in bioretention systems. However, these have shown high treatment efficiencies (Dagenais et al. 2018; Zhang et al. 2014).

As OP is not only present in particulate form but also in dissolved or colloidal state, it is also necessary to remove these from the runoff. These non-particulate compounds can form emulsions at the nanoscale and thus facilitate the leaching of pollutants (Markiewicz et al. 2020). Furthermore, it was found that dissolved and colloidal OP can only be removed to a limited extent by sorption processes on filter media (Flanagan et al. 2019; Kalmykova et al. 2014). In this respect, it is essential to develop appropriate treatment methods to increase the removal efficiency.

2.2.3 Microplastics

The issue of MPs has received considerable attention in recent years. Not only have high concentrations of MPs been found in road dust, but also TRWP has been identified as the main entry route for MP. In this respect, approximately 11,000 tonnes of TRWP and 8,000 tonnes of bitumen are released annually in Sweden (Järlskog et al. 2022a; Järlskog et al. 2022b; Järlskog et al. 2021; Järlskog et al. 2020; Kole et al. 2017; Magnusson et al. 2016; Polukarova et al. 2020).

The term "microplastics" is generally used to refer to solid and insoluble synthetic polymers with a particle size of less than 5 mm (Stöven et al. 2015). A distinction is also made between primary and secondary MP. Primary MP is produced on an industrial scale, mainly in the form of plastic-based granules and pellets (Igalavithana et al. 2022). Secondary MP refers to particles generated through the chemical, physical, and biological degradation and fragmentation of meso- and macroplastic particles. The reduction in particle size primarily occurs due to mechanical forces, UV radiation, oxidation, and exposure to saltwater, leading to the production of a growing population of increasingly smaller microplastic particles (Modak and Basu 2022). Secondary MP is mainly found in urban stormwater in the form of tyre, road, and brake abrasion, plastic waste, paint, and industrial waste. (Smyth et al. 2021) These plastics tend to be polypropylene (PP), polyethylene (PE), polyamide (PA), polyethylene terephthalate (PET), polystyrene (PS), and polyvinyl chloride (PVC), as well as natural rubber (NR), styrene-butadiene rubber (SBR) or butadiene rubber (BR) (Smyth et al. 2021; Molazadeh et al. 2023; Panko et al. 2019; Werbowski et al. 2021).

If wastewater and stormwater are drained separately in cities, the latter is often discharged into the environment and water bodies without pre-treatment (Österlund et al. 2023). The elevated concentration of microplastics (MPs) in urban runoff, coupled with their persistent nature, presents potential risks. For instance, pollutants and pathogens may attach to MP particles, subsequently influencing water quality and ecosystems (Kumar et al. 2020). In addition, the small particles can be ingested by organisms. If the plastic particles contain harmful substances or additives such as phthalates, they can leach out and contaminate organisms (Vinay et al. 2023). But even without the sorption of pollutants, MP can be hazardous. These effects encompass translocation, physiological stress, alterations in energy balance, abnormal metabolism, immune responses, behavioural changes, impacts on fertility, severe intestinal damage, and mortality (Amelia et al. 2021).

In contrast to the effects of MP on fresh and marine waters and their organisms, the stress, transformation, and transport processes in soil or bioretention systems are still largely unknown (Alimi et al. 2018). A few studies have been carried out on the removal efficiencies of MP by biofiltration (Kuoppamäki et al. 2021; Lange et al. 2021; Smyth et al. 2021; Werbowski et al. 2021; Gilbreath et al. 2019). Although the systems were not specifically designed to retain MP particles, removal efficiencies of 100% were achieved for particles >500µm, 81% for MP between 355-500µm and 55% for particles between 125-355µm (Gilbreath et al. 2019). However, in the study by Lange et al. (2022), a large number of particles in the size range of 20-100µm were detected in stormwater and Rödland et al. (2022) also found that >80% of particles in road runoff are <30 µm. It is therefore important to investigate the removal efficiency of smaller particles. Nevertheless, it has already been shown that filter systems contribute positively to the retention of small MP (Rullander et al. 2023). At the same time, the plastic particles can damage plant cells and inhibit plant growth, which can affect treatment efficiency (Huang et al. 2023). The retention of microplastic (MP) particles has been found to occur predominantly within the first 5 cm of the soil layer. Despite this, wet-dry cycles have been observed to increase particle transport to deeper soil layers and promote leaching (O'Connor et al. 2019).

In general, research on the removal of MP from urban stormwater using bioretention filters can be classified as "data poor". The limited number of studies, coupled with restricted comparability of results, requires the development of research initiatives with standardised analytical methods, parameters, and objectives in order to fully understand the retention and transport of MP in bioretention systems (Österlund et al. 2023). In addition, it is essential to investigate the influence of colour, density, surface morphology, shape, size, and weathering processes, as these can influence and change the properties, sorption capacities, and affinity for pollutants of MP (Amelia et al. 2021).

3. Materials and Methods

Increasing the retention and degradation capacity of pollutants and particles from urban stormwater through bioretention filter systems is a current area of research. In order to investigate the effects of drought on the treatment efficiency of different filter sorption materials under field conditions, the present work consists of a field and a laboratory part. The fieldwork was carried out from June 13th 2023 to November 6th 2023. A planted biofilter of each sorption medium (ash, biochar, peat) and a control filter were exposed to a dry period of 13 weeks (June 13th 2023 to September 13th 2023). Following the fieldwork, the samples were prepared and analysed in the laboratory, and the results evaluated.

3.1. Rain Garden Pilot Facility

The pilot rain garden used in this Master's thesis was designed and built by Chalmers University of Technology during spring 2022 in the central part of Gothenburg close to the highway E6 (Johansson et al. 2024). The stormwater used for irrigation is collected in a chamber that receives stormwater from the 5.1 ha catchment area of Gårda (Figure 1). The catchment area consists of a total of 2.1 ha of impervious surfaces in the form of roads (82%), roofs (6%) or other impermeable surfaces (Markiewicz et al. 2017; Björklund et al. 2009). As shown in Figure 1, the rain garden is located next to the E6 highway, which is one of the busiest roads in Sweden with approximately 86,000 vehicles per day. As a result, the E6 contributes significantly to road wear and pollutant emissions to the environment (Björklund et al. 2009).

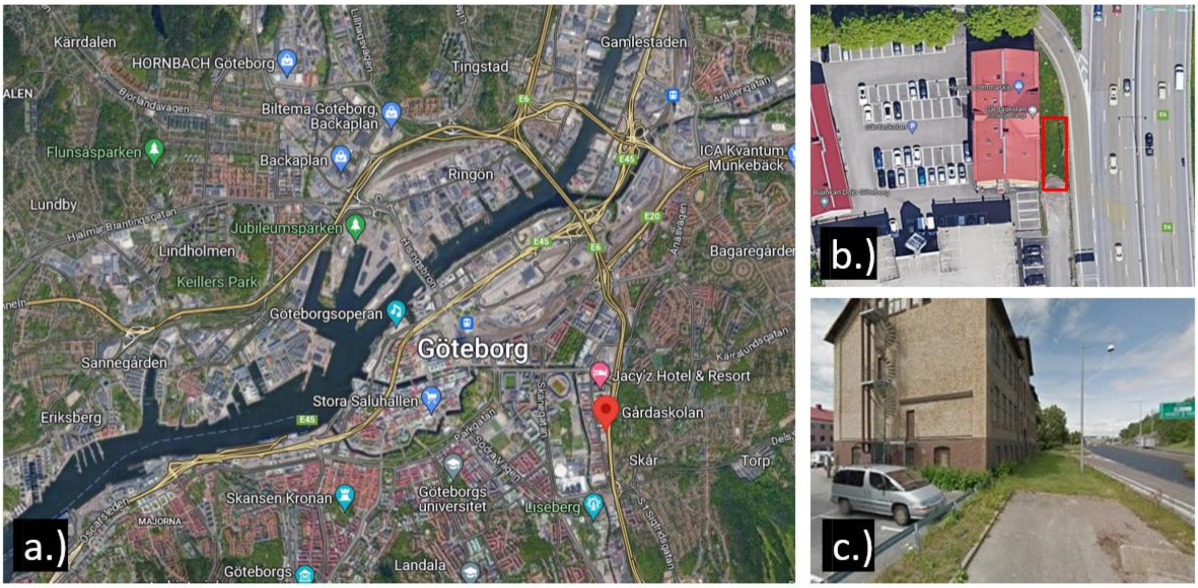

Figure 1 - (a) Map of Gothenburg, on which the Gårda area is marked with a red dot, (b) Aerial view of the construction site for the Gårda rain garden, (c) Photo of the site selected for the rain garden (from Johansson et al. 2024)

The rain garden was constructed on a pilot scale and consists of 13 bioretention filters in polyethylene columns (~980 litres/column). The design of the filtration units is shown in Figure 2. Throughout the test, the filters were watered only from above. To collect the filtered effluent, a perforated drainpipe 110 mm above the ground was used to collect the water into 20 litre canisters. Underneath the pipe, there was a saturated zone, and as described in section 2.2.1, the presence of a saturated zone can have a variety of positive effects on the treatment efficiency (Shrestha et al. 2018; Palmer et al. 2013).

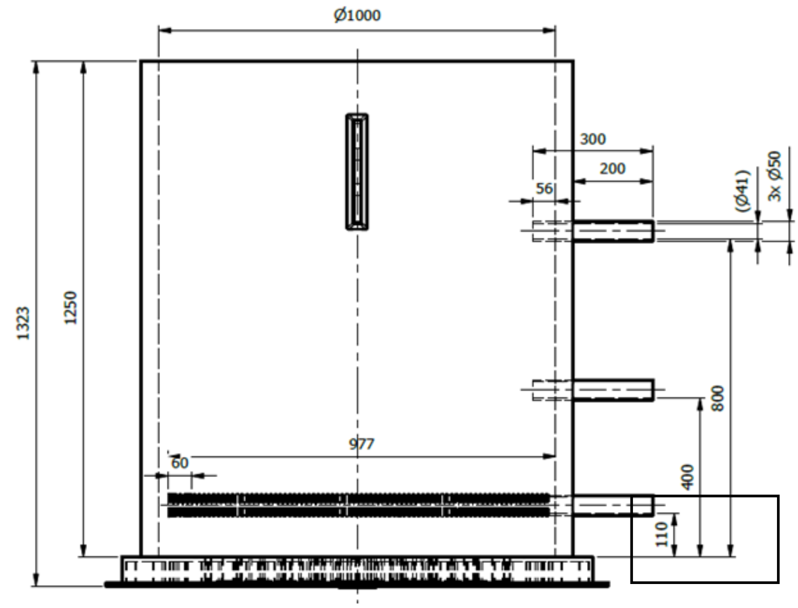

Figure 2 - Structure of the filter column for the bioretention filters in the pilot plant of the Gårda rain garden [in mm] (modified from Johansson et al. 2023)

Three different sorption materials were used in the rain garden: ash, biochar and peat. For each of these materials, there was one bioretention filter without plants and three filters with plants. In addition, a control filter (C) was constructed without a sorption layer but with plants. The species used in this study include thrift (*armeria maritima*), sea buckthorn (*hippophae rhamnoides*), rushes (*juncus effusus*) and red fescue (*festuca rubra*). These non-invasive plants were selected because they do not only tolerate drought, but also resist large amounts of water and increased salinity, and contribute positively to the degradation of pollutants (Johansson et al. 2024). The distribution and location of the various bioretention filters within the rain garden are shown in Figure 3.

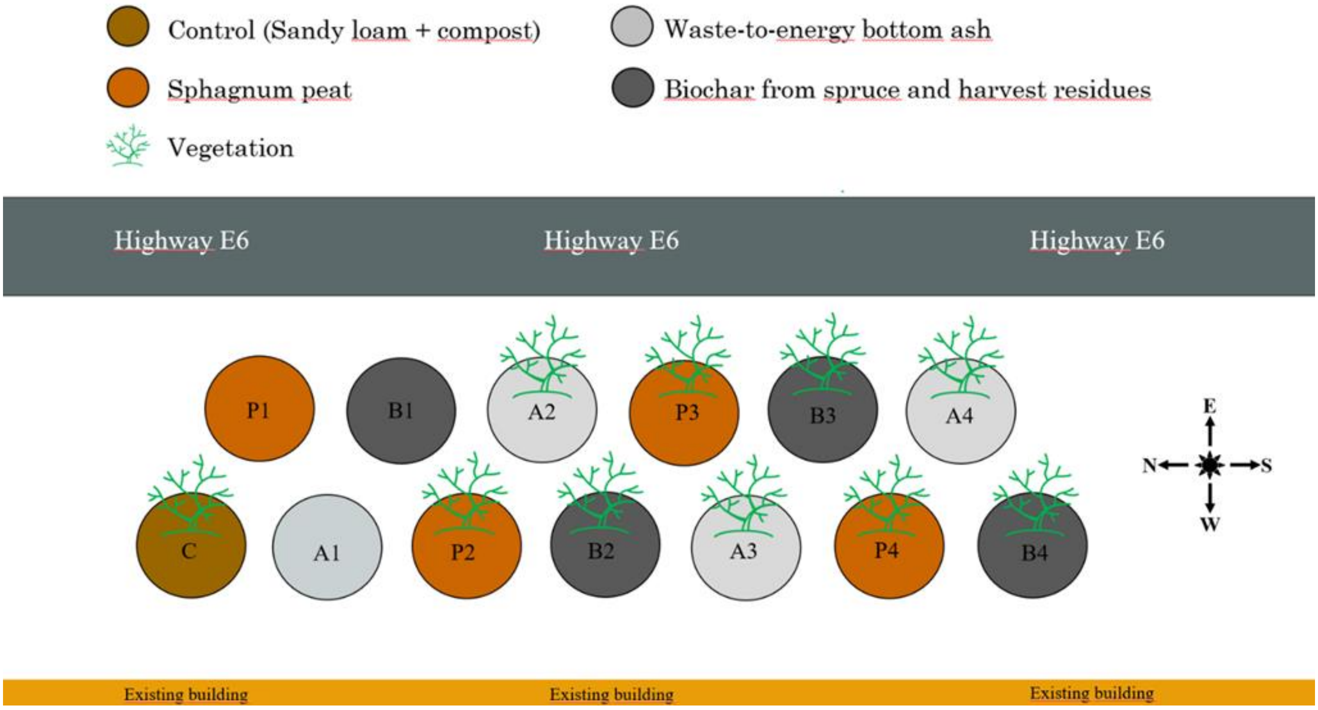

Figure 3 - The position of the various columns, with and without plants, in the rain garden at Gårda (from Johansson et al. 2024)

Filter Bed Structure

When designing bioretention systems, filter materials are important for the retention of pollutants and particles. For this study, the layering of the different soil materials within the bioretention filters was based on a defined principle (see Table 1). The filter bed structure, from top to bottom, consists of a 150 mm thick soil layer of sandy loam with pumice, followed by a layer of sandy loam with pumice, compost and sorbent (550 mm), a sorbent layer of 100 mm, and drainage layers of fine sand (100 mm) and gravel (200 mm). An exception in this respect is the ash filter, whose sorbent layer consists of 50 mm each of peat and biochar. It is therefore the only filter structures that contains all sorption material.

Municipal incineration bottom ash (MIBA), biochar and *sphagnum* peat were used as sorption materials. The selection of these materials was based on the results of previous studies regarding their sorption capacity and retention efficiency. A positive contribution was observed from MIBA in the removal of metals and water-soluble OPs (Karlfeldt Fedje et al. 2021; Markiewicz et al. 2020). Biochar and peat, on the other hand, have been shown to absorb a variety of pollutants from stormwater, including OPs, metals as well as nutrients (Kalmykova et al. 2009). For more information on the properties of different sorbents, see Chapters 2.1.1.1, 2.1.1.2 and 2.1.1.3.

Table 1 - Soil-bed materials in the different bioretention filter columns in the pilot rain garden at Gårda. (modified from Johansson et al. 2024)

Depths from top to bottom	**Filter material**	**Depths from top to bottom**	**Filter material**
(mm**)**	**Control**	(mm)	**A1-A4**
0-150	Sandy loam with pumice stone (Hekla Regnbädd)	0-150	Sandy loam with pumice stone (Hekla Regnbädd)
150-800	A mixture of 85% Hekla Regnbädd and 15% compost	150-700	A mixture of 35% Hekla Regnbädd, 50% MIBA and 15% compost
800-850	Sand (size 0.4–0.8 mm)	700-750	Biochar
850-900	Sand (size 0.8–1.2 mm)	750-800	Peat
900-1000	Gravel (size 3–5mm)	800-850	Sand (size 0.4–0.8 mm)
1000-1100	Gravel (size 4–8mm)	850-900	Sand (size 0.8–1.2 mm)
		900-1000	Gravel (size 3–5mm)
		1000-1100	Gravel (size 4–8mm)
	P1-P4		**B1-B4**
0-150	Sandy loam with pumice stone (Hekla Regnbädd)	0-150	Sandy loam with pumice stone (Hekla Regnbädd)
150-700	A mixture of 60% Hekla Regnbädd. 40% peat	150-700	A mixture of 60% Hekla Regnbädd. 40% biochar
700-800	Peat	700-800	Biochar
800-850	Sand (size 0.4–0.8 mm)	800-850	Sand (size 0.4–0.8 mm)
850-900	Sand (size 0.8–1.2 mm)	850-900	Sand (size 0.8–1.2 mm)
900-1000	Gravel (size 3–5mm)	900-1000	Gravel (size 3–5mm)
1000-1100	Gravel (size 4–8mm)	1000-1100	Gravel (size 4–8mm)

A detailed characterisation of the filter bed materials and information on the selection procedures for the sorbents and plants used in the Gårda rain garden can be found in the corresponding paper by Johansson et al. 2024.

3.2. Fieldwork

The fieldwork within this study included irrigation and on-site sampling.

Irrigation

The stormwater used for irrigation originates from the Gårda catchment area and was channelled into the existing pre-sedimentation facility, consisting of seven chambers connected in series. During the study, the rainwater was pumped up from the first chamber and temporarily stored in IBC tanks. Prior to watering the rain garden, the water in the tanks was mixed using agitators to achieve a uniform water composition for each watering day. The biofilters were then watered manually from above using 10 litre watering cans. It should be noted that natural stormwater was used, so it can be assumed that the composition of the stormwater varies from day to day.

In total, the filters without drought were watered 13 times during the study period from 2023-06-13 to 2023-11-06. Filters C, A4, B2 and P4 (all with plants) were subjected to an artificial drought for three months (2023-06-13 – 2023-09-13). During this period, the filter systems were not irrigated, and rain roofs with open sides were used to block any direct precipitation. The plants only received water from the moisture in the air from the open sides. The 2023-09-04 is an exception as all biofilters were irrigated to protect the plants from total dryness. In order not to distort the results of the study, a small volume of 20 litres was chosen. This volume of water did not generate any effluent and therefore no particles or contaminants were flushed out prematurely. In total, the columns that experienced a drought were watered 8 times after the dry period. The irrigation volume of all biofilters within the study varied between 20-150 litres per filter. In order to make the results more comparable, efforts were made to use the same amount of water per irrigation day and per filter. The water volume used by Johansson et al. 2024 was taken as a reference, which varied between 20 and 70 litres. Only on 2023-09-13 the amount of water supplied to each filter varied, as the first flush was the main focus on that day. The exact amount of water per filter column and day is shown in Table 2.

Table 2 - Irrigation volume per filter and irrigation day

Date/Sample	**C**	**A1**	**A2**	**A3**	**A4**	**B1**	**B2**	**B3**	**B4**	**P1**	**P2**	**P3**	**P4**
2023-06-13	70	70	70	70	70	70	70	70	70	70	70	70	70
2023-07-10	0	40	40	40	0	40	0	40	40	40	40	40	0
2023-07-13	0	90	90	90	0	90	0	90	90	90	90	90	0
2023-07-28	0	40	40	40	0	40	0	40	40	40	40	40	0
2023-08-11	0	40	40	40	0	40	0	40	40	40	40	40	0
2023-08-23	0	40	40	40	0	40	0	40	40	40	40	40	0
2023-09-04	20	20	20	20	20	20	20	20	20	20	20	20	20
2023-09-13	150	50	140	140	140	60	140	140	140	50	140	140	140
2023-09-19	70	70	70	70	70	70	70	70	70	70	70	70	70
2023-10-02	70	70	70	70	70	70	70	70	70	70	70	70	70
2023-10-20	40	40	40	40	40	40	40	40	40	40	40	40	40
2023-10-26	70	70	70	70	70	70	70	70	70	70	70	70	70
2023-11-06	70	70	70	70	70	70	70	70	70	70	70	70	70

Dry period   Sampling days

Sampling

As part of this study, a total of five sampling days were carried out (see Table 2). The purpose of the sampling was to determine the particle and contaminant retention as well as the pollutant removal capacity of the different filter materials. Furthermore, the effects of dryness and the regeneration capacity of the materials could be investigated in the laboratory.

Prior to sampling, the 20 litre collection vessels of all filter columns were emptied. The water samples were then taken directly from the IBC tank for the inlet samples and the outlet samples from the collection vessels of each filter column using a peristaltic pump and analysed on site using the Hanna Multimeter. These samples were collected in 1 litre glass bottles. The stormwater composition of the inlet samples from the IBC tank serves as a reference value by which the removal efficiencies of the different filter bed compositions can be determined. Due to the different infiltration properties of the soils, the runoff volume from the different biofilters varied. Therefore, samples were only taken when there was sufficient flow in the outlet tank of a total volume of over 10 litres. The samples were then transported in cooling bags and the glass bottles were stored in refrigerators until sent to commercial laboratories or analysed at Chalmers.

3.3. Laboratory Work

After collecting the samples, the water probes were analysed in the laboratory. The tests for TSS/VSS, TOC/DOC and cations and anions were carried out by the author herself in the Water and Environment Lab at Chalmers University. Tests related to OP and MP, on the other hand, were carried out at external laboratories.

3.3.1 Suspended Materials

For the determination of Total Suspended Solids (TSS) and Volatile Suspended Solids (VSS), the ESS Method 340.2: Total Suspended Solids, Mass Balance (revised June 1993) was used. The test procedure is shown in Figures 4 for TSS and 5 for VSS.

Total Suspended Solids

For the TSS analysis, glass filter papers with a diameter of 47 mm and a particle retention size of 0.7 µm were used. To avoid the release of loose filter particles during filtration, the filters were first rinsed with Milli-Q water using the vacuum pump. The filters were then baked within the aluminium cups in a muffle furnace at 550°C ± 50°C for 2h and placed in a desiccator to cool down. Once the filters had reached room temperature, they were weighed together with the cup (A). The stormwater samples were then filtered. For this purpose, the filters were placed on the vacuum pump and moistened with MQ. Before adding the water samples, the water was shaken, and the filtration volume was measured (B). These steps are necessary to ensure that the sedimented particles are evenly distributed within the water and that the results are not distorted. Measuring the volume on the other hand is necessary for subsequent calculations. Different sample volumes were used depending on the sample. Either the volume of water was filtered until the paper was saturated or until a volume of ±1 litre was reached. The filter papers were then dried overnight at 105°C, cooled in a desiccator and weighted (C). Afterwards, the TSS content could be calculated according to the following formula:

$$TSS\ \left[\frac{mg}{L}\right] = (A - C) * \frac{1{,}000}{B}$$

With: A = weight of dry filter and cup + residue [mg]

B = volume of sample filtered [mL]

C = weight of filter and cup [mg]

Volatile Suspended Solids

After determining the TSS content, the proportion of VSS in the water samples can be determined. For this purpose, the previously weighed filter papers with the beakers were reheated in the muffle furnace at 550°C ± 50°C for two hours, cooled in the desiccator and then weighed (D). The VSS content was calculated using the following formula:

$$VSS\ \left[\frac{mg}{L}\right] = (A - D) * \frac{1{,}000}{B}$$

With: A = weight of dry filter and cup + residue [mg]

B = volume of sample filtered [mL]

D = weight of baked filter and cup + residue [mg]

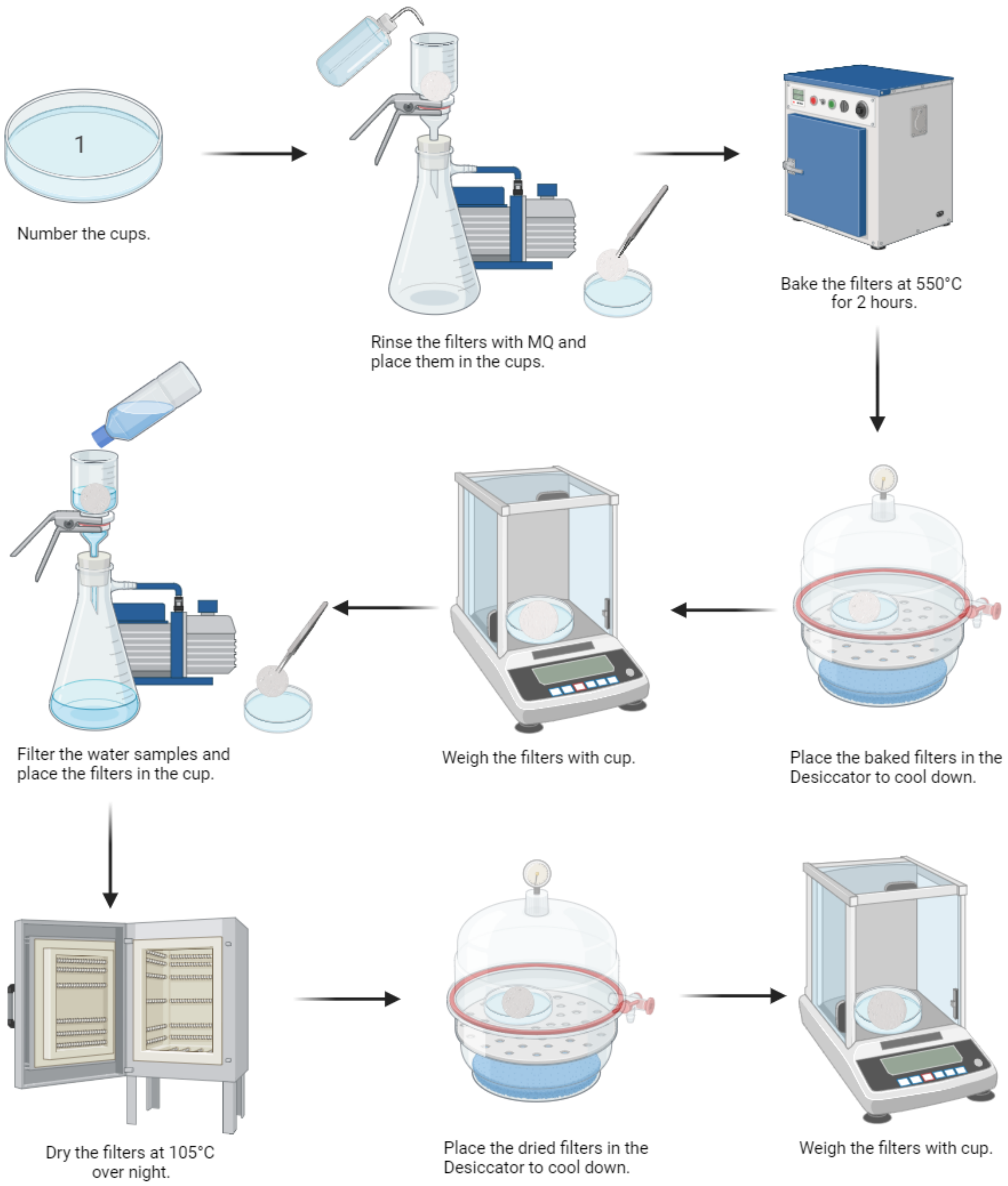

Figure 4 - Experimental procedure for the determination of TSS (created via Biorender.com)

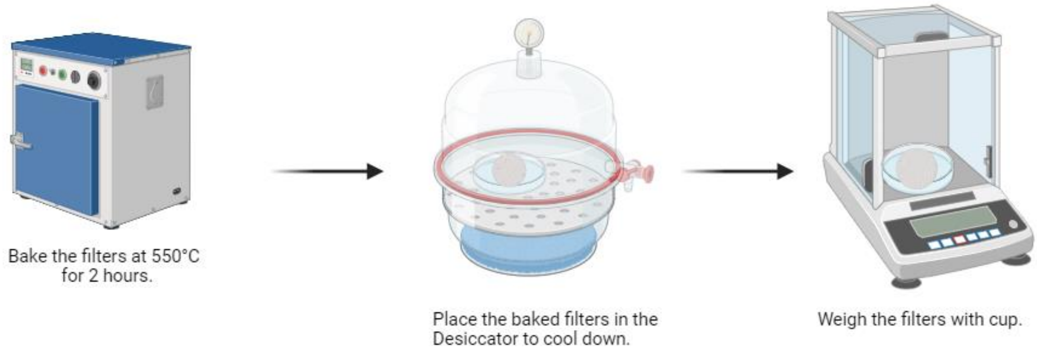

Figure 5 - Experimental procedure for the determination of VSS (created via Biorender.com)

3.3.2 Anions and Cations

Ion chromatography (IC) was used to analyse the anions and cations present in the samples. The first step was the sample preparation (see Figure 6). The stormwater samples were initially filtered through a filter with a pore size of 0.45 µm. This was necessary to remove impurities such as particles, suspended solids or organic matter from the sample. Subsequently, the conductivity of the samples was measured. As the Thermo-Fisher Dionex ICS-900 used is designed for conductivities in the range of 300 ± 100µS, it was necessary to dilute the samples if their conductivity exceeded this value. After diluting, the water samples were filled into the clean IC vials, capped and placed in the autosampler. The analysis was performed using the cation/anion method with an analysis time of 20 min per sample, a sample volume of 0.5 ml, the conductivity detector and a flow rate of 1 ml/min. Samples were tested for the following anions: acetate, bromide, chloride, nitrate, nitrite, phosphate and sulphate. The determination of cations included: ammonium, calcium, lithium, magnesium, manganese, potassium and sodium.

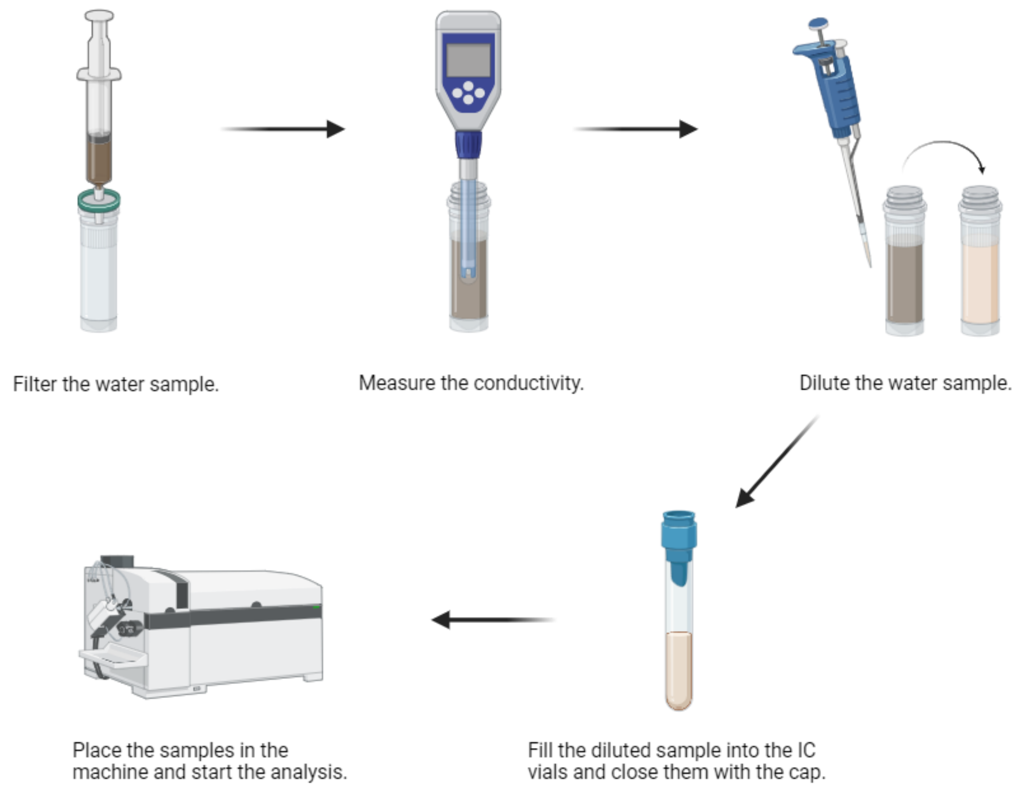

Figure 6 - Sample preparation for the determination of ions using IC (created via Biorender.com)

3.3.3 Organic Carbon and Nitrogen

Total organic carbon (TOC), dissolved organic carbon (DOC) and total nitrogen (TN) were analysed using the 9ml-TC-IC-TN 200mg/L method 415.3 in the TOC-Vcph Shimadzu. The preparation of the samples followed the procedure described in Chapter 3.4.2. However, the filtration step was omitted for the determination of TOC and TN. In addition, all diluted samples were filled into the appropriate sample vials for the Shimadzu.

3.3.4 Organic Pollutants

As part of this study, BTEX, aliphatic, aromatic and polycyclic aromatic hydrocarbons were analysed. All organic pollutants (Ops) were analysed by GC-MS, but based on different methodological principles. The methods used to detect aliphatic (aliphatic >C5 -C8 , aliphatic >C8 -C10 , aliphatic >C10 -C12 , aliphatic >C12 -C16 , aliphatic >C5 -C16 , aliphatic >C16 -C35) and aromatic hydrocarbons (aromatics >C8 -C10 , aromatics >C10 -C16 , methylpyrene/methylfluoranthene, methylchrysene/methylbenz(a)anthracene, aromatics >C16 -C35) were performed according to the SPIMFAB manual and the internal methods of the commercial laboratory. The 16 PAHs (PAH-L: naphthalene, acenaphthylene, acenaphthene; PAH-M = fluorene, phenanthrene, anthracene, fluoranthene, pyrene; PAH-H = benz(a)anthracene, chrysene, benz(b)fluoranthene, benz(k)fluoranthene, benz(a)pyrene, indeno(1,2,3,cd)pyrene, benz(g,h,i)perylene, dibenz(a,h)anthracene) were determined based on US EPA 8270D, US EPA 8082A, CSN EN ISO 6468 and US EPA 8000D.

The selection of the investigated OPs and the analytical methods are based on previous studies by Markiewicz et al, 2017; Björklund et al, 2009; Polukarova et al, 2020; Järlskog et al, 2021 and Johansson et al, 2024. In the above-mentioned studies, these OPs were identified as primary pollutants in road and stormwater runoff in urban areas.

3.3.5 Microplastics

The investigation involved analysing stormwater for ten different types of plastic and rubber particles. These included polyisoprene (PI), polybutadiene (PB), polyethylene (PE), polypropylene (PP), polyvinyl chloride (PVC), polystyrene (PS), polyethylene terephthalate (PET), polyamide 6 (PA6), polymethyl methacrylate (PMMA) and polycarbonate (PC). To determine the respective proportions, the stormwater samples were first filtered through a stainless steel filter (SSF, pore size 10 µm). The extracted filter was then subjected to ultrasonic treatment for ten minutes in a beaker filled with water, rinsed, and washed. KOH was then added until the final concentration of 10% KOH was reached.

Afterwards, the stormwater sample was incubated overnight at 40°C. The filtration process was then repeated and CaCl2 was added as a density separator. In total, the samples were decanted three times. Prior to analysis on the pyrolysis GC/MS instrument (Frontier-labs Pyrolyzer, Agilent GC 8890, Agilent MS 5977b), the samples were filtered again using a 1.6 µm GF/A filter. Finally, the generated results were processed and interpreted using F-Search 2.0. For a more detailed description of the MP analysis method used, see (Johansson et al. 2024).

3.4. Removal Efficiency

The following formula was used to determine the pollutant and particle removal efficiency (RE). It is a rough estimate of the purification performance assuming that the inflow corresponds to the outflow. In this respect, water losses in the form of water absorption by plants, sorption by the filter materials or evaporation losses are neglected and equated to zero.

$$\mathrm{RE}\ [\%] = \frac{C_{Inc.SW} - C_i}{C_{Inc.SW}} * 100$$

With: $C_{Inc.SW}$ *= influent concentration [mg/L]*

C_i *= effluent concentration [mg/L]*

4. Results and Discussion

In order to minimise the deterioration of water quality caused by stormwater discharges, this study is based on the limits set by the City of Gothenburg (see Table 3). It should be noted that the standards may vary depending on the city, region, and country. For those parameters where the City of Gothenburg has not specified a limit value, the assessment is based on the RE alone.

Table 3 - Target values for various particles and pollutants before the discharge of polluted water (Göteborgs Stad 2020)

Substance/ Parameter	Target value
Nitrogen	site-specific if necessary, from 1.25 mg/l
Suspended material	25 mg/l
Benz(a)pyrene, indicator for PAH	0,27 µg/l
TOC	site-specific if necessary, from 12 mg/l

4.1. Suspended Materials

The reduction in suspended solids due to treatment in the various filter columns was determined both visually and in the field using the Hanna Multimeter, as well as in the laboratory using ESS method 340.2: Total Suspended Solids, Mass Balance.

4.1.1 Turbidity and Coloured Particles

In the water samples, a minimisation of visible solid particles and colourants was observed after the treatment of all bioretention filters, see Figure 7 and Table 6 Appendix A. Comparing the samples from all five sampling days, see Appendix A - Figure 14, it becomes apparent that the turbidity and particle content of the incoming stormwater varies significantly. The water sample from 2023-10-02 is visually by far the clearest, with only a few particles settled to the bottom. In contrast, the stormwater sample from 2023-11-06 had the most coloured and visible particles. This was also confirmed by the measurements of the Hanna Multimeter. As the highest turbidity of 410 FBU was obtained on the last sampling day, while the lowest turbidity values were measured on the 2023-09-19 (7.3 FNU). The discrepancies between the visual and technical results may be due to the fact that the visual and technical turbidity measurements were performed on two different water samples taken on the same day. The assumption that the different levels of turbidity depends on the number of previous days of rainfall could only be supported to a limited extent. The water used for irrigation was stored in the IBC tanks and was always from the previous sampling day, as the tanks were refilled on that day. This was done to ensure that a sufficient water amount was available for irrigation on the sampling days. Figure 15 in Appendix A shows that there were high rainfall values of up to 23.5 mm before the 2023-09-19 and on the day itself, which is likely to have significantly minimised the number of particles deposited on the road. Since the stormwater used for irrigations always originates from the previous sampling day, the effects of the increased precipitation before and on 2023-09-19 can be seen in the samples from the 3rd sampling day, 2023-10-02. Although it also rained for at least four consecutive days before the last three sampling days, the rainfall intensity only varied between 0.1 and 7.6 mm. It is unclear to what extent this contributes to minimising the number of particles in the water used for irrigation, or whether it is due to the natural variability of stormwater. Another possible cause of the increased pollution on 2023-11-06 could be the change from summer to winter tyres. However, this is a hypothesis that requires further investigation.

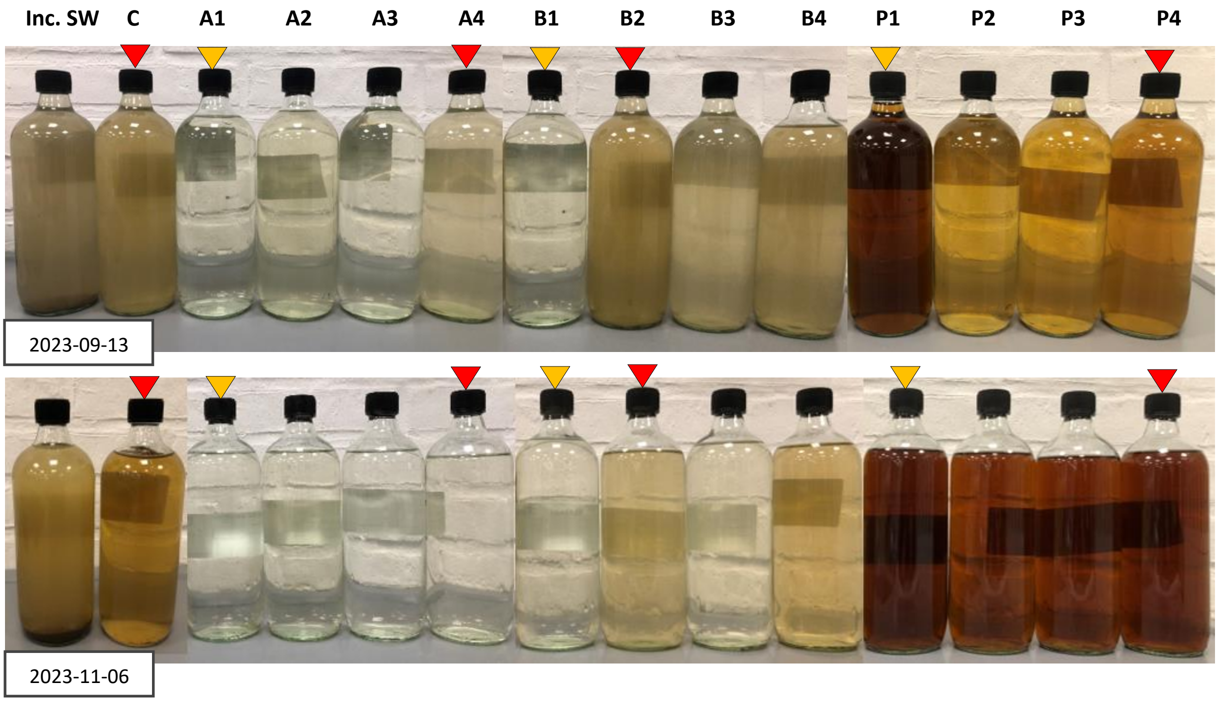

Figure 7 - Comparison of the reduction in coloured particles in all bioretention filters effluents on the first and last sampling day - the red triangle indicates the water samples from the retention filters that were exposed to the dry period, the yellow tringle indicated the filters without plants

Furthermore, the samples from the bioretention filters exposed to the artificial drought generally showed a higher turbidity on the first sampling day than the filters that was continued to be irrigated. However, the difference in turbidity decreased over the study period and samples from the ash and peat filters were barely more coloured than those without exposure to drought, indicating that the filters had regenerated. Despite this, the drought filter A1 consistently showed slightly higher FNU values than the reference filters. In the case of the peat filters, on the other hand, the unplanted filter in particular has higher turbidity values, which shows that the plants have a positive contribution in minimising turbidity in the peat filters. The drought sample of the biochar filter (B2), on the other hand, shows a consistent yellowish discolouration, similar to B4, while the discharge colours of filters B3 and B4 are different, even though they were exposed to the same conditions. The only difference between the two filter columns is that B4 has a male sea buckthorn plant. In the study of Li et al. (2005) it was observed that male sea buckthorn undergoes cold acclimatisation earlier than female plants, resulting in different chemical and biological processes during the seasonal change. This might affect the purification and retention performance of bioretention systems. However, the author is not aware of any studies that have investigated the effect of plant sex on biofilter performances. In addition, this assumption implies that the vegetation plays an extremely important role in the purification performance. However, the unplanted filters showed that high visual improvements can be achieved even without plants. At the same time, it was found that the drought had an influence on the vegetation. As the plants in the dry filters were significantly smaller than those in the reference filters.

It is also noticeable that the colour content of all effluent samples, except for the peat filters, decreased over the course of the study. The water from P2-P4 was yellow to light brown at the beginning, but dark brown at the end of the test series, which can also be observed in the increasing FNU values. There could be several reasons for this. Among other things, the colder water temperature can mean that the organic materials have a lower solubility in water, which means that they are less efficiently dispersed in the water, resulting in a more intense colour. Temperature also affects biological bacterial activity. At colder temperatures, the activity of microorganisms can be impaired, which can lead to a reduction in the decomposition of organic matter. In addition, an increased iron content may contribute to increased colouration of water (Kimura et al. 1993; Laurenson et al. 2013; Johnson et al. 1990; Markiewicz et al. 2020).

In general, visual observation did not reveal any significant differences in the removal of solid particles within the different materials and irrigation conditions. No particles were visible in any of the filter samples, only the colouring varied according to the filter materials used. However, in conjunction with the measured turbidity, the ash filters not only show the least discolouration visually, but also have the lowest turbidity values of all the sorption materials from a technical point of view. The biochar and control filters, on the other hand, show the highest FNU values, which is consistent with the results obtained by Johansson et al. (2024).

4.1.2 Total Suspended Solids

By analysing the suspended solids in the water, the TSS concentration shown in Figure 8 and the removal efficiency (RE) of the different filter materials (see Figure 9) could be determined. It can be seen that the TSS content of all effluents is lower than those of the influents and that solid suspended particles are therefore retained in all filters. It is noticeable that on the first sampling day, all filters without plants have the lowest TSS values and therefore the highest RE of all filters. On the second and third sampling days, this was only applicable to the ash and biochar filters, and from the fourth day onwards, the planted filters showed a better RE. At the same time, the solids content in the water samples exposed to drought had the highest concentration on the first day. Similar results were observed in the studies of Mangangka et al. (2015) and there are several possible reasons for this. For example, plant growth and the associated functions of water absorption, filtration and retention of particles and pollutants may have been affected by the drought. In addition, dirt and contaminants from the air may have accumulated on the filter surface, which were also washed into the retention filters during irrigation, and therefore may have impaired the filter performance. As the filter columns in this study were almost 1.30 metres high and rain roofs were used, it is assumed that this is a rather negligible process. In general, irrigation after a dry period can be considered as a first flush, resulting in an increased washout of contaminants, as dry sediments and pollutants are more mobile and therefore more easily washed out during the first irrigation (Österlund et al. 2023). This phenomenon was also observed in the study by (Sansalone and Cristina 2004). On the second sampling day, P1 and B4 already had a lower RE than the filters exposed to drought. Nevertheless, it can be seen that all the ash filters had the highest RE, which is particularly evident on the last sampling day when the influent contained the highest concentration of TOC. At the same time, the RE of the dry filters decreases on the fifth sampling day, so that they again have the lowest RE compared to the other filters. It can therefore be assumed that the effects of drought are most significant and long-lasting when high levels of pollutants are present in the influent. On this basis, future studies could investigate whether there is a permanent degradation of the bioretention filter performance in relation to influent with high pollutant concentrations, or if the regeneration period is simply longer in this case.

In general, all filters showed a similar RE of over 95% on the last sampling day, which shows the regeneration capacity of the filters exposed to a long period of drought. Regarding the effects of drought, it is clear that the filter with ash had a consistently lower RE than the irrigated filters. However, the differences are marginal and even the A4 filter achieved RE results of over 90% throughout the test period. For peat and biochar, the worst filter performance in terms of TSS was only observed on the first day for the drought-exposed filters. In the overall comparison of all drought filters, it is clear that the control filter had the lowest RE, highlighting the importance of using sorption materials in the construction of rain gardens. The filters A4 and P4, on the other hand, had a continuous RE of over 85% and showed high removal rates despite the drought. Overall, the highest TSS removal performance were achieved with ash filters, followed by peat and biochar. In particular, B4 and C have the lowest RE values. Although the filter composition is different and C has been exposed to drought, these are the only two filter columns with male sea buckthorn plants, which underlines the hypothesis presented in Chapter 4.1.1. regarding the influence of plant sex. These results are also reflected in the observation of compliance with the limits set by the City of Gothenburg (Göteborgs Stad 2020). A target value of 25 mg/L for suspended solids is defined for discharges into receiving water bodies. Filters C and B2-B4 exceed this value on the first sampling day with TSS values between 36.8 and 60.4 mg/L. On the second day, however, only C exceeds the 25mg/l limit with 30.4 mg/l and on the third sampling day C and B4 (25.2 mg/l and 26.8 mg/l). All other filters are within the specified limits.

A detailed analysis of the RE based on the different filter materials and a comparison between the different filters exposed to the drought can be found in Figures 16-19 in Appendix B. When analysing the results, it should be noted that on 2023-10-02 for A1, on 2023-10-26 for B2 and on 2023-11-06 for samples from A1 and P4, algae from the collection tanks were found in the water samples, which may influence the results.

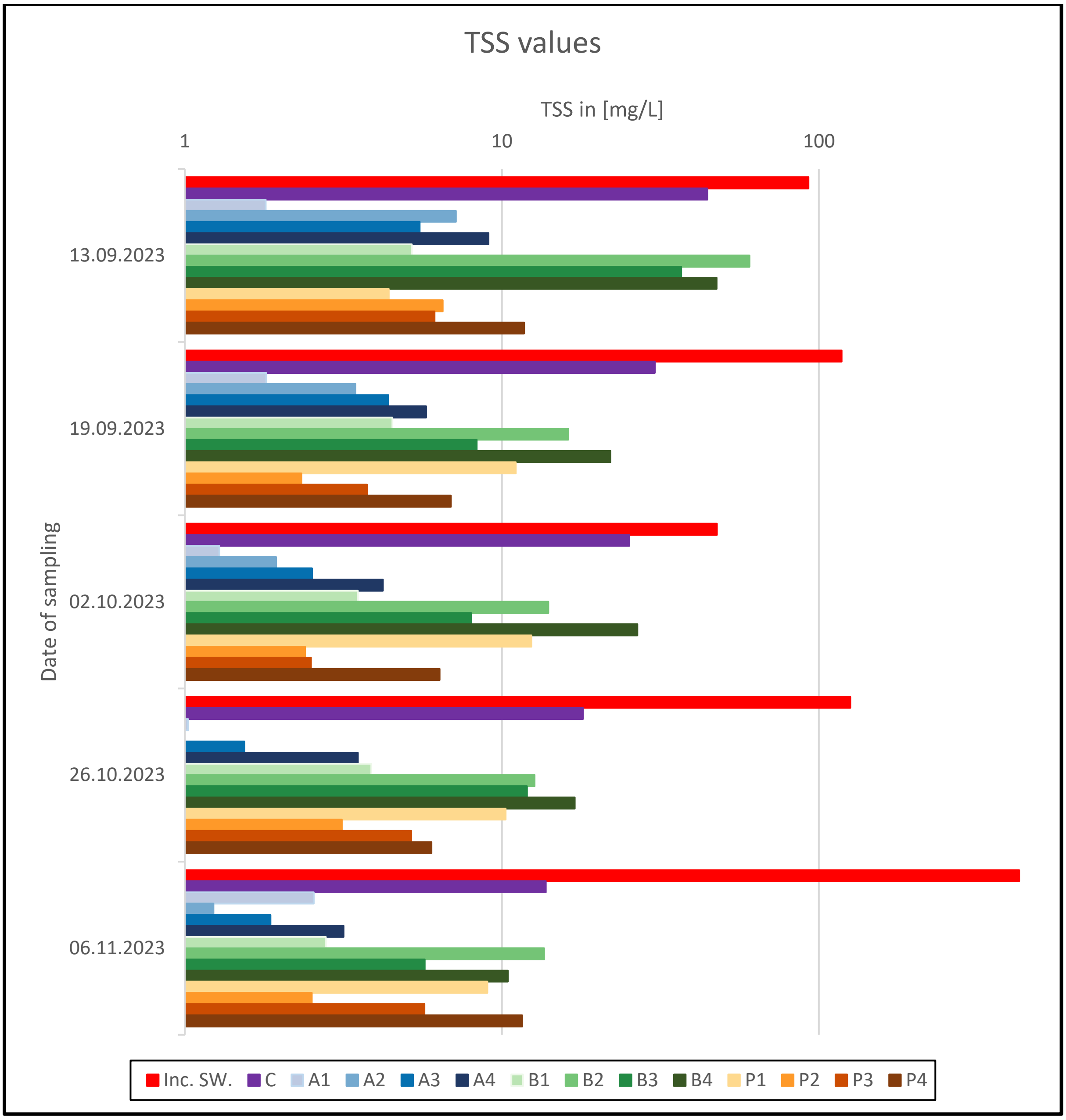

Figure 8 - TSS concentration in influents and effluents at the different sampling days in all the bioretention filters at the Gårda raingarden facility

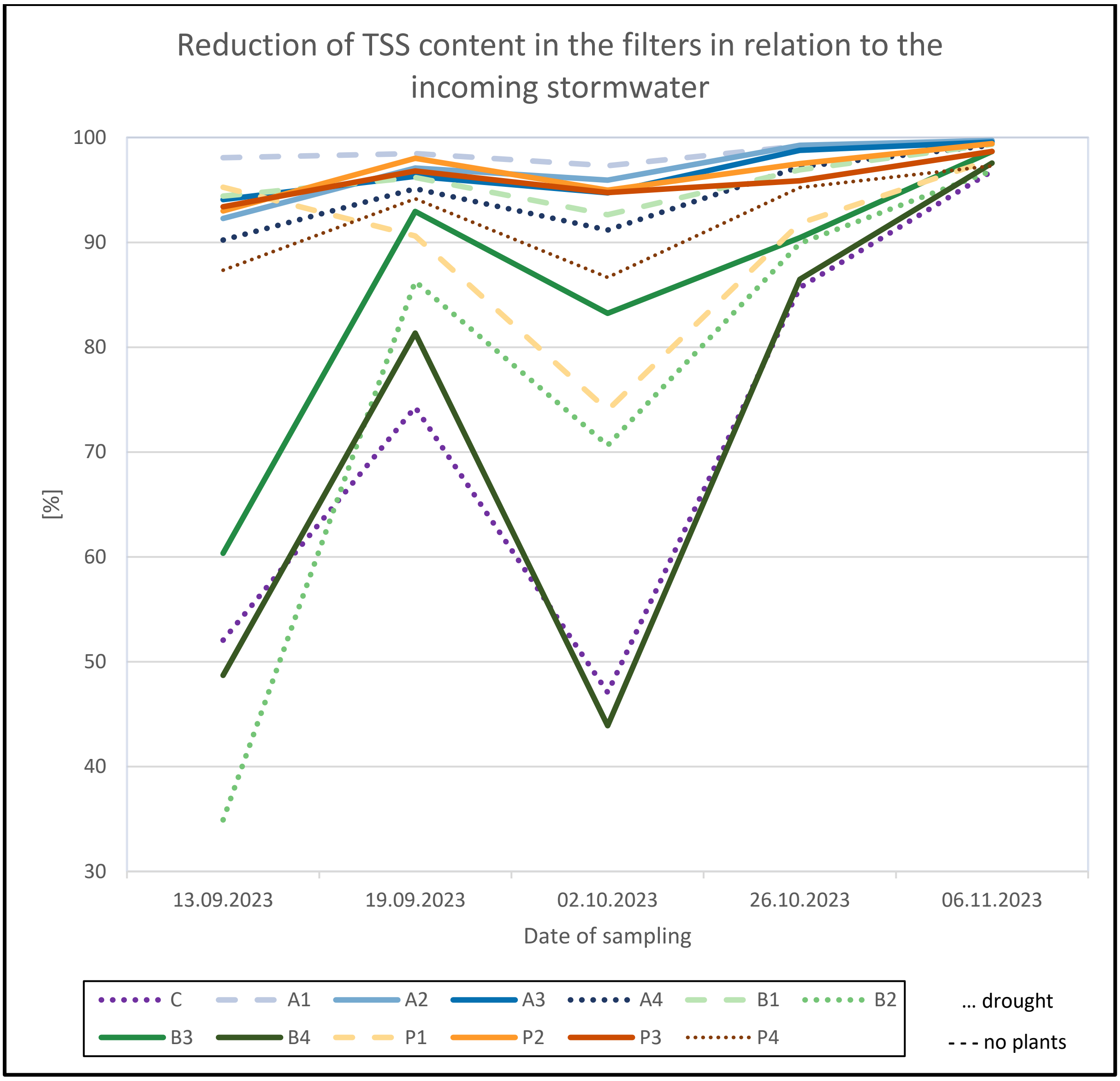

Figure 9 - TSS removal efficiency at the different sampling days in all the bioretention filters at the Gårda raingarden facility

4.1.3 Volatile Suspended Solids

In general, the results for VSS are very similar to those for TSS. However, as VSS only considers volatile suspended solids, i.e. organic suspended solids, and TSS also includes inorganic suspended solids, the proportion of VSS is lower than for TSS. The measured VSS concentrations and RE values of the different filters are shown in Figures 20 and 21, Appendix C, as well as a detailed analysis of the RE based on the different sorption materials and dry filters (Figures 22-25 in Appendix C).

As in the analysis of TSS, ash filters had the highest RE in terms of VSS with over 75%, followed by peat (>45%) and biochar (>15%). The lowest RE was found in the control filter (>2%). These values refer to the lowest RE, which was present on 2023-10-02. As all filters show a decrease in RE values on the third sampling day, it is assumed that the low values were due to the low VSS content in the influent, as this is a percentage observation. On the last sampling day, when a high VSS content was measured in the influent, the RE of all filters was above 90%. It can therefore be assumed that the filters had regenerated after the long period of drought. The filter P4 had the lowest filter performance on this day. The RE stagnated at 90% and was >7% lower than the TSS value. In this respect, it can be assumed that the drought mainly affected the retention of VSS in the peat filter, but not significantly.

In general, as with TSS, the unplanted filters had the best retention on the first day, while the filters exposed to drought had the lowest. Similarly, the drought-exposed materials also had higher VSS levels on the last day than the irrigated filters. The exception is B4, which showed lower RE than B2 on the last day. As described above, this is probably due to the sex of the sea buckthorn plant but requires further investigation. When comparing the drought filters, it becomes clear that the control filter requires the longest regeneration time in terms of VSS. Although all filters had an RE of >90% on the last sample day, the filter performance on the third day varied by more than 55% between C and the other drought filters. Overall, all measured VSS values were below the 25 mg/L limit for suspended solids for the city of Gothenburg (Göteborgs Stad 2020).

4.2. Anions and Cations

Within this study, 13 ions were identified in the water samples, of which ammonium, nitrite, nitrate, and phosphate were analysed in more detail. These four ions were selected because, as described in Chapter 2.2.1., they are nutrients that are part of the nitrogen cycle and/or contribute to the eutrophication of water bodies. In addition, these ions were identified as one of the main pollutants in stormwater in the studies by Goonetilleke et al. (2005) and Liu et al. (2012). The results are illustrated in Table 4. An overview of the concentrations of all measured anions and cations is also shown in Tables 6 and 7 in Appendix D.

Table 4 - Nitrite, nitrate, ammonium and phosphate concentrations in the influents and effluents from the bioretention filters at the Gårda raingarden pilot on the different sampling days (quantification limit: 0.01 mM)

[mM] sampling day	Anions	Inc. SW.	C	A1	A2	A3	A4	B1	B2	B3	B4	P1	P2	P3	P4
1	Nitrite	0.043	0	0	0	0	0	0	0	0	0	0	0	0	0
2	*[NO_2^-]*	0.023	0	0	0	0	0	0	0	0	0	0	0	0	0
3		0.023	0	0	0	0	0	0	0	0	0	0	0	0	0
4		0.021	0	0	0	0	0	0	0	0	0	0	0	0	0
5		0	0	0	0	0	0	0	0	0	0	0	0	0	0
1	Nitrate	0.076	0.144	0.168	0.207	0.220	0.208	0.078	0.128	0.120	0.156	0	0.088	0.084	0.092
2	*[NO_3^-]*	0.036	0.088	0.186	0.200	0.208	0.203	0.084	0.116	0.110	0	0.052	0.084	0.092	0.084
3		0.044	0.104	0.165	0.200	0.192	0.200	0.078	0.100	0.110	0.132	0.046	0	0.088	0
4		0.042	0.084	0.170	0.176	0.168	0.154	0.048	0.092	0.126	0.132	0.025	0	0.063	0
5		0	0.088	0.140	0.168	0.132	0.138	0.046	0.092	0.105	0.115	0.024	0	0.046	0.063
1	Ammonium	0.024	0	0	0	0	0	0	0	0	0	0	0	0	0
2	*[NH_4^+]*	0.057	0	0	0	0	0	0	0	0	0	0	0	0	0
3		0.057	0	0	0	0	0	0	0	0	0	0	0	0	0
4		0.032	0	0	0	0	0	0	0	0	0	0	0	0	0
5		0.456	0	0	0	0	0	0	0	0	0	0	0	0	0
1	Phosphate	0	0.104	0	0	0	0	0	0	0	0	0.084	0	0	0
2	*[PO_4^{3-}]*	0	0	0	0	0	0	0	0	0	0	0.075	0	0	0
3		0	0	0	0	0	0	0.079	0	0	0	0.078	0	0	0
4		0	0	0	0	0	0	0.054	0.101	0	0	0.060	0	0.080	0.110
5		0	0	0	0	0	0	0.055	0.103	0	0	0.066	0.058	0.064	0.083

Inorganic Nitrogen Compounds

Ammonium, nitrite, and nitrate are inorganic nitrogen compounds that are part of the nitrogen cycle. First, the nitrification of ammonia to nitrite takes place, followed by the conversion of nitrite to nitrate by nitrifying bacteria (Payne et al. 2014a). The sequence of this process is also apparent from the results shown in Table 4. Ammonium and nitrite are only present in the incoming stormwater, whereas nitrate is present in all water samples. In addition, nitrate leached from most of the filters (see Figure 10). As there was no nitrate in the influent on the last sampling day, or the concentration was below the limit of quantification, it was not analysed graphically. However, all filters except P2 had negative RE on that day. In general, the largest leaching of nitrate occurred within the ash filters. There might be several reasons for this. On one side, the sorption materials could have different microbial activities, so that the conditions inside the ash filters favour the activity of nitrifying bacteria and thus an increased conversion of ammonium to nitrate takes place. In addition, as described in Chapter 2.1.1.1., ash generally has an alkaline pH, which favours the conversion of nitrogen compounds to nitrate (Karlfeldt Fedje et al. 2021). The RE was almost identical in all ash filters, regardless of whether they had been continuously irrigated or exposed to drought. Only the unplanted filter showed a lower leaching during the first three days. With the exception of the filter B4, the biochar filters also showed consistently negative RE. However, the concentrations were only half of those for the ash filters but showed similar trends. The filter B1 also has the lowest concentrations. The filter with the male plant B4 was again an exception. On the second day of sampling, this filter was the only one in which no nitrate concentration was measured in the outlet. It is unclear why this filter differs so much from the others. The best nitrate removal efficiency was achieved by the peat filters. These are the only filters for which positive RE could be measured. On the first two sampling days, P1 also shows the best RE. From the third day, however, P2 and P4 have higher RE. In general, the RE for nitrate in these two filters are almost identical. This indicates that drought had no effect on the nitrate retention capacity and is therefore valid for all filter compositions tested. However, this contradicts the study by Mangangka et al. (2015), who found an effect of drought on the nitrate treatment performance.

In general, the nitrate RE of 42.5-91.8% achieved by Wang et al. (2017) could not be substantiated in this study. For example, filters P1, P2 and P4 partially showed positive RE for nitrate with values up to 100%. In contrast, the other filters showed nitrate leaching of up to five times the inlet concentration within the first four days of the test. The high leaching may be due, in part, that larger volumes of water were used (~70 litres/filter) than it is the case with most natural rainfall events. The residence time of the water in the system may have been too short to allow processes such as denitrification and assimilation to take place, resulting in increased leaching of dissolved nitrates. According to Lambers et al. (2008)(and Taylor et al. (2005), nitrogen retention depends not only on plant selection and diversity but also strongly on the residence time of the stormwater in the system.

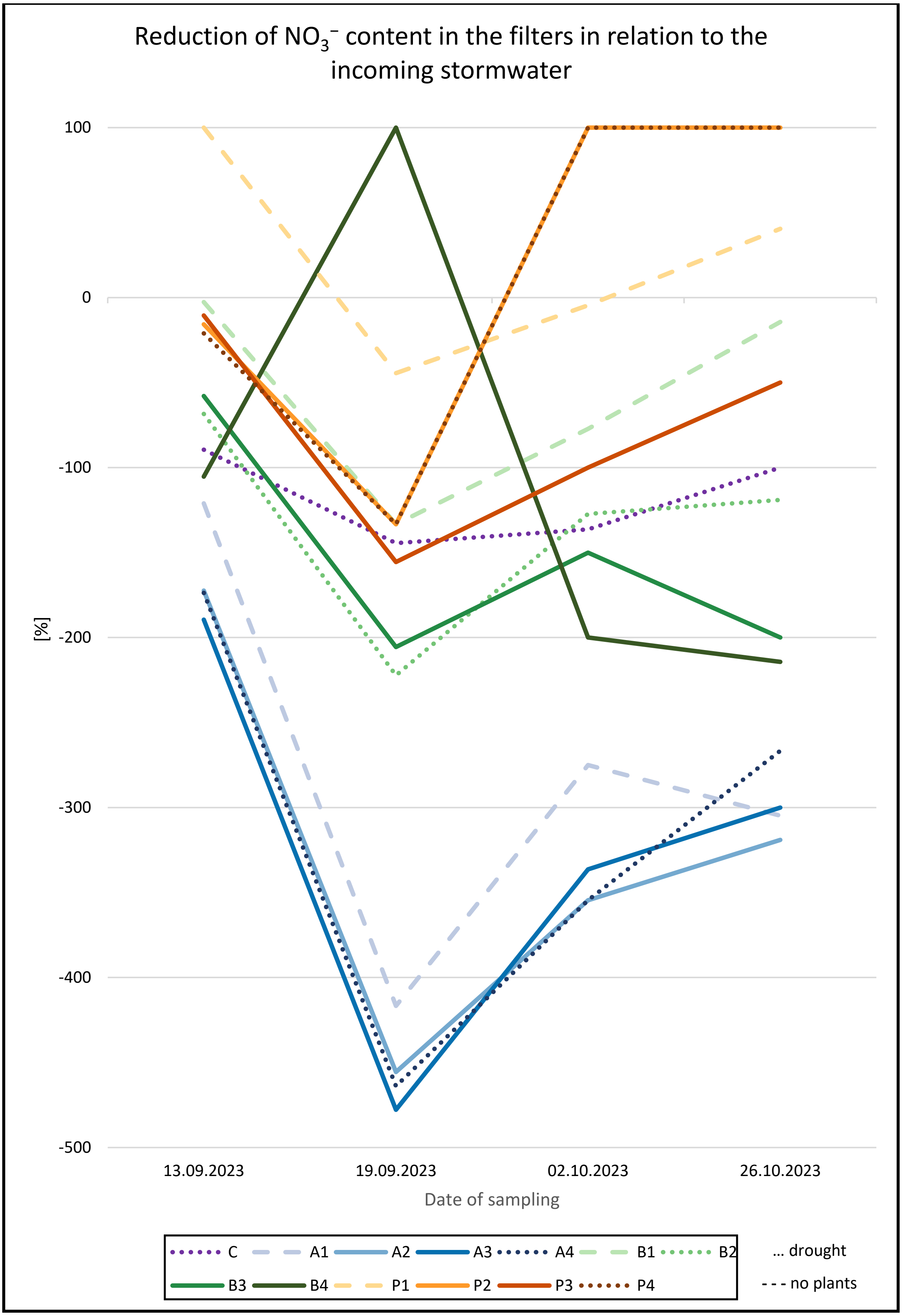

Figure 10 - NO_3^- removal efficiency at the different sampling days in all the bioretention filters at the Gårda raingarden facility

Phosphorus

In contrast to the inorganic nitrogen compounds, all phosphate concentrations in the influent were below the limit of quantification of the analysis method used. The same applied to the effluents from all ash filters. Compared to the other sorption materials, the MIBA ash filters did not leach any phosphate. For the biochar filters, only B1 and B2 were found to leach phosphate. As this was observed in both the unplanted and drought-exposed filters, it is evident that not only plants could counteract phosphate leaching as mentioned in Chapter 2.1.2., but also that dry periods have a negative impact on the treatment efficiency. The same was observed in the peat filters. Although an increase in phosphate concentration in the effluents could be observed for all filters on the last sampling day, P1 was the only filter that had all through the study a leaching of phosphate. In addition, P4 had the highest concentrations of phosphate in the effluents on the last two sampling days. In this respect, an influence of the drought on leaching of phosphate from the filter was observed, especially for the biochar filter, but also for the peat filters. However, it is unclear why the effect is only visible starting from the fourth sampling day and could not be explained based on results from other studies. As the influent concentrations of phosphate were below the limit of quantification on all sampling days in this study, only phosphate leaching could be detected and no reductions as in the study by Zhou et al. (2017), where removal efficiencies of 82-97% were achieved. Jiang et al. (2019), on the other hand, found variable removal rates ranging from -240% to +60% and attributed this to leaching of TP by stormwater runoff.

4.3. Organic Carbon and Nitrogen

The 9ml-TC-IC-TN 200mg/L method of the TOC-Vcph Shimadzu was used to analyse the TOC/DOC and TN content of the influent and effluent. The results were evaluated on the basis of the limits set by the City of Gothenburg and RE.

4.3.1 Total Organic Carbon

The TOC value was used to measure the amount of organic material in the influent and effluent, regardless of the specific type of organic carbon. The data generated and subsequently analysed are shown in Figures 11 and 12 and in the Graphs 26-29 and Table 8 in Appendix E. The target effluent value set by the City of Gothenburg is 12 mg/L (Göteborgs Stad 2020). In general, only a few filters were able to meet the specified limit. The only filters that consistently showed values below the 12 mg/L guideline value in the effluent were A1 and B1. Where A1 indicated a decreasing trend in the TOC content, while B1 fluctuates around the 6 mg/L mark. It is noticeable that both filters are pilot setups without vegetation. This could be because dead plants and roots can contribute to TOC leaching. The decomposition processes of the organic material convert it into dissolved organic compounds, which can lead to an increase in the TOC content (Shrestha et al. 2018). However, this is contradicted by the fact that the unplanted peat filter P1 had by far the highest TOC values. In some cases, the values correspond to 11 times the influent and more than double the TOC concentrations of the planted peat filters. In general, the peat filters showed the highest TOC leaching of all experimental setups. As *sphagnum* peat consists mainly of dead plants with high organic content, the organic matter can be leached out of the peat during rain events (Markiewicz et al. 2020; Kalmykova et al. 2008). However, since filter P4, which was exposed to the drought, had the highest TOC value only on the last day of sampling, and the unplanted filter did not, it can be assumed that when peat is used as a sorption medium, planting reduces the leaching of TOC from the filter material. This agrees with the results of the study by Beral et al. (2022), where the planted systems removed a higher proportion of TOC from the influent than the unplanted filters. In the case of ash and biochar, the plants appear to contribute more to TOC leaching, as previously described. Nevertheless, in addition to filters A1 and B1, a positive RE was also achieved for filters A3 from the third sampling day and A2, A4 and B3 from the last day. Although there is an increase in the TOC content in the other ash and biochar filters, significantly lower values were observed throughout the study than in the peat filters. This indicates that peat is not a suitable sorbent for the removal of TOC from stormwater. The ash and biochar filter materials, on the other hand, showed little difference and low TOC values that either met or exceeded the required limits by a maximum of 10 mg/L, excluding filter B4 with the male plant.

By comparing the TOC values of the dry filters with those of the irrigated filters, it can be seen that the differences are minimal. A2 compared to A4 and filters P2, P3 and P4 show almost identical curves. In addition, A4 and P4 had the lowest RE values compared to the filters with the same sorption materials only on the last day. However, the difference is marginal. B2, on the other hand, had significantly better values than the B4 filter with the male plant. This suggests that dry periods have little or no effect on the TOC content of the effluent. In the Lim et al. (2015) study, on the other hand, an impairment of TOC removal by drought was observed. The different results are probably due to the use of different sorption materials. In this respect, the filter medium and the choice of plants seem to contribute more to TOC removal and leaching than the presence of droughts.

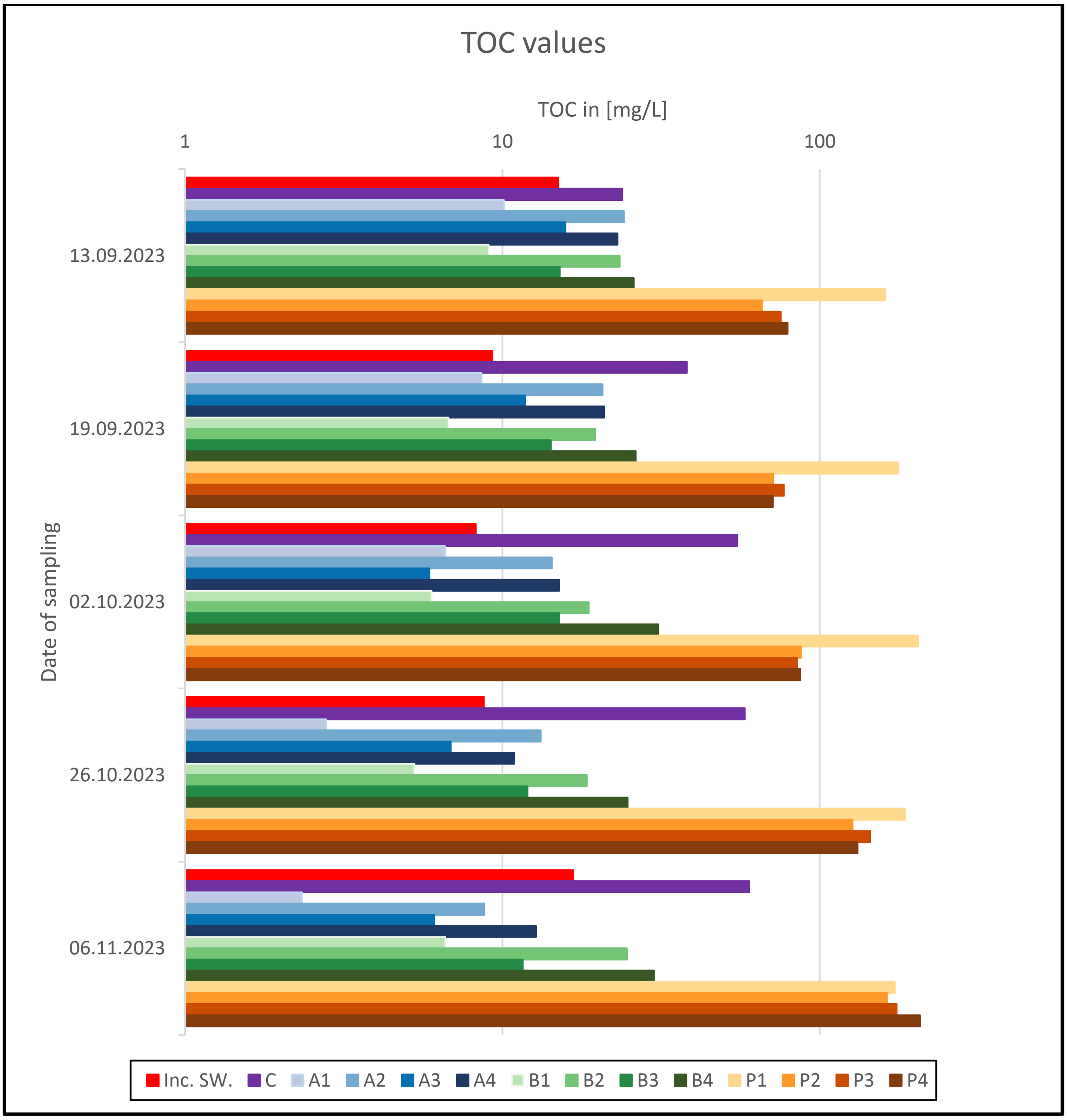

Figure 11 - TOC concentration in influents and effluents at the different sampling days in all the bioretention filters at the Gårda raingarden facility

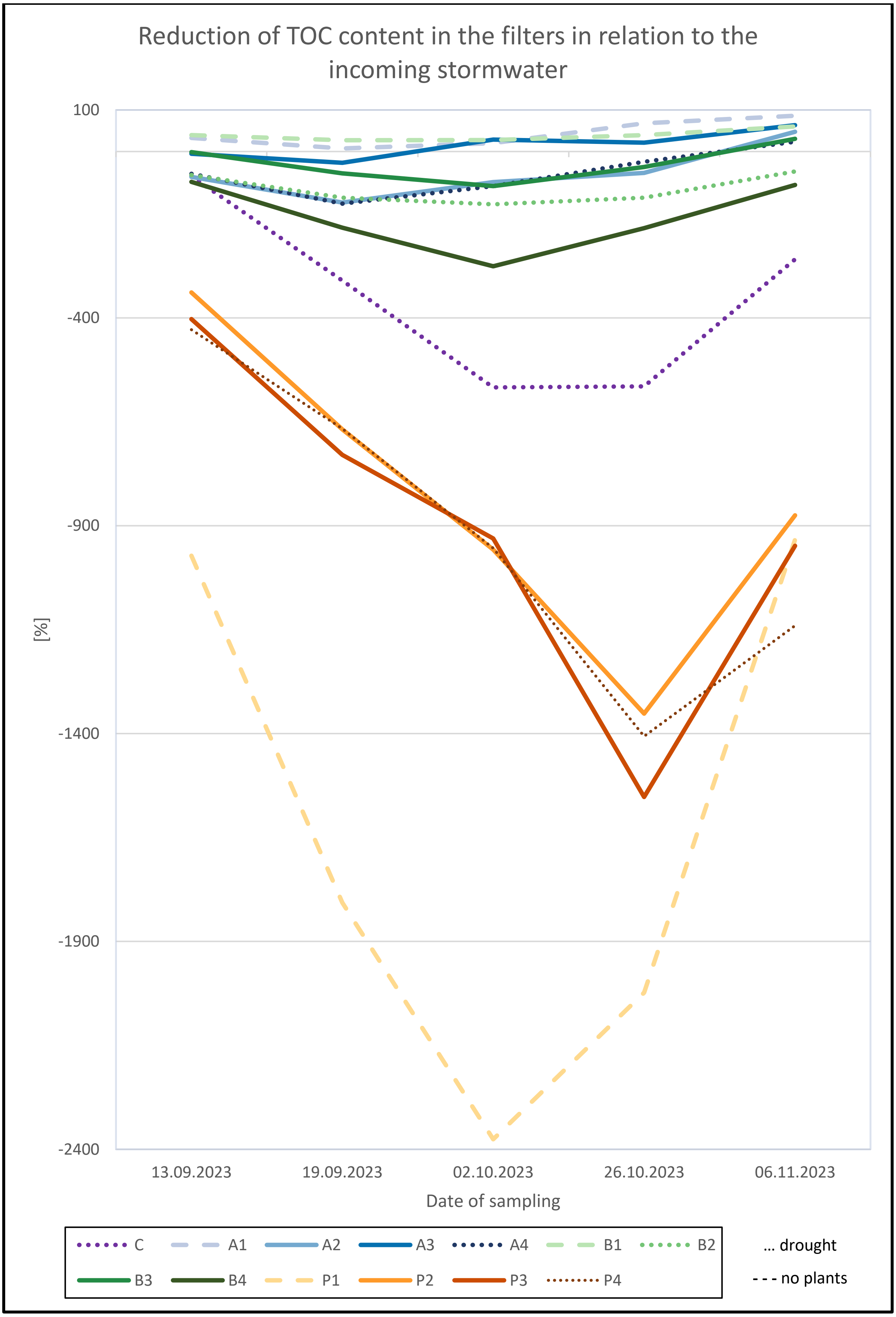

Figure 12 - TOC removal efficiency at the different sampling days in all the bioretention filters at the Gårda raingarden facility Dissolved Organic Carbon

The present DOC values are almost identical to the TOC values. This indicates that the organic carbon in the water occurs mainly in dissolved form, or that the leaching of the filter materials consists primarily of dissolved carbon. In this respect the results and trends are almost identical. The DOC and RE values are shown in Figures 30 and 31, Appendix F. Again, peat showed by far the highest values and a continuous release of DOC, which is consistent with the continuous leaching of DOC from peat filters due to peat decomposition mentioned in Chapter 2.1.1.3 (Kalmykova et al. 2008). Furthermore, as with TOC, A1 and B1 are the only two filters that exhibit a steady reduction in DOC. The filter A3 also showed positive RE values apart from the 4th sample day, A2 only on the last and B3 only on the first test day. However, looking at the Table 9 in Appendix F, it is clear that the DOC values are higher than the TOC values (Table 8, Appendix E), especially on the fourth day of sampling. However, as the TOC value is the sum of DOC and particulate carbon and as the same method of analysis was used, this should not be the case. Therefore, it is assumed that the TOC samples were not completely homogeneous as the particulate components were deposited at the bottom. As a result, the total TOC content was not captured and inaccuracies exist. The ash filter samples from 2023-10-26 are particularly affected by the variations. This is particularly evident when comparing the TOC-RE curves shown in Appendix E with those of DOC (Appendix F). The peat and biochar filters show almost identical curves for TOC and DOC, whereas the ash filters show large deviations.

In general, no serious impairment of the filtration and retention properties of the different sorption materials due to drought could be detected for DOC. However, on the basis of the data generated, the use of peat for DOC treatment is not recommended; in the case of ash and biochar, leaching also occurs in some cases, but to a much more limited extent.

4.3.2 Total Nitrogen

The TN content in the different samples also follows similar RE curves as TOC and DOC (see Figures 35-41 and Appendix G). Since, as described in Chapter 2.2.1., elevated TN concentrations can cause water quality problems, including eutrophication processes, algae blooms and impairment of aquatic ecosystems, the City of Gothenburg has set a limit of 1.25 mg/L for water discharges (Göteborgs Stad 2020). As shown in Table 10, Appendix G, the influent water used for irrigation always exceeds this value. Although the ash and biochar filters consistently reduce the TN content, all ash filters still exceed the limit on the first and second sampling days. On the third day, this is only the case for A1. The filter A3 achieved a complete TN reduction from 2023-10-26 and A2 on the last sampling day. Filters B1 and B3, on the other hand, are the only setups in this study whose effluent concentrations are continuously below the 1.25 mg/L mark. From 2023-10-02, B2 also falls below this limit. The filter B4, on the other hand, only meet the limit on the fourth day of sampling. Similar to TOC and DOC, the peat filters had the highest TN effluent concentrations with values in the range of 1.9-5.1 mg/L. The unplanted peat filter P1 had the highest TN concentrations in the effluent, which is consistent with the results of Zhang et al. (2018) and Morash et al. (2019), which found that plants can increase TN removal. However, this could only be demonstrated to a limited extent in this study, as B1 consistently had the highest TN-RE compared to the planted biochar filters. However, as the amount of water used (~70 litres per irrigation day) exceeds average natural rainfall events, this may lead to a reduction in treatment efficiency. Long and intense rainfall events can result in increased leaching of dissolved nutrients as the retention time in the filter materials is insufficient for the bio- and physio-chemical processes (Shrestha et al. 2018). Thus, the peat filters showed leaching of TN on at least two test days. In general, the RE of the peat filters decreases continuously during the course of the experiment until the fourth sample day, when it reached its minimum of -90% (P2) to -210% (P1). Negative TN removal rates of -178% on average were also measured in the study by Wang et al. (2017). On the fifth sampling day, however, P1-P3 showed positive RE values. It was expected that most of the dissolved nitrogen had already been leached out. Only the filter P4 had a negative RE of -9%, which could be due to the previous drought period. As the fifth sampling day was the last day of the test, it is unclear whether the peat filters would continue to show positive RE in the future or whether this is only due to the comparatively higher TN content in the influent, as a percentage analysis was carried out. In general, all filters had the highest RE on the last sample day, except for the filters P2-P4. The ash and biochar filters reached RE of >70% on this day.

Overall, it is important to keep the TN content of the effluent as low as possible in order to counteract or minimise eutrophication processes in the water bodies. In this respect, ash and biochar are recommended as sorbents to reduce TN from stormwater. All these filters reduced TN levels compared to the influent, regardless of whether they were continuously irrigated or experienced a dry period, and therefore all had a positive RE balance, even if the limit values were not always met. On the other hand, the peat filters and the control filter showed an increased nitrogen accumulation. In this respect, peat was less suitable for the treatment of TN, as well as TOC and DOC. In general, no real impact of the dry period on the TN removal properties could be determined for all sorption materials. Only P4 showed a higher value on the last sample day compared to the other peat filters. As the values on the other sampling days differed only marginally from those of the other peat filters, this is considered negligible.

4.4. Organic Pollutants

As most of the OP concentrations measured in this study were below the limit of quantification, it was not possible to make a precise quantitative analysis. Furthermore, it cannot be excluded that the external laboratory had made changes to the analytical method during the study period. These changes could have a direct impact on the results of the present OP analysis and its interpretation.

To ensure the validity and reproducibility of the results, further analysis and verification of the samples will be required in another study. This could include reanalysis using standardised analytical methods and, if necessary, collaboration with the external laboratory to clarify method changes.

4.5. Microplastics

In total, the following eight MP and two rubber particle components were analysed in the stormwater of the Gårda area within this study: polyethylene (PE), polypropylene (PP), polystyrene (PS), polymethylmethacrylate (PMMA), polycarbonate (PC), polyvinylchloride (PVC), polyethylene terephthalate (PET), polyamide 6 (PA6), polybutadiene (PB) and polyisoprene (PI). The concentrations of particles >10µm and their percentual composition in the influent are shown in Figure 13. When analysing the results, it should be noted that the stormwater stagnated in the sedimentation chamber before being pumped into the IBC containers. Therefore, it is assumed that particularly heavy polymer and rubber particles would have settled to the bottom, resulting in not capturing all of the particles. However, it is noticeable that on 2023-09-13 (~∑27µg/L) almost 46 times less MP mass was present in the stormwater than on 2023-11-06 (~∑1253µg/L). In particular, the number of PE, PVC and PI particles had increased compared to the other polymers to the first sampling day. This could be due to the change between summer and winter tyres during the study which would increase the concentrations of the tyre particles. Other causes could be the density and weight of the different particles, so that significantly heavier MP particles could have been present in the stormwater on the first sampling day, which could have resulted in the majority of particles being settled in the sedimentation chamber or, in general, fewer particles being deposited on the road surface. However, it is unlikely that the difference is that significant. The exact cause is unknown and therefore only assumptions could be made. A correlation with rainfall events in the area could not be established (see Figure 15, Appendix A), however, the MP content also varied in the studies by Johansson et al. (2024) and Lange et al. (2021). Thus, in the study by Lange et al. (2021), which examined the highway runoff in Sundsvall, no rubber particles of the size of 100-300 µm were detected on two of the nine sampling days. On the other days, the concentrations also varied greatly, so that the number of rubber particles fluctuated between <0.31 and 740 particles/L. In addition, slight fluctuations within the MP concentrations were detected (<0.38 - 6.2 particles/L).

On both sampling days, PI (>44%) and PVC (~30% and ~40%) were predominantly identified in the influent. The PI and PB present can be attributed to tyre abrasion. However, the PB concentration in the water sample of 2023-11-06 was remarkably low. As the pilot facility is located next to the E6, an increase of the tyre-related MP concentrations was to be expected, but the results were in line with the results of the studies by Johansson et al. (2024) and Järlskog et al. (2020) in other parts of the city of Gothenburg. Overall, PE (2.7; 165µg/L), PVC (7.8; 499µg/L) and PI (11.8; 588µg/L) were quantified on both sampling days. However, PP (1.7; <0.5µg/L), PA6 (0.1; <0.1µg/L) and PB(2.5; <1µg/L) were below the limit of quantification on the last sampling day, as were PS (<1µg/L), PMMA (<0.2µg/L), PC (<1µg/L9) and PET (<0.3µg/L), which were consistently below the detection limit. Thus, the influent sample of 2023-09-13 consists of a total of four MP types and two rubber components, whereas the influent sample of 2023-11-06 consists of only two MP components and PI. Such a variation in the percentage distribution and occurrence of MP and rubber types was also observed in the study by Blomqvist et al. (2023), in which the snow on the plough edges of various roads in Karlstadt was investigated. In addition, PE and PI were identified as some of the most notable particles, which is consistent with this study, as these, along with PVC, are the only particles detected on both days.

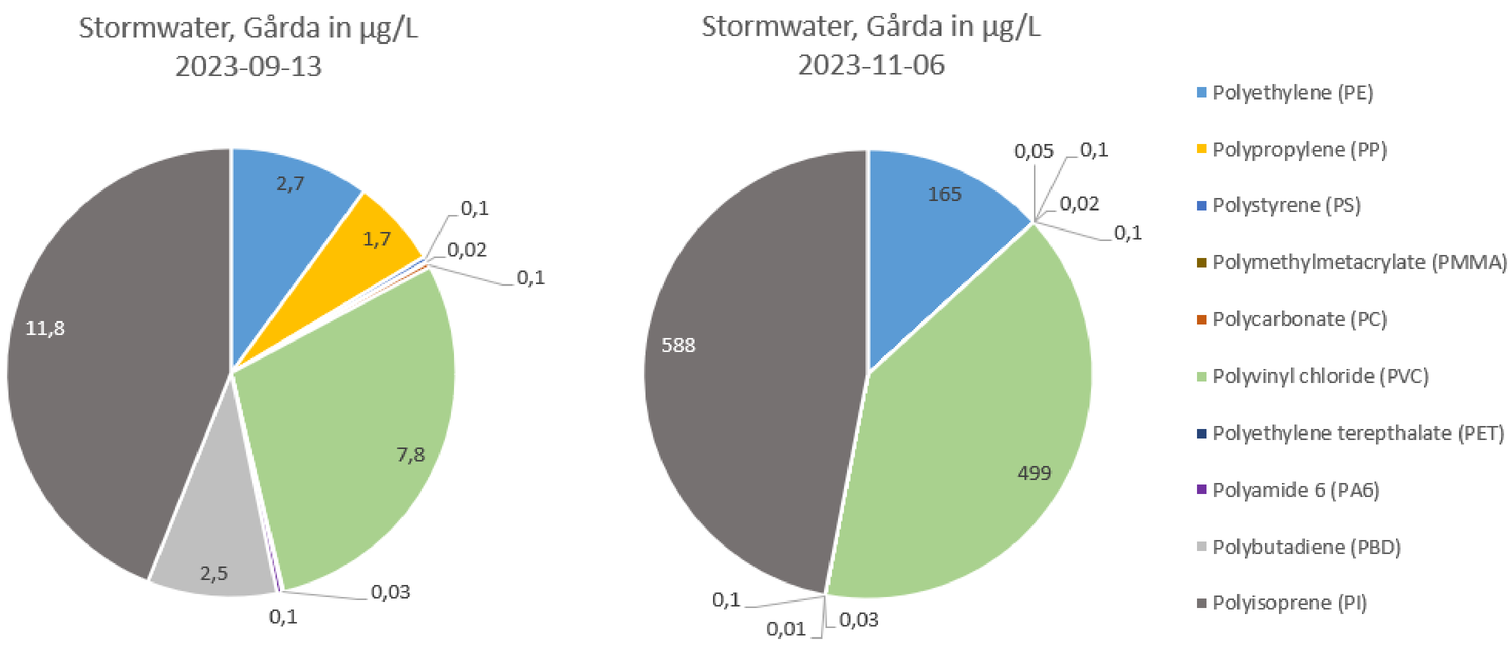

Figure 13 - Concentrations and relative compositions of microplastic polymers and rubber components in stormwater from the Gårda catchment area

Microplastic Removal

The individual concentrations in the effluents and the RE of the different filters are shown in Table 5. For cost reasons, only eight water samples were analysed for MP on the first and last sampling days. The analysis includes the influent, the filters exposed to drought (C, A4, B2 and P4) and a planted reference filter of each sorption medium (A3, B3 and P3).

In general, PS, PMMA, PC and PET were not higher than the limit of quantification in the influent or effluent of any sample on either of the two test days. The same was observed for PA6 and PBD on 2023-11-06. For PP, on the other hand, a very high leaching was found in all samples, regardless of the sorbent materials tested. This may be related to the sampling containers used. In addition to PP, only PE leached on the first sampling day in filters A3 (-88.89%) and B3 (-3.7%). Both are the reference filters that were continuously irrigated. The filters exposed to the dry period showed a better RE in this respect, which contradicts the results of O'Connor et al. (2019), were it was observed that wet-dry cycles favour the leaching of MP particles. Because the RE values on the last sampling day are only slightly different, that much higher RE values are present and that only two sampling days were carried out in total, no statement can be made about the cause. Properties similar to those for PE were also found for PI and PVC. The RE properties of A3 for PVC and PI and B3 for PI are also lower on the first sampling day than on the latest, but the discrepancies are smaller. On the other hand, this phenomenon was not observed for the peat filters. Thus, P3 and P4 had almost identical RE values on both test days.

As only two samples per filter were analysed, measurement errors cannot be excluded and identified. In this respect, it is assumed that the drought had no effect on the retention capacity. This is because, apart from the inconsistencies of A3 and B3 mentioned above, the RE values of all filters are either identical or only slightly different for the various MP particles and rubber components. In general, the majority of filters retained >90% of the MP particles, which illustrates the removal efficiency of biofilter systems even for different filter materials and is consistent with the results of Gilbreath et al. (2019); Kuoppamäki et al. (2021); Lange et al. (2021, 2022); Smyth et al. (2021); Werbowski et al. (2021) and Johansson et al. (2024). An evaluation of the different sorbents, on the other hand, is of limited validity. However, based on the available results, peat has the best retention properties in relation to MP, followed by biochar and ash.

In general, it should be noted that the results presented are based on data from only two sampling days, which limits the transferability and representativeness of the outcomes. A more detailed study with a larger sample size is needed to make more general and reliable statements.

Table 5 - MP concentration and removal efficiency on the first and last day of sampling (below the limit of quantification)*

sampling day		*effluent [µg/L]* *RE [%]*	**Inc. SW.**	**C**		**A3**		**A4**		**B2**		**B3**		**P3**		**P4**	
			infl.	***effl.***	***RE***	***effl.***	***RE***	***effl.***	***RE***	***effl.***	***RE***	***effl.***	***RE***	***effl.***	***RE***	***effl.***	***RE***
	1	PE	2.7	<1*	96.3	5.1	-88.9	1.3	51.9	2	25.9	2.8	-3.70	<1*	96.3	<1*	96.3
	5		165	<1*	99.9	<1*	99.9	<1*	98.7	<1*	99.9	<1*	99.94	1.1	99.3	1.8	98.9
	1	PP	1.7	32.6	-1817.7	13.5	-694.1	11.3	-564.7	15.3	-800	15.8	-8294	7.1	-317.7	27.2	-1500
	5		<0.5*	98.6	-197100	0.9	-1700	3.5	-6900	11.6	-23100	41.3	-82500	19.9	-39700	5.7	-11300
	1	PS	<1*	<1*	0	<1*	0	<1*	0	<1*	0	<1*	0	<1*	0	<1*	0
	5		<1*	<1*	0	<1*	0	<1*	0	<1*	0	<1*	0	<1*	0	<1*	0
	1	PMMA	<0.2*	<0.2*	0	<0.2*	0	<0.2*	0	<0.2*	0	<0.2*	0	<0.2*	0	<0.2*	0
	5		<0.2*	<0.2*	0	<0.2*	0	<0.2*	0	<0.2*	0	<0.2*	0	<0.2*	0	<0.2*	0
	1	PC	<1*	<1*	0	<1*	0	<1*	0	<1*	0	<1*	0	<1*	0	<1*	0
	5		<1*	<1*	0	<1*	0	<1*	0	<1*	0	<1*	0	<1*	0	<1*	0
	1	PVC	7.8	<3*	96.2	5.2	33.3	<3*	96.1	<3*	96.2	<3*	96.1	<3*	96.2	<3*	96.2
	5		499	<3*	99.9	<3*	99.9	<3*	99.9	<3*	99.9	<3*	99.9	<3*	99.9	<3*	99.9
	1	PET	<0.3*	<0.3*	0	<0.3*	0	<0.3*	0	<0.3*	0	<0.3*	0	<0.3*	0	<0.3*	0
	5		<0.3*	<0.3*	0	<0.3*	0	<0.3*	0	<0.3*	0	<0.3*	0	<0.3*	0	<0.3*	0
	1	PA6	0.1	<0.1*	90	<0.1*	90	<0.1*	90	<0.1*	90	<0.1*	90	<0.1*	90	<0.1*	90
	5		<0.1*	<0.1*	0	<0.1*	0	<0.1*	0	<0.1*	0	<0.1*	0	<0.1*	0	<0.1*	0
	1	PBD	2.5	<1*	96	<1*	96	<1*	96	<1*	96	<1*	96	<1*	96	<1*	96
	5		<1*	<1*	0	<1*	0	<1*	0	<1*	0	<1*	0	<1*	0	<1*	0
	1	PI	11.8	1.9	83.9	4.3	63.6	1.2	89.8	1.4	88.1	<1*	99.2	<1*	99.2	<1*	99.2
	5		588	<1*	100	<1*	100	<1*	100	<1*	100	<1*	100	<1*	100	<1*	100

4.6. Comparison of Effluent Parameters

In order to provide a general answer to the research questions posed in Chapter 1.2, the previously analysed results of the drought filters compared to the reference filters are summarised in Table 11 Appendix H.

In general, a regeneration of the filter properties was observed for all sorption materials used during the duration of the test. At the beginning of the study, the dry filters were impaired in terms of TSS, VSS, TOC and DOC. By the end of the study, however, there were no or only marginal deviations to the reference filters. Although there was a decrease in RE on the fifth sampling day, it is assumed that this is attributable to the increased pollutant content of the influent. As a result, the effects of the drought can be detected most significantly and for the prolonged period, mainly at high influent pollutant concentrations. However, no effects of drought were found for the other pollutants or particles analysed. In addition, the influence of the drought on B2 can only be assessed to a limited extent, since the analysis values of the reference filters B3 and B4 differed in some cases considerably, so that the RE of B2 often ranged between those. However, as the experimental setups of B3 and B4 are identical, only the sex of the sea buckthorn plants differs (B3 female, B4 male), it is recommended to carry out further studies on the influence of the sex of the plants on the purification performance in bioretention systems. In general, no differences were found in the regeneration capacity of the different materials after the dry period. Although the different sorption materials have different filter properties, the assessment of the regeneration capacity is based on the equivalent reference filters.

When comparing the different filter materials in terms of functionality and pollutant removal, it becomes clear that the ash filter removes seven of the ten pollutants and particles listed in Table 11, Appendix H most effectively. This result is also reflected by the ash filters in the overall comparison of the different sorption materials. It is striking that the ash filters are the only set-ups that contain all three sorption materials. In this respect, the use of ash filter assemblies for bioretention filters is recommended. In addition, increased nitrification takes place within the ash filters. Although nitrite and ammonium removal are increased, nitrate accumulation is the highest of all filters. Ash filters can therefore contribute to degradation in nitrogen-polluted areas, but it is essential to monitor redox conditions and control nitrate levels.

Peat and biochar filters, on the other hand, generally perform almost identically. Although the peat filters have the best removal characteristics and the lowest accumulation of nitrate, PE and PI, the removal efficiencies of TOC, DOC and TN are inadequate. However, as nitrogen leads to eutrophication in water bodies (see chapter 2.2.1.) and can therefore cause considerable ecological damage, it must be reduced and controlled (Morash et al. 2019). The same applies to DOC, as changes in DOC can release previously sorbed metals. In addition, DOC and OP are directly related, as DOC can act as an adsorbent for OP, thereby facilitating the transport of organic pollutants in the system (Laurenson et al., 2013). Overall, the use of sorbents improved the retention efficiency compared to the control filter, therefore their use is highly recommended within bioretention filters.

In contrast to the results of Mangaka et al. (2015), who found a strong influence of drought on the purification performance of bioretention filters, this study showed that the different RE are mostly based on the different properties of the sorption materials rather than on the dry period and is therefore in line with the study by Lucke et al. (2015). In this respect, when designing a bioretention system, it is important to know the composition of the influent and to select the sorbent with the best filtering properties for the pollutants to be reduced. If the composition is unknown or a general pollutant reduction is desired, it is recommended to use the ash filter design based on the results obtained. In addition, a combination of the sorbents investigated in this study can be analysed in further research in order to optimise the retention of pollutants. To ensure the effectiveness of retention filters both after droughts and in the long term, regular inspections and maintenance must also be carried out.

5. Conclusion and Outlook

This work highlights the efficiency of biofiltration systems for pollutant treatment of urban stormwater using different sorption media under drought conditions and shows that the different RE is mainly due to the different properties of the sorption materials and not caused by the dry period.

The filter performance of all sorption media and the control filter was only impaired with regard to TSS, VSS, TOC and DOC during the first irrigation after the drought period. In addition, the drought affected the plant growth and resulted in smaller plants than in the reference filters. The efficiency of MP and nutrient removal remained unchanged. Overall, a regeneration of all filters was observed within this study period and no differences in the regeneration capacity of the different sorption media were observed. However, long-term effects of drought on the purification performance were noted with increased pollutant concentrations in the influent. Furthermore, the results indicate that the organic carbon in the influent is mainly present in dissolved form and that the leaching of the filter materials consists mainly of dissolved carbon.

Overall, microplastic particles >10µm were effectively removed in all bioretention filters. PE and PP are the only polymers that were leached in this study, which is probably due to the sampling containers used.

When comparing the different sorption media, the ash filters generally showed the best removal performance. The control filter, on the other hand, showed the lowest removal efficiencies, emphasising the importance of using sorbent media in rain gardens. In addition, the analysis in this study indicates an influence of the sex of the sea buckthorn plant on the removal performance.

Based on the results of this research, there are several opportunities for further investigations. Firstly, this study should be repeated for OP, NP and metals to obtain a comprehensive understanding of the main pollutants in stormwater. To ensure the comparability of the studies conducted, it is also necessary to establish a standardised method of analysis, including the parameters and sample sizes to be investigated. Secondly, more studies need to be carried out under field conditions and with natural stormwater to investigate the efficiency of treatment under real conditions. Other possible research topics include the influence of the sex of the sea buckthorn plant on the purification performance and the long-term ecological effects of prolonged drought on the bioretention filters.

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Appendix

Appendix A – Turbidity and Coloured Particles

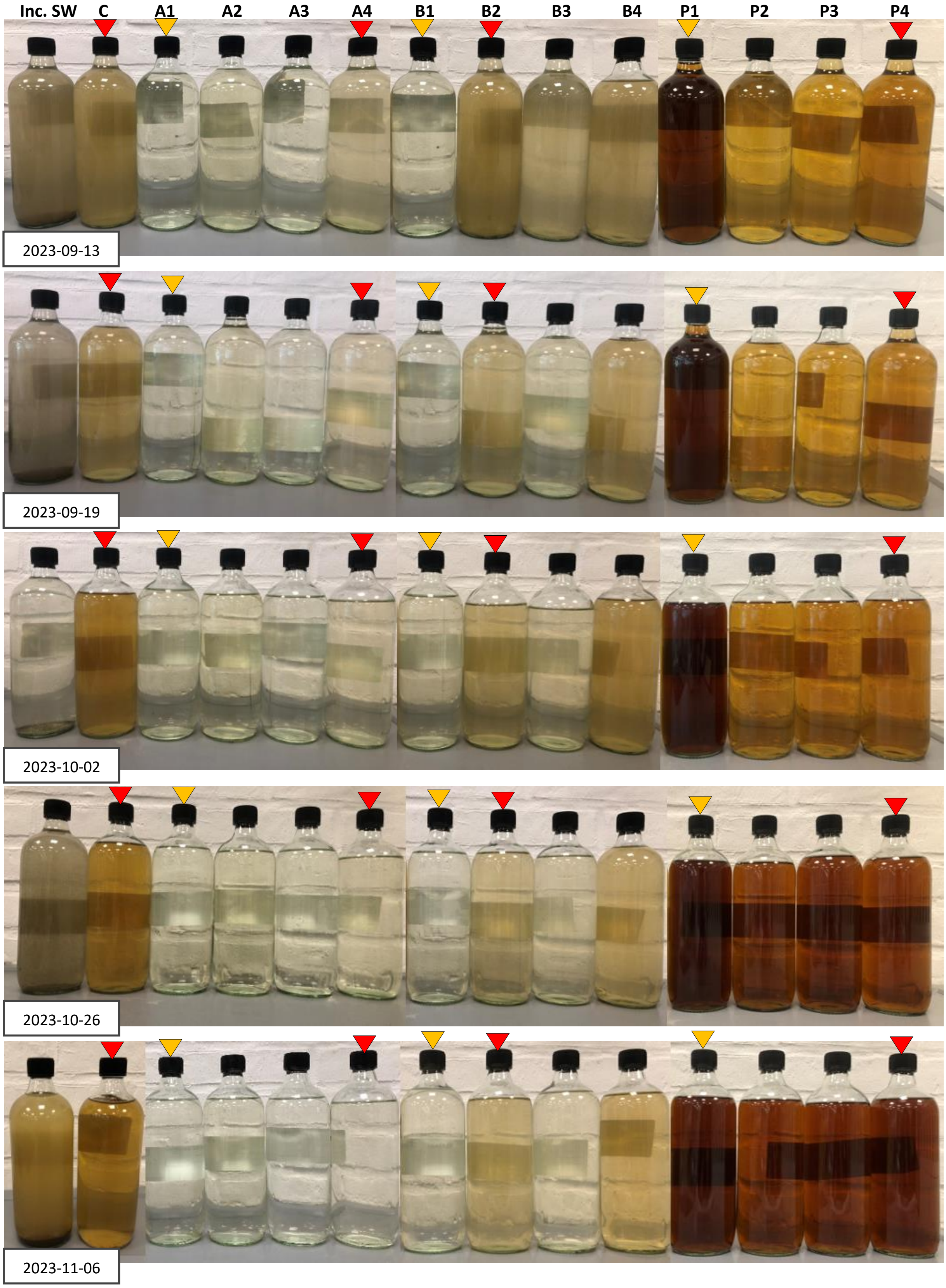

Figure 14 - Comparison of the reduction in coloured particles in all bioretention filters effluents on the first and last sampling day - the red triangle indicates the water samples from the retention filters that were exposed to the dry period, the yellow tringle indicated the filters without plants

Table 6 - Turbidity values of the inflow and all bioretention filter effluents

FNU	**13.09.2023**	**19.09.2023**	**02.10.2023**	**26.10.2023**	**06.11.2023**
Inc. SW	124	7,3	47,1	96,6	410
C	99,3	29,9	45,3	26,7	24,8
A1	1,2	1,1	1,5	0,4	0,4
A2	11,7	3,3	2,3	1,3	1,1
A3	4,9	2,7	2,7	1,1	1,3
A4	34,2	9	8,3	5,2	6,2
B1	8,5	5,9	8	7	6,6
B2	111	30,6	29,1	23,6	33,2
B3	75,5	11,4	17,3	23,6	13,7
B4	91,2	32,1	45,1	33,4	8,89
P1	6,4	15,1	19,4	22,5	23,9
P2	10,9	2,6	2,3	3,5	4,5
P3	0,6	4,1	3,5	9,2	10,7
P4	-	12	12	9,4	16,1

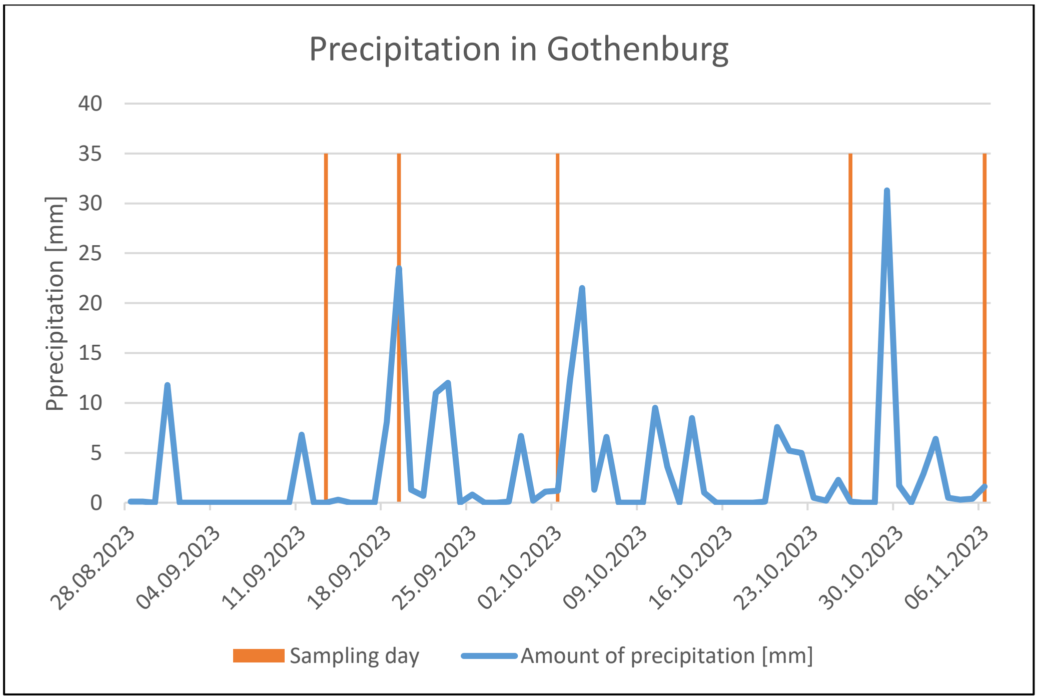

Figure 15 - Daily precipitation in Gothenburg between 2023-08-28 and 2023-11-06 measured at precipitation station Gothenburg A (71420) 2m above ground level; based on values from (SHMI)

Appendix B – Total Suspended Solids

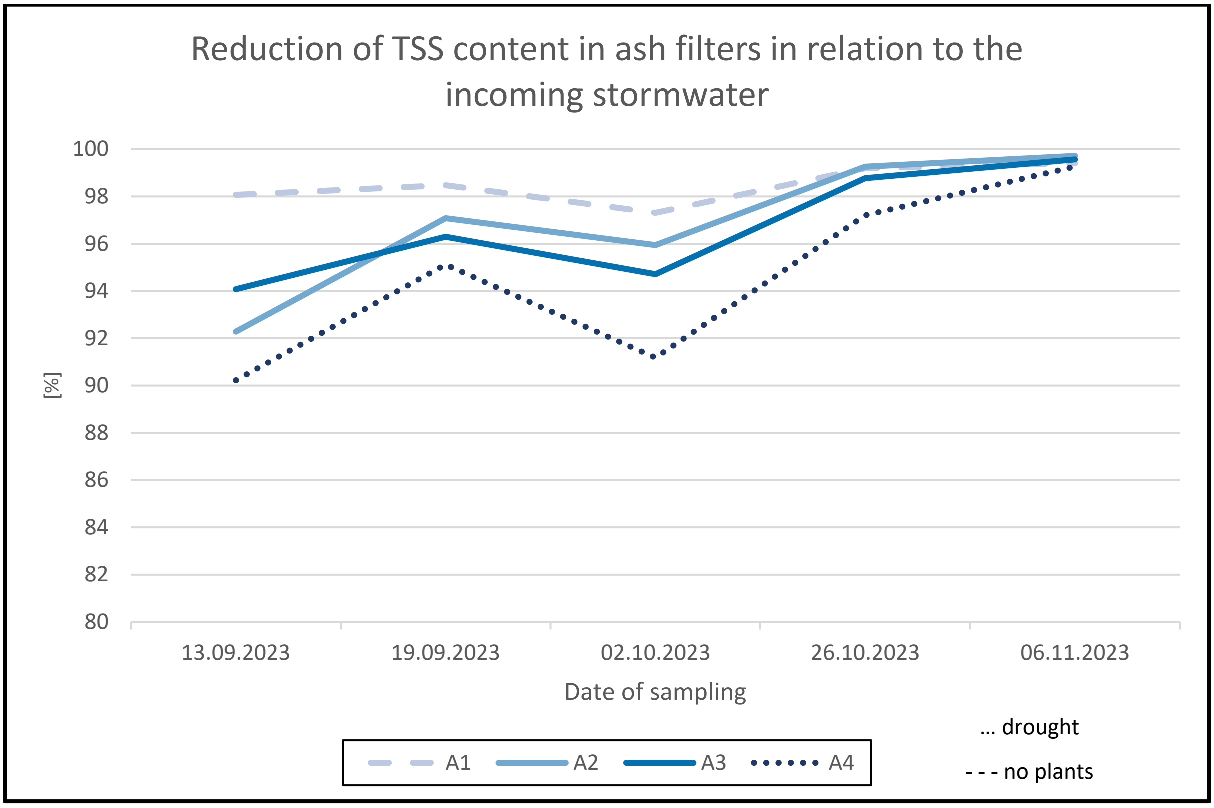

Figure 16 - Reduction of TSS content in ash filters in relation to the incoming stormwater

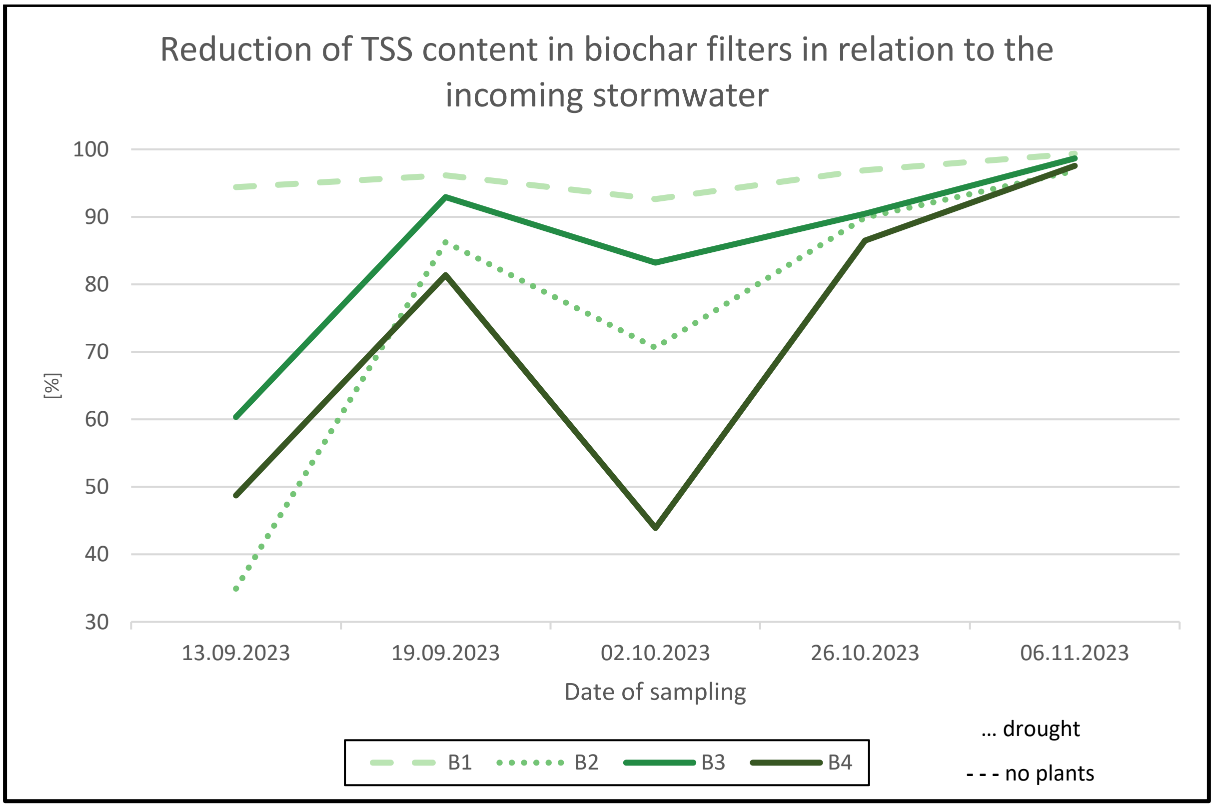

Figure 17 - Reduction of TSS content in biochar filters in relation to the incoming stormwater

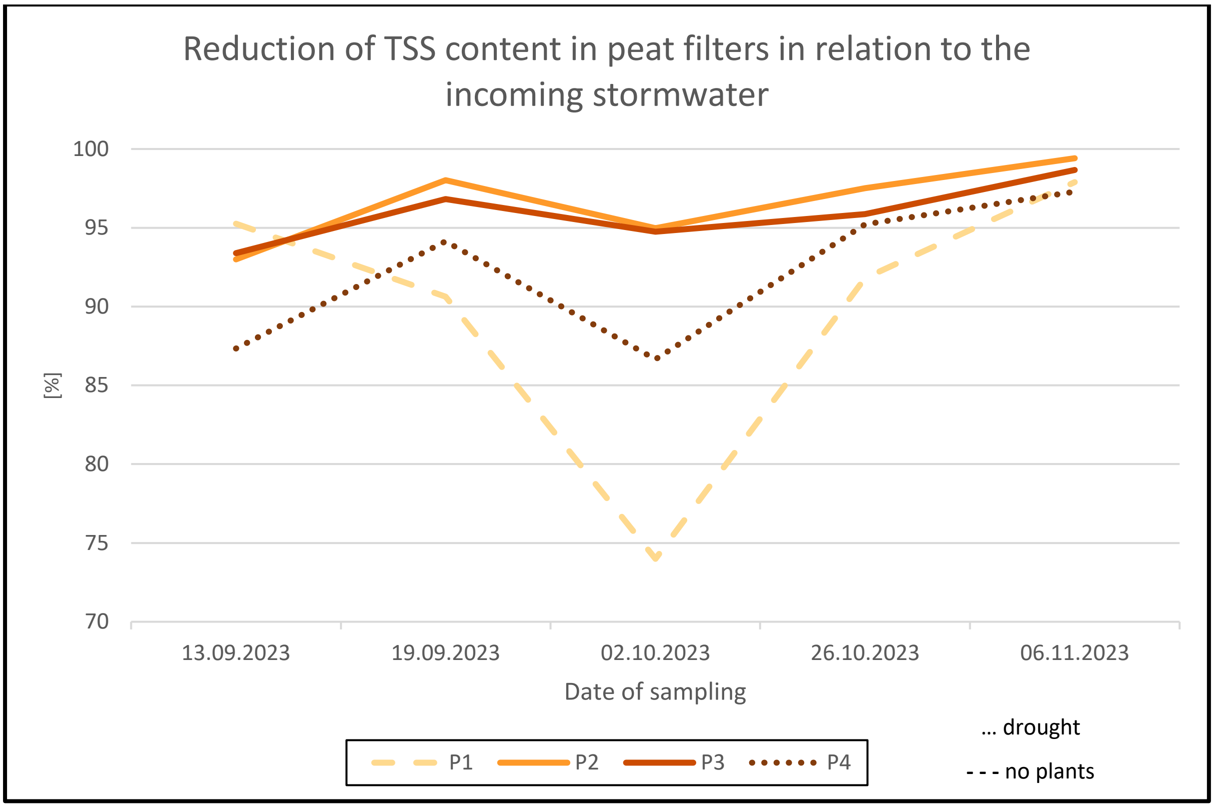

Figure 18 - Reduction of TSS content in peat filters in relation to the incoming stormwater

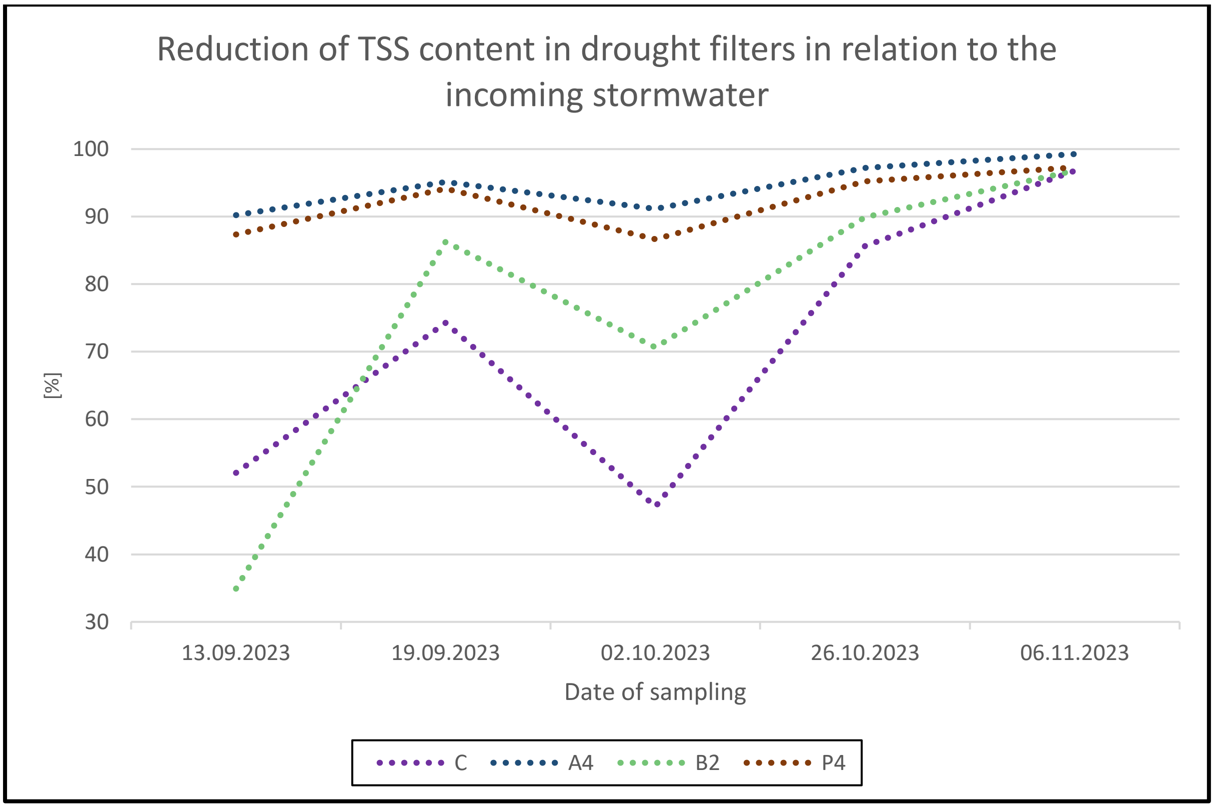

Figure 19 - Reduction of TSS content in drought filters in relation to the incoming stormwater

Appendix C – Volatile Suspended Solids

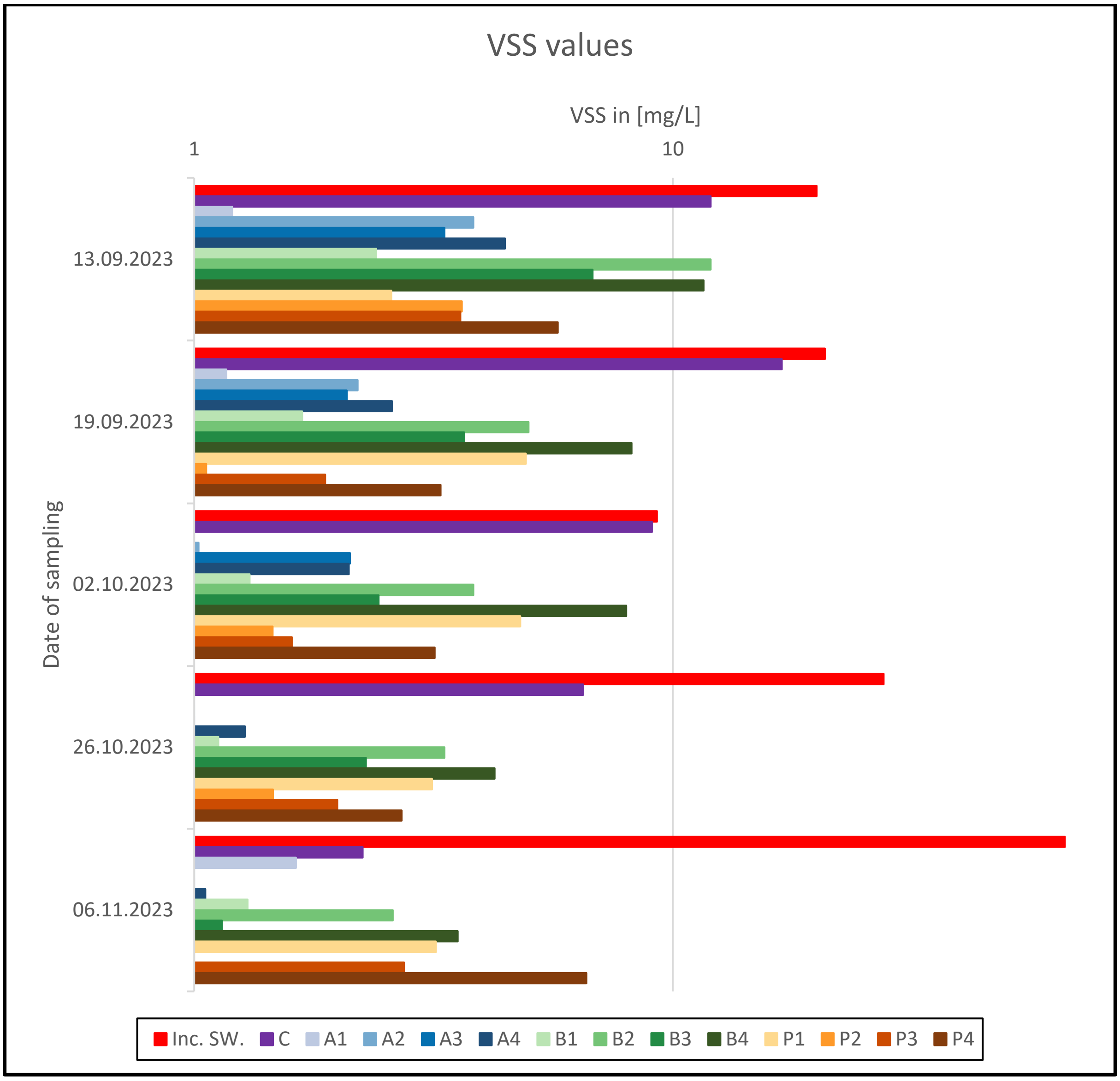

Figure 20 - VSS concentration in influents and effluents at the different sampling days in all the bioretention filters at the Gårda raingarden facility

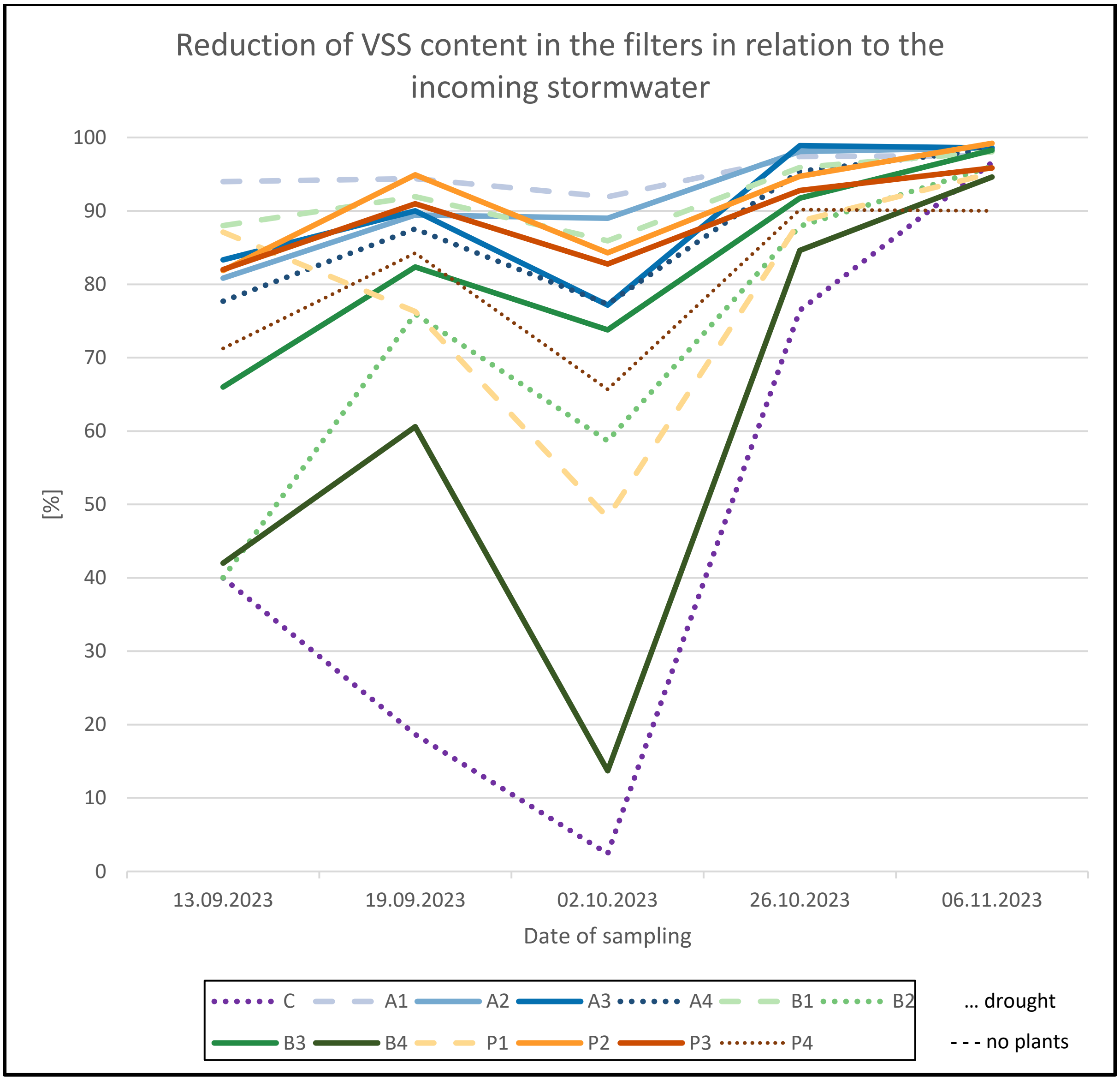

Figure 21 - VSS removal efficiency at the different sampling days in all the bioretention filters at the Gårda raingarden facility

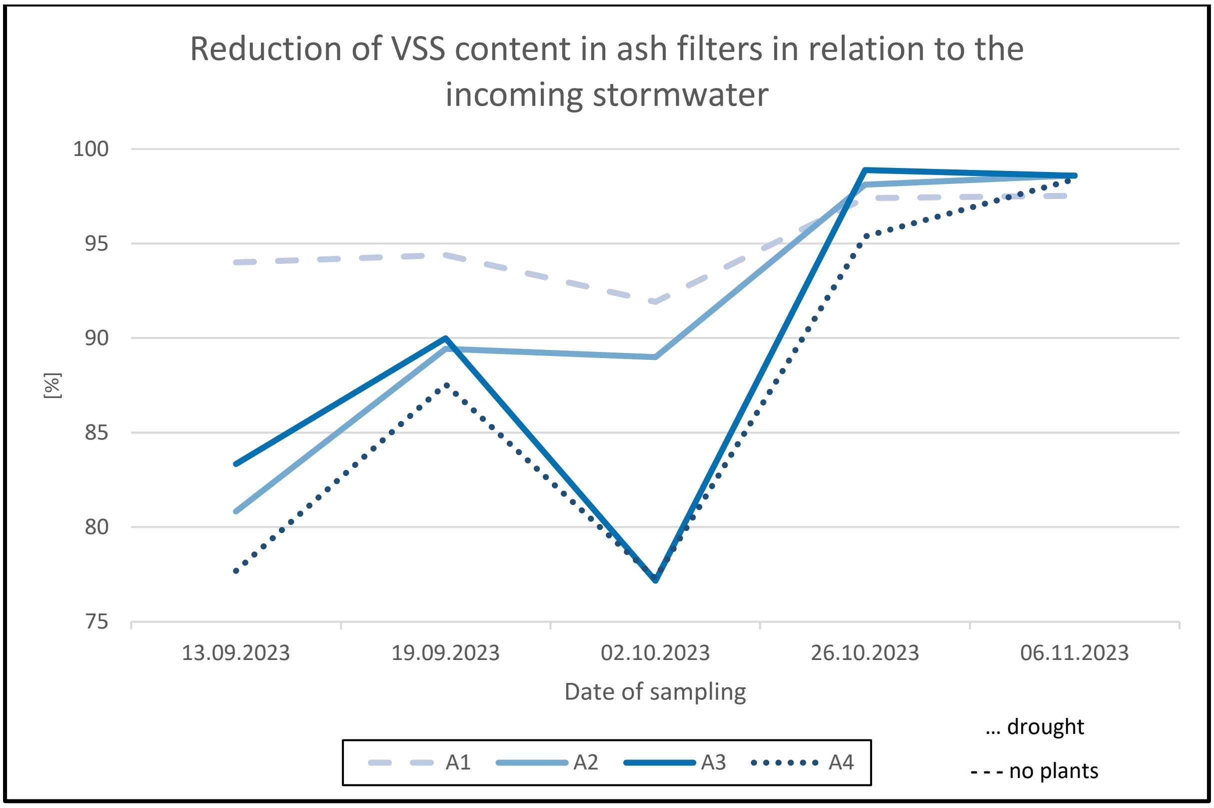

Figure 22 - Reduction of VSS content in ash filters in relation to the incoming stormwater

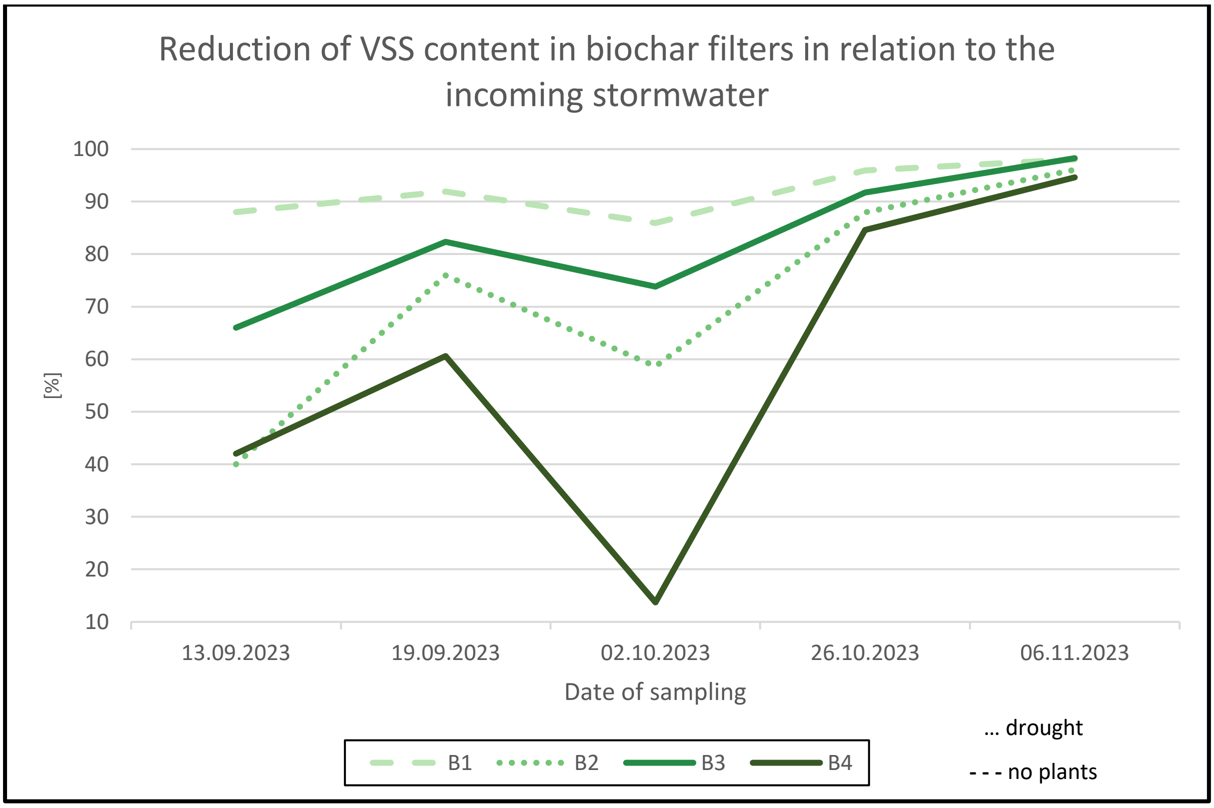

Figure 23 - Reduction of VSS content in biochar filters in relation to the incoming stormwater

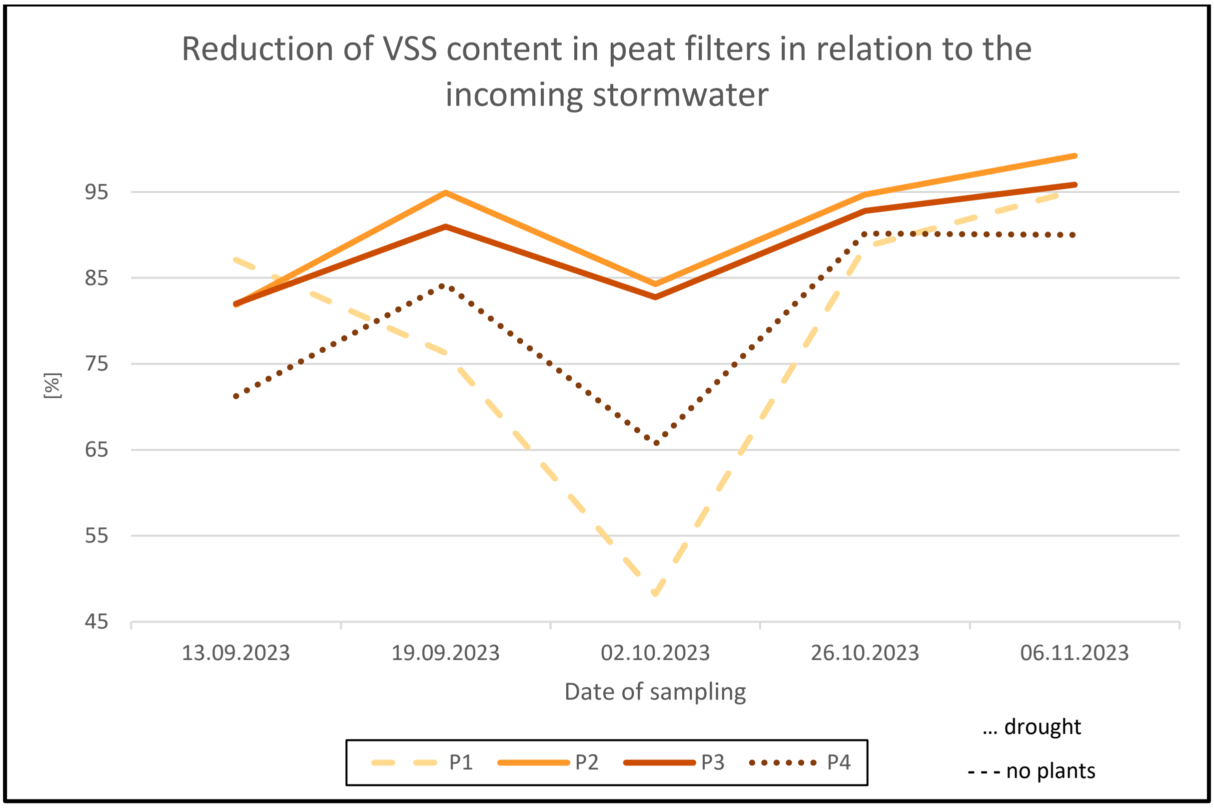

Figure 24 - Reduction of VSS content in peat filters in relation to the incoming stormwater

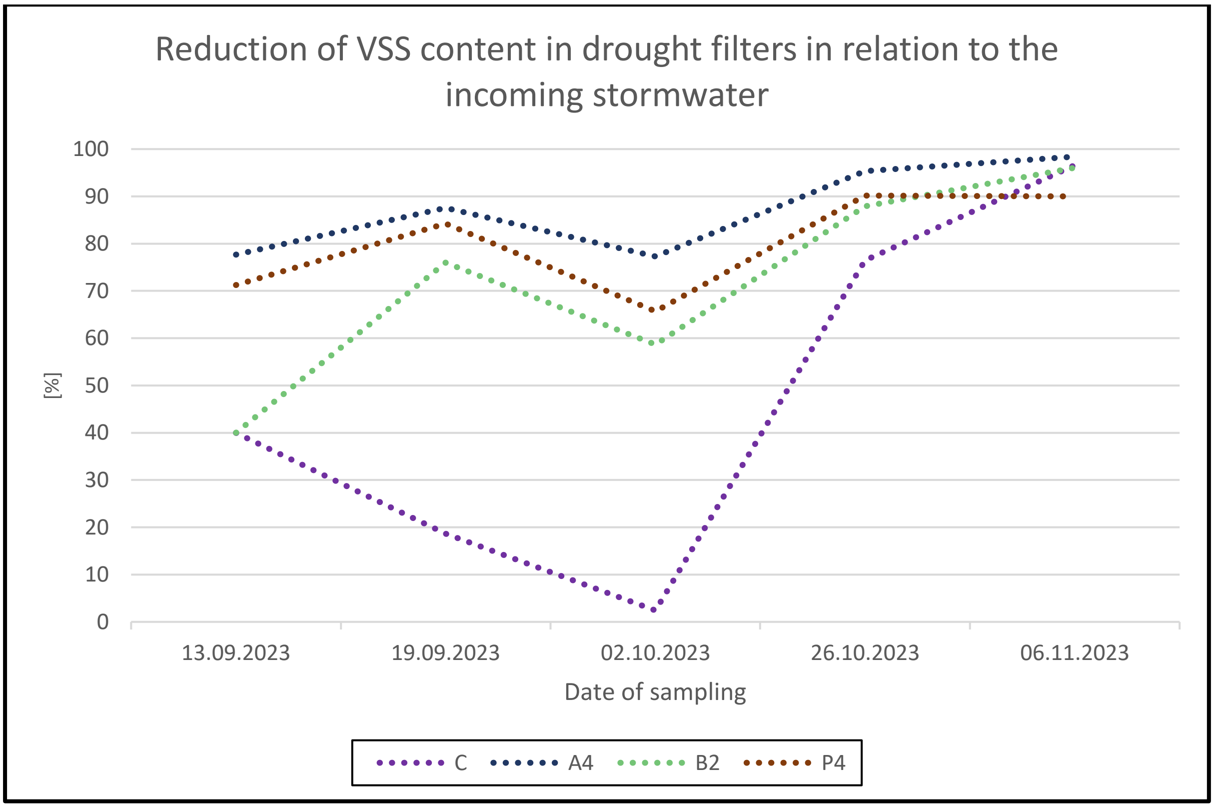

Figure 25 - Reduction of VSS content in drought filters in relation to the incoming stormwater

Appendix D – Anions and Cations

Table 7 - Anions content in the influents and effluents on the different sampling days

[mM] sampling day		Anions	Inc. SW.	C	A1	A2	A3	A4	B1	B2	B3	B4	P1	P2	P3	P4
	1	Acetate	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	2		0	0	0	0	0	0	0	0	0	0	0	0	0	0
	3		0	0	0	0	0	0	0	0	0	0	0	0	0	0
	4		0	0	0	0	0	0	0	0	0	0	0,005	0	0	0
	5		0	0	0	0	0	0	0	0	0	0	0,023	0,015	0,015	0
	1	Chloride	3,961	9,307	5,593	14,66	9,614	12,07	5,325	7,632	8,909	15	47,67	11,28	11,03	12,04
	2		0,782	10,17	5,057	11,14	7,105	9,112	4,583	7,981	10,72	13,02	6,21	11,2	11,96	11,34
	3		0,317	8,612	3,673	11,42	6,909	8,12	3,588	8,358	10,26	12,18	4,545	9,885	9,834	12,21
	4		0,248	8,082	2,319	9,642	5,456	6,157	2,577	8,209	9,569	11,08	2,327	5,809	6,328	10,38
	5		3,291	7,375	1,498	6,475	3,451	3,992	2,054	7,022	8,178	10,14	1,403	4,945	5,748	9,037
	1	Nitrite	0,043	0	0	0	0	0	0	0	0	0	0	0	0	0
	2		0,023	0	0	0	0	0	0	0	0	0	0	0	0	0
	3		0,023	0	0	0	0	0	0	0	0	0	0	0	0	0
	4		0,021	0	0	0	0	0	0	0	0	0	0	0	0	0
	5		0	0	0	0	0	0	0	0	0	0	0	0	0	0
	1	Bromide	0	0	0	0	0	0	0	0	0	0	29,39	0,176	0,092	0
	2		0	0,156	0,456	0,176	0	0,182	0,075	0,104	0,11	0,168	0,06	0,088	0,088	0,096
	3		0	0	0	0	0	0	0	0	0	0	0	0	0	0
	4		0,023	0	0	0	0	0	0	0	0	0	0,025	0	0	0
	5		0,046	0	0	0	0,132	0	0	0	0	0	0,022	0	0	0,069
	1	Nitrate	0,076	0,144	0,168	0,207	0,22	0,208	0,078	0,128	0,12	0,156	0	0,088	0,084	0,092
	2		0,036	0,088	0,186	0,2	0,208	0,203	0,084	0,116	0,11	0	0,052	0,084	0,092	0,084
	3		0,044	0,104	0,165	0,2	0,192	0,2	0,078	0,1	0,11	0,132	0,046	0	0,088	0
	4		0,042	0,084	0,17	0,176	0,168	0,154	0,048	0,092	0,126	0,132	0,025	0	0,063	0
	5		0	0,088	0,14	0,168	0,132	0,138	0,046	0,092	0,105	0,115	0,024	0	0,046	0,063
	1	Phosphate	0	0,104	0	0	0	0	0	0	0	0	0,084	0	0	0
	2		0	0	0	0	0	0	0	0	0	0	0,075	0	0	0
	3		0	0	0	0	0	0	0,079	0	0	0	0,078	0	0	0
	4		0	0	0	0	0	0	0,054	0,101	0	0	0,06	0	0,08	0,11
	5		0	0	0	0	0	0	0,055	0,103	0	0	0,066	0,058	0,064	0,083
	1	Sulfate	0,132	0,207	1,036	2,616	8,07	4,31	0,194	0,206	0,175	0,175	0,155	0,139	0,129	0,189
	2		0,068	0,184	0,803	2,013	5,239	3,452	0,186	0,213	0,171	0,16	0,11	0,137	0,132	0,173
	3		0,059	0,144	0,745	2,242	6,137	4,218	0,174	0,189	0,142	0,157	0,099	0,113	0,129	0,146
	4		0,062	0,123	0,708	2,178	5,786	4,092	0,123	0,184	0,157	0,155	0,069	0,102	0,092	0,122
	5		0,099	0,131	0,608	1,762	4,382	2,957	0,107	0,202	0,135	0,139	0,069	0,079	0,07	0,101

Table 8 - Cations content in the influents and effluents on the different sampling days

[mM] sampling day	Cations	Inc. SW.	C	A1	A2	A3	A4	B1	B2	B3	B4	P1	P2	P3	P4
1	**Litium**	0	0	0,022	0,023	0,029	0,018	0	0	0	0	0	0	0	0
2		0	0	0,021	0,023	0,028	0,013	0	0	0	0	0	0	0	0
3		0,0006	0	0,022	0,027	0,033	0,018	0	0	0	0	0	0	0	0
4		0	0	0,02	0,034	0,032	0,018	0	0	0	0	0,0005	0	0	0
5		0,001	0	0,021	0,027	0,026	0,016	0	0	0	0	0	0,001	0	0
1	**Sodium**	4,183	8,365	8,686	17,7	12,64	16,18	6,539	9,576	10,89	17,02	33,18	7,475	7,25	9,518
2		1,069	9,466	8,391	15,4	9,708	13,96	5,849	10,55	12,67	16,45	5,51	7,685	8,001	9,322
3		0,487	9,157	7,408	16,27	10	14,36	5,12	11,36	13,07	15,85	4,788	6,961	7,252	9,903
4		0,403	9,322	6,201	15,73	9,914	13,46	3,973	11,88	14,19	16	3,148	5,482	5,666	9,493
5		3,927	9,025	5,329	13,09	7,734	11,06	3,461	11,49	13,81	15,89	2,531	4,76	5,091	8,462
1	**Ammonium**	0,024	0	0	0	0	0	0	0	0	0	0	0	0	0
2		0,057	0	0	0	0	0	0	0	0	0	0	0	0	0
3		0,057	0	0	0	0	0	0	0	0	0	0	0	0	0
4		0,032	0	0	0	0	0	0	0	0	0	0	0	0	0
5		0,456	0	0	0	0	0	0	0	0	0	0	0	0	0
1	**Potassium**	0,172	0,577	1,309	1,507	1,409	1,228	0,833	0,522	1,285	1,084	17,98	1,051	1,033	0,661
2		0,092	0,841	2,152	1,417	1,173	1,153	0,739	0,782	1,466	1,112	0,608	0,93	1,04	0,654
3		0,049	0,682	1,192	1,469	1,333	1,256	0,716	0,861	1,547	0,934	0,543	0,814	1,026	0,642
4		0,046	0,634	0,967	1,278	1,15	1,054	0,469	0,742	1,310	0,847	0,314	0,609	0,69	0,492
5		0,187	0,619	0,828	1,18	1,033	0,958	0,433	0,662	1,239	0,858	0,256	0,516	0,584	0,419
1	**Magnesium**	0,118	0,256	1,884	2,223	2,7	0,912	0,48	0,192	0,4	0,378	0,552	0,352	0,364	0,296
2		0,051	0,336	1,704	1,8	2,04	0,833	0,432	0,26	0,44	0,288	0,16	0,356	0,372	0,24
3		0,033	0,36	1,74	2,28	2,416	1,104	0,441	0,316	0,51	0,366	0,14	0,324	0,364	0,268
4		0,033	0,44	1,645	2,688	2,52	1,204	0,392	0,376	0,534	0,396	0,091	0,276	0,3	0,296
5		0,166	0,468	1,408	2,429	2,124	1,032	0,362	0,356	0,495	0,355	0,065	0,21	0,228	0,201
1	**Manganese**	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2		0	0	0	0	0	0	0	0	0	0	0	0	0	0
3		0,0003	0,016	0	0	0	0	0	0	0,005	0,01	0	0,002	0	0
4		0	0,064	0	0	0	0	0	0	0	0	0	0,013	0	0
5		0	0	0	0	0	0	0	0	0	0	0	0	0	0
1	**Calcium**	0,612	0,732	5,043	4,872	10,33	5,303	1,324	0,537	1,143	0,904	1,154	0,948	1,135	1,11
2		0,474	1,024	4,513	4,035	7,892	4,651	1,179	0,749	1,274	0,746	0,712	1,014	1,214	0,992
3		0,31	1,071	4,353	4,851	9,172	5,914	1,206	0,905	1,431	0,883	0,645	0,983	1,238	1,158
4		0,361	1,327	4,017	5,193	9,232	6,085	1,106	1,078	1,534	0,95	0,537	0,959	1,115	1,316
5		1,06	1,309	3,486	4,603	7,692	5,025	1,056	0,995	1,637	1,147	0,356	0,7	0,867	1,035

Appendix E – Total Organic Carbon

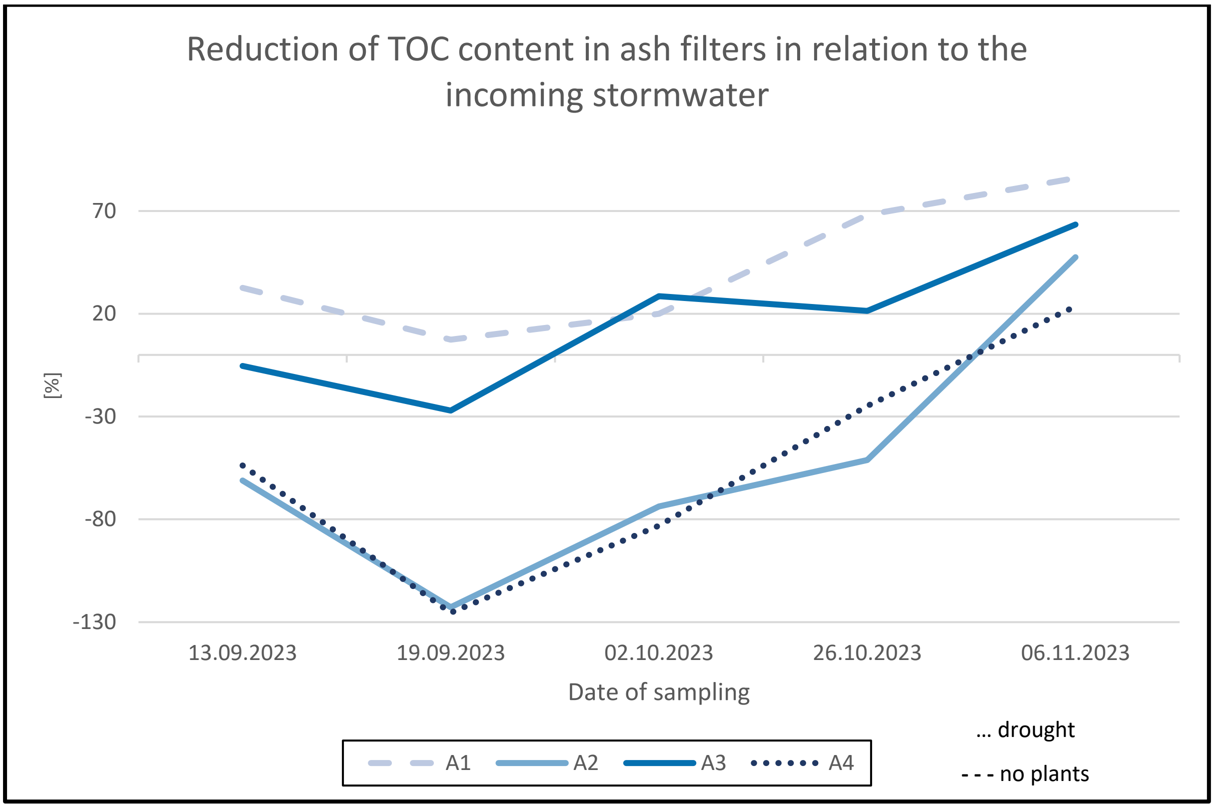

Figure 26 - Reduction of TOC content in ash filters in relation to the incoming stormwater

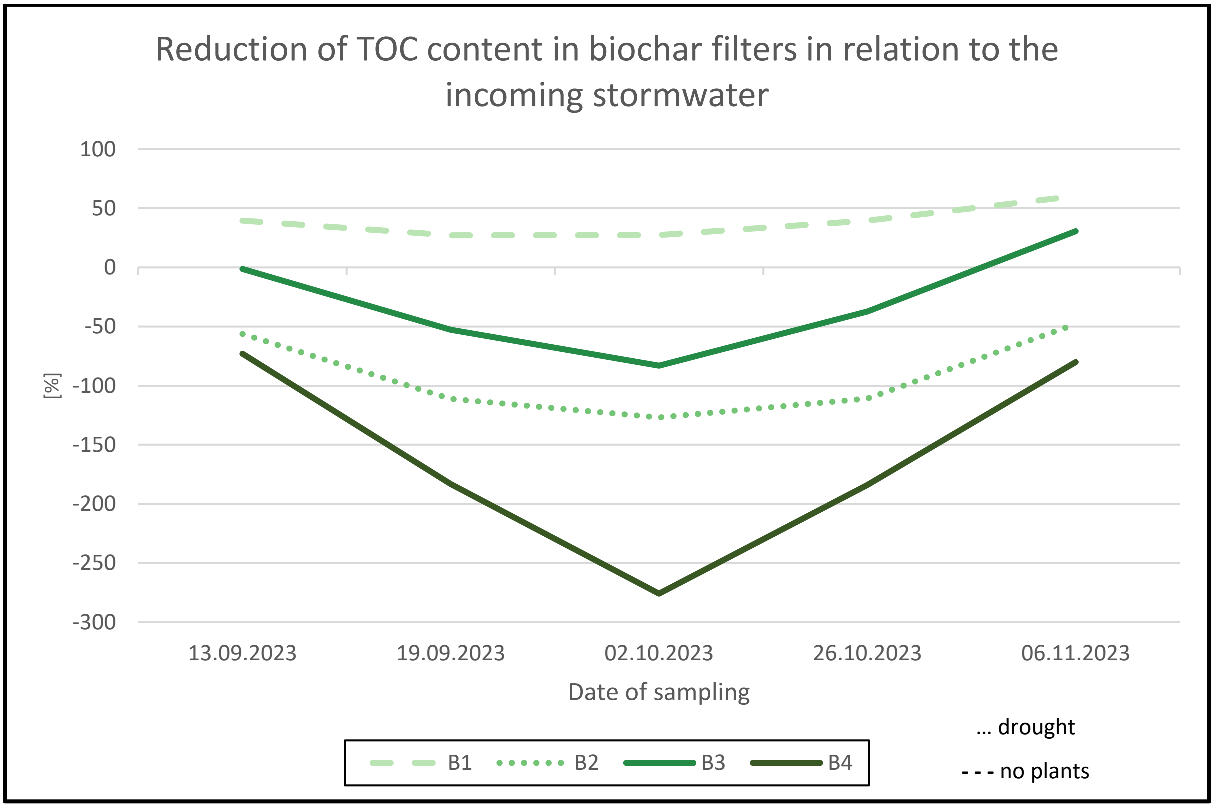

Figure 27 - Reduction of TOC content in biochar filters in relation to the incoming stormwater

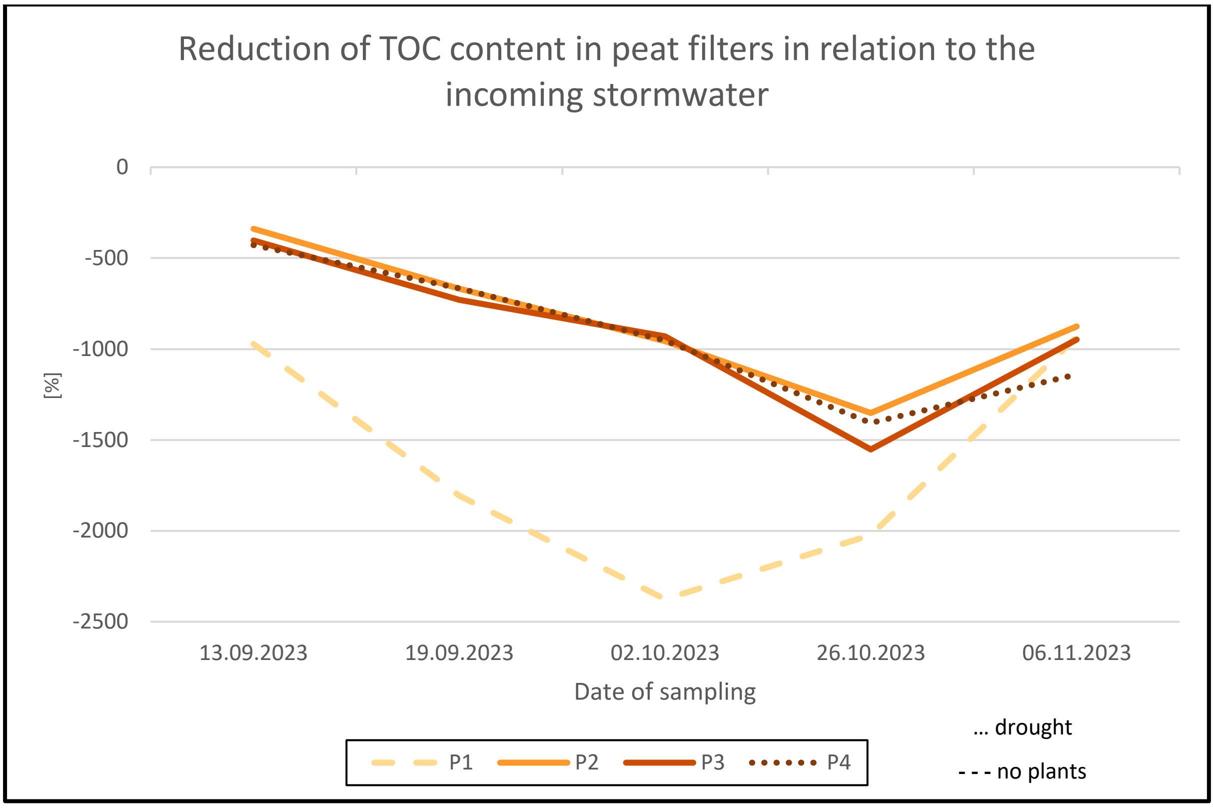

Figure 28 - Reduction of TOC content in peat filters in relation to the incoming stormwater

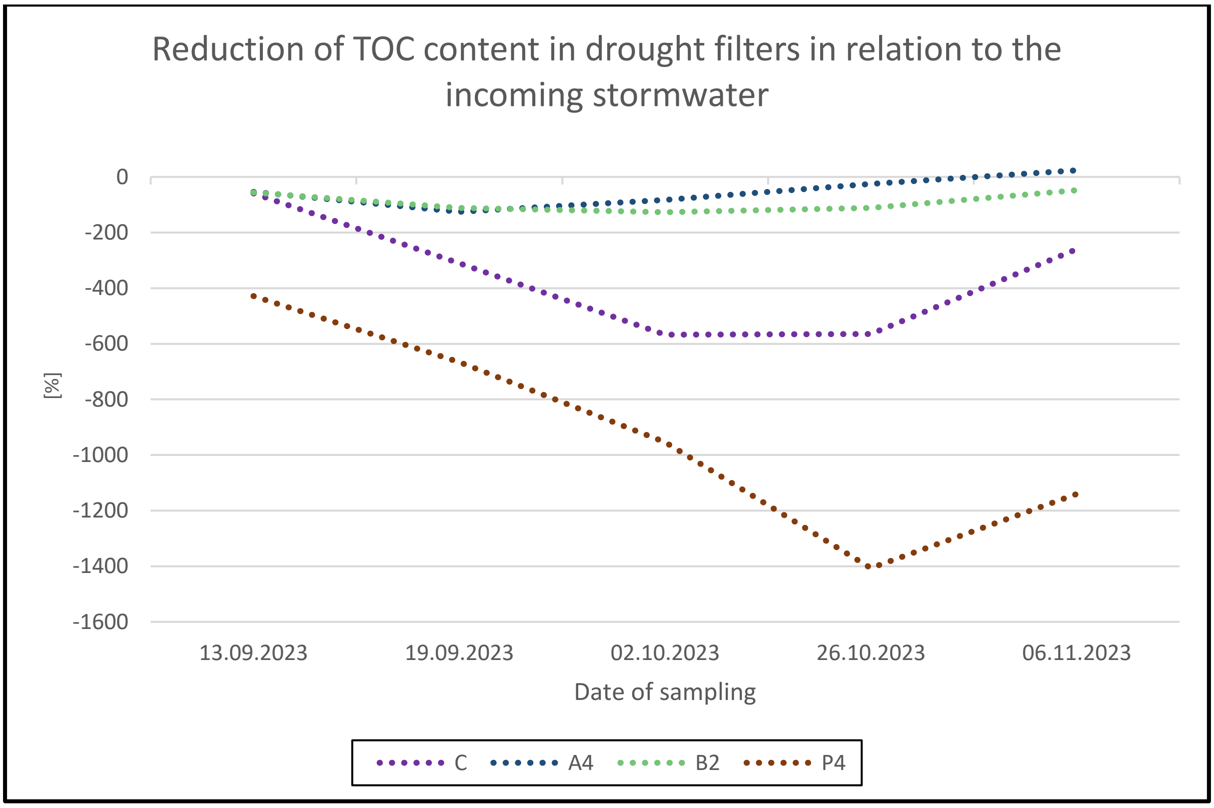

Figure 29 - Reduction of TOC content in drought filters in relation to the incoming stormwater

Table 9 - TOC concentrations of the various water samples (green: compliance with the limit value 12mg/L; red: leaching)

TOC in [mg/L]	2023-09-13	2023-09-19	2023-10-02	2023-10-26	2023-11-06
Inc. SW.	14.990	9.293	8.248	8.738	16.690
C	23.860	38.130	55.040	58.080	60.000
A1	10.110	8.604	6.600	2.791	2.334
A2	24.140	20.700	14.328	13.208	8.757
A3	15.790	11.810	5.890	6.869	6.108
A4	23.050	20.950	15.104	10.906	12.750
B1	9.054	6.769	5.988	5.282	6.618
B2	23.420	19.620	18.720	18.440	24.692
B3	15.170	14.200	15.105	11.988	11.580
B4	25.930	26.330	31.014	24.828	30.080
P1	160.700	177.200	204.200	185.500	172.500
P2	65.800	71.420	87.210	126.870	162.760
P3	75.400	77.110	85.000	144.390	174.980
P4	79.230	71.220	86.920	131.640	207.030

Appendix F – Dissolved Organic Carbon

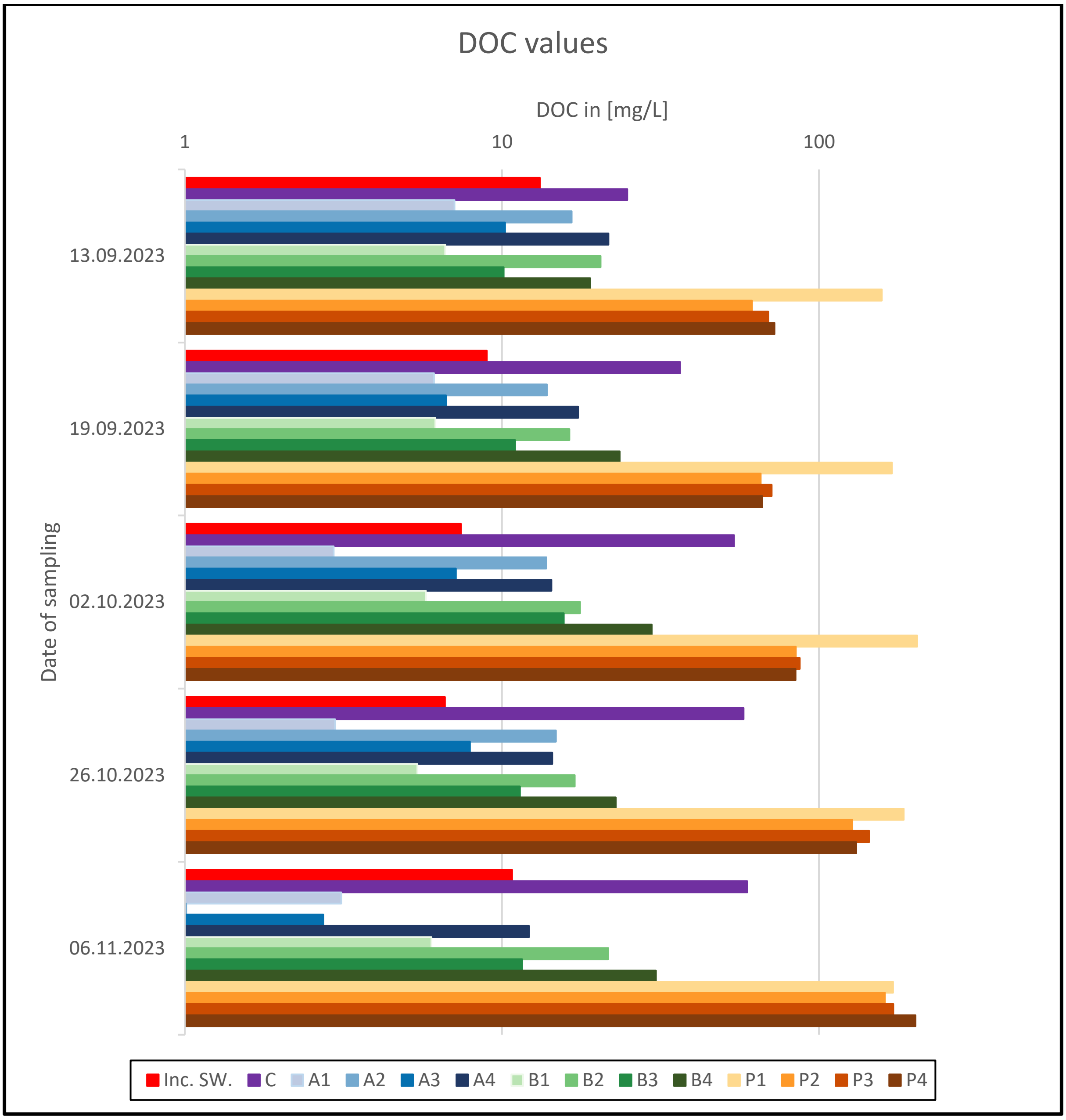

Figure 30 - DOC concentration in influents and effluents at the different sampling days in all the bioretention filters at the Gårda raingarden facility

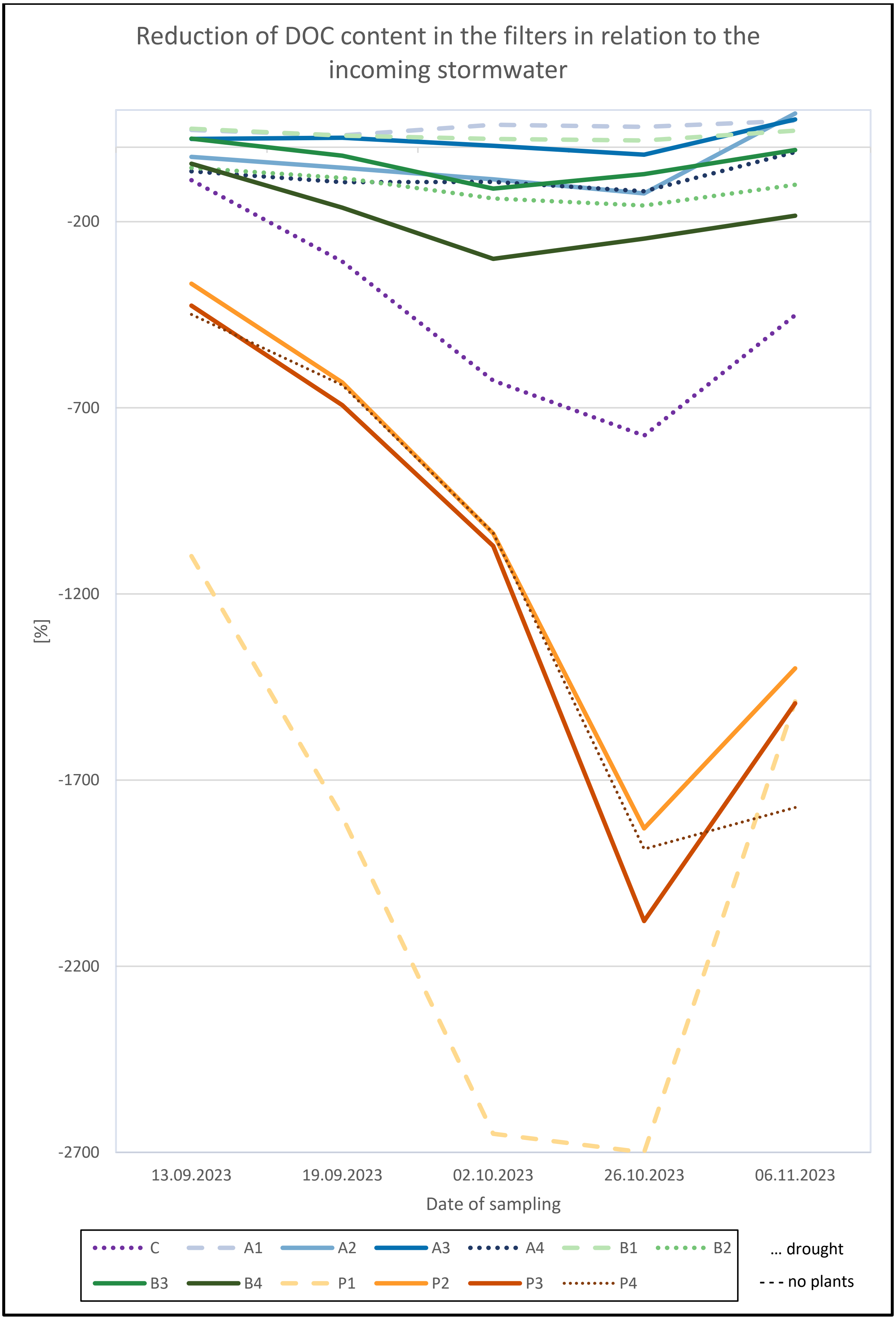

Figure 31 - DOC removal efficiency at the different sampling days in all the bioretention filters at the Gårda raingarden facility

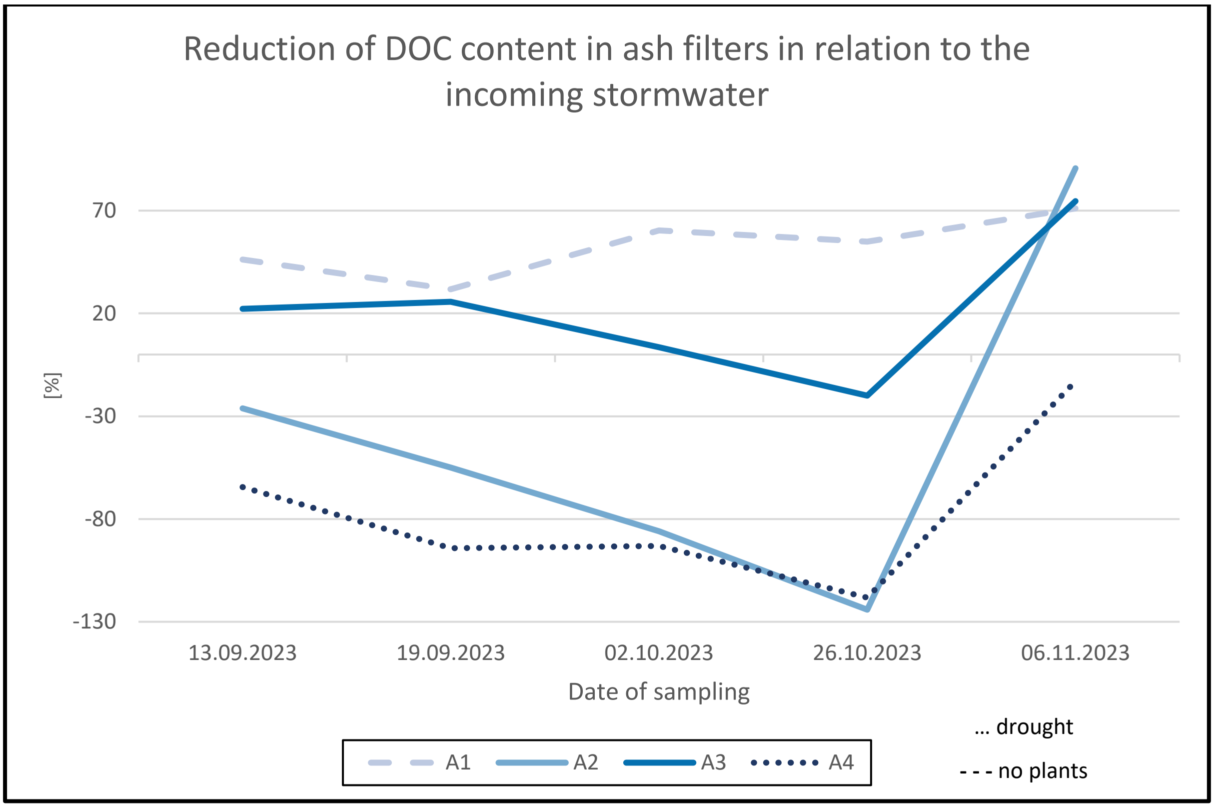

Figure 32 - Reduction of DOC content in ash filters in relation to the incoming stormwater

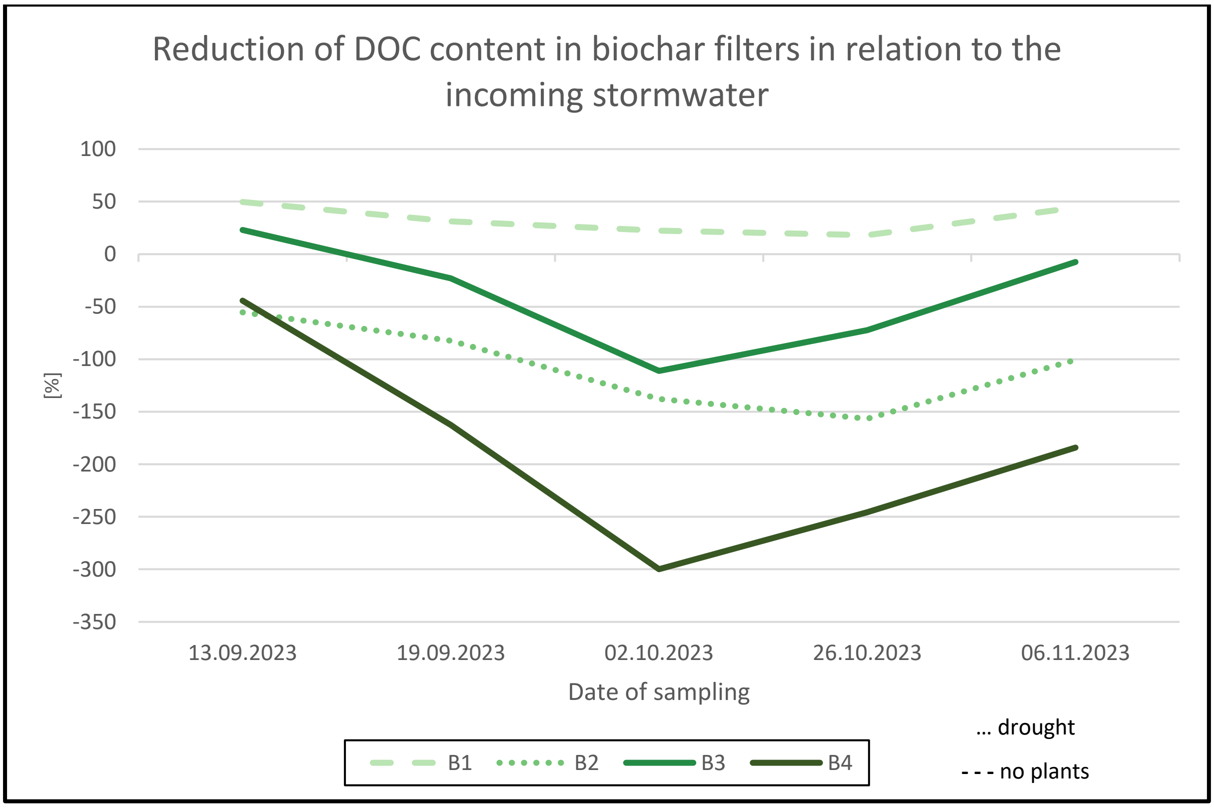

Figure 33 - Reduction of DOC content in biochar filters in relation to the incoming stormwater

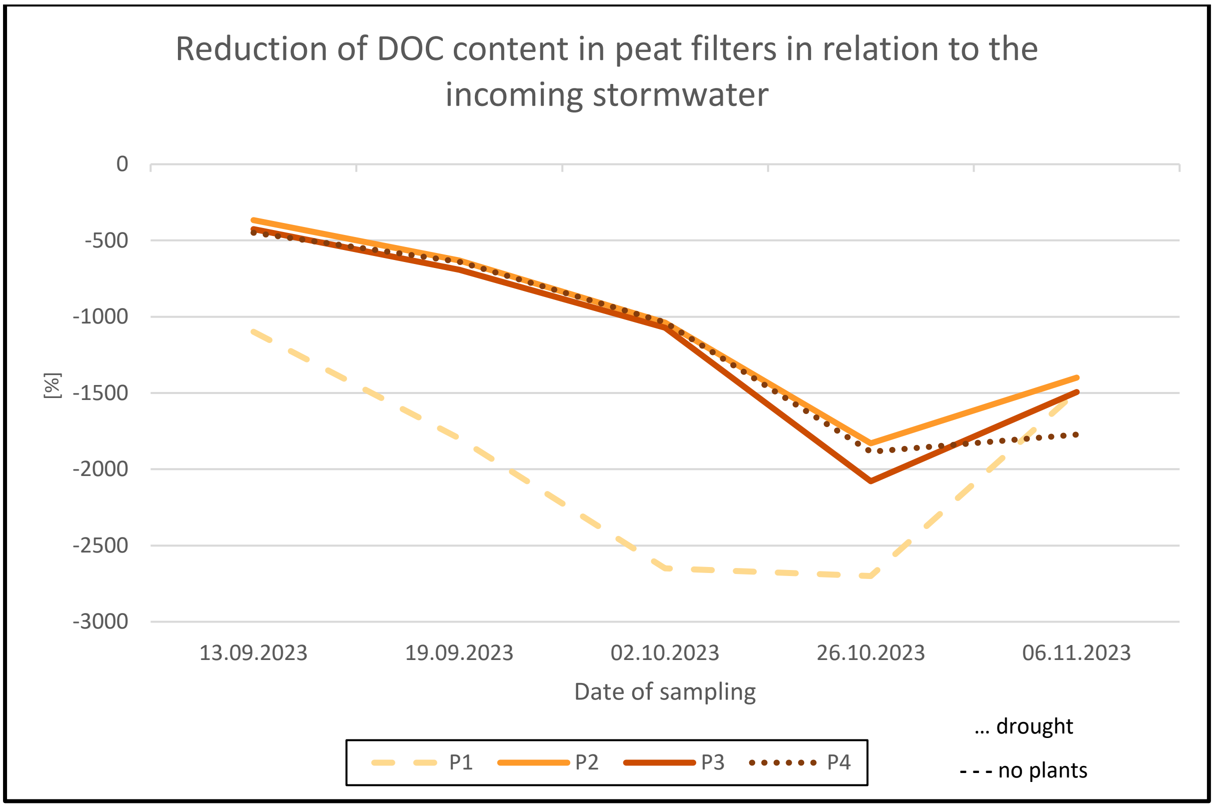

Figure 34 - Reduction of DOC content in peat filters in relation to the incoming stormwater

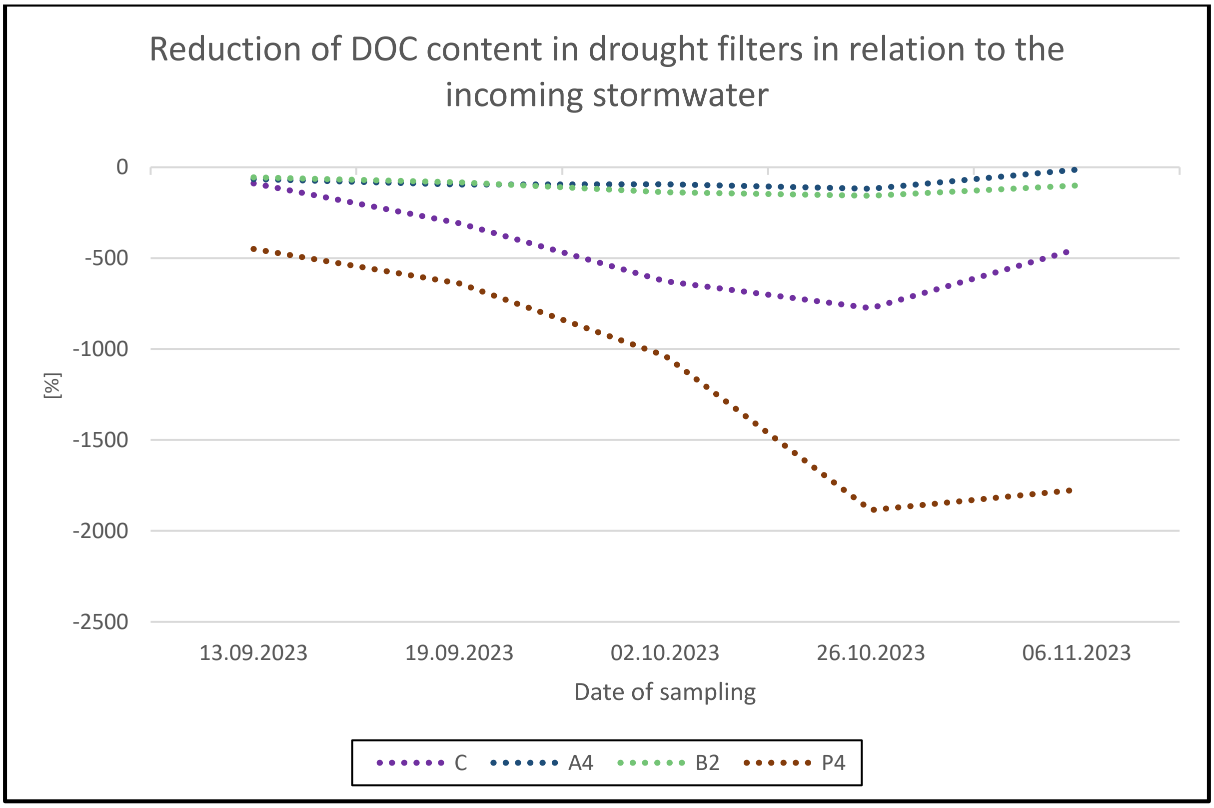

Figure 35 - Reduction of DOC content in drought filters in relation to the incoming stormwater

Table 10 - DOC concentrations of the various water samples (yellow: higher DOC values than TOC values; red: leaching)

DOC in [mg/L]	**2023-09-13**	**2023-09-19**	**2023-10-02**	**2023-10-26**	**2023-11-06**
Inc. SW.	13.168	8.951	7.426	6.607	10.768
C	24.828	36.464	53.960	57.800	59.360
A1	7.086	6.108	2.946	2.978	3.112
A2	16.605	13.872	13.808	14.808	1.009
A3	10.230	6.660	7.156	7.929	2.731
A4	21.664	17.381	14.336	14.420	12.168
B1	6.630	6.165	5.757	5.412	5.990
B2	20.456	16.316	17.652	16.956	21.616
B3	10.135	11.005	15.680	11.394	11.580
B4	18.996	23.496	29.694	22.836	30.600
P1	157.800	169.920	204.200	185.000	171.100
P2	61.440	65.520	84.480	127,500	161.520
P3	69.200	70.960	86.960	143.940	171.640
P4	72.320	66.160	84.400	131.160	201.750

Appendix G – Total Nitrogen

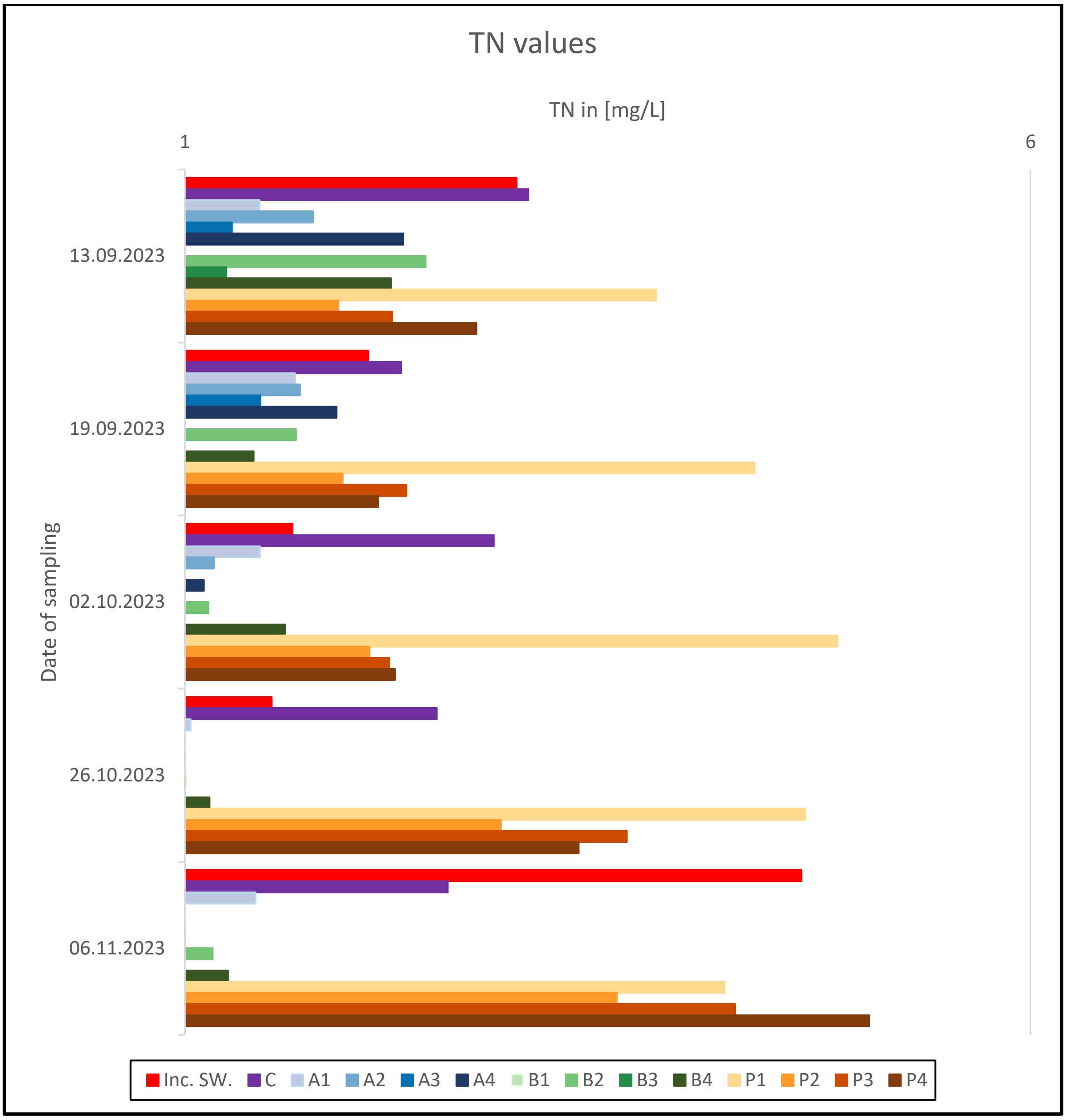

Figure 36 - TN concentration in influents and effluents at the different sampling days in all the bioretention filters at the Gårda raingarden facility

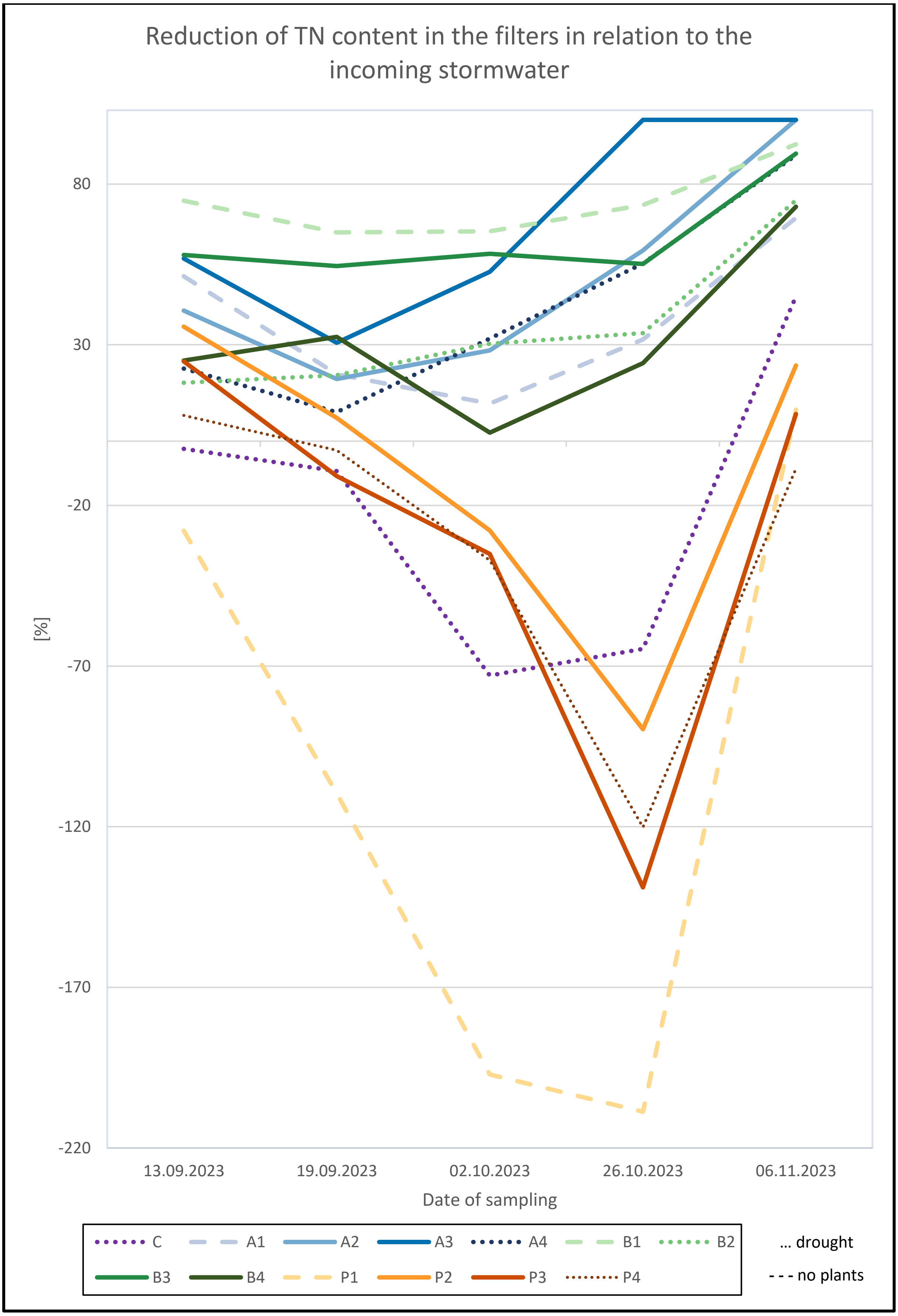

Figure 37 - TN removal efficiency at the different sampling days in all the bioretention filters at the Gårda raingarden facility

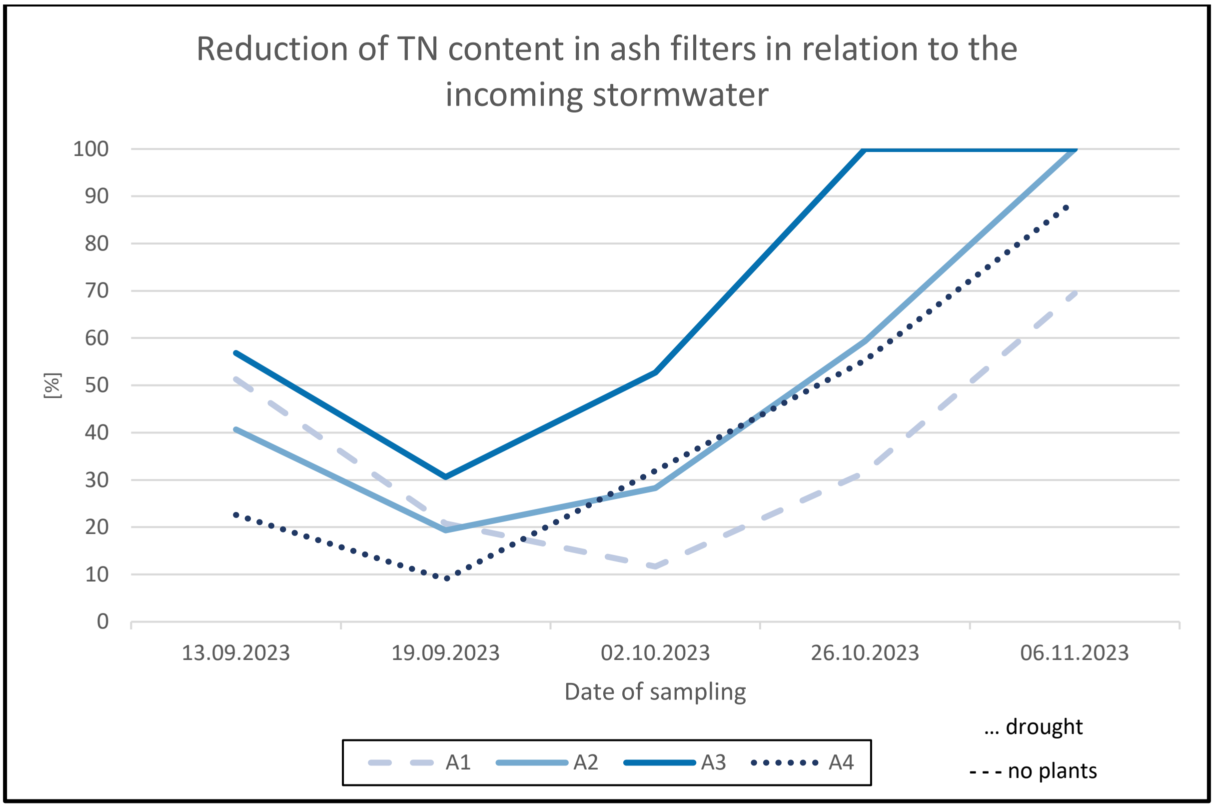

Figure 38 - Reduction of TN content in ash filters in relation to the incoming stormwater

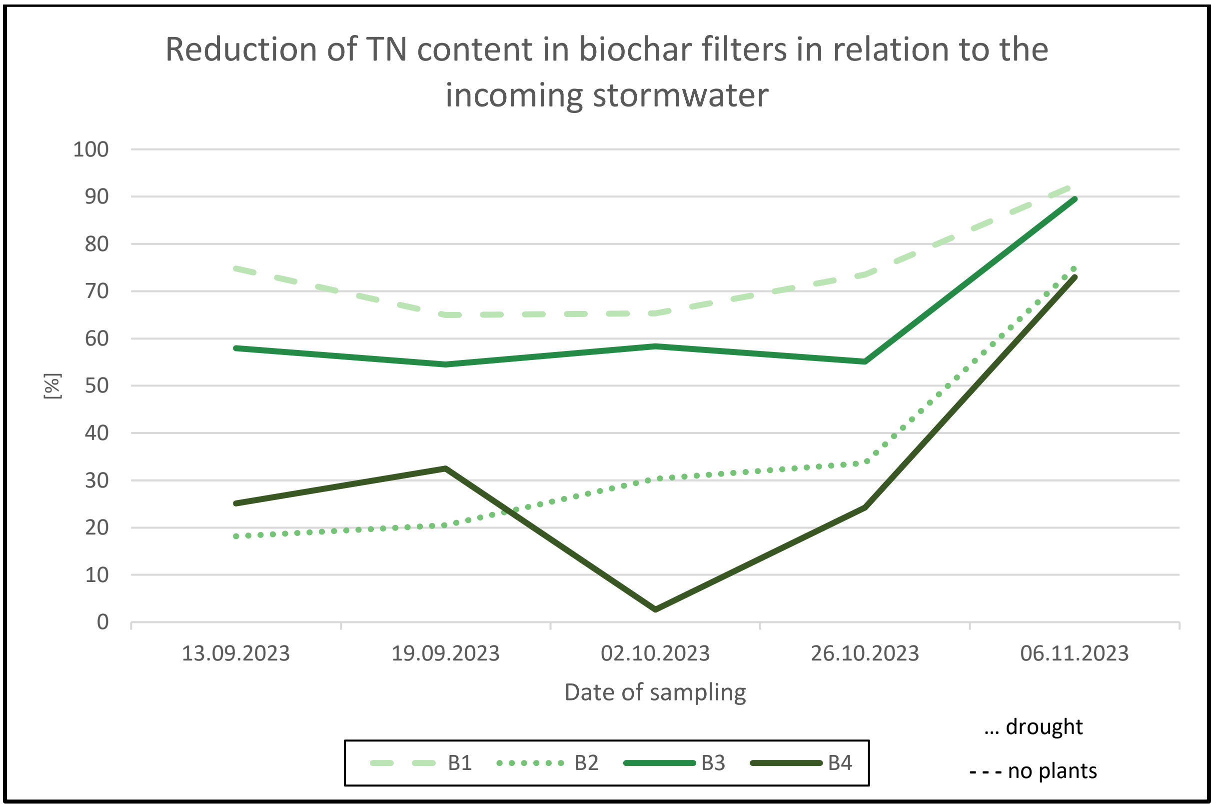

Figure 39 - Reduction of TN content in biochar filters in relation to the incoming stormwater

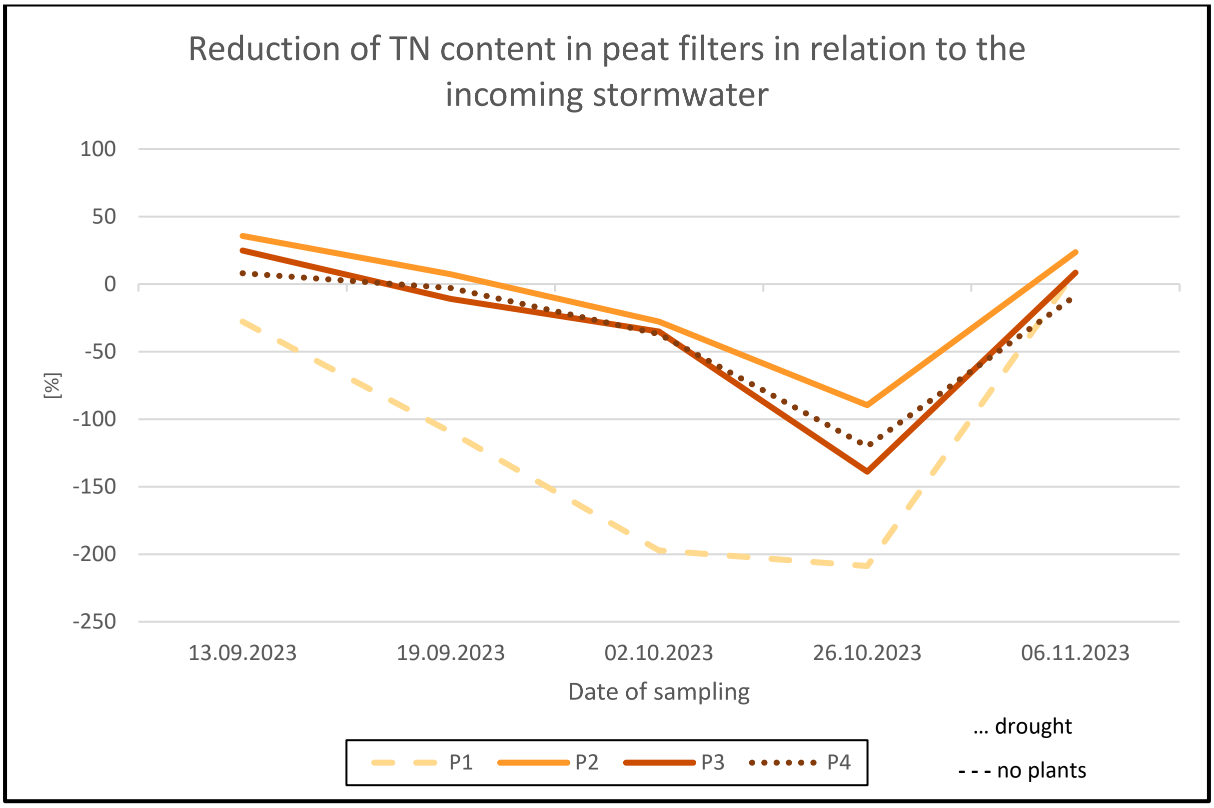

Figure 40 - Reduction of TN content in peat filters in relation to the incoming stormwater

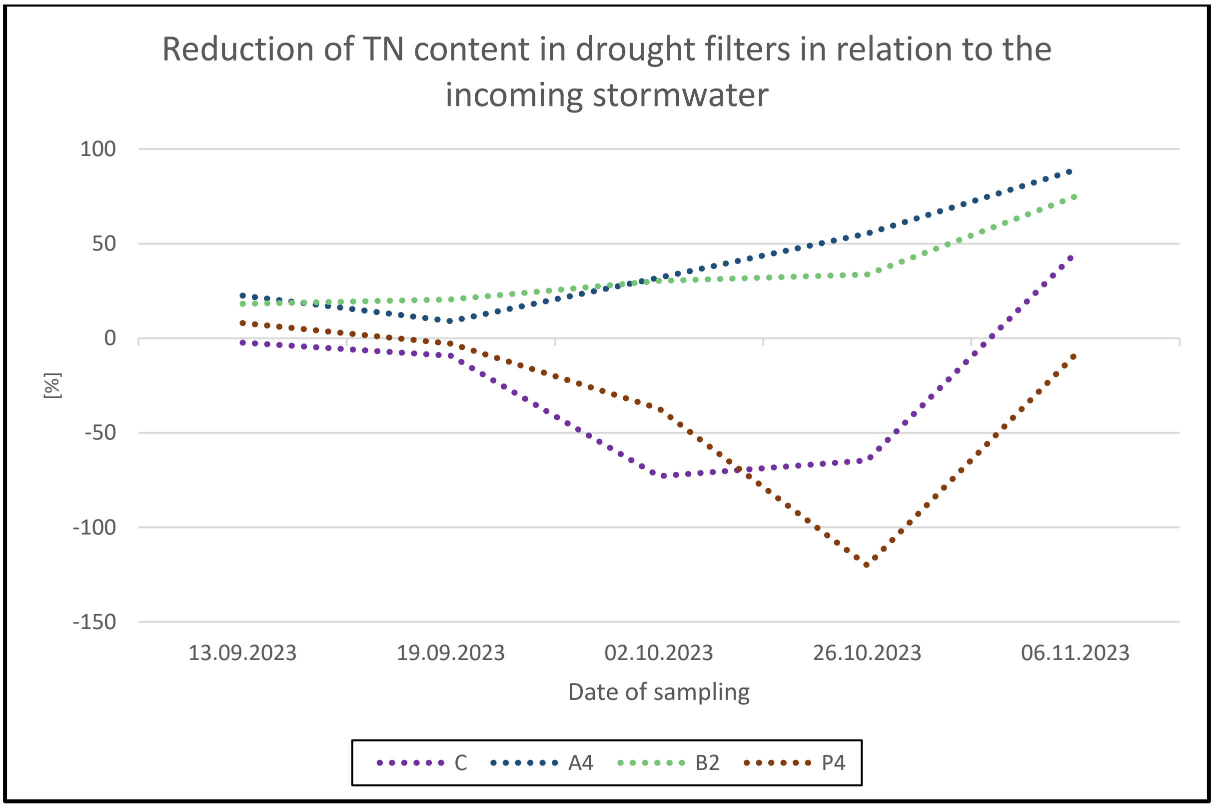

Figure 41 - Reduction of TN content in drought filters in relation to the incoming stormwater

Table 11 - TN concentrations of the various water samples (green: compliance with the limit value 1,25mg/L; red: leaching)

TN in [mg/L]	2023-09-13	2023-09-19	2023-10-02	2023-10-26	2023-11-06
Inc. SW.	2.961	2.084	1.635	1.512	4.646
C	3.031	2.278	2.828	2.489	2.555
A1	1.442	1.651	1.444	1.034	1.419
A2	1.757	1.681	1.173	0.615	0.000
A3	1.278	1.446	0.773	0.000	0.000
A4	2.292	1.896	1.113	0.676	0.511
B1	0.746	0.730	0.567	0.401	0.354
B2	2.423	1.657	1.139	1.003	1.164
B3	1.246	0.948	0.681	0.679	0.488
B4	2.218	1.407	1.592	1.145	1.256
P1	3.786	4.368	4.858	4.668	4.191
P2	1.906	1.933	2.090	2.868	3.554
P3	2.226	2.311	2.209	3.612	4.254
P4	2.724	2.143	2.242	3.329	5.046

Appendix H – Comparison of Effluent Parameters

Table 12 - Comparison of drought filters

	A4	B2	P4	Recommendation based on the results
Turbidity	- Higher turbidity than reference filter at the beginning - Minimization of the discrepancies during the study - Regeneration of the filter	- Higher turbidity than reference filter at the beginning - Clear with yellowish discolouration like B4; B3, however, without colouration	- Higher turbidity than reference filter at the beginning - Minimisation of the discrepancies in the course of the study - Regeneration of the filter	
TSS	- Positive RE - Compliance with the limit value - TSS content higher than reference filter, but continuous RE >90% - Effect of drought highest at high pollutant concentrations, lower RE than reference filter - Regeneration of the filter	- Positive RE - Compliance with the limit value from day 2 - TSS content on day 1 & 5 higher than reference filter - Effect of drought highest at high pollutant concentrations, lower RE than reference filter - Regeneration of the filter	- Positive RE - Compliance with the limit value - TSS content on 1st & 5th day higher than reference filter - Continuous RE >85% - Effect of drought highest at high pollutant concentrations, lower RE than reference filter - Regeneration of the filter	1. A4 2. P4 3. B2 4. C
VSS	- Positive RE - Compliance with the limit value - Effects of drought highest at high pollutant concentrations, lower RE than reference filter - Regeneration of the filter	- Positive RE - Compliance with the limit value - Continuously lower RE than B2 but higher than B4 - Influence of drought cannot be assessed - Regeneration of the filter	- Positive RE - Compliance with the limit value - Effects of drought highest at high pollutant concentrations, lower RE than reference filter - Regeneration of the filter	1. A4 2. P4 3. B2 4. C
IC – *Nitrite*	- Positive RE - Below the detection limit - No effect of the drought	- Positive RE - Below the detection limit - No effect of the drought	- Positive RE - Below the detection limit - No effect of the drought	No detectable differences
Nitrate	- Leaching - Similar to the reference filters; no effect of drought	- Leaching - Minimally better values than the reference filters (exception: day 2 B4) - No effect of the drought	- First c, then complete removal - Worse than P2, but partially better properties than P3 - Influence of drought cannot be assessed	1. P4 2. B2& C 3. A4
Ammonium	- Positive RE - Below the detection limit - No effect of the drought	- Positive RE - Below the detection limit - No effect of the drought	- Positive RE - Below the detection limit - No effect of the drought	No detectable differences

Phosphate	- Neither removal nor leaching - Below the detection limit - No effect of the drought	- Leaching - No leaching in reference filters, negative RE only from the 4th sample day onwards - Effect of the drought	- Leaching - Slightly worse values than reference filter, negative Re from the 4th sample day onwards - Minor effect of the drought	1. A4 2. C 3. B2 & P4
TOC	- Leaching + 5th day positive RE - Exceeding the limit value except on day 4 - Impact of drought only at high pollutant concentrations	- Leaching - Exceeding the limit value - Continuously lower RE than B2 but higher than B4 - Influence of drought cannot be assessed	- Extreme leaching - Exceeding the limit value - Minor impact of drought only at high pollutant concentrations	1. A4 2. B2 3. C 4. P4
DOC	- Leaching + 5th day positive RE - Impact of drought only at high pollutant concentrations	- Leaching - Continuously lower RE than B2 but almost always higher than B4 - Influence of drought not assessable	- Extreme accumulation - Minor impact of drought only at high pollutant concentrations	1. A4 2. B2 3. C 4. P4
TN	- Positive RE - Compliance with the limit value from day 3 - Low impact of the drought	- Positive RE - Compliance with the limit value from day 3 - Similar behaviour to B4 but lower RE than B2 - Influence of drought cannot be assessed	- Except for 1st day leaching - Exceeding the limit value - No effect of the drought	1. A4 2. B2 3. C 4. P4
OP	- Most of the OP values below the detection limit, no analysis	- Most of the OP values below the detection limit, no analysis	- Most of the OP values below the detection limit, no analysis	-
MP – *PE*	- Positive RE - 1st day significantly better value than reference filter - No effect of the drought	- positive RE - 1st day significantly better value than reference filter - No effect of the drought	- Positive RE - No impact of the drought	1. P4 &C 2. A4 3. B2
PP	- Extreme leaching - Higher leaching than reference filter	- Extreme leaching - Significantly lower leaching than reference filter	- Extreme leaching - Significantly lower leaching than reference filter	1. A4 2. B2, P4 & C
PVC	- Positive RE - 1st day significantly better value than reference filter	- Positive RE - No impact of the drought	- Positive RE - No impact of the drought	No detectable differences
PI	- Positive RE - 1st day better value than reference filter	- Positive RE - 1st day better value than reference filter	- Positive RE - No impact of the drought	1. P4 2. B2 & A4 3. C
PA6 & PBD	- Positive RE - No impact of the drought	- Positive RE - No impact of the drought	- Positive RE - No impact of the drought	No detectable differences