LICENTIATE THESIS

Components for area-efficient stormwater treatment systems

Ivan Milovanović

Urban Water Engineering

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Components for area-efficient stormwater treatment systems

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Preface

This licentiate thesis presents a summary of my research carried out in the Urban Water Engineering group at the Department of Civil, Environmental and Natural Resources Engineering at Luleå University of Technology. This study was funded by the Swedish Research Council FORMAS, project numbers 2016-20075 and 2016-01447 and by Vinnova as part of the DRIZZLE Centre for Stormwater management (grant number 2016-05176) and GreenNano (grant number 2018-00441), whose support is greatly appreciated. The work was also carried out as a part of a research cluster Stormwater&Sewers, a collaboration between Swedish municipalities of Luleå, Skellefteå, Östersund, Boden, and the water ytilities Vakin, MittSverige Vatten & Avfall, VASYD, the Swedish Water and the Urban Water Engineering research group.

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Ivan Milovanović

Luleå, May 2021

Abstract

With progressing urbanisation, treatment of urban stormwater is a vital issue that should be addressed to ensure good water quality in receiving water bodies. Treatment may be performed near the source, with different filter systems using various filter materials, or by using an end-of-pipe method, e.g. a stormwater pond. One constraint in the urban environment is the lack of available space in developed areas, where stormwater treatment facilities are needed the most. Methods developed to treat the stormwater runoff have been the focus of previous studies but the increasing standards of water quality and increasing land constraint pressures demand the further development of stormwater treatment systems. Both laboratory and field experiments are necessary to understand and improve the treatment processes as well as to evaluate how the implemented methods perform under field conditions. The aim of the thesis was to increase the knowledge about the components in stormwater treatment systems that can be used in area-efficient treatment facilities. In order to compare four potential stormwater filter materials (peat, bark, air-blown polypropylene and milkweed), column experiments were carried out using synthetic stormwater that simulated road runoff. Experiments were carried out to evaluate the impacts of the ageing of synthetic stormwater quality during laboratory testing, including dissolved metal concentrations and their impact on the estimation of filter efficiency. In a field study, a full-scale application of a zeolite filter installation was investigated, with a focus on service life and the efficiency of treating copper roof runoff. In order to further investigate a novel sedimentation device, a bottom grid structure (BGS), promoting sediment settling in a smaller area of a stormwater pond, a hydraulic modelling study was conducted to investigate the impact of the cell geometry of the structure on sediment settling and the impact of the structure on pond maintenance and sediment resuspension.

The column tests of four different filter materials showed that bark and peat had higher treatment efficiency for dissolved metals than milkweed and polypropylene, with the order of efficiency being peat>bark>milkweed>polypropylene. All four of the filter materials showed a total metal reduction of over 70%, which could be due to the separation of particle-bound metals in the columns. The ageing of the synthetic stormwater showed that dissolved metals, particularly copper, decreased in concentration, quite rapidly. During one experiment run, the dissolved copper concentration was reduced to 15% of its initial value. In order to account for the concentration changes an equation was proposed that normalised the concentration of dissolved metal over the duration of the experiment. During the observation period of 16 months, the zeolite installation removed 52% to 82% and 48% to 94% of total and dissolved copper, respectively. However, the effluent concentrations were still high (360-600 μg/l). There was also an indication of the decreasing filter performance over time with a prediction that the treatment level of total copper would drop to approximately 25% by the end of the service life of three years. The hydraulic experiments on a scaled model of a BGS showed that wider cells were on average 13% more efficient in trapping the particles than the narrower variant. The cell wall angle also had an impact (tilted walls added to the sedimentation efficiency), although the applicability of such cell structures can be questioned, as this cell shape may hinder maintenance efforts. It was also hypothesised that the inclusion of the BGS in the pond reduces the area needed for sediment settling, thus making the pond more area-efficient and easier to include in an urbanised setting.

Sammanfattning

Med den pågående urbaniseringen är dagvattenrening en viktig del för att försäkra en god vattenkvalitet i recipienter. Dagvattenrening kan utföras nära källan genom att använda olika filtersystem med olika filtermaterial, eller genom att använda en end-of-pipe metod, exempelvis en dagvattendamm. Ett hinder när det gäller stadsmiljö är platsbristen i bebyggda områden, just där reningen behövs som mest. Metoder för behandling av dagvatten har avhandlats i tidigare studier, men med ökande krav på rening för att förbättra vattenkvalitet samt brist på tillgänglig ytor för rening krävs det ytterligare utveckling av dagvattenreningssystemen. Både laboratorieförsök och fältförsök är nödvändiga för att förstå och förbättra reningsprocesserna samt utvärdera hur de implementerade metoderna presterar under naturliga förhållanden. Syftet med detta arbete var att öka kunskapen om komponenter i dagvattenreningssystem som kan användas i yteffektiva reningsanläggningar. För att jämföra fyra potentiella filtermaterial för dagvatten (torv, bark, luftblåst polypropylen och sidenört), utfördes kolonnexperiment med syntetiskt dagvatten framställt att efterlikna trafikdagvatten Experiment, uppställda i laboratoriemiljö, utfördes för att utvärdera hur kvaliteten på syntetiskt dagvattnet påverkas av åldrande, inkluderat lösta metallkoncentrationer, samt dess inverkan på bedömning av filtereffektivitet. I ett fältförsök undersöktes en fullskalig tillämpning av en installation med zeolitfilter, med fokus på livslängd och reningseffektivitet gällande takavrinning från koppartak. För att ytterligare undersöka en ny typ av sedimentationsapparatur som främjar sedimentering i ett mindre område i en dagvattendamm, ett så kallat sedimentationsraster, utfördes en hydraulisk modelleringsstudie för att utforska betydelsen av rastrets cellgeometri och dess inverkan på underhåll och återsuspension av sediment.

Kolonnexperimenten med fyra olika filtermaterial visade att bark och torv hade högre reningseffektivitet för lösta metaller än sidenört och polypropylen, med effektivitetsgraden i storleksordning torv>bark>sidenört>polypropylen. Alla fyra filtermaterial visade på en total metallreduktion om mer än 70 %, vilket förklarades med separation av partikelbundna metaller i kolonn. Åldrandet av syntetiskt dagvatten visade att koncentrationen av lösta metaller, i synnerhet koppar, minskade ganska snabbt. Under ett specifikt försök reducerades halten löst koppar till 15 % av det ursprungliga värdet. För att ta höjd för koncentrationsförändringarna föreslogs en ekvation som normaliserade koncentrationen av lösta metaller. Under observationsperioden på 16 månader avlägsnade zeolitfiltret 52–82 % och 48–94 % av totalt respektive löst koppar. Dock var kopparkoncentrationen i det behandlade vattnet fortfarande hög, 360-600 μg/l. Det fanns också en indikation på minskad reningsförmåga över tid och en prognos över filtrets förmåga att rena totalt koppar visade på en minskning till nära 25 % då livstiden på tre år uppnåtts. De hydrauliska experimenten på en nerskalad modell av sedimentationsrastret visade att bredare celler var i genomsnitt 13 % mer effektiva på att fånga partiklar jämfört med en smalare variant. Cellernas vägglutning var också av betydelse (lutande väggar ökade sedimentationseffektiviteten), även om nyttan med sådana väggar kan ifrågasättas då underhållsarbetet försvåras. Användandet av ett raster i en dagvattendamm kan minska den yta som krävs för sedimentation, vilket kan göra dammen mer yteffektiv och lättare att inkludera i en urbaniserad miljö.

Table of Contents

List of papers

Paper I Milovanović, I., Herrmann, I., G., Hedström, A., Nordqvist, K., Müller, A., Viklander, M.
Synthetic stormwater for laboratory testing of filter materials.
Submitted to Environmental Technology

Paper II Milovanović, I., Hedström, A., Herrmann, I., Viklander, M.
Performance of a Zeolite Filter treating Copper Roof Runoff
Submitted to Urban Water Journal

Paper III Milovanović, I., Vojtěch, B., Hedström, A., Herrmann, I., Picek, T., Marsalek, J. & Viklander, M. (2020).
Enhancing stormwater sediment settling at detention pond inlets by a bottom grid structure (BGS)
Water Science and Technology, 81(2), 274-282

Paper No	Development of idea	Research study design	Data collection	Data processing analysis and interpretation	Manuscript preparation (writing and data presentation)	Submission and response to reviewers
1	No contribution	No contribution	No contribution	Shared responsibility	Shared responsibility	-
2	No contribution	Shared responsibility	Responsible	Responsible	Shared responsibility	Responsible
3	No Contribution	Contributed	Shared responsibility	Shared responsibility	Shared responsibility	Shared responsibility

Responsible – developed, consulted (where needed) and implemented a plan for completion of the task.
Shared responsibility – made essential contributions towards the task completion in collaboration with other members in the research team
Contributed – worked on some aspects of the task completion
No contribution – for valid reason, has not contributed to completing the task (e.g. joining the research project after the task completion)
N/A – Not applicable

1. Introduction

Urbanisation is an ongoing phenomenon where the population shifts from rural to urban areas. In 2007, for the first time, more people lived in towns and cities than outside on the countryside, and by the middle of the 21st century, it is expected that 68% of mankind will live in urban areas (Kundu and Pandey, 2020). One of the many effects of this is the change in the hydrological cycle. The increase of paved surfaces, associated with urbanisation, leads to the increase of stormwater runoff quantity and decrease in stormwater quality.

Historically, the focus was on safely handling stormwater flow, but now more attention is paid to stormwater quality and its impact on receiving water bodies (Fletcher et al., 2015). Hvitved-Jacobsen et al. (2010) identified and divided the pollutants in stormwater into six categories: solids, metals, biodegradable organic matter, organic micropollutants, pathogenic microorganisms and nutrients. Metals are pollutants of particular interest due to their toxicity. Kayhanian et al. (2008) identified that copper and zinc are primary causes of toxicity to fish, and are often found in urban stormwater runoff. Stormwater treatment systems are used to mitigate the impact of the stormwater pollution on the environment. In this thesis, the focus will be on stormwater treatment systems based on filtration, and sedimentation of particles.

Stormwater ponds are among the most used stormwater treatment systems (Fletcher et al., 2015) and their main purpose is to allow for peak flow retention and sedimentation of solids in stormwater (Persson, 1999). However, because the removal of pollutants from stormwater takes place via the sedimentation, ponds are not efficient in treating dissolved pollutants (Buren et al., 1997). Stormwater filter systems are often used as a method of treating both particulate and dissolved pollutants from urban runoff (Hatt et al., 2008). Their downside, compared to ponds, is that they require much more frequent maintenance in order to operate efficiently. These two systems could be combined in a so-called treatment train, which utilises stormwater ponds to remove coarser sediment particles, which could clog the filter system, from the stormwater, as well as filters to remove the dissolved pollutants.

One of the limitations in selecting and designing stormwater treatment systems is the area needed for their function. Shortage of space is often cited as one of the reasons for not implementing stormwater control measures in existing urban infrastructure (Faram and Andoh, 2007; Cettner et al., 2014). Thus, it is important to consider the area-efficiency of these measures, and possible ways for how to reduce the footprint required while maintaining the necessary level of stormwater treatment.

1.1. Aim and research objectives

The aim of the thesis is to provide better knowledge about the components of the area-efficient facilities used for stormwater treatment. Particular attention was paid to the treatment of total and dissolved metals and the removal of total suspended solids.

Furthermore, this thesis contributes with a discussion related to the area-efficiency aspect of stormwater treatment systems, and their implementation in existing urban infrastructure. Besides summarising the knowledge, the licentiate thesis will offer further improvements to these facilities and provide recommendations for future application.

The research objectives of the thesis were:

1. to assess possible improvements of stormwater ponds with respect to sediment settling using a bottom grid structure,

2. to analyse the effectiveness of different filter materials to treat stormwater in laboratory and field conditions,

3. to provide comparison between different methods for stormwater treatment.

The thesis consists of three Papers (Paper I-III). In Paper I, suitability of four different filter materials was tested in laboratory setting. The filter materials in question was peat, bark, milkweed and polypropylene. Paper II focused on a full-scale application of a zeolite filter installation. Lifetime and efficiency in treating total and dissolved Cu and Zn from roof runoff of the filter were investigated. Paper III focused on laboratory experiments with bottom grid structures, more specifically, on how does the geometry of the bottom grid structure influence sediment settling.

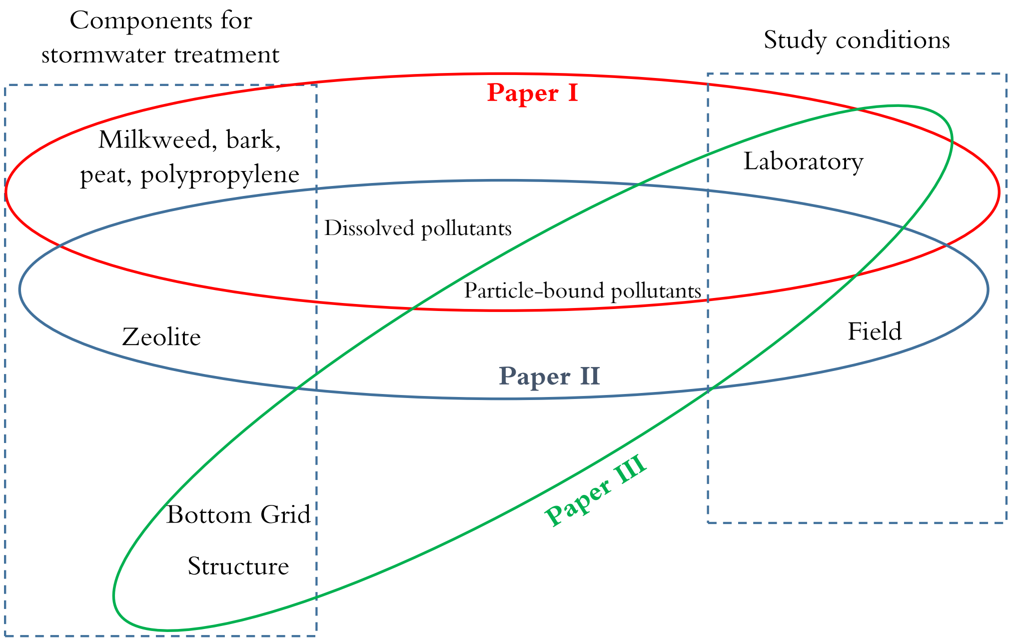


Figure 1: Synthesis of the licentiate.

Table 1: Summary of papers.

Paper	Paper I	Paper II	Paper III
Component Studied	Filter materials (Milkweed, Bark, Peat, Polypropylene)	Zeolite filter	Bottom Grid Structure
Type of pollutants	Particulate/Dissolved metals	Particulate/Dissolved metals	Solids
Water	Synthetic Stormwater	Copper roof runoff	Neralite spiked water
Setting	Laboratory	Field	Laboratory

The thesis consists of seven chapters: Chapter 1 provides the introduction of research topic and presents aims and objectives of the thesis. Chapter 2 focuses on the previous research about filtration stormwater treatment systems and stormwater ponds, including the aspect of area-effectiveness. Chapter 3 presents the methods used to in laboratory and field experiments on filter materials used in stormwater treatment systems, and a description of the hydraulic model of a bottom grid structure. Chapter 4 presents the results obtained in those experiments. Chapter 5 discusses the results obtained in laboratory and field studies by comparing it to the previous studies done, as well as discussing the area needed for the operation of stormwater treatment systems investigated. Chapter 6 presents the main conclusions drawn from the thesis, and finally, chapter 7 provides the list of references used in this thesis. The three journal papers are appended at the end.

2. Background

This section presents a brief literature overview of the methods for stormwater treatment, focusing specifically on pollutants in stormwater, stormwater ponds, stormwater filters, and area constraint issues.

2.1. Pollutants in stormwater

Stormwater runoff is considered to be a significant transport vector of pollutants, which leads to the worsening of water quality in the receiving water bodies (Lee et al., 2007; Kayhanian et al., 2008). The contaminants found in stormwater include metals, nutrients, polycyclic aromatic hydrocarbons, (PAHs), chlorinated benzenes, bacteria and microplastics, among others (Tsanis et al., 1994; Marsalek et al., 1999; Galfi et al., 2016; Müller et al., 2020). Contaminants found in urban stormwater can roughly be divided into particulate-associated pollutants and dissolved ones (Makepeace et al., 1995). The dissolved pollutants are sometimes further classified into colloid and truly dissolved pollutants (Lindfors et al., 2020). Suspended solids represent one of the main pollutants in stormwater primarily because of other pollutants (such as metals, phosphorus and organic compounds) that adsorb to particles (Herngren et al., 2005; Horowitz et al., 2008; Wakida et al., 2013; Borris et al., 2016). Metals belong to one of the most studied stormwater pollutants and the stormwater runoff is considered to be one of the main sources of metal pollution to natural water (Gnecco et al., 2005; Barbosa et al., 2012; Huber et al., 2016c). Some of the major sources of metal pollution are atmospheric deposition, vehicular transportation activities and metallic building envelopes (Gunawardena et al., 2013; Müller et al., 2019). Some of the metals that are of the greatest concern include copper (Cu), zinc (Zn), lead (Pb), cadmium (Cd), chromium (Cr) and nickel (Ni) (Makepeace et al., 1995). Although there is an established correlation between solids in stormwater and metal concentration (Beck and Birch, 2012; Djukić et al., 2016), still a considerable fraction of metals in stormwater can be found in dissolved phase, which treatment systems that focus on sedimentation or mechanical filtration of particles cannot address.

2.2. Stormwater treatment systems

Stormwater treatment systems are used in order to manage the stormwater pollution described above, and thus relieving the pressure on the receiving water bodies. Stormwater treatment facilities can either be located close to the source of pollutants or as an end-of-pipe treatment Measures to decrease the pollutants in stormwater can be divided into structural, where a system is implemented to treat the stormwater, or non-structural, aiming at reducing the pollution by controlling the source through various legal measures such as town planning controls, pollution prevention procedures, education and regulatory controls (Taylor and Wong, 2002). Some of the structural stormwater methods are presented in Table 2. In this thesis, focus is on stormwater ponds and filtering systems.

Table 2: Structural stormwater management and treatment methods (adapted from Georgia Stormwater Management Manual, table 3.1.1 (2015)).

Structural measure	Description
Stormwater Pond	Constructed retention basins that have a permanent pool of water.
Stormwater Wetlands	Constructed wetland systems that consist of combination of marsh areas, open waters and semi-wet areas
Enhanced Swales	Vegetated open channels that are designed to capture and treat stormwater in dry or wet cells formed by dams
Bioretention Systems	Shallow stormwater basins or landscaped areas which utilize engineered soils and vegetation to capture and treat stormwater runoff by filtration. Runoff is then either returned to the pipe system, or percolated through the ground.
Filtering systems	Filtering systems that use material that is able to provide enhanced removal of contaminants.

2.2.1. Treatment in stormwater ponds

Stormwater ponds are one of the most encountered stormwater control measures (Fletcher et al., 2015). Ponds are end-of pipe stormwater treatment system, meaning that they are located at the down-stream area of the catchment. Main function of earlier stormwater ponds was their ability to prevent flooding by reducing the flow peak of the from intensive rains, but later pond systems also have been constructed to enhance sedimentation, therefore improving stormwater quality. Most important aspects of the stormwater pond design is the settling area and pond shape (Persson, 1999; Al-Rubaei et al., 2017). The land area recommended for a stormwater pond (percentage of impervious watershed) has been estimated to approximately 1–2 % of the catchment area, which can be problematic to achieve, given densely built urban areas (Persson et al., 1999; Johnson, 2007). In a study where eight different ponds in Sweden were evaluated with respect to pollutant removal efficiency, Persson and Pettersson (2009) found that stormwater ponds were able to remove 38 – 83 % of the TSS. The metal removal rates varied, both between the metals and the different ponds (6-85%). It was shown that removal was higher for the ponds that had a specific pond area above 200 m^2 ha^{-1} (corresponding to 2 % of the catchment area).

Besides sediment settling and flood mitigation, ponds offer a place for recreation, , carbon sequestration, space for wildlife and other ecosystem services, which are also a reason for including ponds in urban infrastructure (Lawrence et al., 1996; Moore and Hunt, 2012).

As with any other stormwater control system, in order to ensure that stormwater ponds function properly, it is important to maintain the pond systems at required intervals (Erickson et al., 2013). One of the main reasons for the decrease of hydraulic and treatment performance of stormwater ponds is the accumulation of sediment, which effectively reduces the storage volume of the stormwater pond (Al-Rubaei, 2016). This leads to reduced retention capacity, thus it is important to remove accumulated sediment to ensure proper operation of stormwater ponds. In a study conducted in 2017, it was found that out of 25 ponds surveyed in Sweden, 54% required at least minor maintenance (Al-Rubaei et al., 2017). In order to ensure that ponds are accessible for required maintenance, an easy access to the ponds is required. This is not always the case, as the same survey by Al-Rubaei et al. (2017) found that some ponds were designed in a way that made access for inspection and maintenance difficult.

One way to reduce the required area and increase the sedimentation effectiveness of ponds is an implementation of additional devices in stormwater ponds. The most common structure associated with stormwater ponds are forebays. Forebay is a pool located near the inlet structure which is designed to both increase sedimentation efficiency of a stormwater pond, and to allow for easier access for maintenance (Johnson, 2007; Blecken et al., 2017). Another way to provide enhanced pollution removal capabilities of pond systems can be to introduce floating wetlands, a system of floating plants where the water flows through the roots, which capture finer particles that would not settle in the pond (Headley and Tanner, 2006; Johnson, 2007)

A novel sedimentation device, a bottom grid structure (BGS), has been suggested to increase the particle sedimentation in stormwater ponds by introducing vertical vortexes in the flow as the water passes over the structure thus guiding the sediments into the cells (He and Marsalek, 2014; He et al., 2014). Given that the concept of the bottom grid structure is relatively new, only two studies have been carried to investigate its effectiveness in enhancing sedimentation in stormwater ponds (He and Marsalek, 2014; He et al., 2014). The complex structure of the BGS and it's interaction with the sediment in the flow means that numerical modelling of that behaviour is difficult, so previous studies have been carried out in hydraulic laboratory and in field conditions. In a hydraulic laboratory experiment, He and Marsalek (2014) showed that the enhancement of sedimentation rate was proportional to the flow speed along the top of the BGS, before stabilising at a maximum value. The BGS was found to increase the sedimentation for 10 – 30%, compared to the experiments with bare bottom, and with the devices performance increasing as the flow increased. It was hypothesised that the increase was both due to the function of vortex generation, and shielding settled sediment from resuspension that would have been caused by more powerful flow. In order to further assess the impact of a BGS on sedimentation in stormwater ponds and to be able to suggest the design of a BGS, more detailed investigations into effects of cell design on the sedimentation effectiveness and reduction of sediment resuspension, as well as full-scale experiments of BGS performance would be required (He and Marsalek, 2014; He et al., 2014).

2.2.2. Filter materials in stormwater treatment

Filter systems are used for stormwater treatment before the stormwater release in receiving water body (Kandra et al., 2014).The level of treatment of pollutants depends both on the characteristics of the filter system, the filter material and the targeted pollutants.

Various filter systems can be applied to treat both particulate-bound and dissolved pollutants from the stormwater. The filter systems include small filter units such as in gullypot filters, but metal-adsorbing filter materials can also be added to the soil medium of bioretention systems to increase the retention of metals (Søberg et al., 2019)

Filter effectiveness has been found to depend on numerous parameters. In a batch experiment where 10 different filter material for bioretention systems were tested, the authors found that material with higher pH, lower organic content and higher specific surface tended to increase the treatment of dissolved metals (Søberg et al., 2019). The filter performance also depends on factors such as filter design and inflow concentration. Färm (2002) found that the treatment of metals from a synthetic stormwater depended on the hydraulic load, where the treatment effectiveness increased with decreased hydraulic load in the experiments. Similar results were found by Brown et al. (2000), who found that the metal treatment efficiency of peat correlated to the loading rates, where the treatment level decreased with higher loading rates. Removal of solids from the stormwater also depends on physical characteristics of both pollutants and filter material such as pollutant particle size and material pore size (Clark and Pitt, 2012). This also means that the filtration ability of filter material will reduce with the operation due to clogging. Therefore proper maintenance of filter systems is required.

The filter materials used in stormwater treatment facilities can roughly be divided into organic materials such as compost, bark, peat and biochar and inorganic materials such as sand, minerals and iron-based materials (Okaikue-Woodi et al., 2020). The processes that lead to dissolved metal removal from stormwater are adsorption, precipitation, ion exchange and chemisorption (Reddy et al., 2014). Biochar has been studied and suggested as a filter material for stormwater treatment due to its low cost, and effectiveness in removing both organic pollutants from stormwater via diffusion, and metals via complexation (Okaikue-Woodi et al., 2020). Bark has been a popular filter material due to its availability and the capacity to treat pollutants from the stormwater. From the tree biomass, bark has shown the highest capacity of metal sorption (Şen et al., 2015). Metal removal efficiency of bark depends on the tree species, pre-treatment of bark, pH of influent, and influent concentration, but the reported rate of removal varied between 50 and 99% (Gaballah and Kilbertus, 1998). Peat is another organic filter material which is created by the decomposition of vegetation in marches, bogs or swamps (Spedding, 1988). Peat has been utilised as a filter material in stormwater and waste water treatment due to its capacity to remove metals from the influent (Brown et al., 2000).

One group of filter materials known for their cation exchange properties are zeolites. Zeolites are porous natural minerals mined throughout the world and they have been used in stormwater and wastewater treatment systems for treating pollutants such as metals, ammonium and bacteria (Hedström, 2001; Pitcher et al., 2004; Gray, 2012; Li et al., 2016). Both natural zeolites such as chabasite, mordenite, clinoptilolite and artificial zeolites have been studied in a number of laboratory experiments to determine their efficiency in removing pollutants from stormwater. Sometimes the zeolites are pre-treated with a sodium chloride solution to increase the cation exchange capacity(Li et al., 2011). Ion exchange ability of zeolites have been previously confirmed in numerous studies (Ćurković et al., 1997; Doula et al., 2002; Pitcher et al., 2004; Athanasiadis et al., 2007). In a study by Pitcher et al., (2004), a synthetic zeolite MAP and modernite were tested for their ability to treat dissolved metals from the stormwater,. The results of the study showed that zeolites were able to remove more than 42-90 % of studied metals (Zn, Cu, Pb, Cd). Färm (2002) found that a mixture of a silicate rock and zeolite was able to reduce concentration of metals in water, but that level of reduction depended significantly on the hydraulic loading rate. In a field experiment carried out by Athanasiadis et al. (2007), an infiltration system including clinoptilolite as a filter materialwas able to reduce the copper from a copper roof runoff up to 96 %, during the 30–month long investigation period. The concentration of copper decreased in the percolated water to a value lower than the discharge level set by German Federal Soil Protection Act and Ordinance.

The filter effectiveness in treating pollutants from stormwater has been determined in controlled environment (batch tests and column experiments) or in field setups. In the experiments, researchers have used either real stormwater (Sansalone, 1999; Barrett, 2010), collected in situ, or made an approximation of the stormwater by creating synthetic stormwater (Blecken *et al.*, 2009; Reddy *et al.*, 2014; Björklund and Li, 2015 among others). There are various levels of approximations of real stormwater. One example is when the synthetic stormwater based on a single pollutant, such as copper, zinc, or another metal (Huber et al., 2016a; Norman, 2018). This approach is suitable when testing how a specific pollutant is treated by a filter material. Next level in approximation of stormwater is a combination of metals that is dissolved in water, which helps to illuminate how different metals compete for adsorption sites on filter material (Sounthararajah et al., 2017; Haile and Fuerhacker, 2018). However, there are fewer studies that tried to mimic stormwater by including sediments in synthetic stormwater (Søberg *et al.*, 2017; Ng *et al.*, 2018).

Due to differences in conditions in the laboratory and field, pilot scale tests are often required to validate findings from batch and column experiments. However, field studies are fewer in number compared to those carried out in laboratory. In a critical review article on contaminant removal by filter materials (Okaikue-Woodi et al., 2020), only seven out of 38 filter material studies, had been evaluated in field.

2.3. Area limitations in urban stormwater treatment

In 2015, UN adopted the Sustainable Development Goals (SDGs) as a "blueprint to achieve a better and more sustainable future for all" (United Nations, 2015). The SDGs serve to promote multidisciplinary cooperation and a holistic approach to addressing global challenges. Since the need for stormwater treatment facilities is greatest in highly populated areas, it is not possible to plan their construction without a holistic approach. Links between water and the SDGs have been previously established. Stormwater treatment facilities address several SDGs, specifically SDG 6 (clean water and sanitation), SDG 11 (sustainable cities and communities), SDG 9 (industry, innovation and infrastructure), SDG 13 (climate action), SDG 14 (life below the water), SDG 15 (life on land), SDG 12 (responsible consumption and production), SDG 17 (partnership for the goals), SDG 3 (good health and wellbeing and SDG 8 (Decent work and economic growth) (Blecken, 2018). SDG 11 – Sustainable cities and communities aims to make cities inclusive, safe, resilient and sustainable. However, interviews with water professionals in Sweden revealed that they were doubtful whether it would be possible to utilise sustainable stormwater ideas due to problems concerning the absence of available land and the cost of implementing such programs (Cettner et al., 2014).

Recently, more effort has been put to enumerate the economic value of stormwater facilities. Numerous tools and models are available to assess the benefits from green stormwater infrastructure (Joksimovic and Alam, 2014; Eckart et al., 2018; Hoang et al., 2018; Bixler et al., 2020; Hamann et al., 2020). B£ST (Benefits Estimation Tool), for example, divides the benefits of blue green infrastructure in several categories such as Air Quality, Amenity, Biodiversity, Flooding, Groundwater recharge, Recreation, Tourism, Water quality and Carbon reduction and sequestration, among others (Ashley et al., 2018).

With the stated concerns of the absences of available land, and the fact that the stormwater systems are needed the most in the areas that are mostly populated, it becomes clear that the improvement of area-efficiency of stormwater treatment systems would make them easier to integrate in the existing city infrastructure.

3. Methods

In order to answer the research questions, two general approaches were taken, laboratory experiments (Papers I and III) and a field experiment (Paper II). The laboratory experiments with synthetic stormwater (Paper I) were carried out in the laboratory of Luleå University of Technology during the period June – July 2016 and January – February 2017, the field-testing of a zeolite filter (Paper II) was conducted during the period March 2018 – March 2020 in Stockholm. The laboratory experiment on a bottom grid structure (Paper III) was carried out at the Czech Technical University in Prague in 2018. First, study design is explained in Section 3.1, followed by analytical procedures used for the analysis (Section 3.2), and then by data analysis in Section 3.3.

3.1. Study design

3.1.1. Filter column experiment (Paper I)

Eight filter columns with an inner diameter of 74 mm (Figure 2) were filled with four different filter materials (milkweed (M), bark (B), peat (P) and polypropylene (PP) Figure 3). The volume of filter material added to each column was approximately 0.3 L. In order to ensure an even flow distribution through the cross section, glass beads were placed above and below the filter material. In order to reduce the dispersion of the filter material, a piece of geotextile was placed between the glass beads and the filter material. This type of geotextile is often used in full-scale applications of stormwater gully pot filters (Paul and Tota-Maharaj, 2015).

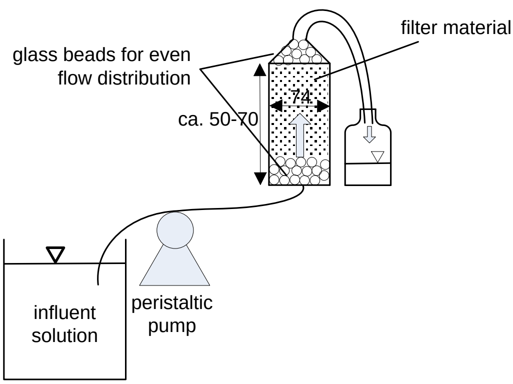


Figure 2: Experimental set-up for the column experiment (dimensions in mm).

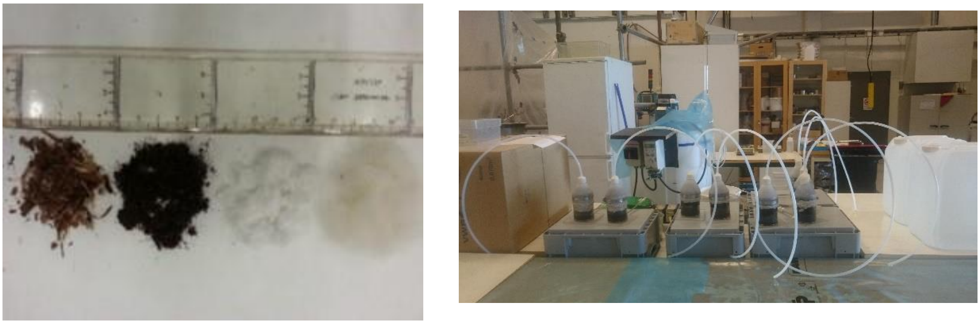


Figure 3: Filter materials used in the column experiment (from left to right), Bark, Peat, Milkweed and Airblown Polypropylene (left) and column experiment set-up (right).

In order to achieve a similar filter bed volume in each column, different masses of filter material were used: 9 g and 13 g of milkweed, 50 g and 45 g of bark, 51 g and 64 g of peat, and 22 g and 24 g of polypropylene, for the duplicate columns. The bark used was the commercial product Zugol (Zugol, n.d.), which consisted of 85–90% pine bark and 10-15% wood fibre with a density of 0.25 kg/dm^3. Milkweed, peat and polypropylene were acquired through commercial means. The choice to include these filter materials was driven by the desire to test proven stormwater treatment materials (bark, peat) with emerging materials (milkweed, polypropylene) in a complex synthetic stormwater solution. Bark and peat have been proven to be a low-cost sorption material in previous studies which focused on their applicability in stormwater treatment (Färm, 2002; Al-Faqih et al., 2008; Kalmykova et al., 2009; Björklund and Li, 2015; Ilyas and Muthanna, 2017). Milkweed and polypropylene have previously been used for the treatment of oil from water (Praba Karana et al., 2011; Li et al., 2012), and polypropylene has been a subject of studies in stormwater treatment experiments (Lee et al., 2005).

The first phase of the experiment consisted of three consecutive days of loading, after which the columns were left for four days to rest. After three weeks, the experimental set-up was left to rest for six months. After that, the second phase of loading commenced and continued for 5 weeks with the same loading and resting periods as during the first phase of the study. During the loading, synthetic stormwater was pumped from the chamber with influent solution by peristaltic pumps with plastic tubing through the columns in an up-flow mode. During the first phase, the flow was set at 0.005–0.008 L min^{-1}, which corresponded to 0.07–0.11 m^3 m^{-2} h^{-1}. During the second phase, the flow was increased to 0.012–0.014 L min^{-1}, which corresponded to 0.16–0.20 m^3 m^{-2}h^{-1}.

Samples from the influent chamber were taken from each daily batch at the beginning of a run and at the end of it, the day after. The effluent water from each column was collected in separate effluent chambers. Samples from the effluent chambers were taken each day at the end of the loading phase. Samples were analysed on the total and dissolved metal content, TSS, and pH, and during the first week of the experiments, particle size distribution was also determined.

The influent solution was a synthetic stormwater prepared to simulate heavily polluted stormwater runoff. The synthetic stormwater was prepared by adding metal solutions to tap water, alongside oil and collected sediment from an underground sedimentation basin in Stockholm, Sweden. Achieved concentrations for TSS, total and dissolved metals are presented in Table 3.

Table 3: Achieved chemical and physical characteristics of the influent batches of synthetic stormwater used for filter column experiment with standard deviations in brackets (n=24).

Parameter	Total concentration	Dissolved concentration
TSS (mg L^{-1})	140 (18)	-
Cd (μg L^{-1})	0.96 (0.1)	0.61 (0.1)
Cr (μg L^{-1})	14.5 (1.2)	5.4 (0.4)
Cu (μg L^{-1})	117 (20)	17.8 (9.1)
Ni (μg L^{-1})	11.6 (1.0)	7.08 (0.5)
Pb (μg L^{-1})	23.9 (2.0)	4.8 (2.1)
Zn (μg L^{-1})	374 (21)	161 (16)

3.1.2. Synthetic stormwater ageing (Paper I)

The effect of ageing on the quality of the synthetic stormwater was evaluated in the short term (one-day experiments) and the long term (eleven days). The purpose was to demonstrate how the synthetic stormwater quality, especially the dissolved metal concentrations, changed after the preparation of synthetic stormwater. During the short-term experiment, samples of the influent were taken at 15, 100, and 1200 minutes after the preparation of synthetic stormwater and they were analysed for metals (described in Section 3.2). Samples from the synthetic stormwater were also taken at the beginning and the end of the loading phases, as described in 3.1.1. For the eleven-day experiments, samples were taken on day 2, day 3, day 5, day 8 and day 11, and they were analysed for TOC, DOC, TSS, pH, Turbidity, and metals further described in Section 3.2.

3.1.3. Full-scale zeolite filter study (Paper II)

In 2017, a new copper roof was installed as a part of the renovation of Stockholm National Museum. In order to treat the runoff from the roof, a commercial stormwater filter system with zeolite as filter material was installed in the garden of the museum. Due to the size of the roof, a filter system including five units in parallel was installed. The filter system was put into operation in July 2018. The runoff from the roof, as well as the runoff from a small part of the park in which the filter unit was located, was first collected in a collection tank and pumped into a storage tank. From the storage tank, the stormwater was transferred to an influent sampling tank (Figure 2). From this tank, the water was pumped through the filter units, in an up-flow mode. The pump was operated automatically; starting once a certain depth of water level had been reached in the storage tank, and stopped once the water level fell to the stop point. This pump operation pattern was reflected in the effluent flow, which showed varying flow intensities and peaks during the sampling occasions. After the filter units, the water was transported by gravity through pipes to an outlet manhole for sampling (Figure 4).

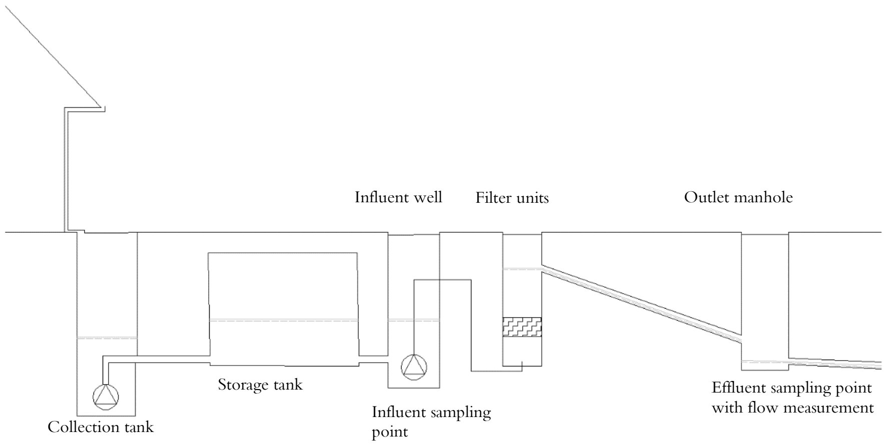


Figure 4: The zeolite filter installation for treatment of run-off with high copper concentrations.

The five filter units were operating under the loading rate of 6.7 $m^3\ m^{-2}h^{-1}$. Each unit had a diameter of 1 m, height of 0.5 m, and a filter element weighing 66 kg (3P Technik, 2020). The zeolite used as the filter material was a synthetic zeolite with the chemical formula $Na_2O{\star}Al_2O_3{\star}2.4SiO_2{\star}nH_2O$ (Gmbh, 2010).

In order to assess the treatment capacity of the zeolite filter units, seven sampling occasions were carried out between December 2018 and March 2020. Samples were taken from the influent (2–7 samples), and effluent (3–6 samples), see Table 4.

The sampling was carried out simultaneously at the influent well and at the outlet manhole. The first inflow sample was taken while the pump in the influent well was being turned on, and the first outlet sample was taken when the first flow was detected in the outlet manhole. Originally, the plan was to take automated flow-proportional samples, but the variability of the flow proved to be too great to allow for that.

At both the inlet and the outlet sampling points, samples were taken time-proportionally. At the outlet, additional samples were taken to cover the peaks of the flow described above. Influent samples were taken using a Rüttner water sampler. At the outlet, sampling was performed manually, by placing a container underneath the outlet pipe to collect the water, at pre-determined time intervals. The intervals of the time-proportional sampling were determined in order to obtain the planned number of time-proportional samples (5) for each occasion. On the first two sampling occasions, a limited volume of water was stored in the storage tank on the sampling day, and therefore, fewer time-proportional samples were taken. After each sampling occasion, the samples were filtered in situ through 0.45 µm filters in order to obtain samples for the dissolved metal and dissolved organic carbon analyses.

Table 4: Number of samples taken from the influent and effluent of the zeolite filter on each sampling occasion.

	Sampling occasion						
	1	2	3	4	5	6	7
Date	03-12-18	06-03-19	02-05-19	27-06-19	09-09-19	07-11-19	19-03-20
Influent samples	2	4	7	5	5	5	5
Effluent samples	4	3	6	5	5	5	5
Peak effluent samples	0	0	3	3	4	2	3
Sampling duration (minutes)	45	45	60	40	40	40	40

3.1.4. Physical hydraulic model for bottom grid structure (Paper III)

In order to assess the effect of different BGS cell geometries on sediment entrapment, a scaled hydraulic model was constructed. The model was built in the Hydraulics Laboratory of the Czech Technical University in Prague. The water supply system for the hydraulic laboratory was connected to an inlet tank. A synthetic sediment solution was added to the inlet pipe, using a peristaltic pump at different rates, to ensure a constant inflow sediment concentration of 100 mg L^{-1} in order to simulate stormwater sediment. The PVC powder Neralite was used with a specific gravity of 1.32 and d_{50} of 143 µm. In order to ensure that the sediment did not settle on the bottom of the chamber, sediment solution was constantly stirred. The chamber holding the synthetic sediment solution was placed on electronic weighing scales that continuously measured the dose of sediment.

The BGS model (Figures 3, 4 and 5) with a width of 0.5 m and a length of 1 m was placed in an existing flume. Since the diameter of the pipe feeding the water with the sediment to the BGS model was 100 mm, and the width of the BGS model was 500 mm, there was a noticeable effect of expansion. To test the effect of expansion on the sediment transport, two set-ups were used. The first set-up was with a sudden expansion from the inlet pipe to the flume, and the second with a channel diffuser with length of 0.5 m (Figure 4). A multiple slots weir controlled the water depth in the BGS tank where it was possible to fit or remove metal planks to ensure a constant water level at different flow rates.

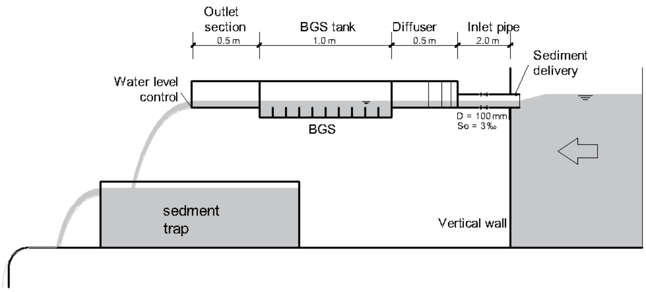


Figure 5: Cross section of the BGS model.

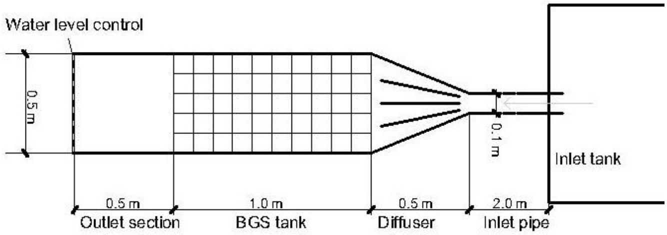


Figure 6: Plan view of the BGS model.

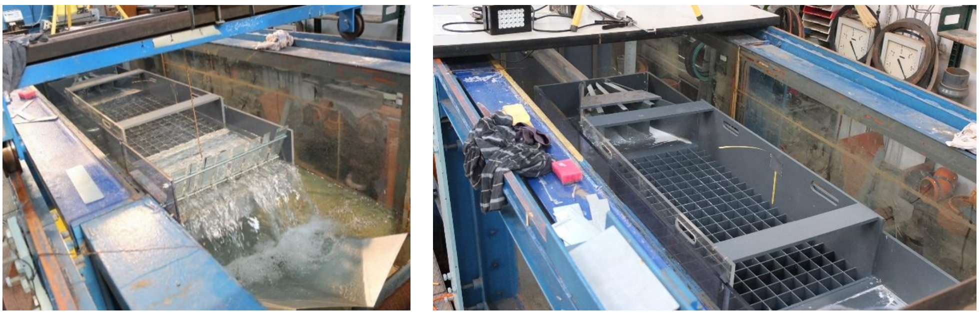
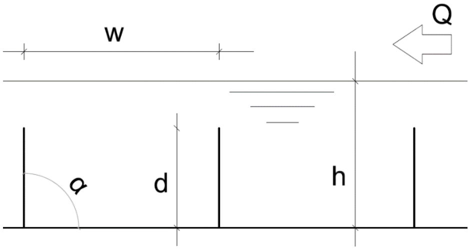


Figure 7: BGS model during one of the runs (left) and during a break in between the runs (right).

The discharge from the water supply system through the BGS model was measured using an electromagnetic flowmeter, Krohne Waterflux 300, with an accuracy of 1% of the measured value. The flow was measured with another flow measurement device – a Thompson weir with a level meter (accuracy 2-5%). To avoid sediment settling in the inlet pipe and also to ensure a subcritical flow regime throughout the BGS tank, flow rates from 1 to 4 L s^{-1} were selected. The factors evaluated for the BGS structure were cell widths, depths, and cross-wall angles (Figure 8).



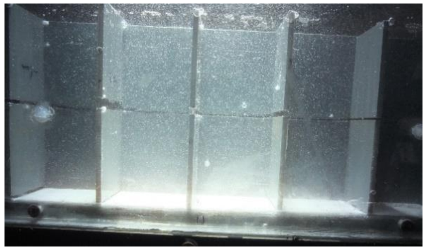


Figure 8: BGS cells and their features varied in the experiments, cell width (w), cell height (d) cross-wall angle (α) and flow depth (h) (left) and a photo of the cells during a model run (right).

In total, 24 runs of the BGS modelling experiment were carried out for selected flow conditions (Table 5).

In order to test whether the differences obtained during the BGS runs with different parameters were due to uncertainty of measurements and variability of runs, run 23 (Table 5) was replicated five times with the following settings: d = 10cm, w= 5 cm, α = 90°, and Q = 3 L s^{-1}.

Table 5: Settings of the investigated factors in the BGS experiments.

Run	Inlet transition	Bottom arrange-ment[a]	Cell width w [cm]	Cell depth d [cm]	Cross-wall angle α [°]	Flow-rate Q [l/s]	Flow velocity V [m/s]	Flow depth h [cm]
1	sudden[b]	BGS	5	5	90	2	0.08	5
2	sudden	BGS	5	5	90	1	0.04	5
3	sudden	BGS	5	5	90	1	0.03	7.5
4	sudden	BGS	5	5	90	2	0.05	7.5
5	sudden	BGS	5	5	90	4	0.11	7.5
6	sudden	smooth	-	-	-	1	0.03	7.5
7	sudden	smooth	-	-	-	2	0.05	7.5
8	diffuser[c]	smooth	-	-	-	2	0.05	7.5
9	diffuser	smooth	-	-	-	3	0.08	7.5
10	diffuser	smooth	-	-	-	4	0.11	7.5
11	diffuser	BGS	5	5	90	2	0.05	7.5
12	diffuser	BGS	5	5	90	3	0.08	7.5
13	diffuser	BGS	5	5	90	4	0.11	7.5
14	diffuser	BGS	5	10	90	2	0.05	7.5
15	diffuser	BGS	5	10	90	3	0.08	7.5
16	diffuser	BGS	5	10	90	4	0.11	7.5
17	diffuser	BGS	5	10	90	1	0.03	7.5
18	diffuser	BGS	10	10	90	2	0.05	7.5
19	diffuser	BGS	10	10	90	3	0.08	7.5
20	diffuser	BGS	10	10	90	4	0.11	7.5
21	diffuser	BGS	5	10	60	2	0.05	7.5
22	diffuser	BGS	5	10	120	2	0.05	7.5
23	diffuser	BGS	5	10	120	3	0.08	7.5
24	diffuser	BGS	5	10	60	3	0.08	7.5

[a] Bottom of the BGS tank; [b]Sudden expansion – the inlet pipe was connected directly to the BGS tank; [c]Diffuser expansion

3.2. Laboratory analyses

In both Paper I and Paper II, total suspended solids were analysed by filtration through glass fibre filters according to the relevant standard (European Committee for Standardisation, 2005). Turbidity was analysed by the HACH 2100N Turbidity meter in the laboratory. pH values were measured in situ for Paper I and II using the WTW pH3110 set, which was calibrated at the start of each sampling occasion. Particle size distribution (PSD) was analysed with a laser scattering particle size distribution analyser, Horiba LA-960. The metal analyses were carried out at an accredited laboratory according to the Swedish Standards Institute, (2009). Total metal concentrations were determined using ICP-SFMS according to SS-EN ISO 11885:2009 (Swedish Standards Institute, 2009). The reporting limit for copper was 1 µg L^{-1}. Müller et al. (2019) observed the copper concentration in copper roof runoff to be approximately 3000 µg/l, indicating that the reporting limit above was sufficient. Concentrations of dissolved metals were determined using ICP -SFMS according to SS-EN ISO 17294-2:2016. Samples for dissolved metals were filtered through a 0.45 µm filter prior to analyses. Metals included in the analyses were (Fe, Mg, Na, Al, As, Ba, Cd, Co, Cr, Cu, Hg, Mn, Mo, Ni, Pb, V and Zn)

The uncertainty of each analysis was reported by the laboratory and it takes into account instrument instability, uncertainty in balances, volumetric equipment, and errors in the calibration standards (Joint Committee For Guides In Metrology, 2008). When calculating average uncertainty for a sampling occasion in the zeolite filter study, mean concentrations and their uncertainties were obtained using Equations 1 and 2.

$$\overline{c_i} = \frac{\sum_1^n c_{i,n}}{n} \qquad (eq\ 1)$$

$$\overline{u_i} = \frac{\sqrt{\sum_1^n u_{i,n}^2}}{n} \qquad (eq\ 2)$$

Where, $\overline{c_i}$ was the mean concentration for the sampling occasion *i* in which a total number of *n* samples was taken, and $\overline{u_i}$ was the uncertainty for the same sampling occasion. The uncertainty was then compared to the variability of samples taken, and a larger one was used.

At the end of a BGS model run, sediment was vacuumed from the BGS cells using a peristaltic pump and placed in a bucket that was allowed to dry in the oven (at 50 °C). After the drying, the sediment weight was measured and the trapping efficiency was calculated by comparing the mass of the sediment fed into the model with the mass of dry sediment.

3.3. Data analyses

For the filter systems (Paper II), the performance was not only dependant on the amount of time they were in operation, but also perhaps more related to the amount of water treated by the filter, expressed in bed volumes treated. That was calculated according to Equation 3.

$$n_0 = \frac{V_w}{V_f} \qquad (eq\ 3)$$

where, n_0 was the number of bed volumes treated, V_w was the volume of the treated water at different sampling occasions and V_f was the volume of the filter. By estimating the treated bed volumes, as well as the level of treatment on each sampling occasion, it was possible to determine the filter ageing effects on the treatment efficiency.

The volume of treated water was determined by analysing daily precipitation data from a Swedish Meteorological and Hydrological Institute weather station (station number 98210), which was located approximately 2 km from the field site.

In order to evaluate whether there was a significant difference between the copper concentrations of the time-proportional samples and the peak flow samples in the effluent of the zeolite filter (Paper II), a Welch's test was used (Ruxton, 2006) where the null hypothesis was that there was no significant difference between the peak samples and time-proportional samples.

In the instances where the concentrations of total and dissolved metals and TSS were under the limit of detection (LOD) (Papers I and II), an additional analysis was conducted in order to identify the frequency of this occurrence in relation to the total number of samples. In cases where the number of samples under LOD represented less than 15% of the total number, half of the LOD was set as the value for those samples under LOD, following the advice of the EPA (US EPA, 2006). When the number of samples under LOD exceeded 15% of the total number of samples, the value of sample was set at the LOD, as this reflects the "worst case scenario" in filter treatment performance.

In order to assess the performance of the filter columns in treating dissolved metals from synthetic stormwater, it was necessary to determine the influent concentration of dissolved metals. This was challenging because the sediment added to the synthetic stormwater quickly adsorbed the dissolved metals. To take this into account, the following calculation procedure was suggested and used. Time-weighted average of the dissolved metals in the influent was calculated using the data from the short-term ageing experiment described in 3.1. Time-weighted average concentrations were calculated according to Equation 4 and essentially represented the integral of the curve representing the dissolved metal concentration in the influent throughout the duration of each loading phase.

$$C_{avg} = \frac{\sum_{i=1}^{3} \frac{(C_i + C_{i-1})}{2}(t_i - t_{i-1})}{T} \qquad (eq\ 4)$$

where C_{avg} was the average influent concentration, C_i was the concentration of the element in question at the time step t_i and the total experiment time was T. The proposed coefficient that would account for the change in dissolved concentration in synthetic stormwater was obtained using Equation 5.

$$k_{c,m} = \frac{\sum_{j=1}^{3} \left(\frac{C_{avg,j}}{C_{0,j}} \right)}{3} \qquad (eq\ 5)$$

where $k_{c,m}$ stands for the adjustment coefficient for metal m, $C_{0,j}$ stands for initial concentration for the experiment day *j*, and $C_{avg,j}$ for the time-weighted average concentration for the experiment day *j*, obtained by Equation 4. Inflow concentrations of dissolved metals for the column experiment were then determined by multiplying the initial concentrations by the corresponding coefficient $k_{c,m}$. Metal removal efficiencies were then calculated by comparing the time-weighted average synthetic stormwater concentrations and effluent concentrations throughout the experiment (Equation 6).

$$R_m = \frac{\sum_{i=1}^{n} \left(\frac{({C_{in,i}}^m * {k_c}^m) - {C_{e,i}}^m}{({C_{in,i}}^m * {k_c}^m)} \right) \times 100}{n} \ [\%] \qquad (eq\ 6)$$

Where, R_m represents the removal efficiency for metal *m*, ${C_{in,i}}^m$ is the initial influent concentration for the metal *m*, and batch *i*, ${k_c}^m$ is the adjustment coefficient for the metal *m*, and $C_{e,i}$ is the effluent concentration of the metal *m* for batch *i*, and n is the total number of experiment days.

4. Results

This section will present the results from the laboratory and field experiments in the thesis and will focus on total and dissolved Cu, and Zn effluent concentrations, total and dissolved Cu and Zn treatment, and BGS model results, as well as ageing experiments with synthetic stormwater.

4.1. Ageing of synthetic stormwater

During the 11-day ageing experiment of the synthetic stormwater (Figure 9) a peak was observed at day 8.

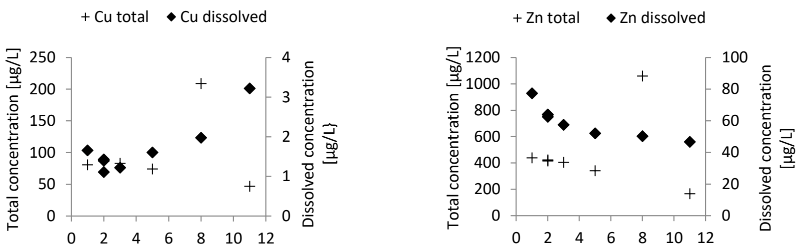


Figure 9: Concentrations of total and dissolved Cu (left) and Zn (right) in the synthetic stormwater during the 11–day ageing experiment.

There was a decrease in the dissolved concentration of Zn during the experiment period, where the concentration decreased from 77 to 46 µg L^{-1}. For dissolved Cu it seems that there was an increase of the dissolved concentration over the duration of the experiment, although low dissolved concentrations, close to LOD, might introduce some uncertainty into the results. For both total Cu and total Zn, there was a peak on day 8 of testing which may be explained by the sampling on that day, which in turn, may be explained by the accumulation of sediment near the outlet of the tank with synthetic stormwater. Low dissolved Cu (and to lesser extent Zn) concentrations may be explained by the fast decrease of the dissolved concentration noticed in the short-term ageing experiment (Figure 10).

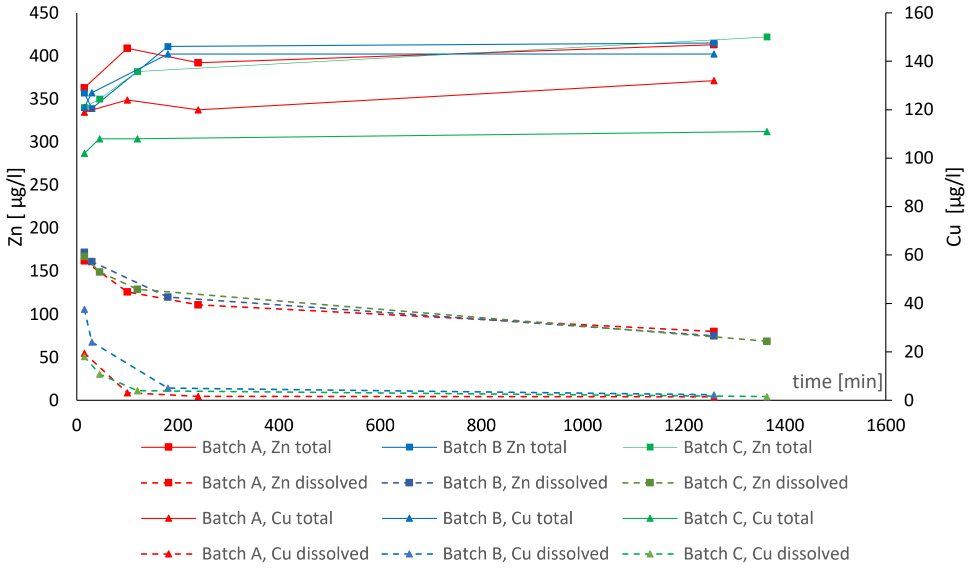


Figure 10: Change in dissolved Cu and Zn concentration in the inflow batches during the short-term ageing experiment.

Dissolved concentration rapidly decreased in the first 200 minutes following the mixing. In the case of dissolved Cu, on average there was an 85% decrease in the first 200 minutes after the batch was prepared. Concentration of dissolved Zn was more stable, and it decreased by 28% in the first 200 minutes. In order to adjust for this decrease in the dissolved metal concentration, the coefficient described in 3.1.2 was applied when calculating treatment efficiency for the column experiment.

4.2. Total and dissolved metal concentrations in the influents and effluents of the studied filters

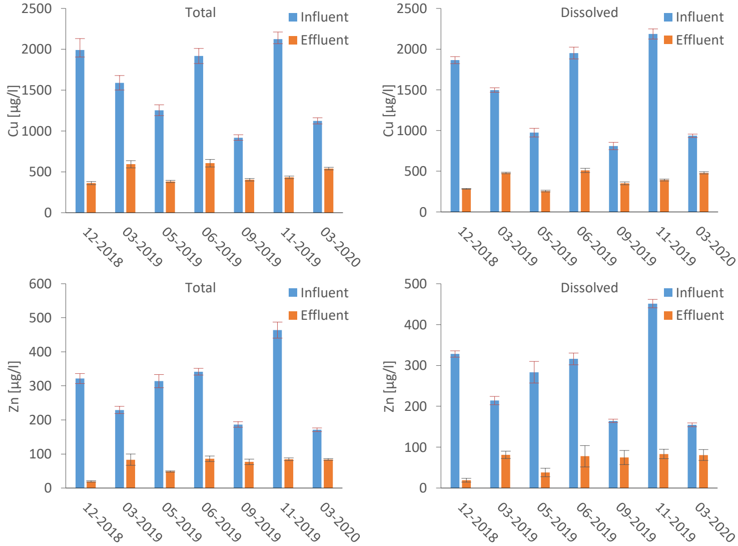


Figure 11: Average total and dissolved concentration of Cu and Zn in the influent and effluent of the zeolite filter for seven sampling occasions.

The concentrations of Cu in the influent of the zeolite filter ranged from 916 µg L^{-1} to 2124 µg L^{-1} with 93% of it being dissolved (Figure 11). The effluent concentrations ranged from 360 µg L^{-1} to 600 µg L^{-1} and the dissolved fraction accounted for 80% of total Cu (Figure 11). As for Zn, concentrations in the influent ranged from 190 µg L^{-1} to 460 µg L^{-1}, and the effluent values were 20 µg L^{-1} to 80 µg L^{-1}. The dissolved phase accounted for 93% of both the influent and the effluent.

Concentrations of total and dissolved metals analysed in samples taken from the synthetic stormwater batch and the effluent from the columns containing milkweed, bark, peat and polypropylene are presented in Figure 12.

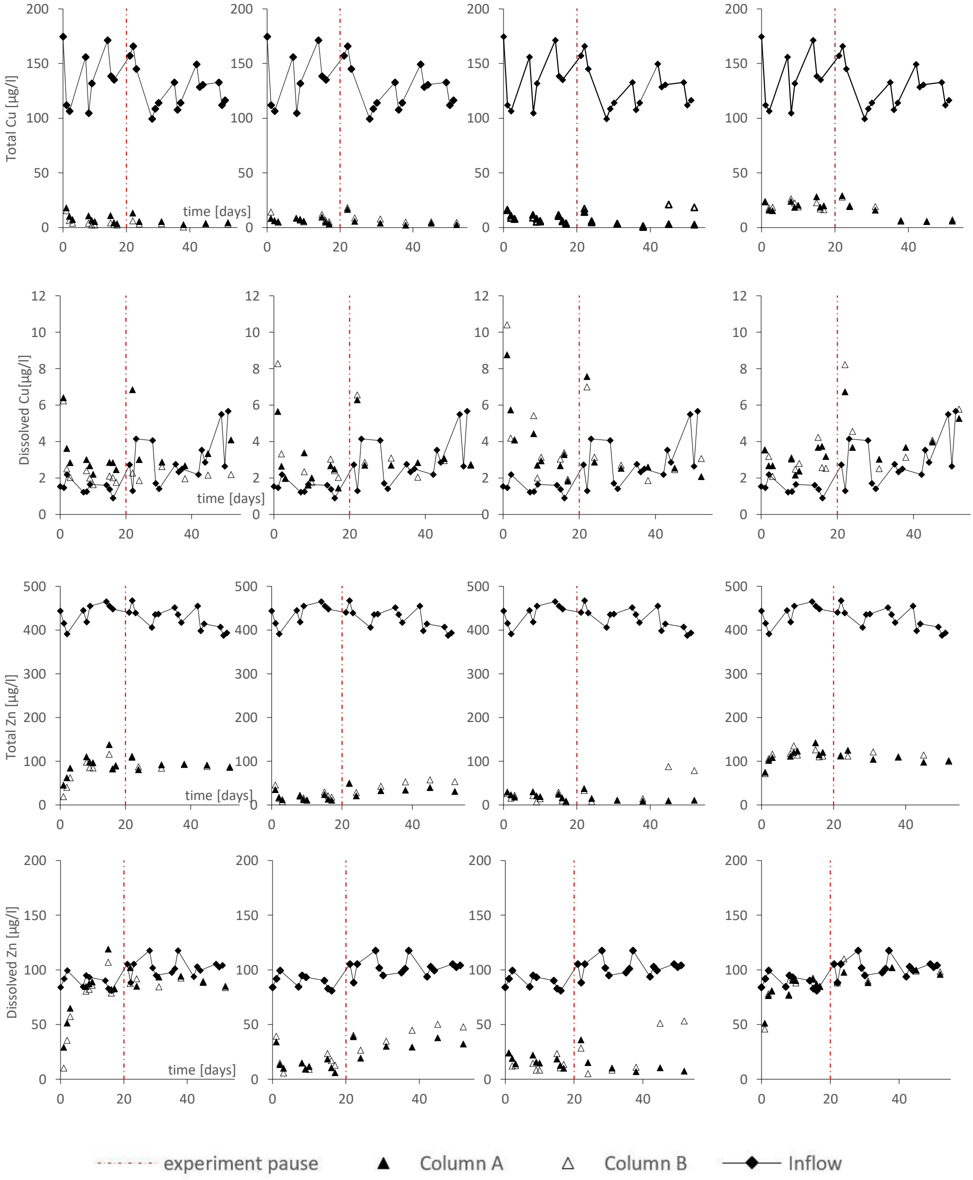


Figure 12: Total and dissolved concentrations of Cu and Zn in the column experiment in the inflow and the outflow from duplicate columns. The red line indicates the break in the experiment. Order of charts from left to right: Milkweed, Bark, Peat, Polypropylene.

When comparing metal concentrations in the influents of column and field experiment, it can be seen that the Cu concentration was about 10 – 20 times higher in the roof-runoff of the field experiment than in synthetic stormwater used for the column experiments. This is expected, since influent for the column experiments (Paper I) was created to simulate a polluted road runoff, which by nature is more diverse in pollutant loading than the copper roof runoff (Paper II). The zinc concentrations in the influent to the filters were similar (387–467 µg L^{-1} in column experiment and 186–464 µg L^{-1} in the roof runoff).

4.3. Metal removal efficiency of the investigated filters

In general, treatment efficiency of the zeolite filter ranged from 52-82% for total Cu and 49-85% for dissolved Cu (Table 6). There was a noticeable trend of a decrease in treatment efficiency. Regarding the zinc treatment, efficiency varied between 51-94% for the total Zn, and 48-94% for the dissolved Zn (Table 6).

Table 6: Treatment efficiency of the zeolite filter (%) during each event for total and dissolved copper and zinc.

	Copper		Zinc	
	Total	Dissolved	Total	Dissolved
03-12-18	82	85	94	94
06-03-19	63	68	64	62
02-05-19	69	74	84	87
27-06-19	68	74	75	75
09-09-19	56	57	59	55
07-11-19	80	82	82	82
19-03-20	52	49	51	48

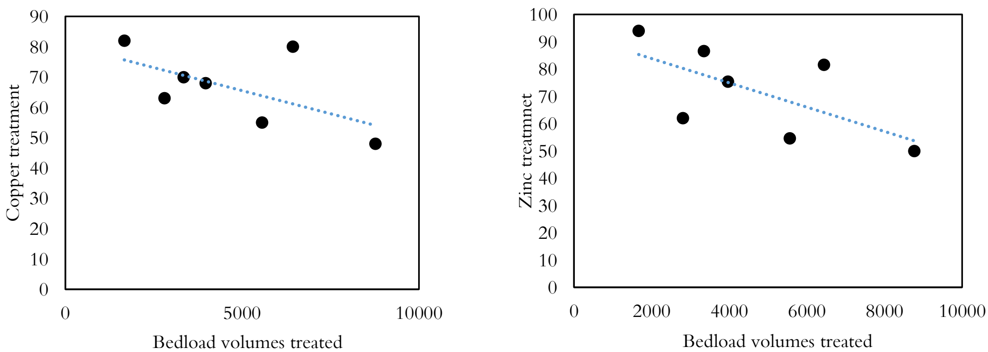


Figure 13: Change in the treatment efficiency of total copper (left) and zinc (right) in relation to total bed load volumes treated over time.

There is a noticeable drop in treatment efficiency for both total Cu and Zn throughout the duration of the experiment conducted for Paper II, with treatment efficiency dropping from 82% and 92% to 48% and 50% for total Cu and Zn, respectively.

In the column experiment the level of total treatment of Zn was considerably higher than dissolved for all of the studied filter materials (74–95%). The same was true for total Cu, where removal varied from 86–96%.The high level of treatment could be attributed to the filtering process, where most of the particulate-bound metals were removed by removing the sediment from the synthetic stormwater. Figure 14 shows the removal efficiency for total and dissolved Cu and Zn for the column experiment.

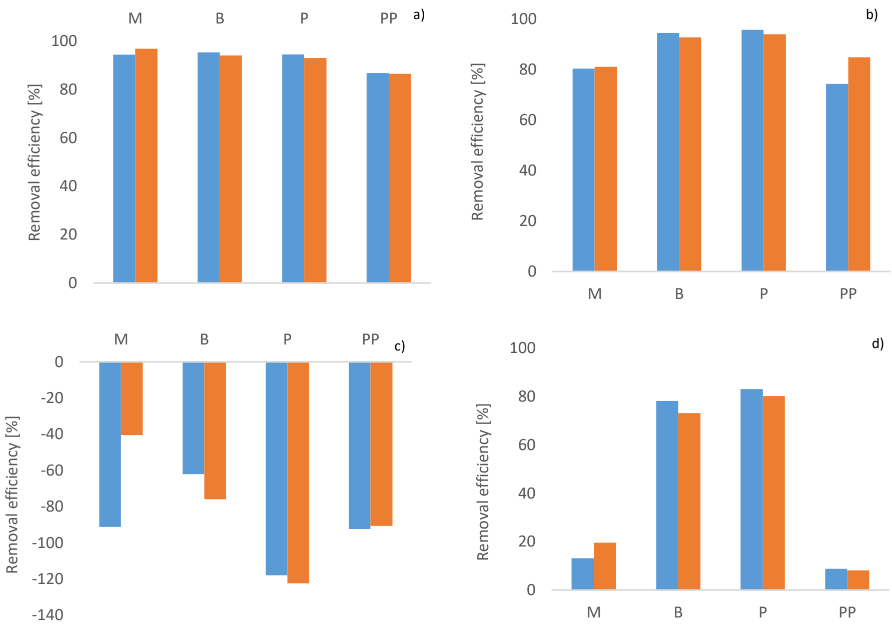


Figure 14: Removal efficiency of milkweed (M), bark (B), peat (P) and polypropylene (PP) with respect to total Cu (a), total Zn (b), dissolved Cu (c) and dissolved Zn (d). Left and right bars represent two replicates for each filter material.

The negative removal rate for Cu treatment across all filter materials may indicate that the coefficient used to adjust the starting inflow concentration of dissolved copper may overestimate the concentration, since no change in pH was detected between the inflow and outflow samples.

It is difficult to make a direct comparison of filter treatment efficiency between the column experiments (Paper I) and the field experiment (Paper II) due to the differences in filter media, and due to inherent differences between field conditions and laboratory conditions, such as hydraulic loading, approximation of stormwater, and the generally more controlled conditions present in the laboratory study. However, some observations could be made. The treatment of total and dissolved Zn could be compared due to the similarity of total Zn concentrations. The treatment of dissolved Zn by "traditional" stormwater filter materials (bark and peat) in the column experiment was approximately 75 and 81%, respectively. The treatment of total and dissolved Zn by the zeolite filter was comparable, 51–94% and 48–94%, respectively, over the observed period. These three materials showed a much higher treatment level than milkweed and polypropylene, which could remove dissolved Zn from the synthetic stormwater by 16 and 8%, respectively.

4.4. BGS Modelling results

The most influential factor for all of the runs was discharge, as it dictated the streamwise velocity in the model (Figures 16-18). With the increase in velocity, if all of the other parameters remained the same, trapping efficiency decreased. The runs with the sudden transition set-up showed a very uneven flow pattern in the BGS tank, with a dominant flow jet passing through the middle of the tank, and two noticeable asymmetrical backflow jets forming on the sides of the tank. There was a noticeable pattern of cells devoid of sediment on the pathway of the jet, indicating that the high velocity prevented sediment deposition, and the backflow jets promoted suspended sediment settling (Figure 15).

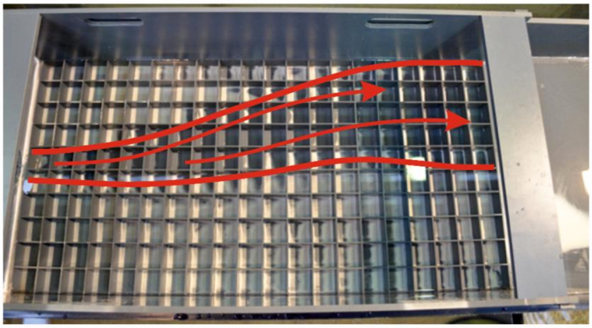


Figure 15: Photo of the BGS grid with a visible region of cells devoid of sediment, simulating the jet stream trajectory.

The expansion used in runs 8-24 reduced this negative occurrence, and allowed a relatively uniform velocity distribution along the BGS tank. This also increased sediment settling efficiency of the BGS, for Q = 2 l/s, the runs with the diffuser were 45% more effective than the sudden expansion ones, whereas for Q = 4 l/s, the BGS effectiveness of the diffuser proved to be more than 70% more effective than the run with the sudden expansion. When comparing the effectiveness of various BGS configurations to the bare bottom runs, in general, BGS runs were 25% more efficient in removing sediment. The only flow for which this was not the case was Q = 4 l/s, where the bare bottom run removed about 10% more sediment than the best preforming BGS run. However, it has to be noted that this was the only case for all of the experiment set-ups in which the performance increased with the flow, so the validity of this point is questionable.

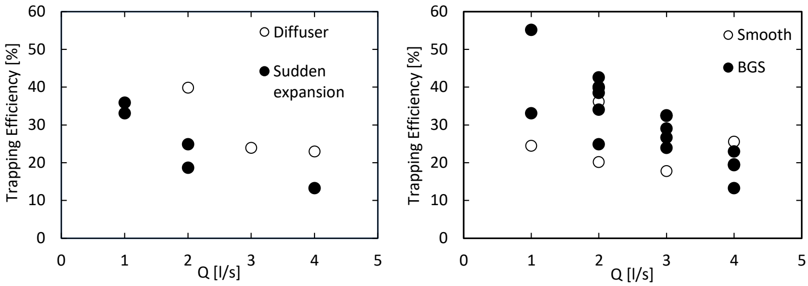


Figure 17: Sediment trapping efficiencies with and without the diffuser (cell width and depth 5 cm, cross-wall angle 90°), and (right) Sediment trapping efficiencies in runs with and without BGS (smooth bottom runs).

Besides vertical walls, BGS models with cross-wall angle-cell inclinations of 60° and 120° were also tested. Cross-walls with both angles performed better than the 90-degree walls. There was a noticeable amount of sediment settled on the wall surface, which could be the reason behind the increase of the settling efficiency. This was again more noticeable for the higher flow rate .

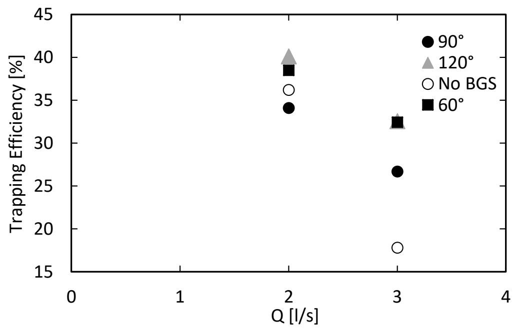


Figure 18: Sediment trapping efficiency for varying wall angles.

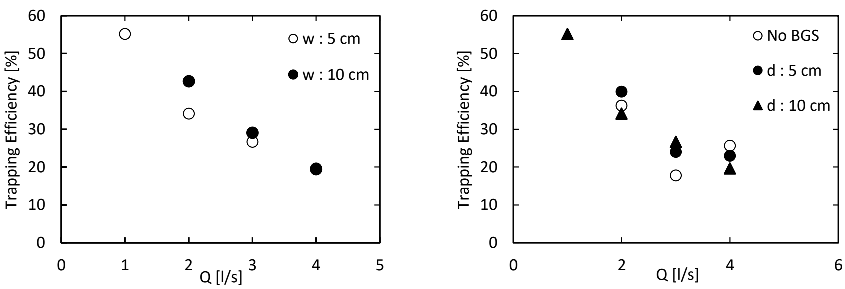


Figure 16: Sediment trapping efficiency for cell depths of 5 and 10 cm, and the cross-wall angle of 90° (left); Sediment trapping efficiency for no BGS, the cell width of 5 cm and two depths (5 and 10 cm), and the cross-wall angle of 90° (right).

Model set-ups with BGS cell sizes of 10 cm were on average about 10% more effective than the runs with a cell width of 5 cm (Figure 16, left). This was more noticeable for lower flow rates, while for Q = 4 l/s, there was no difference between the different cell widths. Within the range of tested depths, there was no difference in the sedimentation rate (Figure 16, right).

The repeatability test was carried out for d = 10 cm, w= 5 cm, $\alpha = 90°$, and Q = 3 L/s. The documented sediment trapping efficiency was 28% with a standard deviation of 0.5%. The result indicated that the differences in the performance in the various BGS settings were not due to the uncertainty of experimental runs.

5. Discussion

In this chapter, the results that were presented in the previous chapter are discussed within the context of previous studies. In Section 5.1, filter material effectiveness and the application of stormwater filters are discussed. Section 5.2 discusses the preparation and ageing of synthetic stormwater. Section 5.3 presents the impact of a bottom grid structure (Paper III) on stormwater treatment capacities of stormwater ponds.

5.1. Impact of BGS cell design of sedimentation effectiveness and suggestion for future studies

One of the main design parameters of a stormwater pond is the area needed for the sedimentations of particles (Persson, 1999). Urbanisation is one of the leading causes of increased stormwater runoff, and the decrease of its quality (Walsh et al., 2005) and available area in urban space are often limiting factors. Based on results from field measurements (He et al., 2014), the implementation of a BGS could increase the equivalent settling area by 5 to 60 times.

The hydraulic modelling study helped to determine the impact of BGS cell parameters on the sedimentation efficiency of the structure. However, the results are not directly translatable to the field conditions. In order to translate experiment results from the model to the prototype, it is necessary to keep the relationship between the inertia and gravity forces identical. This relationship is expressed by the Froude number (eq. 8).

$$Fr = \frac{V^2}{gh} \qquad (eq\ 8)$$

Where V is flow velocity, g is the acceleration of gravity, and h is the water depth. The Froude number should be same on both the model and the prototype. Since the gravity acceleration is the same on both the model and the prototype, it leads to Equation 9.

$$Fr_m = Fr_p \rightarrow \frac{{V^2}_m}{h_m} = \frac{{V^2}_p}{h_p} \qquad (eq\ 9)$$

In the experiments conducted on the BGS model, it was assumed that the length scale $L^\star = Lp/Lm = 10$, where $L^\star$ is the scale factor, Lp represents lengths in the prototype and Lm stands for lengths on the model. That would mean that the prototype would have a length of 10 m, a width of 5 m and a cell wall height of 50–100 cm. Modelling sediment transport was challenging. Studied conditions did not allow the requirements for the sediment properties (density and particle diameter) to be fully satisfied, thus the model was suitable for providing qualitative comparisons between different cell designs, but not for quantified results.

Gathered data from the experiment suggest that the width of the cell impacts the settling efficiency, with wider cells providing enhanced sedimentation (Figure 16). Another factor that proved important was cell wall angle, where experiment set-ups with inclined walls provided on average 20% better sediment removal rate. However, the drawback of this configuration might be too great, as it would likely be difficult to maintain such a structure, given the geometry of the cell walls.

One of the key factors often neglected is the maintenance of stormwater control measures. According to (Al-Rubaei et al., 2017), half of the inspected 25 stormwater ponds in Sweden were in need of maintenance. The inclusion of a BGS system could address this issue, as most of the sediment could be expected to settle in the cells. From there it could be extracted by means of hydraulic dredging, a practice that is used to maintain stormwater ponds (Drake and Guo, 2008). The BGS would also reduce the size of the area in need of dredging, compared to dredging the whole pond or forebay.

The inclusion of the BGS in the stormwater pond should reduce sediment resuspension, which is important since stormwater pond sediment can contain elevated levels of metals such as Cr, Cu and Pb (Marsalek and Marsalek, 1997).

Finally, the next research phase should focus on optimising the geometry of the BGS cells, and then testing it in field conditions, since it will always be challenging to fully simulate the complex behaviour of sediment transport over such a structure in laboratory conditions.

5.2. Filter effectiveness in metal treatment from stormwater

5.2.1. Treatment efficiency of filter materials

Peat

Peat filters have been the subject of numerous studies that focused on their capacity to remove pollutants from wastewater and stormwater (Couillard, 1994; Crist et al., 1996; Brown et al., 2000; Kalmykova et al., 2009). In a field study investigating treatment from highway runoff, Zhou et al. (2003) determined that the filtration system containing peat as filter material was able to remove 90% of total Zn, and 70% of dissolved Zn, from the runoff. Similarly, in the column study, peat columns removed 94% of total Zn, and 81% of dissolved Zn, over the duration of the experiment, and it was the most effective filter material in removing dissolved Zn from the synthetic stormwater.

Bark

Bark has shown high capacity to remove metal ions from waste and stormwater (Vázquez et al., 2002; Jang et al., 2005; Genç-Fuhrman et al., 2007; Nehrenheim and Gustafsson, 2008). In the column study, bark was the second most effective material at treating dissolved zinc from the synthetic stormwater, with an average efficiency of 75%, over the duration of the experiment.

Milkweed

Milkweed as a material has been proven to have high oil sorption capacity. In a study that compared multiple sorbents (milkweed, kapok, cotton, wool, polypropylene and kenaf), milkweed showed the highest sorption capacity (Choi, 1996). In a study that assessed the sorption capacities of different filter materials at various concentrations of Cu and Zn, milkweed was found to be able to reduce the concentration of Zn in cases where the influent concentration did not exceed 500 µg. In the column experiment described in this thesis, Milkweed was not able to achieve the same level of removal of dissolved Zn as bark and peat. It should be noted that due to the volume limitation, the mass of used milkweed was several times lower than bark and peat. Milkweed columns were packed with 9 and 13 g of filter material, while bark and peat had 35 and 50 g, and 51 and 64 g, respectively. When adjusted for the mass of the filter material (Table 4, Paper I), milkweed showed a comparable reduction to bark and peat. It should also be noted that milkweed columns achieved the highest removal of the dissolved Cu.

Polypropylene

The main reason to include polypropylene in the study is its capacity to remove oil from the stormwater runoff (Praba Karana et al., 2011). In the column study described in 3.1.1, polypropylene showed the lowest removal efficiency of the four tested filter materials, removing 8% of the dissolved Zn from the synthetic stormwater. The batch studies conducted by Norman (2018), also showed that the material demonstrated no sorption capacity. In previous studies, the examined polypropylene has been modified. For example, Mavlyankariev and Rhee (2007) showed that when coated with Manganese Dioxide, polypropylene granules show high removal efficiency for Pb, Cu, Cd and Zn.

Zeolite

The inflow for the field experiment in Paper II was runoff from a copper roof. The quality of inflow was comparable to similar studies. In the influent, the total Cu concentration was in the range of 900–2100 µg L^{-1}, with 93% of it found in the dissolved fraction. This is in line with other studies concerning Cu roof runoff. Athanasiadis et al. (2007), reported the event mean concentration of approximately 1600 µg L^{-1} and in a study investigating runoff from a Cu roof in Stockholm, Jönsson (2013) reported the concentration ranges of 945–3400 µg L^{-1}.

In a study by Athanasiadis *et al.*, (2007), it was shown that a filter system with zeolite installed as a barrier material was able to reduce the total Cu up to 97%. This level of treatment has not been reached in the study in Paper III. However it is important to point out the differences between the two systems. The filter system, as described in 3.1.2, contains a pump that allowed the stored water to be treated. In the study referenced above, water was allowed to flow gravitationally through the system, which could explain higher treatment efficiency. The average treatment of total and dissolved Cu in the field experiment was 69, and 73%, respectively (Table 5 and Figure 13). Given the level of the inflow concentration of Cu, this meant that the level of Cu in the effluent exceeded the recommendation by the authorities multiple times (City of Gothenburg, 2020).

In order to reduce the impact of the pollutants of the receiving water body, a filter system should provide a high level of treatment throughout its service life. According to the current trend of decreasing performance (Figure 13) this would not be the case. The service life of the filter system studied in (Paper II) is three years (3P Technik, 2020). If the current trend were to be continued, the treatment level of total Cu and Zn would drop to 25%, and 7% by the end of the stated service life.

An explanation for lower treatment efficiency in the filter system evaluated in this thesis compared to previous studies, such as (Athanasiadis et al., 2007) is that the hydraulic loading of the filter system is too high to allow efficient metal removal. In a study that investigated the performance of columns packed with different filter materials (peat, zeolite and calcium silicate rock), Färm (2002) found that there was a significant decrease of metal treatment by the columns with the increase of hydraulic load. The decrease of the performance in that study was observed when the hydraulic loading increased above 3.5 $m^3\ m^{-2}h^{-1}$, which is almost two times lower than the load in the analysed filter system presented in this thesis (6.7 $m^3\ m^{-2}h^{-1}$). Future studies of the impact of the hydraulic load on the treatment performance of the zeolite filter installation could demonstrate how much of an effect a decrease in hydraulic load would have on the improvement of metal removal.

5.2.2. Applicability of filters for stormwater treatment.

Filter materials can be used either to enhance stormwater treatment in other stormwater treatment systems such as bioretention systems, as stand–alone solutions such as gully-pot filters or as a part of a treatment train, a sequence of stormwater control measures that aims to maximise the control of pollutants from the runoff (Wong et al., 2002). The treatment train approach aims to utilise the specific strengths of stormwater treatment methods to achieve more efficient stormwater treatment. For example, clogging can reduce the effectiveness of infiltration trenches, permeable pavements and other filtration based stormwater control measures (Hatt et al., 2007; Ziyath et al., 2011; Blecken et al., 2017). That could be prevented if the filtration step was preceded by a system efficient in removing larger solids from the stormwater, such as a stormwater pond. Ponds are efficient in removing solids from the stormwater, and their insufficient dissolved pollutant treatment would be remedied by the filter systems.

Bark, Peat and Zeolite showed high dissolved metal removal, and could be considered as suitable materials in these filter installations. Milkweed and polypropylene have not shown as good removal of pollutants, so it is likely that they would need to be adapted before their use as stormwater filter material.

5.3. Synthetic stormwater as a proxy for stormwater in experiments

Synthetic stormwater is a model of the stormwater runoff found in situ that is used for various laboratory studies in order to test different filter materials to determine their effectiveness in removing various pollutants. The advantage of using synthetic stormwater over natural, or collected, stormwater is that it allows better repeatability and in general more controlled experiments. The disadvantage is that synthetic stormwater will never truly replicate the chemical, physical and biological characteristics of real stormwater (Hatt et al., 2007). Synthetic stormwater used in Paper I was created to simulate the runoff from a polluted road. Metal concentrations were targeted to be in the range of those found in the literature (Makepeace et al., 1995; Davis et al., 2001; Camponelli et al., 2010). When compared to the other studies that also used synthetic stormwater, the values for heavy metals also fell within the range of the values used by other researchers

(Haselbach et al., 2014; Lim et al., 2015; Borris et al., 2016; Genç-Fuhrman et al., 2016; Huber et al., 2016b; Rycewicz-Borecki et al., 2016; Søberg et al., 2017).

The ageing of SW is an important factor to consider when taking into account filter effectiveness. Often a filter performance is judged based on the reduction of pollutant concentrations. Commonly, one way to determine the performance of the filter is to compare the concentration in the inflow and at the outflow from the filter set-up (Genç-Fuhrman et al., 2007; Blecken et al., 2009; Søberg et al., 2019), usually by following a formula similar to the one presented in 3.3.3 (Equation 6). Thus determining the importance of the initial concentration when estimating filter effectiveness. As shown in Paper I, there is a rapid change in dissolved metal concentration upon mixing. In Paper I, a time-weighted average concentration was introduced to account for this change. Figure 19 presents the impact on filter efficiency of different methods of accounting for the change in the synthetic stormwater. The methods are as follows: (I) "Starting" uses the dissolved concentration of the metal at the start of the experiment as the inflow concentration. (II) "Mid" uses the average of the concentration at the start of the experiment, and one found in samples at the end of the experiment. (III) "Integral" uses the time–weighted coefficient defined in Chapter 3.1.2. (Equations 4–6).

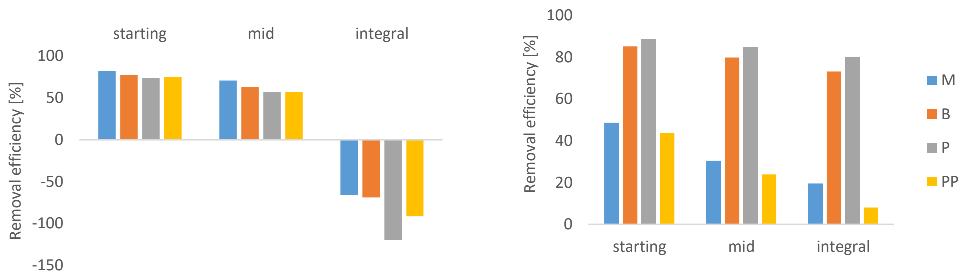


Figure 19: Impact of different correction methods of inlet concentrations on dissolved Cu (left) and Zn (right) treatment efficacy. Starting – initial concentration of dissolved metals. Mid – mean of the initial concentration of dissolved metals and the measured concentration at the end of the experiment, and Integral – adjusting the initial concentration with time–weighted coefficients.

The difference between the different methods depends on how quickly the level of dissolved metal concentration changed in the synthetic stormwater. It was far more noticeable for Cu, where the Starting scenario resulted in 73.7% treatment level of peat columns, the Mid scenario resulted in 56.8% treatment, and the Integral scenario with time-weighted coefficient indicated a negative removal of -120%. The negative removal of Cu might indicate that the time resolution of the method was too low, since the pH of the influent and effluent solutions did not vary significantly (7.7 in synthetic stormwater, 7.2 in peat column outflow). More detailed analysis (more frequent time–steps) could have resulted in a more precise method to determine inflow concentration of dissolved C. The difference was present in other metals as well, although it was not as pronounced. In the case of dissolved Zn, the three methods did not differ as much. When observing the effectiveness of peat columns, the use of Starting, Mid and Integral methods resulted in 88, 84 and 80% Zn removal, respectively.Since the 11–day ageing experiment showed little change in the metal concentration over the first 6 days of the duration of the experiment, and the shorter ageing experiment showed that most of the reduction in dissolved metal concentration took place in the first three hours following the mixing, it is recommended in future similar studies that water should be mixed at least three hours before the start of the experiment, to allow a more stable inflow concentration. Furthermore, it is also recommended to conduct test runs before the actual experiment in order to determine the amount of metal salts needed for spiking the synthetic stormwater.

In the BGS modelling study, water spiked with Neralite was used to simulate stormwater runoff. Since the sediment used in the modelling study had a uniform diameter, effects of particle size have not been addressed. Laboratory studies conducted by He and Marsalek (2014) indicated that the structure was more successful in trapping larger particles than smaller ones, particularly under higher flow conditions.

5.4. Stormwater treatment and area efficiency

An integral approach to stormwater management in relation to other urban infrastructure is needed, so it is important to determine, and where possible reduce, the area that stormwater systems require to function properly. Stormwater systems are most needed in urban environments, where space requirements have to be balanced between different functions. In one of the studies included in this thesis, the need for the surface area was bypassed by placing the collection and storage tank, as well as the five filter units, below the ground. This is, however, not always possible, since the area below the ground is often occupied by pipes, cables and other utilities. The total area occupied by the filter installation was approximately 60 m^2. The combined surface of the roof and the part of the park that was treated was approximately 4800 m^2, so the area needed for treatment amounted to 1.25% of the catchment, which is comparable to the area typically needed for stormwater ponds (Persson et al., 1999). If the hydraulic loading of the filter was to be decreased, in order to improve the treatment efficiency, the required area of the filter installation would increase even further. Another way to utilise filter media consists of stand–alone solutions such as gully pot filters or catch basin inserts (Lau et al., 2001; Färm, 2004). These filters are located in gully pots and are intended to treat stormwater at the source. The advantage of these systems is that they can be easily retrofitted in existing urban drainage infrastructure, and that they do not have a land footprint of their own. However, the disadvantage is that these systems are easily clogged and require frequent maintenance (Atlanta Regional Commission, 2001).

One of the main design parameters of a stormwater pond is the area needed for sedimentation of particles (Persson, 1999). Based on results from a field experiment it has been hypothesised that the implementation of a BGS could decrease the required equivalent settling area by 5 to 60 times (He et al., 2014), which would make implementation of stormwater ponds in urban areas more feasible.

6. Conclusions

The aim of this licentiate was to summarise and provide better knowledge about the components of area-efficient stormwater treatment facilities, as well as to further increase the knowledge about laboratory experiments to evaluate potential filter materials that could be used in said facilities. The following conclusions can be drawn:

The experiments that compared the ability of different filter materials to treat total and dissolved metals from synthetic stormwater showed that the order of efficiency of filter materials were: peat>bark>milkweed>polypropylene. All of the filter materials exhibited high total metal removal, although that was likely to be the case due to the column set-up, where the combination of filter materials, geotextile and glass beads, removed a considerable amount of total solids that had pollutants attached to them. Bark and peat columns achieved 75% and 81% dissolved metal removal, respectively. While milkweed and polypropylene showed a far lower removal efficiency (16% and 8%, respectively), milkweed showed a removal rate comparable to bark and peat, when adjusted for the mass of the filter present in the columns.

The field study that investigated the Cu and Zn removal efficiency of a zeolite filter treating runoff from a copper roof showed that the treatment level for total and dissolved Cu ranged between 52–82% and 48–94%, respectively. Zn removal was estimated at 49–85% and 48–64% for total and dissolved zinc, respectively. Besides a relatively high Cu removal rate, effluent concentrations still exceeded the recommended effluent values suggested by environmental authorities by more than 30 times. The trend in the treatment over time also indicated that the performance would continue to deteriorate. High hydraulic loading, compared to the similar studies, was identified as a potential reason for the lower Cu removal rate.

In order to further the understanding of the dissolved metal behaviour in synthetic stormwater, an 11-day ageing experiment was conducted, as well as a one-day ageing experiment. The 11-day ageing experiment showed that dissolved metals remained stable used up to 6 days after mixing. The triplicates of the one-day experiment showed a rapid change in dissolved metal concentration shortly after the mixing, with the most rapid changes detected in dissolved Cu, where the concentration decreased by 85% in the first 200 minutes following the preparation of synthetic stormwater. It was shown that the different methods of calculating filter effectiveness led to drastically different results for removal, thus a time-weighted coefficient was proposed that would account for the changes of the dissolved metals in synthetic stormwater and provide more precise treatment values.

In order to improve settling efficiency of stormwater ponds, at the same time as reducing the area for settling, a novel device bottom grid structure was investigated in a series of tests performed in a hydraulic laboratory, on a scaled model. The tests showed that the larger cells were more effective in capturing model sediment, increasing the efficiency by 13%. Inclined walls increased the efficiency of the BGS model, although their implementation in prototype and full scale might prove problematic due to maintenance requirements.

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Paper I

Ivan Milovanović, Inga Herrmann, Annelie Hedström, Kerstin Nordqvist, Alexandra Müller, Maria Viklander

Synthetic Stormwater for Laboratory Testing of Filter Materials

Submitted to Environmental Technology

For the full licentiate thesis, including the paper "Synthetic Stormwater for Laboratory Testing of Filter Materials" please contact the author (ivan.milovanovic@ltu.se) for a physical copy of the licentiate.

Paper II

Ivan Milovanović, Annelie Hedström, Inga Herrmann, Maria Viklander

Performance of a Zeolite Filter Treating Copper Roof Runoff

Submitted to Urban Water Journal

For the full licentiate thesis, including the paper "Performance of a Zeolite Filter Treating Copper Roof Runoff" please contact the author (ivan.milovanovic@ltu.se) for a physical copy of the licentiate.

Paper III

Ivan Milovanović, Vojtěch Bareš, Annelie Hedström, Inga Herrmann, Tomas Picek, Jiri Marsalek, Maria Viklander

Enhancing stormwater sediment settling at detention pond inlets by a bottom grid structure (BGS)

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Enhancing stormwater sediment settling at detention pond inlets by a bottom grid structure (BGS)

Ivan Milovanović, Vojtěch Bareš, Annelie Hedström, Inga Herrmann, Tomas Picek, Jiri Marsalek and Maria Viklander

ABSTRACT

Stormwater sediments of various sizes and densities are recognised as one of the most important stormwater quality parameters that can be conventionally controlled by settling in detention ponds. The bottom grid structure (BGS) is an innovative concept proposed in this study to enhance removal of stormwater sediments entering ponds and reduce sediment resuspension. This concept was studied in a hydraulic scale model with the objective of elucidating the effects of the BGS geometry on stormwater sediment trapping. Towards this end, the BGS cell size and depth, and the cell cross-wall angle were varied for a range of flow rates, and the sediment trapping efficiency was measured in the model. The main value of the observed sediment trapping efficiencies, in the range from 13 to 55%, was a comparative assessment of various BGS designs. In general, larger cells (footprint 10×10 cm) were more effective than the smaller cells (5×5 cm), the cell depth exerted small influence on sediment trapping, and the cells with inclined cross-walls proved more effective in sediment trapping than the vertical cross-walls. However, the BGS with inclined cross-walls would be harder to maintain. Future studies should address an optimal cell design and testing in an actual stormwater pond.

Key words | hydraulic scale modelling, sediment settling, sediment trapping efficiency, stormwater ponds

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INTRODUCTION

Progressing urbanisation leads to profound changes of the urban water cycle manifested by increased surface runoff and deterioration of runoff quality by discharges of various pollutants, including stormwater sediments (Walsh *et al.* 2005). In this context, stormwater sediments represent a broad spectrum of particle sizes, including total suspended solids (TSS) and bedload sediment, and impact both the water quality in the receiving waters and operation of drainage systems. Former impacts include TSS interference with quality processes in the water column, impairment of aquatic biota (Bilotta & Brazier 2008), and transport of attached chemicals and faecal microorganisms (USEPA 1983). Bedload sediment size classes comprising sand and fine gravel may cause blockage of conveyance elements, and reduction of water and sediment storage volumes in drainage facilities, including the ponds. Consequently, controls of runoff peaks and stormwater sediments have been among the highest priorities of stormwater management since the early 1970s. Since then, tens of thousands of stormwater detention ponds have been built worldwide for controlling runoff peaks by storage and removing stormwater sediment by settling.

Well-functioning stormwater ponds remove high quantities of incoming TSS and coarse sediment (Yousef *et al.* 1994; Pettersson 1999), which deposit and spread throughout the pond. For restoration of the pond design conditions and reduction of the risk of contaminated sediment resuspension (Bentzen 2010) and washout during high flows (Karlsson *et al.* 2010), pond sediments need to be removed and safely disposed of at time intervals as short as 16–17 years (Rishon 2013). Such a task represents one of the most

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costly items in pond maintenance (Al-Rubaei *et al.* 2017). To reduce the costs of pond sediment management, the first step was to introduce sediment forebays into pond design, with forebays occupying up to one third of the total permanent pool area (MOEE 2003). Recognizing that even the forebay cleanout is an onerous task, pre-treatment of stormwater immediately upstream of the pond, by swales or oil/grit separators, was recommended (MOEE 2003). Another way of achieving this objective would be to incorporate a sediment trap, with a small footprint, immediately downstream of the stormwater inlet into the pond.

Non-proprietary devices comprising bottom grid (or cellular) structures for enhancing settling and protecting the settled sediment against resuspension in confined waters were proposed; for example, by He & Marsalek (2014) and Simpson *et al.* (2018). Both structures enhanced suspended sediment settling by secondary currents in low velocity fields, and reduced the risk of resuspension of the settled sediment by confinement within the grid (He & Marsalek 2014). The main difference between the two structures is the cell shape: rectangular for BGS and a honeycomb in the cellular structure. The bottom grid structure (BGS) was further tested in the field, and in comparison to settling on the bare bed, it increased the sediment removal rate by a factor ranging from about 4 to 11, for various particle size ranges (He *et al.* 2014). Such promising results led to the idea of pre-treating stormwater entering the pond by a BGS device, which is easier and less expensive to manufacture than the cellular design. Thus, to reduce sediment spreading throughout the pond or forebay and lower the maintenance costs, it is proposed here to place a BGS structure downstream of the pond inlet, leading to the following benefits: (i) coarser sediment immobilization in the BGS with a small footprint, where it would be protected against washout and could be inexpensively removed by common municipal equipment, and (ii) such operations would reduce the frequency of pond dredging, which is relatively expensive and produces negative impacts on the downstream environment.

The objectives of the study reported on here were to: (i) comparatively assess the feasibility of using BGSs of various geometries to entrap and immobilize incoming stormwater sediment and (ii) suggest future research and development.

MATERIALS AND METHODS

Experiments with the BGS sediment trap were conducted in a hydraulic scale model. For model construction and testing, the following steps were taken: (i) a scale model was designed assuming a geometric scale 1:10 applied to hypothetical prototype dimensions (the inlet sewer D = 1 m, and BGS cells 0.5 × 0.5 m, 0.5 m deep), (ii) model sediment was chosen, and (iii) sediment trapping experiments were conducted in the model for selected flow rates, sediment fluxes and BGS cell designs. Further details follow.

Hydraulic scale model and model sediment

The hydraulic scale model, shown in Figure 1, was built in the Hydraulics Laboratory of the Czech Technical University in Prague and placed in an existing 1-m flume, which was fed water from the laboratory water supply system (LWSS) via an inlet tank. The discharge in the LWSS was measured using a MID flowmeter, Krohne Waterflux 300 (accuracy <1% of the measured value). Another flow measurement device, a Thompson weir (90° notch; accuracy 2–5%), was placed at the inflow to an existing 1-m wide

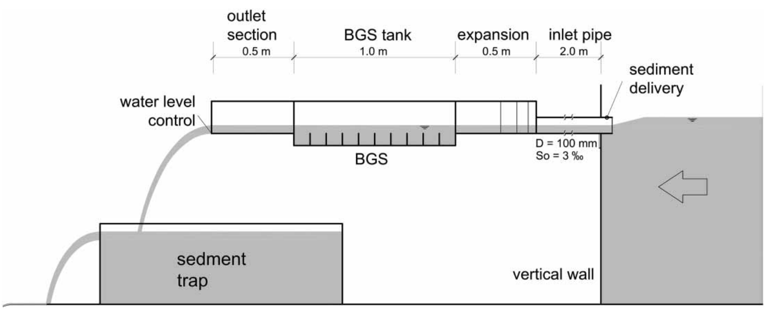

Figure 1 | Experimental setup of the BGS model.

flume. From the inlet tank, water flowed through a 100 mm PVC pipe ($L = 2$ m, $S = 0.3$ ‰) to the BGS tank (i.e. a settling tank fitted with BGS on the bottom). Two types of transition between the pipe outfall and the BGS tank were tested: (i) a sudden expansion (i.e. the inlet pipe opening was flush with the upstream BGS tank headwall) and (ii) a channel diffuser transition, 0.5 m long. The outer diffuser walls formed an angle of 40°, and insertion of three flow distribution baffles formed four flow channels with an expansion angle of ~10° (Figure 2).

The BGS tank, shown in Figure 2, was 1.0 m long and 0.5 m wide, and on its bottom rested the grid structure, with a basic cell size of $50 \times 50 \times 50$ mm ($L \times W \times D$), subject to modifications during selected runs. The water depth in the model was controlled by a sill at the downstream end of the BGS tank, and by an inclined multiple-slots weir located at the downstream end of the outlet section, about 0.5 m downstream of the sill. Two initial runs with a depth of 5 cm indicated that a greater depth was needed for flow calming, recognizing that the BGS tank also functions as an inlet stilling basin. The addition of the BGS structure should increase the sediment trapping and protect the trapped sediment against a washout (He & Marsalek 2014). Consequently, the remaining runs were done with a constant depth of 7.5 cm.

The selection of a model sediment represents a compromise between the specifications of 'ideal' material properties (sizes in low tens of μm, density slightly exceeding that of water), the practicality of running settling experiments and retrieving the settled sediment from the model, and availability of suitable materials on the market. After such considerations, a granular PVC 'powder' NERALIT® (specific gravity = 1.32, $d_{50} = 143$ μm, settling velocity $D_{50} = 0.0033$ m/s calculated and 0.0034 m/s measured) was selected as the model sediment. The choice of model sediment concentration was governed by practicality of experimental methods, subject to two constraints: (a) working with a sufficiently large mass of material in individual experiments to ensure accuracy of measurements and, at the same time, (b) avoiding the interference of excessive suspended sediment mass with flow dynamics. The choice of 100 mg/L met such conditions (as would probably do some other concentrations as well).

During experimental runs, the model sediment was introduced into the model upstream of the inlet pipe, at a rate producing a nominal sediment concentration of 100 mg/L in the model inflow. For this purpose, a stock of water/sediment mixture was prepared, with the model sediment concentration of $C = 250$ g/L placed in a continuously stirred container and pumped by a peristaltic pump (ISMATEC MCP/BVP 360) at calculated rates, which would produce the inflow sediment concentration of 100 mg/L. As a mass balance check, the actual mass of sediment delivered in individual runs was verified from continuous readings of an electronic weigh scale (KERN 572, resolution of 0.05 g, representing 0.1–0.3% of the total mass of sediment used in runs with $Q = 1$–4 L/s) placed under the stock container.

Experimental conditions

As commonly done in comparative testing of settling structure geometries (Stovin & Saul 1994), investigations are done in a steady flow regime with constant concentrations of model sediment. Towards this end, a range of flow rates (1–4 L/s) was selected to avoid sediment settling in the inflow pipe (starting at about $Q = 1$ L/s) and maintain a subcritical flow regime in the model for about $Q \leq 4$ L/s). For the range of model flows studied, $Q = 1$–4 L/s, a subcritical flow regime in the BGS tank, required the flow depth of 7.5 cm in the tank, and consequently, this depth was maintained in all runs by downstream flow controls (i.e. the sill at the downstream end of the BGS tank and the downstream weir). A

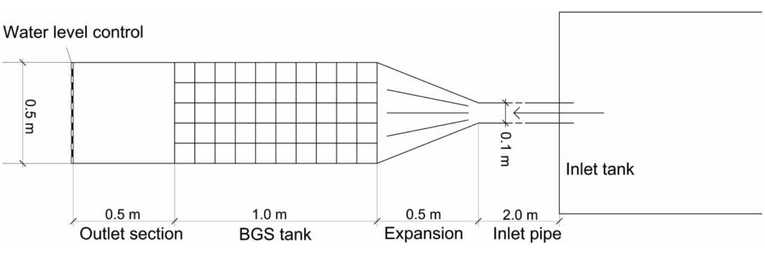

Figure 2 | Plan view of the experimental set-up of the BGS model.

laboratory protocol for model runs comprised the following steps: (a) prepare the water-model sediment mixture, (b) set up the hydraulic conditions in the model (the flow rate and depth) and start feeding in the water-sediment mixture, (c) run a preselected experimental scenario for durations of 30–50 minutes, (d) after finishing the run, retrieve the settled sediment from the BGS by a peristaltic pump, decant the water-sediment mixture, dry sediment in the oven (at 50 °C), and weigh the dry sediment, and (e) calculate the trapping efficiency $E_{tr} = M_{tr}/M_{in}$, where M_{tr} is the sediment mass trapped in the BGS and M_{in} is the mass of sediment fed into the BGS model. In total, 24 runs of the BGS model were carried out for various flow conditions (two flow depths, two transitions from the inlet pipe to the BGS tank, four flow rates), and combinations of cell widths, depths, and cross-wall angles (see Figure 3 for notation and Table 1 in the next section).

Uncertainties in trapping efficiencies E_{tr} were relatively low, because they represent values averaged over the experiment durations of 30–50 minutes. Furthermore, the total mass of sediment fed into the BGS during each run (180–720 g, for 30 minutes) was weighed accurately (0.5 g), so the uncertainty in M_{in} could be neglected. The remaining source of uncertainty was the trapped (retrieved) mass of sediment, which could be underestimated (i.e. incomplete retrieval and processing), or overestimated (should there be some sediment leftovers from previous runs, or incomplete sediment drying). M_{tr} uncertainty was conservatively estimated at 5–10%, and this estimate also represents the uncertainty in E_{tr} and its estimated magnitude was further supported by the results of repeated runs presented in the Results and Discussion section. Finally, this simplified consideration of uncertainties is acceptable in the context of study objectives constituting a comparative assessment of BGS geometries.

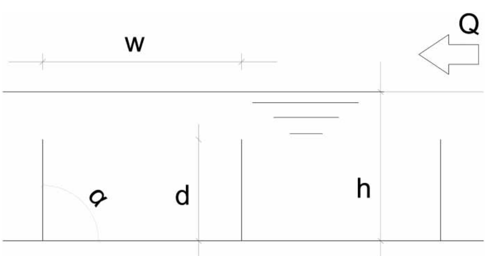

Figure 3 | BGS cells and their features varied in the experiments: cell width – w, flow depth – h, cell height – d and cross-wall angle – α.

Model similarity: flow and sediment transport

The dominant forces driving flow through the BGS model are those of gravity and inertia, for which the similarity between the model and prototype is achieved by maintaining identical Froude numbers in the model and prototype. However, the sediment transport similarity is much more challenging, because of complexities resulting from the need to reproduce not only the forces of gravity and inertia, as in the Froude similitude, but also viscous forces. For the conditions studied, this was not feasible and, consequently, the model was deemed as providing qualitative comparisons of various scenarios, but not fully quantified results.

RESULTS AND DISCUSSION

The presentation of results starts with the hydraulics of the BGS (settling) tank followed by sediment trapping. The results of all 24 experimental runs are presented in Table 1.

Initial runs of the BGS model with low flow depth (5 cm; runs 1 and 2) and a sudden transition from the inlet pipe to the BGS settling tank (runs 1–7) indicated a highly agitated flow in the tank, with a high velocity jet passing through the tank and two large backflow eddies forming on both sides of the jet. Such conditions were disruptive for effective separation of the incoming sediment from water and, consequently, the first steps were to correct this situation by: (i) ensuring subcritical flow through the facility by adjusting the flow depth to 7.5 cm and (ii) providing a hydraulically effective transition for the pipe inlet to the BGS tank by a diffuser. Such measures were fine-tuned by running the model with a smooth bottom in the BGS tank (i.e. without the BGS) in runs 6–10 and produced a quasi-uniform distribution of flow across the tank width. The testing of the BGS cell geometries followed (runs 3–5 and 11–24).

Experimental results documenting the benefits of the effective inflow transition and the presence of the BGS structure on the tank bottom are presented in Figure 4. There is a significant difference between the results with and without the diffuser transition (Figure 4(a)): for Q = 2 l/s, the diffuser proved to be 45% more effective than the sudden transition, while for Q = 4 l/s, the relative difference increased to 73%.

In general, runs with the BGS were about 25% more efficient in removing sediment from the flow than those without the BGS (Figure 4(b)), except for run 10 with the highest discharge (Q = 4 l/s), which was done with a

Table 1 | Experimental run parameter settings and the corresponding trapping efficiencies

Run	Inlet transition	Bottom arrangement[a]	Cell width w [cm]	Cell depth d [cm]	Cross-wall angle α [°]	Flow-rate Q [l/s]	Flow velocity V [m/s]	Flow depth h [cm]	Trapping efficiency E_{tr} [%]
1	Sudden[b]	BGS	5	5	90	2	0.08	5	19
2	Sudden	BGS	5	5	90	1	0.04	5	36
3	Sudden	BGS	5	5	90	1	0.03	7.5	33
4	Sudden	BGS	5	5	90	2	0.05	7.5	25
5	Sudden	BGS	5	5	90	4	0.11	7.5	13
6	Sudden	Smooth	–	–	–	1	0.03	7.5	25
7	Sudden	Smooth	–	–	–	2	0.05	7.5	20
8	Diffuser[c]	Smooth	–	–	–	2	0.05	7.5	36
9	Diffuser	Smooth	–	–	–	3	0.08	7.5	18
10	Diffuser	Smooth	–	–	–	4	0.11	7.5	26
11	Diffuser	BGS	5	5	90	2	0.05	7.5	40
12	Diffuser	BGS	5	5	90	3	0.08	7.5	24
13	Diffuser	BGS	5	5	90	4	0.11	7.5	23
14	Diffuser	BGS	5	10	90	2	0.05	7.5	34
15	Diffuser	BGS	5	10	90	3	0.08	7.5	27
16	Diffuser	BGS	5	10	90	4	0.11	7.5	20
17	Diffuser	BGS	5	10	90	1	0.03	7.5	55
18	Diffuser	BGS	10	10	90	2	0.05	7.5	43
19	Diffuser	BGS	10	10	90	3	0.08	7.5	30
20	Diffuser	BGS	10	10	90	4	0.11	7.5	20
21	Diffuser	BGS	5	10	60	2	0.05	7.5	39
22	Diffuser	BGS	5	10	120	2	0.05	7.5	40
23	Diffuser	BGS	5	10	120	3	0.08	7.5	33
24	Diffuser	BGS	5	10	60	3	0.08	7.5	32

[a]Bottom of the BGS tank.
[b]Sudden expansion – the inlet pipe was connected directly to the BGS tank.
[c]Diffuser expansion.

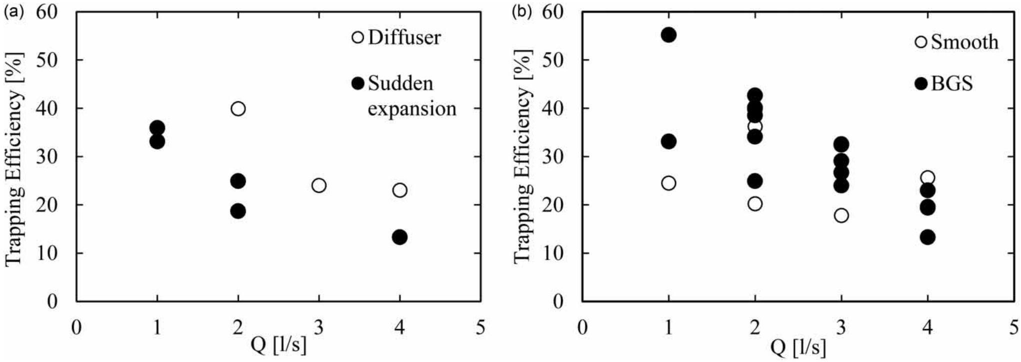

Figure 4 | (a) Sediment trapping efficiencies with and without the diffuser (cell width and depth 5 cm, cross-wall angle 90°) and (b) sediment trapping efficiencies in runs with and without BGS (smooth bottom runs).

smooth bottom and was about 11% more efficient than the best run with the BGS (i.e. run 13). Note, however, that run 10 was the only one among the routine runs in which the trapping efficiency for a particular BGS tank arrangement increased with an increasing discharge (i.e. compared to run 10, for $Q = 3$ l/s), which raises some doubts about the validity of this data point. One should also recognize that both variants; that is, with or without the BGS, use the same (BGS) tank promoting favourable settling conditions. The addition of the grid structure yields another benefit – protection of the deposited sediment against scouring.

Observation of flow patterns and sediment transport in the model indicated the presence of horizontal rollers in individual cells, with water moving downward along the downstream cross-wall, then in the upstream direction as a counter-current along the cell bottom, and finally upward along the upstream cross-wall and exiting from the cell. Such rollers entrained the sediment and moved it along a similar trajectory. While this flow pattern brings sediment into cells, it also tends to wash it out. Because of gravity, this pattern promotes overall particle settling, because along the downstream wall, flow and particles fall velocity act in the same direction, but along the upstream wall the particle washout is resisted by its gravity. Therefore, particles should settle near the downstream wall faster than they are being ejected by the fluid (Pedinotti *et al.* 1992). While it is conceivable that flow baffles preventing sediment washout could be fitted inside the cells, such a system would become complex and defeat the feasibility of easy maintenance and sediment removal.

Throughout all runs, the most influential factor for the sediment trapping efficiency was the discharge; that is, the streamwise flow velocity, with trapping efficiencies decreasing with increasing velocity. Figure 5 demonstrates the effect of flow velocity on the sediment trapping efficiency for a particular cell geometry ($w = 5$ cm, $d = 10$ cm, and $\alpha = 90^\circ$).

Effects of various cell geometries on trapping efficiency were examined in runs 11–24, by changing the cell footprint size, depth, and cross-wall angle.

Cell footprint size and depth. BGS configurations with a cell size of 10 cm were on average 13% more effective in trapping sediment than the 5 cm cells (Figure 6(a)). A possible explanation of this observation may be that larger cells allowed a better development of the horizontal-axis rollers, which contributed to trapping more sediment. Within the range of the depths tested (5 and 10 cm), the cell depth did not seem to influence the efficiency of the BGS significantly (Figure 6(b)).

Cross-wall angle. Besides vertical walls, BGSs with cross-wall angle inclinations of 60° and 120° were also

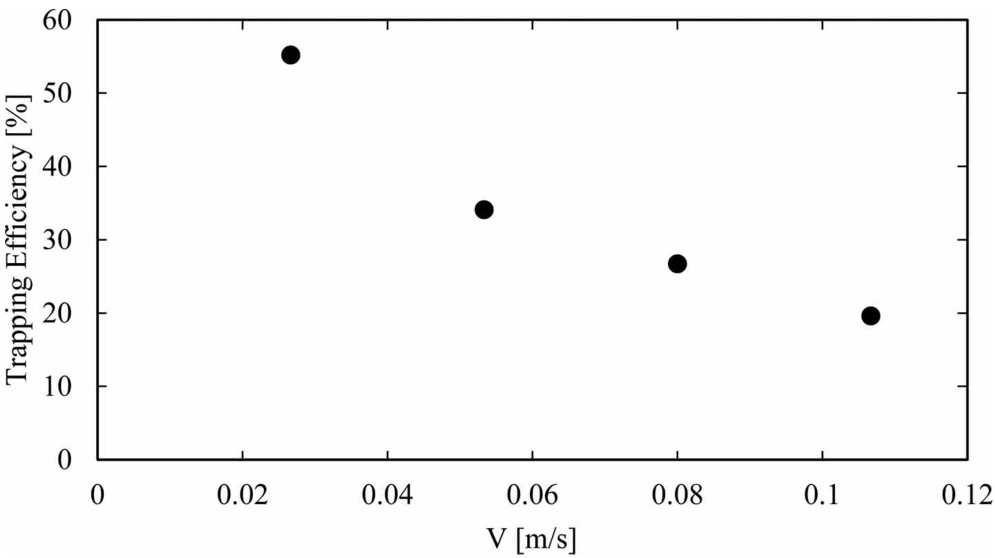

Figure 5 | Sediment trapping efficiency under various flow velocities (cell width 5 cm, cell depth 10 cm, cross-wall angle 90°).

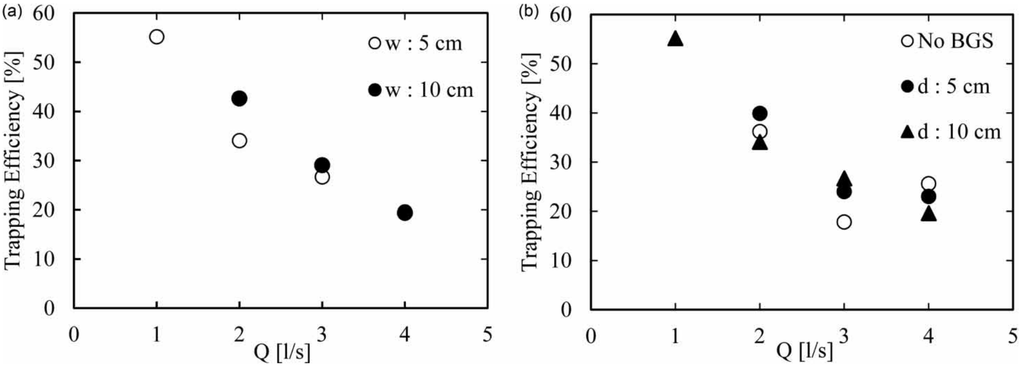

Figure 6 | (a) Sediment trapping efficiency for cell depths of 5 and 10 cm, and the cross-wall angle of 90°; (b) sediment trapping efficiency for no BGS, a cell width of 5 cm, two depths (5 and 10 cm), and a cross-wall angle of 90°.

tested, following up on the idea of settling enhancement with lamella plates. Cross-walls with both angles performed comparably and removed about 20% more sediment than those with 90° walls (Figure 7). This result was explained by the increased wall surface on which the sediment could settle. While this configuration might be beneficial for improving settling, its drawback would be the maintenance of the inclined walls, which would be more challenging compared to cells with vertical walls, particularly where the sediment removal would be done by the suction of sediment from the BGS cells.

In support of the discussion of experimental uncertainties (see Methods), a typical run with w = 5 cm, d = 10 cm, $\alpha = 90°$, and Q = 3 L/s was repeated five times. For the conditions addressed, the repeatability tests showed a very close agreement characterized by a mean trapping efficiency of 28% and a standard deviation of 0.5%. This suggests that, for the case tested, the differences in trapping efficiencies among the different experiments cannot be attributed to uncertainties in experimental techniques. However, further testing of repeatability for different conditions would be useful to examine whether the level of repeatability would remain the same.

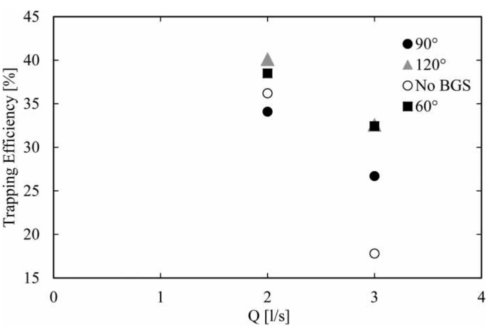

Figure 7 | Sediment removal efficiency for different cross-wall angles, a cell depth of 10 cm, and a cell width of 5 cm.

Future research. This section explores two classes of challenges encountered in studies of sediment removal by small engineering structures: (a) modelling of sediment transport in hydraulic scale models and (b) testing of small engineering structures in the field.

The experimental study reported on here expanded the knowledge of sediment settling in engineered facilities inserted into stormwater ponds, and demonstrated the challenges encountered in scale modelling of settling structures, including the attempts to attain model similitude for sediment settling in complex flow fields (i.e. as opposed to quiescent conditions in settling basins). The experience gained here was similar to the findings in the literature, best summarized by Gill & Pugh's (2009) statement that scale modelling of sediment transport in small engineering structures offers 'limited precision at best'. Consequently, the results reported herein should be understood as informative and best suited for comparisons of design alternatives, as was the case in the earlier studies of a similar nature (Stovin & Saul 1994; Dufresne *et al.* 2010; He & Marsalek 2014). Within the realm of such limitations, the study results indicate that an in-pond sediment trap (BGS) can concentrate the settling of coarser particles to a relatively small area, from which the sediments could be inexpensively removed using conventional municipal equipment (vacuum suction trucks).

In field installations, stormwater would enter the BGS facility in the form of hydrographs with varying discharges and sediment concentrations. Consequently, trapping efficiencies would vary with the varying Q, and the sediment influx and characteristics, including the particle size distributions and densities. Such conditions were tested by He *et al.* (2014) in a simplified field experiment, in which the grid structure was mimicked by batteries of open-top plastic containers ($22 \times 22 \times 11$ cm, $L \times W \times H$) attached to the pond bottom at 10 m downstream from the pond inlet. The results showed that these containers representing individual cells retained 4–11 times more sediment mass (with particle diameters $D < 250$ μm and $D < 32$ μm, respectively) than the bare pond bottom over a period of three months. The capture of very fine particles ($D < 32$ μm) was particularly surprising and could result from the settling of flocculated particles after the cessation of runoff. These findings also point out the importance of field testing of sediment trapping devices. The effect of particle sizes on sediment trapping was not addressed in our experiments using a model sediment with a single particle size. In laboratory experiments performed by He & Marsalek (2014), it was noted that the BGS trapped higher rates of larger particles than smaller ones, especially for higher flow rates. Finally, consideration of using the BGS sediment trap in new or retrofitted ponds would require a site-specific assessment of cost and benefits.

CONCLUSIONS

A scale-model study of the Bottom Grid Structure (BGS) sediment trap at the inlet of a stormwater impoundment produced information on the feasibility of such a pre-treatment of stormwater entering the impoundment, with the ultimate objective of reducing maintenance costs. At this phase of research, the following conclusions can be drawn: (a) the BGS would effectively confine coarser sediment (bedload) deposits to a small area in the pond; (b) as tested in this study, the BGS effectiveness was improved by placing it into an inlet stilling basin comprising a diffuser with wing walls, an apron and a water level control sill; and (c) concerning the settling cell geometry, larger cells (10 cm, compared to 5 cm cells) appeared more effective, increasing the efficiency by 13%; no strong influence of the cell depth was noted, and while inclined cross-walls improved settling in the model, their use might be counterproductive because of the costlier maintenance requirements. The next research phase should focus on the need for the stilling basin structure and the optimal geometry of BGS cells. Ultimately, the BGS concept should be tested in the field.

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