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Urban Flow-Through Facilities' Media Compositions for Stormwater Quality and Quantity Improvements

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Urban flow-through facilities' media compositions for stormwater quality and quantity improvements

By

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Urban stormwater management is evolving toward sustainable approaches which rely on dispersed small-scale bioretention BMPs. One such BMP is the flow-through planter, commonly applied in areas where infiltration into *in situ* soil is restricted or not possible. A project was developed to evaluate 18, vertically scaled flow-through mesocosms. Three replicates of six treatments, including four soil mixtures containing varied percentages of sand, compost and topsoil, were tested for orthophosphate and nitrate removal, volume reduction capabilities, and peak flow attenuation through the application of a synthetic solution over a simulated 2-inch, Type II storm event. Runoff volume was significantly ($p < 0.05$) reduced compared to controls. Nutrient levels observed along the hydrograph at different time-steps and flow rates revealed patterns not apparent in cumulative results. The observation of preferential flow patterns along with variability in nutrient removal across treatments highlights the need for design modifications of flow-through facilities.

DEDICATION

I dedicate my research and thesis to my parents Nancy and Jim Overbey and sister Alyson Overbey for their endless encouragement, support, and love. Their emotional support and friendship is the reason I have made it to where I am today. I also dedicate my work to my grandparents Nona Faye and Jim Overbey and Lorraine and Jim Thompson. They laid the path for the success of all their grandchildren and I will forever be grateful for their love and support. Lastly, I thank my partner in life and fiancé, Jason Brandt for his unconditional love and faith in me. His willingness to listen and offer advice when I needed it most kept me focused and determined throughout graduate school.

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER I

INTRODUCTION

Stormwater runoff management is evolving from centralized detention methods toward sustainable approaches which rely on dispersed small-scale bioretention solutions (Debo and Reese 2003). These solutions are used to target both water quantity and quality problems that contribute to the degradation of receiving surface waters. One such bioretention solution is the flow-through planter, commonly applied in urban areas where infiltration into *in situ* soils is restricted or not possible (BES 2006). Flow-through planters are a relatively new bioretention tool and with further research, show potential for stormwater runoff mitigation in the urban environment.

This study was developed to examine various bioretention soil mixtures in flow-through planters with the hope of determining which soil mixtures were most effective for volume, peak flow, and nutrient reductions. Though there have been studies that have examined bioretention soil media for water quality and quantity improvements, no studies to date have applied a simulated hydrograph or have explored the structural design of the flow-through planter.

The need for further research to determine ideal soil mixture composition, especially in regards to nutrient reductions was outlined by Davis et al. (2009) and Roy-Poirier et al. (2010). The lack of literature on the effects of an actual storm event on volume and nutrient reduction in bioretention highlight the importance of this study.

Additionally, bioretention best management practices (BMPs) such as raingardens are effective for volume and nutrient reduction; however a limited number of studies have focused on flow-through planters which are more conducive to constrained urban spaces than more land intensive stormwater management solutions (BES 2010).

This thesis is organized into the following chapters: Literature Review, Methods and Materials, Results, Discussion, and Conclusions. The literature review outlines existing literature focused on stormwater management, bioretention components, bioretention soil media recommendations, bioretention field and laboratory research for water quantity and quality, and geochemical processes responsible for nutrient fate in soils. The Methods and Materials chapter discusses the experimental design, procedures for data collection, application of synthetic stormwater solution and simulated hydrograph, and analysis of collected data. The Results chapter presents water quality and quantity results and comparisons between soil media treatments. The Discussion chapter relates the results to findings from literature reviewed. Finally, the conclusions chapter discusses overall observations, recommendations for future research, limitations, and applications for landscape architects.

CHAPTER II

LITERATURE REVIEW

Emerging sustainable approaches to stormwater runoff management have been at the forefront of BMP research for the past decade. This literature review explores a wide range of research related to the components of sustainable stormwater management, specifically the application of bioretention as a small-scale stormwater BMP for water quantity and quality improvements. A brief overview of stormwater management and the need for a shift to more sustainable small-scale bioretention solutions is followed by a look at research related to the application of bioretention facilities. Research limitations regarding the application of bioretention for runoff volume reduction, peak flow mitigation, and nutrient removal are discussed. Additionally, a brief overview is given on biogeochemical processes responsible for achieving nutrient reduction in soil.

2.1 Stormwater Management

The properties of stormwater and how it travels over land depend on the types of surfaces it encounters. When a storm event occurs on land that is undeveloped, processes such as interception with vegetation, filtration into the soil, evapotranspiration, and overland flow clean stormwater and reduce peak flows (Horner 1994). However, in developed or urban conditions, stormwater encounters impervious surfaces such as rooftops, roadways, lawns, and parking lots which limit natural filtration processes and increase peak flows (Bannerman et al. 1993; Horner et al. 1994).

In a comprehensive history of municipal stormwater management, Debo and Reese (2003) describe paradigm shifts in stormwater practices. Prior to the 1950's the primary goal in the management of stormwater runoff was to direct it away from urban areas as quickly and as efficiently as possible. This was achieved by conveying stormwater and untreated sewage together in combined stormwater and sewage systems directly to nearby water bodies such as streams, rivers, and lakes. With this combined system, however, concerns for untreated sewage, contaminated runoff, and toxic discharges brought about an effort to separate stormwater from raw sewage. In the 1960's municipalities began separating stormwater from sewage pipes to increase the capacity of sewage treatment plants during storm events (Debo and Reese 2003).

In response to flood concerns, cities and states in the 60's and 70's began to adopt ordinances regulating peak flow and volume control of stormwater. This was accomplished by incorporating large detention basins within developments that store stormwater on site and release it slowly over time (Debo and Reese 2003). However, this conventional approach to stormwater management did not manage for water quality (PGC 1999), contributing to widespread surface water impairment.

In 1962, Rachael Carson in her book "Silent Spring" made the connection between pollutants that enter surface waters and the detrimental effects they have on aquatic, animal and human health. This book heightened awareness and inspired public concern for the health of the environment, especially surface waters. It also helped to initiate the modern environmental movement, which eventually led to the development of the Clean Water Act (CWA) in 1972. The CWA established basic regulations for pollutant discharge into surface waters and mandated standards for water quality, which have been implemented over time in several phases (EPA 2000a).

In 1990, Phase I of the National Pollutant Discharge Elimination System (NPDES), which regulates stormwater runoff under the CWA, was implemented. Under Phase I any building project over 5 acres and cities with populations over 100,000 were required to obtain a stormwater discharge permit (EPA 2000a). In 1999, Phase II expanded the program to include building projects 1 acre or more and medium sized cities. The NPDES program led to the adoption of structural and non-structural BMPs to treat stormwater from developed areas.

In response to the NPDES program, cities began to adopt water quality standards in addition to flood control ordinances to reduce total maximum daily loads (TMDL's) as defined by the EPA and began monitoring surface water pollution. These water quality and quantity standards are typically met with large, concentrated detention facilities and other structural BMPs (EPA 1999a). As these programs evolved, cities began to refine ordinances and approaches to meet their individual stormwater management needs. Today, green infrastructure practices have emerged to manage stormwater holistically with small, dispersed facilities which focus on managing water quality and quantity at the source (Holman-Dodds 2007; Wise 2008).

2.1.1 Stormwater Quantity

The addition of roadways, parking lots, lawns, and rooftops increases impervious surface area in urban environments and drastically changes the hydrology of a watershed (Booth and Jackson 1997). Impervious surfaces prevent water from infiltrating into the soil which greatly increases the volume and rate of stormwater runoff created during storm events (Schueler 1994; Arnold and Gibbons 1996). Schueler (1994) explains how urban runoff is conveyed swiftly to local surface waters, resulting in higher peak flow rates which are responsible for environmental degradation such as:

- down cutting and widening of stream banks which cause overall stream instability;
- increased sedimentation and erosion;
- losses of instream habitat structures;
- accumulated pollutants which disrupt aquatic ecosystems and biodiversity of water bodies;
- increased stream temperatures which result in higher air temperatures and disruption of aquatic life;
- decreased downstream water quantity and quality which effect fisheries productivity.

The development of at least 10% impervious surfaces can negatively impact urban stream health and water quality (Booth and Reinelt 1993; Schueler 1994; Booth and Jackson 1997).

2.1.2 Stormwater Quality

Stormwater runoff from urban environments travels over impervious surfaces, collecting pollutants, debris, and sediment it comes in contact with (Booth and Reinelt 1993; EPA 2003). The primary sources of pollutants in urban environments include:

- nutrients and pesticides from lawns and gardens;

- sediment eroded from construction sites and bare earth;
- heavy metals from atmospheric deposition (both natural and from vehicles), factories, and roof shingles;
- oil, grease, and other hydrocarbons from motor vehicles;
- road salts from ice removal;
- viruses, bacteria and nutrients from pet waste and failing septic systems; and
- thermal pollution from impervious surfaces such as streets and rooftops.

When a storm event occurs after a dry period, pollutants that have settled on impervious surfaces wash off in concentrated amounts as stormwater runoff during the first 1.27-2.54 cm (0.5-1.0 in.) of an event, creating what is known as the "first flush" (Lee et al. 2000). The first flush carries the highest amount of concentrated pollutants and should be targeted with sustainable stormwater management approaches which mitigate resulting adverse effects such as the degradation of fish and wildlife populations, killing vegetation, impairing drinking water, and rendering recreational areas unsafe (EPA 1996; Lee et al. 2002).

Nutrients carried in stormwater such as phosphorus (P) and nitrogen (N) are of concern when they reach surface waters in elevated concentrations. Overloading of P and N in waters leads to eutrophication and subsequent degradation of sensitive receiving aquatic ecosystems such as streams, estuaries, lakes and larger bodies of water (Carpenter et al. 1998). The adverse environmental effects of urban stormwater runoff are increasingly being managed with sustainable stormwater management strategies and green infrastructure practices (Debo and Reese 2003).

2.2 Sustainable Stormwater Management

After the CWA and both phases of the NPDES were implemented, states were required to implement structural and non-structural BMPs in order to reduce point and nonpoint source pollution and peak flows (EPA 2000b). Mitigating both water quantity and quality problems in the urban environment where space is limited can be complex; however, there are a number of innovative approaches and tools that have been developed which assist stormwater managers and designers with this challenge.

The Center for Watershed Protection names three specific approaches which are most commonly used in stormwater management: low impact development (LID), green infrastructure (GI), and environmental site design (ESD). While all three approaches have slightly different meanings, they all have the same core goals. Each approach addresses stormwater sustainably, on a site-by-site basis, and at the source, all by relying on the cumulative impacts of many small-scale BMPs (EPA 2000b; CWP 2013).

These sustainable stormwater management (SSWM) approaches also provide greater opportunity for designers to integrate stormwater solutions into the urban environment, utilizing stormwater as an amenity instead of considering it a burden. Utilizing stormwater as an amenity provides designers with additional tools to create aesthetically pleasing art features, also known as Artful Rainfall Design (Echols and Pennypacker 2008). For the purpose of this research, SSWM is the general term that will be used when referring to these approaches.

Several aspects of SSWM set it apart from traditional stormwater management practices. One distinct difference is managing stormwater where it falls (as close to the source as possible) in order to mimic predevelopment hydrology and encourage infiltration into *in situ* soils (Holman-Dodds et al. 2007). Traditional approaches redistribute stormwater volume over time; however, they do not reduce the volume of stormwater runoff that leaves a site. Sustainable approaches do not place management solely on centralized detention, rather they redistribute the cumulative impact of a storm event onto several decentralized BMPs (Boller 2004). Small-scale BMPs are adaptable and can be retrofitted into existing urban conditions which allows management of stormwater as close to the source as possible (i.e. directly adjacent to buildings, parking lots, and streets) (BES 2008).

Sustainable stormwater management is also site-specific. When stormwater is managed with unique site conditions in mind, considerations such as local soil conditions, rainfall patterns, and pollutants can be accounted for (Holman-Dodds et al. 2007). In contrast, traditional stormwater management approaches are more of a “one size fits all” solution which controls primarily for water quantity but do not typically take into consideration specific site conditions (Boller 2004). Additionally, traditional approaches utilize large detention facilities which temporarily store stormwater and eventually move it away from the site, drastically changing site hydrology as well as transporting contaminants downstream (Debo and Reese 2003).

Best management practices that help support the goals of SSWM are compact and easily integrated into existing urban conditions. Managing at the source with SSWM BMPs helps to mitigate and prevent a number of negative environmental impacts while providing visual amenities in areas that would otherwise be mostly impervious. Specific SWWM BMPs that are used to help mimic predevelopment hydrology and filter out harmful urban stormwater pollutants include but are not limited to:

- green roofs,
- permeable pavement,
- urban forestry,
- sand filters,
- rainwater harvesting, and
- bioretention (i.e. raingardens, bioswales, stormwater planters, etc.).

2.3 Bioretention Based BMPs

The most utilized SSWM BMPs are bioretention based facilities, commonly referred to as raingardens, bioswales, bioretention cells or stormwater planters (Davis et al. 2009). Bioretention facilities are small basins or depressions that typically consist of gravel with a perforated underdrain pipe, an amended soil media layer, a layer of mulch, and appropriate native vegetation (EPA 1999b). Compared to large detention basins, which are land intensive, bioretention facilities provide the opportunity for stormwater management design solutions in tight urban spaces (Holman-Dodds 2007; LID 2007). The compact size and flexible design of bioretention facilities can help to integrate stormwater management into the urban environment where stormwater treatment previously did not exist. Small-scale bioretention facilities are easily adaptable to both residential and commercial lots and are commonly situated on streetscapes, next to buildings, adjacent to parking lots, in parking lot islands, and in residential yards (EPA 1999b; PGC 2007). Designing BMPs that fit in these smaller spaces allows management at the source where stormwater can be intercepted and treated before reaching storm drains, detention ponds, or water bodies (PGC 2007).

Bioretention based BMPs utilize plants, soils, and microbes to cleanse stormwater runoff and reduce volume and runoff rates to predevelopment conditions (EPA 1999b; BES 2008).When stormwater runoff enters a bioretention facility it percolates through the mulch and soil layers where it is absorbed by the soil, taken up by plants, evaporates and exfiltrates to surrounding soils (EPA 1999b). As runoff is absorbed into the media and filtered, pollutants bind to soil particles where biological, physical, and chemical processes can occur which break down or retain pollutants. The filtering of stormwater through soil media is key to removing pollutants and, therefore, is a critical layer in the bioretention facility. Encouraging environmental processes such as absorption, adsorption, and filtration helps mediate both stormwater quantity and quality issues that occur as a result of urban stormwater runoff (Booth and Jackson 1997).

2.3.1 Bioretention Components

Bioretention facilities are constructed in layers, each one having specific functions that contribute to the management of stormwater runoff (BES 2008; Hunt et al. 2011). Some layers are constant across each type of bioretention facility, while a few layers vary depending on the infiltration configurations which are discussed later. The constant layers of bioretention facilities are a reservoir, plant material, mulch and soil media. The variable layers are a soil filtering layer (filter fabric), gravel and underdrain (Hunt et al. 2011).

The reservoir is the area located between the planting surface and the crest of the side walls or berm surrounding the facility. This area of the facility allows entering stormwater to pond as it infiltrates into the soil media. The size of the reservoir can be designed with a range of depths which are typically between 15.24 cm and 30.48 cm (6.0

in. and 12.0 in.) depending on the sizing requirements (BES 2008). The larger the reservoir volume, the larger event the facility is capable of managing.

The soil media layer is an amended engineered soil mix that is typically composed of a mixture of sand, topsoil, and compost (EPA 1999b; PGC 2007; BES 2008; Hunt et al. 2011). The soil media provides numerous benefits such as infiltration, detention, retention, enables plant establishment and growth, and increases chemical and microbial processes which breakdown nutrients and other urban pollutants (EPA 1999b). The specific percentages of sand, topsoil and compost vary greatly between sources and are discussed later in this section. Other soil media specifications include minimum and maximum pH values, range of allowable clay content, and minimum and maximum infiltration rates (LID 2003; PGC 2007; BES 2008).

The depth of the media layer ranges from 30.48-60.96 cm (12.0-24.0 in.) (LID 2003; PGC 2007; BES 2008; Hunt et al. 2011). However, most recommendations specify a minimum of 45.72 cm (18.0 in.) to provide stormwater adequate contact time with the soil for pollutant removal, volume reduction and peak flow reduction (PGC 2007). Water residence time in the soil is critical for chemical and biological processes to occur for nutrient reduction (DeBusk and Wynn 2011). The soil media layer is also responsible for reducing some volume by absorbing water in void space and retaining it in the system (Roy-Poirer et al. 2010; Hunt et al. 2011).

Mulch is placed over the soil mix to hold the soil in place, to control weeds, and to keep the soil from drying out. Mulch has also been shown to reduce pollution by retaining some heavy metals, hydrocarbons, and nutrients (Hunt et al. 2011). Placing mulch in bioretention facilities also stabilizes plants, which are an integral part of bioretention facilities.

Appropriate plantings in bioretention facilities also provide the added benefits of increased aesthetic value, possible wildlife habitat, pavement shading, nutrient reduction and improved air quality (LID 2003; PGC 2007). Bioretention facilities are periodically inundated with stormwater, requiring plants that can tolerate both wet and dry conditions. Native plants are encouraged as they should be able to withstand local climate conditions and often require less irrigation (PGC 2007).

Any water not up taken by plants, evaporated, or absorbed by soil, continues to percolate down through the facility. If *in situ* soils have adequate infiltration rates and there is not concern for contamination, water is allowed to continue to filter through the layers and exfiltrate into surrounding soils. However, if these conditions are not met, an underdrain layer which is composed of components including: a gravel layer, perforated underdrain pipe, soil filtering layer, and/or overflow pipe must be installed (BES 2008).

If a bioretention facility is fitted with underdrain components it will contain a gravel layer at the bottom which primarily serves as an added storage layer (BES 2008). When the rate of the water entering the system exceeds the rate at which it can exfiltrate through *in situ* soils or drain through an underdrain pipe, the gravel layer provides temporary storage. Depth specifications of the underdrain gravel layer typically range between a minimum of 30.4 -60.96 cm (12.0-24.0 in.) (LID 2003; PGC 2007; BES 2008).

If exfiltration into existing, surrounding soils is not allowed or limited, a perforated underdrain pipe can be fitted within the gravel layer. This perforated pipe runs the facility length and empties into a storm drain or a receiving water body (PGC 2007).

If a storm event occurs that exceeds the capacity of the bioretention facility, an overflow pipe or spillway is necessary. This overflow pipe leads from the underdrain layer back up to the surface where water is directed to a storm drain or away from the system (PGC 2007; BES 2008).

As water filters through the soil layer it can carry with it small soil particles that can result in clogging of the underdrain gravel layer and perforated pipe. In order to prevent sediment from the soil media layer from clogging the underlying gravel layer, filter fabric or pea gravel is placed between the soil layer and gravel layer. Specifications for filter fabric require that is a non-woven geotextile fabric, which allow water to freely pass through while blocking finer particles from leaving the system (BES 2008).

2.3.2 Variations in Structural Design

Bioretention based BMPs have two primary structural variations for creating the reservoir: basin or planter. A basin uses sloped sides to create the desired reservoir depth, a 3:1side slope ratio is common for creating basins with bioretention soil mixes to reduce the chances of erosion (BES 2008). In contrast, planters use structural side walls to create the desired reservoir depth. Side walls can be constructed out of any structural material such as concrete, dry-laid stone, or timber.

Due to the cross-sectional differences between the two structural variations, a planter will manage a larger event in the same land area or, conversely the same event in a smaller amount of land area than a basin (BES 2008; Gallo et al. 2012). This makes the planter ideal for tight, urban spaces. However, the additional materials to construct side walls make planters a more costly option than basins.

2.3.3 Variations for Infiltration Configurations

In general, there are three infiltration variations for bioretention facilities: full infiltration, partial infiltration or flow-through. The required configuration depends on local site conditions. Infiltration into *in situ* soils is not possible when the BMP is located adjacent to a building foundation, native soils consist primarily of heavy clays, the site is potentially contaminated (such as at a fueling station or industrial site), the facility is on top of a structure such as a roof, the water table is too high (BES 2008), or if karst conditions exist in the area (DeBusk and Wynn 2011). If at least one of these site conditions exist, a flow-through bioretention facility is required. As water moves through the facility it is absorbed by the soil media layer or taken up by plant material. Water that is not absorbed drains through the soil layer to a gravel layer where it is collected by an underdrain system that moves the treated runoff away from the system (BES 2008).

According to PGC's bioretention manual (2007), full infiltration is the most frequently constructed configuration of a bioretention facility in residential settings as it is the easiest to implement. The effectiveness of a system depends on the infiltration rate of the native soils, as faster soil infiltration rates generally mean a more effective facility in terms of water volume reduction and pollutant removal. Full infiltration systems found primarily in residential areas are commonly referred to as raingardens.

A partial infiltration configuration is used when the native soils have some infiltration capabilities but the rate of infiltration is slower than that of the bioretention media (BES 2008). Similar to the flow-through facility, water not absorbed in the soil media layer drains through the soil to a gravel layer which contains an underdrain configuration that carries treated runoff away from the facility. The partial infiltration facility, however, requires the underdrain configuration to be located at the top of the gravel layer instead of at the bottom. This configuration allows for additional storage at the bottom of the facility, which permits some amount of water to filter over a longer period of time (Hunt et al. 2011).

2.3.4 Sizing Bioretention Based BMPs

A popular method for calculating urban runoff hydrology due to its relative simplicity is the Santa Barbara Urban Hydrograph (SBUH) method (Debo and Reese 2003). The SBUH is based on several variables such as pervious and impervious land areas, time of concentration calculations, runoff curve numbers applicable to the site, and storm size (BES 2008). Additional variables include reservoir depth and infiltration rate of the designed soil media (Gallo et al. 2012).The method utilizes synthetic runoff curves based on geographic location to predict rainfall distribution and intensity (USDA 1986). To size BMPs with the SBUH, a time-step based spreadsheet is used to calculate inflow and outflow every 10-minutes and determine the required facility size (BES 2008).

According to Gallo et al. (2012), sizing small-scale bioretention facilities in different regions of the country requires specific climatic considerations. Typically bioretention is used to manage 2.54-5.08 cm (1.0-2.0 in.) storm events which cover the water quality event but not necessarily water quantity requirements. In the Pacific Northwest bioretention can be sized for larger events due to less intense rain events, which allows for the management of both stormwater quantity and quality. Other regions of the country experience more intense rain events which make sizing bioretention facilities for quantity control more difficult.

2.4 Current Bioretention Soil Media Recommendations

Federal, state and city level guidelines for bioretention soil mixtures vary across the United States. Differing recommendations attempt to balance the primary design objectives of high enough infiltration rates to provide adequate drainage, low enough infiltration rates to allow for adequate contact time between runoff and soil media for pollutant removal, and plant and soil health (Hinman 2009).

Most bioretention recommendations provide general soil type specifications such as the use of sandy loam or loamy sand due to their adequate infiltration rates (LID 2003; PGC 2007; BES 2008). Guidelines additionally specify a limit for soil pH and clay content as soil acidity affects the ability of the soil to adsorb P and the microbiological activity occurring within the soil (O'Leary 2002). Recommendations range between soil pH levels of 5.5 and 7.0 (LID 2003; PGC 2007; BES 2008) which is the pH range at which P is most available for plant uptake (Busman et al. 2002). Clay content recommendations vary depending on desired infiltration rates and targeted pollutants with suggested proportions ranging from less than 5% to as much as 15% (LID 2003; PGC 2007). Several municipalities recommend a more specific breakdown of soil media composition which consists of 50-60% sand, 20-30% leaf compost, and 20-30% topsoil (Davis and McCuen 2005; PGC 2007). Portland Oregon's 2008 Stormwater Management Manual, which is widely referenced across the U.S., recommends a loamy soil, sand and compost that is 30-40% compost by volume, and have a pH between 6.0 and 8.0 (BES 2008).

2.5 Bioretention Research

Gaps exist in bioretention research related to a range of design and performance issues such as pollution prevention and removal, peak flow reduction, soil/filter media composition, treatment processes, retention, and time of concentration issues (Davis et al. 2009). Hunt et al. (2011) further summarized specific research findings and outlined bioretention design recommendations that attempt to balance hydrologic performance and water quality goals. Hydrologic performance and nutrient removal capabilities of bioretention facilities are promising and rely on similar variables, but research outlining the actual removal efficiency of sediment and pollutants by bioretention systems differs, likely because of the wide range of experimental methodologies. Experiments differed and included variations in media compositions, input load concentrations and volumes, inflow rates, media depths, facility configurations, and whether it was conducted in a lab or field.

Lab and field experiments testing a range of variables affecting bioretention have focused on water quality and quantity results from either actual rainfall events or application of a synthetic stormwater solution. Compositions of synthetic stormwater created for experiments ranged widely and have included nutrients, suspended solids, heavy metals, hydrocarbons, and pathogens. However, most common pollutants tested in all experiments were nutrients (specifically, P and N). Field experiments have focused on the performance of installed bioretention cells, while lab experiments were typically conducted in soil columns, box studies or mesocosms. Literature reviewed for the purposes of this research will focus on bioretention soil media performance for volume reduction, peak flow mitigation, and P and N removal.

2.5.1 Volume and Peak Flow Impacts

Bioretention combines natural and engineered processes to manage stormwater runoff with the goal of restoring a site's predevelopment hydrology (EPA 1999b). Outflow volume and peak flow reduction are two important components for reaching these goals and are directly linked to pollutant load reduction (Li and Davis 2009). Bioretention can reduce runoff volume by promoting exfiltration of filtered runoff into the surrounding *in situ* soils and by storing runoff in the engineered soil media until it can be evaporated or assimilated by plants (EPA 1999b; PGC 2007; BES 2008). Peak flow mitigation has proven to be more difficult in bioretention systems due to high infiltration rates of engineered soil media and other design restrictions such as media depth and reservoir depth (Hunt et al. 2011); however, peak flow reductions were observed in field studies conducted by Davis (2008), and Hunt et al. (2008).

Stormwater volume reduction and peak runoff rates are strongly influenced by the total amount of rainfall, maximum intensity of the storm event, volume and depth of bioretention soil (Hatt et al. 2008; Davis 2008; DeBusk and Wynn 2011). Significant volume (97%) and peak flow (99%) reductions found in a bioretention field experiment by DeBusk and Wynn (2011) were attributed to deeper than standard soil depths (0.6-1.2 m). Brown and Hunt (2010) examined the impact of media depth on hydrology by evaluating two loamy-sand filled bioretention cells at varied depths (0.6 m and 0.9 m (2.0 ft. and 3.0 ft.)), and results indicated deeper media depths achieved greater runoff volume reductions (44% reduction for 0.9 m (3 ft.) media compared to 21% reduction for 0.6 m (2 ft.) of media). Deeper media and increased media volume in bioretention systems allows for increased amounts of area exposed to *in situ* soils for exfiltration, additional contact time for media to absorb runoff, and increased residence time to allow for evapotranspiration and uptake by plants (Davis 2008; Li and Davis 2009; Brown and Hunt 2010; Debusk and Wynn 2011).

Various studies have noted the effects of bioretention on peak flow reduction. A comparison study between a bioretention cell that was constructed with an impermeable liner and one that was not indicated hydrologic performance was much worse in the lined cell (Li et al. 2009). Davis (2008) observed significant peak flow reductions (44-63%) were also observed in two installed bioretention cells that were surrounded by liners but they still performed worse than similar unlined cells in the bioretention study by Li et al. (2009). Additionally, greater inflow rates and volumes decreased the effectiveness of bioretention for hydrologic performance, indicating bioretention is much more successful at volume reduction at lower inflow rates and inflow volumes (Li et al. 2009; DeBusk and Wynn 2011).

Another concern related to hydrologic performance of bioretention is the occurrence of preferential flow paths within bioretention soil media. A study conducted by Hsieh and Davis (2005) showed preferential flow paths in lab columns with a depth of 0.8 m (2.63 ft.) resulting in decreased runoff contact time with the media which reduced pollutant reduction. Higher than expected infiltration rates and preferential flow paths in soil media have also been noted in installed flow-through facilities and lab experiments in the Portland, Oregon area (BES 2009; BES 2010). Hunt et al. (2011) point out that media and drainage configuration should be designed to minimize relatively high flow rates and preferential paths through the bioretention facility due to the direct relationship between hydrologic performance and nutrient reduction capabilities. This emphasizes the importance of bioretention soil volume as well as depth.

2.5.2 Phosphorus Removal

Phosphorus is the limiting nutrient for production of algal growth in freshwater ecosystems. Overloading of P in freshwater can lead to eutrophication resulting in depleted dissolved oxygen, which negatively impacting water quality (Brady and Weil 1996; Reddy and DeLaune 2008). Geochemical processes of absorption and/or precipitation are the primary processes for PO_4^{3-} retention in soils (Bohn 2001). Adsorption processes in bioretention mesocosm studies conducted by Lucas and Greenway (2008) were found to be the primary reason for PO_4^{3-} removal. Vegetated mesocosms, however, showed significant total P retention in unsaturated bioretention media when compared to barren mesocosms, illustrating the contribution of plants for the removal of P (Lucas and Greenway 2008).

Phosphate ions adsorb strongly to soil particles, especially finer particles such as clay and silt (Busman et al. 2002). Sorption of P is associated with clay content due to the density of sorption sites; which is why sandy soils have less absorption capabilities compared to soils with higher clay content (Ige et al. 2007). Adsorption of P to soil particles was also recognized as the primary removal mechanism in column experiments, especially when slower percolation rates existed (Erickson et al 2007; Hsieh et al. 2007b). Lucas and Greenway (2008) found that mixtures with greater organic matter reduced P at a greater rate than those with 100% sand. Increased contact time in the media and higher silt, clay and organic matter contents can enhance P adsorption capacity in bioretention facilities (Li and Davis 2009).

The ability of soil to adsorb P is also a function of soil pH, as adsorption of P to soil particles steadily decreases as pH values increase (Goldberg and Sposito 1985; Busman et al., 2002). Uptake by plants utilizing P for growth can occur if the soil pH is between 6.0 and 7.0 (Busman et al. 2002). Therefore, soil recommendations for bioretention typically range between 6.0 and 7.0 for ideal P uptake by plants.

Phosphorus removal by bioretention has been successful in a number of lab and field studies (Davis et al. 2006; Hunt et al. 2006; Lucas and Greenway 2008; Hatt et al. 2009). Total P removals in a bioretention box study (similar to flow-through configurations) indicated high removal rates (70-85%) of P (Davis et al. 2006). Davis et al. (2006) varied the inflow rate and duration applied which were found to affect nutrient removal, with higher flow rates decreasing the effectiveness of the system to remove nutrients. Hunt et al. (2006) reported P load reductions in field bioretention of 65% and concluded that the baseline content of P in the treatment media greatly influenced the total P removal rate of the bioretention facility.

2.5.3 Nitrogen Removal

Nitrate removal performance in bioretention studies was highly variable due to a possible lack of necessary geochemical processes provided by internal water storage and anaerobic conditions (Hunt et al. 2011). Nitrate is highly mobile as water passes through soil (O'Leary et al. 2002). Nitrate does not sorb to soil particles as P does, making, therefore NO_3^--N removal in bioretention is more difficult to achieve (Brady and Weil 1996; Davis 2008). Production of NO_3^--N occurs when aerobic conditions for nitrification are present with nitrification occuring rapidly in soils that are warm, moist, and well aerated (O'Leary et al., 2002). Denitrification, the conversion of NO_3^--N to nitrogen (N_2)

gas, occurs when anaerobic conditions, created by a saturated zone within soil, exist (Reddy and DeLaune 2008; Hunt et al. 2011).

Successful NO_3^--N removal results in bioretention studies were observed when a saturated layer was added to the design which promotes denitrification processes necessary for the breakdown of NO_3^--N (Kim et al. 2003; Passeport et al. 2009). Soils with adequate organic matter and an anaerobic zone are expected to perform well for NO_3^--N removal due to microbial activities that rely on organic matter for denitrification processes (Reddy and DeLaune 2008). Additionally, lower inflow or slower infiltration rates increase retention in the soil media, which provides additional time for possible denitrification processes or uptake by plants (Hunt et al. 2008). Plants assimilate N in the form of NO_3^--N, indicating possible increased NO_3^--N removal when plants are present in bioretention systems (O'Leary et al. 2002).

A bioretention box study performed by Davis et al. (2006) noted variable N removal results with NO_3^--N increases in effluent observed in most cases. Observed NO_3^--N production was attributed to nitrification processes that were suspected to have occurred in the shallower parts of the facility. Increased flow rates applied to the bioretention boxes may have also resulted in decreased nutrient removal in these box studies (Davis et al. 2006). Depending on flow rates and soil oxidation-reduction (redox) characteristics, it is possible that some denitrification processes could occur in bioretention soil media (Davis et al. 2006). Nitrate increases were also noted in a field experiment conducted by Brown and Hunt (2010). This field experiment illustrated the importance of deeper media depths in achieving greater reductions in runoff volume and NO_3^--N.

Variable NO_3^--N reductions were also observed in bioretention field studies in North Carolina (Hunt et al. 2006). The decreases in NO_3^--N were attributed to small pockets of saturated soil which could have provided denitrification processes. Similar variable results of NO_3^--N were also reported in a field experiment by Dietz and Clausen (2006), even when a designed saturated water zone was included. Kim et al. (2003) found that the addition of newspaper shreddings for an additional carbon source promoted conditions conducive for denitrification processes in a column study, with removal rates of NO_3^--N as high as 80%.

Davis (2007) also experimented with field-scale bioretention cells, with one cell having a designed saturated anoxic layer and the other no anoxic layer. While there were no significant differences found between the two cells, naturally occurring saturated zones were observed in the cell not designed with an intended saturated zone. In both cells 90-95% of NO_3^--N was removed. Bioretention studies containing a saturated zone within the lower portion of the soil media resulted in increased residence time of water and therefore NO_3^--N, which allowed for denitrification processes to occur (Hunt et al. 2006; Hsieh et al. 2007a; Hunt et al. 2008; Li and Davis 2009).

2.5.4 Literature Strengths and Gaps

The literature examined in this review covers the history of stormwater management, current sustainable practices and research of bioretention solutions for water quantity and quality improvements. The emerging practice of sustainable stormwater management has warranted a range of research topics which have provided insight for BMP applications. Bioretention laboratory and field research has been extensive for the past decade and comprehensive recommendations are beginning to emerge (Hunt et al. 2011). Factors such as flow rates, flow duration, bioretention media depth, media composition, media volume, and site conditions have all been identified as key components critical to the mediation of urban stormwater quality and quantity problems. For the purposes of this study, bioretention literature focusing on nutrient results as well as volume retention and varied flow rates were outlined. To date, no literature has applied a real-world simulated hydrograph or examined a replicated flow-through planter in a controlled lab experiment which is a primary reason that the current study was developed.

CHAPTER III

METHODS AND MATERIALS

3.1 Introduction

In the summer of 2012, a controlled bioretention soil experiment was conducted at Mississippi State University's (MSU) South Research Farm in Starkville, Mississippi. The experiment was placed in an indoor aquaculture research barn, which allowed for a controlled environment that eliminated most external environmental variables such as rainfall, wind, and atmospheric pollution.

Mesocosms were constructed to be replicates of, vertically scaled flow-through planters following specifications provided by the Portland, Oregon's Bureau of Environmental Services (BES). Portland's specifications were selected because they are widely referenced in SWMMs across the country. Additionally flow-through planters have a single release point which allow for controlled measurements of outflow. Mesocosms were vertically scaled to a fourth of the BES typical flow-through planter design detail to be able to recreate the design proportions of the planters in a lab setting (BES 2008) (Figure 3.1). Sizing the mesocosm layers to a fourth the size of a typical flow-through planter also allowed for a more manageable experiment size in a controlled environment that could be easily replicated and modified in the future as needed.

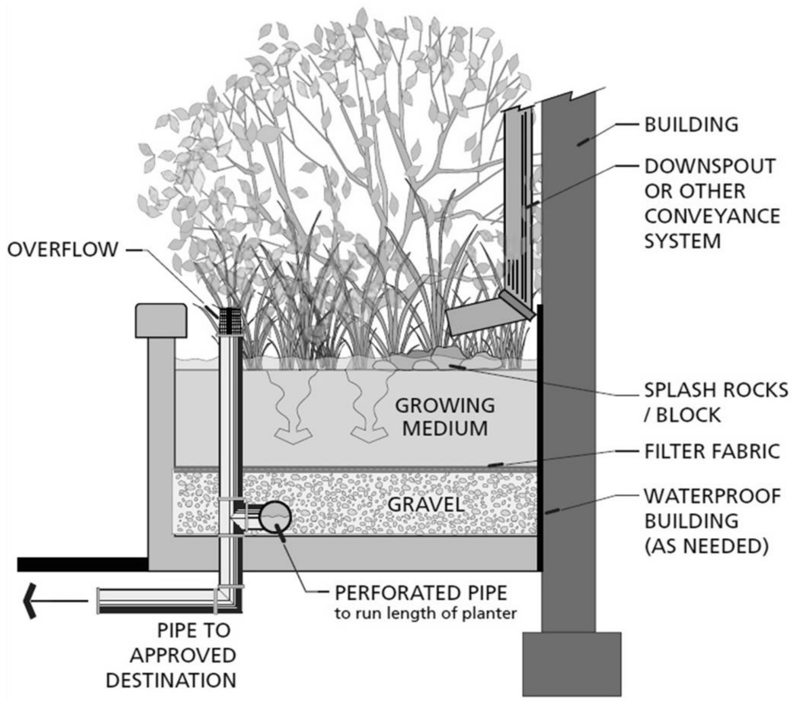


Figure 3.1 Vertical section view of a flow-through planter

Notes: Typical flow-through planter section provided by Portland's BES stormwater management manual (2008).

This study examined volume, peak flow, PO_4^{3-} and NO_3^--N retention capabilities of various bioretention soil mixtures by applying a synthetic stormwater runoff solution over a hydrograph to replicate an actual storm event. Change in volume of all treatments was calculated and compared to the set of controls to determine volume reduction potential of each soil media mixture. Concentrations and load changes for PO_4^{3-} and NO_3^--N were calculated for each treatment and compared to the controls for cumulative results. Concentration and load change results for treatments at each individual time-step and between treatments at each time-step of the hydrograph were also examined. Additionally, the hydrograph was regrouped by time-steps to determine how particular soil treatments performed at different times along the hydrograph.

3.2 Flow-Through Mesocosm Design

3.2.1 Aquaria

Eighteen flow-through mesocosms were assembled during June and July 2012 with aquaria on loan from the Department of Wildlife, Fisheries & Aquaculture at Mississippi State University. The inner dimensions of the tanks were 76.2 cm x 30.48 cm x 45.72 cm (30.0 in. x 12.0 in. x 18.0 in.), which was ideal for the construction of mesocosms one-fourth the minimum depth of flow-through planters used in practice. The treatment replicates included a set of controls (which consisted solely of a perforated PVC pipe), an underdrain assembly (which consisted of a gravel layer, filter fabric and perforated PVC pipe), and four different soil treatments.

The aquaria came with a 1.9 cm (0.75 in.) PVC outflow pipe attachment, which was previously cut into the glass and measured 3.81 cm (1.5 in.) from the bottom of the tank. A modification was necessary to allow the underdrain pipe to rest on the bottom of the facility. This required a 3.81 cm (1.5 in.) layer of sand, covered by plastic sheeting to raise the bottom of the mesocosm to the outlet elevation. The space below the outflow pipe of the tank was eliminated from the design to prevent water from settling at the bottom of the mesocosms.

3.2.2 Layers

Specifications replicated for this experiment are from Portland Oregon's SWMM. Details from Portland's SWMM are referenced in other cities SWMMs and are primarily implemented in the greater Portland, Oregon area. Design specifications for flow-through planters outlined by Portland's SWMM are (from bottom to top of flow-through planter) a perforated underdrain PVC pipe fit at the bottom of the facility, a 30.48 cm (12.0 in.)

layer of gravel, a layer of unwoven geotextile filter fabric, a minimum of 45.72 cm (18.0 in.) of soil media, and 30.48 cm (12.0 in.) of reservoir depth. An overflow pipe connects the perforated under drain PVC pipe with the surface for flood prevention in the event stormwater volume exceeds the size of the facility or the infiltration capabilities of the flow-through planter (BES 2008). The layered vertical elements of the flow-through planters were quartered in size resulting in: a 7.62 cm (3.0 in.) layer of gravel; 11.43 cm (4.5 in.) of soil media and 7.62 cm (3.0 in.) of reservoir depth (Figure 3.2). Each mesocosm was fitted with a 1.9 cm (0.75 in.) inside diameter outlet and PVC underdrain running the length of the facility. A “T” PVC fitting was attached to the underdrain and connected to a reverse bend overflow set at 7.62 cm (3.0 in.) above the soil media layer to act as the overflow. Figure 3.3 A and B show the actual soil layers as they existed in the mesocosms.

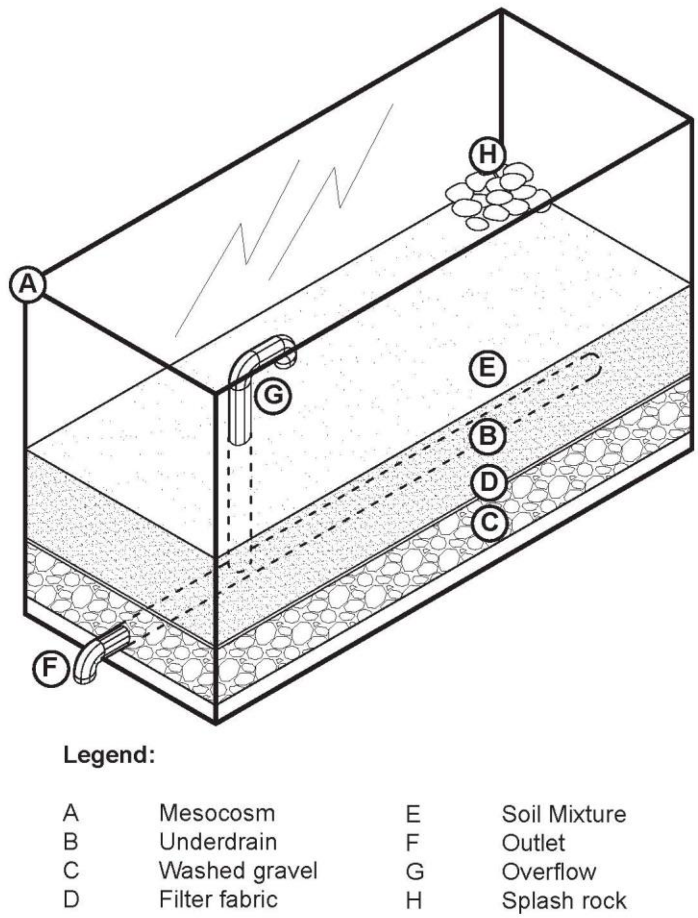


Figure 3.2 Mesocosm diagram

Notes: This mesocosm diagram displays the components of a flow-through planter used in the experiment. The detail was based on Portland Oregon's BES specifications.

a) 

b) 

Figure 3.3 Mesocosm front and side views

Notes: These photos show a front view (a) and side view (b) of one of the mesocosms containing soil.

3.2.3 Soil Media Treatments

Soil media used in the mesocosms were purchased from a local supplier in Starkville, Mississippi and the selected soil types were sent to the Mississippi State University Bost Extension Soil Testing Lab where they were analyzed for organic matter content, pH levels and texture. Soil was chosen based on availability of local materials and criterion provided by the LID Center and Portland's BES SWMM, with both requiring a pH level between 5.5 and 7.5. Based on soil recommendations found in literature, treatments with varying soil media mixtures were developed to test for water volume reduction, peak flow attenuation, PO_4^{3-} and NO_3^--N removal capabilities. The different soil media selected for mixtures in this study consisted of percentages sand, topsoil, and compost. Four varying soil media mixtures were created which are referred to throughout this study by their sand content percentage. Three replicates of each treatment were created ranging from 25% sand with equal parts top soil and compost to 100% sand (Table 3.1).

Table 3.1 Description of native soils and soil treatments

Soil	Sand (%)	Topsoil (%)	Compost (%)	Clay (%)	Silt (%)	Sand (%)	Organic Matter (%)	pH level	Texture
	Treatment Mixtures								
Native Topsoil	n/a	n/a	n/a	2.50	42.25	55.25	1.34	4.7	Loam
Native Sand	n/a	n/a	n/a	1.25	4.25	94.50	0.05	6.4	Sand
Native Compost	n/a	n/a	n/a	n/a	n/a	n/a	17.03	5.5	n/a
25% Sand Mixture	25	37.50	37.50	1.25	27.00	71.75	n/a	n/a	Loamy Sand
50% Sand Mixture	50	25.00	25.00	1.25	15.50	83.25	n/a	n/a	Loamy Sand
75% Sand Mixture	75	12.50	12.50	1.25	13.75	85.00	n/a	n/a	Loamy Sand
100% Sand Mixture	100	0.00	0.00	1.25	5.50	93.25	n/a	n/a	Sand

Notes: Native soils were sent for preliminary tests to determine if pH levels were within the acceptable range. Texture analysis was not possible for the compost.

Each soil mixture treatment was uniformly hand-mixed for desired percentages and hand compacted to simulate compaction that would occur during real world installation. A small pile of washed river rocks was placed in the corner of each mesocosm containing soil to create an energy disperser where the synthetic stormwater inflow would be directed into the mesocosm (Figure 3.4). No plant materials were used in the facilities in order to focus results on the performance of individual soil mixtures.

Figure 3.4 Splash rock

Notes: Washed river rocks were used as an energy disperser in the corner of each mesocosm that contained soil.

3.2.4 Filter Fabric

The filter fabric (Mirati, 140NL, TenCate, Inc., Pendergrass, Georgia) used was specified in Portland's SWMM and was donated by TenCate, Inc. Filter fabric is required to be non-woven geotextile which is designed specifically for filtration (BES 2008). The non-woven fabric was used due to its ability to allow water and fine sediment through while blocking larger sediments which could clog the drainage rock and perforated underdrain pipe. The fabric was cut to fit the dimensions of the mesocosms and was taped into place above the drainage rock (Figure 3.5).

Figure 3.5 Filter fabric layer

Notes: Non-woven geotextile keeps soil particles from clogging the underdrain pipe.

3.2.5 Drainage Rock

The gravel layer consisted of 1.27-2.54 cm (0.50-1.0 in.) washed aggregate (referred to as gravel in this document) acquired from a local distributer. Typically, limestone gravel is used for drainage rock; however, it was not used for this experiment in order to avoid leaching. Three inches of washed gravel was placed on and around the perforated PVC drainage pipe at the bottom of the mesocosm (Figure 3.6).

Figure 3.6 Washed aggregate underdrain

Notes: Washed aggregate as the underdrain material surrounding the perforated PVC underdrain pipe.

3.2.6 Underdrain and Overflow

Underdrain materials consisted of a 1.9 cm (0.75 in.) PVC pipe which ran the length of the mesocosm and was attached to the existing 1.9 cm (0.75 in.) outflow attachment. A drill press was used to perforate the PVC pipes to allow enough drainage for the system. These holes were on the top and bottom of the pipe to maximize their collection potential and prevent flooding of the system. The underdrain pipe was also fitted with an overflow pipe which reached to the surface of the mesocosm and was set 7.62 cm (3.0 in.) above the soil layer to allow for 7.62 cm (3.0 in.) of reservoir depth (Figure 3.7 a and b).

a)  b) 

Figure 3.7 Underdrain configuration and overflow pipe

Notes: These photos show the underdrain configuration (a) and the overflow pipe (b).

3.3 Synthetic Runoff

3.3.1 Composition of Runoff

Synthetic runoff was composed of non chlorinated well water obtained from a well on MSU's South Research Farm. The well water was delivered to a 500 gallon storage chamber where it was dosed with a 2 ppm concentration of dipotassium phosphate (K_2HPO_4) and a 2 ppm concentration of potassium nitrate (KNO_3). In order to keep the application of synthetic stormwater consistent during the delivery process, it was continually mixed inside the storage chamber with a 0.5 horsepower bilge pump placed at the outer edge of the bottom of the chamber.

3.3.2 Delivery Components

The synthetic runoff was delivered from the 1893 L (500 gallon) storage chamber via 0.95 cm (0.375 in.) clear vinyl tubing (Figure 3.8). Manually controlled, variable rate pumps (QV300, Fluid Metering, Inc., Syosset, NY) regulated the flow at which the runoff volume was delivered to the mesocosms (Figure 3.9). The end of the vinyl tubing was placed in the corner of each mesocosm over the splash rock where applicable (previously shown in Figure 3.4).

Figure 3.8 Mixing chamber

Notes: Mixing chamber is shown with the clear vinyl tubing which was used to distribute the synthetic stormwater solution to the mesocosms.

Figure 3.9 Variable flow rate pumps (QV300, Fluid Metering, Inc., Syosset, NY)

Notes: The variable flow rate pumps were used to simulate a Type II, 5.08 cm (2.0 in.) storm event.

Synthetic runoff was delivered over time with the pumps to match a hydrograph for a Type II, 5.08 cm (2.0 in.) storm event. For the experiment, a time-step model based on the Santa Barbara Urban Hydrograph (SBUH) Method was modified to predict flow from a hypothetical impervious area into the mesocosm. The model, developed by the Landscape Architecture Department at Mississippi State University, uses synthetic rainfall curves developed by the Soil Conservation Service to predict rainfall in one of the four U.S. climatic regions (Gallo et al. 2012). Using curves from a Type II event and an assumed 5.08 cm/hr. (2.0 in./hr.) infiltration rate through the soil media, the model predicted that the mesocosm could manage 3.40 m^2 (11.17 $ft.^2$) of impervious area (0.98 CN) before reaching the mesocosms' maximum ponding depth of 7.62 cm (3.0 in.). Due to the low flow at the ends of the model, it was decided to focus on the middle 4.5 hrs. of the 24 hr. hydrograph generated by the model. During the 4.5 hrs. synthetic stormwater event, inflow varied from 32-640 mL/min. The model resulted in an expected total of 33,280 mL of solution delivered to each mesocosm over the 4.5 hr. experiment.

3.3.3 Pump Delivery

Nine variable rate pumps were calibrated for a maximum flow rate of 646 mL/min. This amount was based off the condensed hydrograph calculations where 646 mL/min was the amount delivered at the peak of the 4.5 hr. event. For each 10-minute time-step during the simulated storm event, the flow rate control box was manually adjusted. Table 3.2 illustrates the flow rate for each 10-minute time-step for the 4.5 hrs./240 min. experiment.

Table 3.2 Experiment timeline

Time-Steps (min.)	Pump Speed (%)	Flow Rate (mL/min.)	WQ Sample	Overall Flow (%)
0	5%	32		0.97%
10	5%	32		0.97%
20	5%	32		0.97%
30	6%	39	WQ	1.18%
40	7%	45		1.37%
50	7%	45		1.37%
60	10%	65	WQ	1.97%
70	12%	78		2.37%
80	12%	78		2.37%
90	54%	349	WQ	10.59%
100	98%	633		19.21%
110	100%	646		19.60%
120	60%	388	WQ	11.77%
130	19%	123		3.73%
140	19%	123		3.73%
150	15%	97	WQ	2.94%
160	10%	65		1.97%
170	10%	65		1.97%
180	9%	58	WQ	1.76%
190	7%	45		1.37%
200	7%	45		1.37%
210	6%	39	WQ	1.18%
220	6%	39		1.18%
230	6%	39		1.18%
240	5%	32	WQ	0.97%
250	5%	32		0.97%
260	5%	32		0.97%
270	0%	0	WQ	0%
280	0%	0		0%

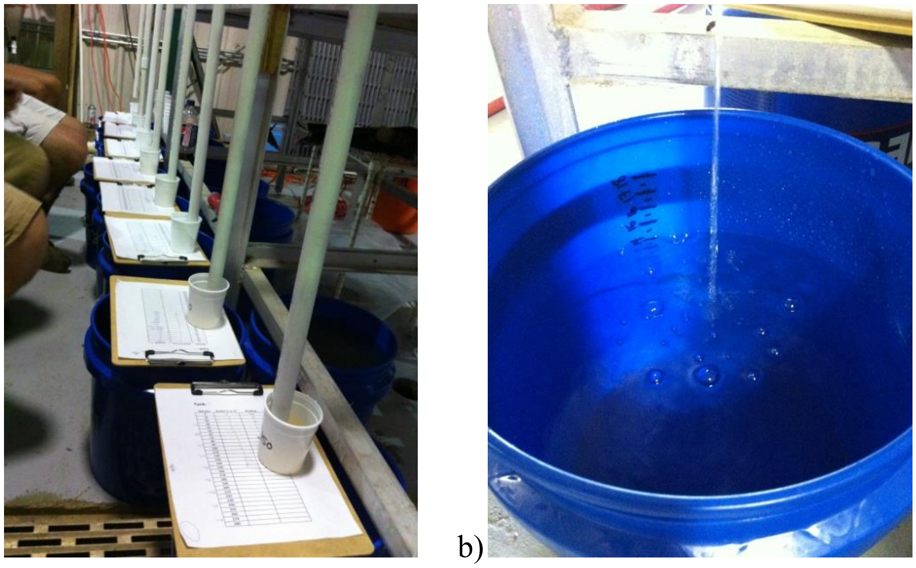
Notes: This table outlines the time-steps over the 4.5 hrs. hydrograph, flow rates at each time-step, volume delivered to the mesocosms at each time-step, and when water quality (WQ) samples were collected.

3.4 Execution of Experiment

Prior to running the experiment, each mesocosm was flooded with non-chlorinated well water and then allowed to dry for two weeks. This was done to rinse finer soil materials from the system and to provide some extra compaction that would occur during installation. The systems were completely dry by the time the experiment took place. The experiment was broken down into two 4.5 hrs. simulated rain events due to the limited number of control pumps. The first run was conducted on nine of the mesocosms and the second run was conducted on the remaining nine mesocosms.

3.4.1 Water Analysis

Water quality samples were collected in 250 mL polyethylene cups (Fisher Scientific, Pittsburgh, PA) every 30 minutes if outflow was occurring. Samples were immediately placed on ice until they could be transported to a refrigerated unit for analysis within 24 hrs. Water samples were filtered (0.45µm) and prepared for flow injection analysis (Lachat FIA 8500, Loveland, CO) which tested samples for NO_x [NO_3-N+ NO_2^--N] (cadmium reduction) and PO_4 concentrations (molybdenum blue / ascorbic acid). Outflow volume measurements were taken every 10 minutes using graduated 5-gallon buckets marked at 500 mL and 1000 mL (Figure 3.10).



a) b)

Figure 3.10 Water quality and outflow volume collection methods

Notes: These photos show the water quality sample collection method (a) and outflow volume method (b).

3.4.2 Volume Analysis

Outflow volume measurements were scheduled for collection 28 times during the experiment (Table 2). Volume change was calculated at each 30-minute time-step for the duration of the experiment. Percent volume change was calculated with the following equation:

$$\text{Volume Change (\%)} = \frac{(\mu V_t)-(\mu V_c)}{(\mu V_c)} \tag{1}$$

Where μV_t represents the mean volume of the treatment replicates and μV_c represents the mean volume of the control replicates.

3.4.3 Water Quality Analysis

Water quality samples were scheduled for collection nine times during the experiment (Table 2). However, several of the replicates did not have sufficient flow for a water quality sample collection until the 90-minute time-step, so concentration means did not include the first 90 minutes of any treatment. Mean concentration differences were calculated at each 30-minute time-step and for the overall experiment. Mean concentration differences were calculated for PO_4^{3-} and NO_3^--N using the following equation:

$$\text{Concentration Change (\%)} = \frac{\mu C_t - \mu C_c}{\mu C_c} \tag{2}$$

Where μC_t represents the mean concentration value of the treatment replicates and μC_c represents the mean concentration value of the control replicates. Additionally, load changes were calculated for each 30-minute time-step and for the overall experiment. Percent load change was calculated for PO_4^{3-} and NO_3^--N with the following equation:

$$\text{Load Change (\%)} = \frac{(\mu C_t * \mu V_t) - (\mu C_c * \mu V_c)}{(\mu C_c * \mu V_c)} \tag{3}$$

Where μC_t represents the mean concentration value of the treatment replicates, μV_t represents the mean volume of the treatment replicates, μC_c represents the mean concentration value of the control replicates, and μV_c represents the mean volume of the control replicates.

3.4.4 Statistical Analysis

The total outflow loads and concentrations were calculated for PO_4^{3-} and NO_3^--N for each set of treatments and compared to the control replicates to determine the cumulative changes and individual time-step changes. Similarly, the total outflow volume was subtracted from the total inflow volume at each time-step of the hydrograph and cumulatively (over entire event). To determine how each treatment performed during different sections of the hydrograph, water quality and quantity results were compared in grouped time-steps. The data were tested for normality using a Shapiro-Wilk test (IBM SPSS version 20). Data found to be normal were tested with a one-way ANOVA, while data found to be non-normal were tested with a Kruskal-Wallis test. All tests were conducted with an assumed alpha level of 0.05.

CHAPTER IV

RESULTS

4.1 Summary of Data

Differences between the treatments for runoff volume retention, peak flow delay, and nutrient changes are presented in this chapter. Volume changes were analyzed cumulatively for differences between treatments, at individual time-steps along the hydrograph, and by comparing volume reduction along the rising limb against the falling limb of the hydrograph. Outflow rates were analyzed at the 120-minute time-step to determine if peak flow was significantly reduced by any of the treatments. Nutrient concentrations and loads were analyzed cumulatively, comparatively, at individual time-steps between treatments along the hydrograph, and by comparing changes at different grouped parts of the hydrograph. A summary of the data is provided below (Table 4.1).

Table 4.1 Data summary for mean cumulative percentages for all mesocosms

Treatment	Mean Volume Change (%)		Mean NO_3^--N Concentration (%)		Mean NO_3^--N Load (%)		Mean PO_4^{3-} Concentration (%)		Mean PO_4^{3-} Load (%)	
25% Sand	-9.63		43.20		29.57		-6.71		-61.63	
	-9.63	-10.71	70.48	43.05	104.09	53.14	-5.96	-11.59	-19.15	-41.34
	-12.87		15.47		25.78		-22.09		-43.23	
50% Sand	-15.58		0.98		-57.31		-63.67		-86.00	
	-11.89	-14.42	32.51	22.49	20.39	-7.15	1.80	-19.04	-12.36	-39.45
	-15.78		33.99		15.48		4.75		-19.99	
75% Sand	-19.79		33.21		-0.01		22.14		5.23	
	-13.63	-17.74	21.03	25.66	5.17	1.04	1.33	7.49	-13.72	-6.49
	-19.79		22.74		-2.03		-0.99		-10.97	
100% Sand	-18.61		14.54		-5.43		22.86		-3.75	
	-17.64	-20.02	13.04	13.59	-2.00	-6.62	0.80	21.32	-11.30	-3.44
	-23.80		13.20		-12.42		40.30		4.72	
Gravel	-8.77		19.21		9.85		51.84		38.08	
	-6.06	-7.72	4.91	13.20	6.77	7.29	-59.82	-13.95	-23.94	12.80
	-8.33		15.48		5.26		49.81		24.26	

Notes: Mean percentages for individual mesocosms and overall percentage means for each treatment compared to the controls.

4.2 Volume Change

Mean cumulative volume reduction for each of the soil treatments ranged from -11 ± 1% to -20 ± 2 %, with increasing levels of sand appearing to provide the greatest retention (Figure 4.1). One-way ANOVA results indicated significant differences between treatments (F = 28.247, $p \leq 0.001$) in total volume reductions (Table 4.2). Results from a post-hoc Tukey HSD test showed significant differences in volume reductions between the controls and all other treatments. Post-hoc results also indicated that the 100% and 75% sand mixture treatments retained significantly greater amounts of water than the 25% sand mixture treatment, but they did not differ significantly from each other or from the 50% sand mixture.

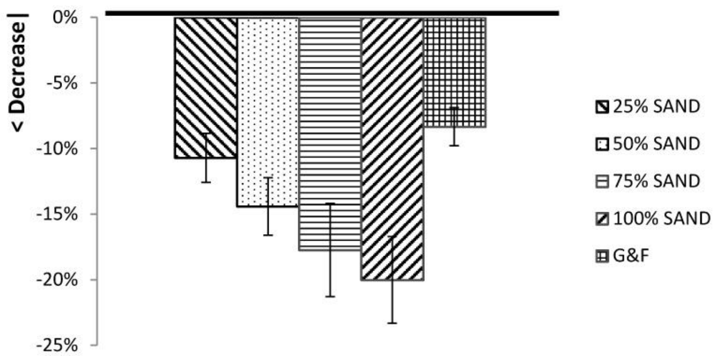


Figure 4.1 Mean cumulative volume percentage changes for all treatments

Notes: Average volume changes for each treatment are shown with standard error bars and are being compared to the controls, which are represented by the 0% line.

Table 4.2 One-way ANOVA test table for volume differences between treatments

Treatments	25% Sand	50% Sand	75% Sand	100% Sand	Gravel	Volume Mean Change (%)	Standard Error (%)
25% Sand						-10.71	± 1.08
50% Sand	------					-14.42	± 1.26
75% Sand	$p = 0.032$	------				-17.74	± 2.05
100% Sand	$p = 0.005$	------	------			-20.02	± 1.91
Gravel	------	$p = 0.043$	$p = 0.003$	$p \leq 0.001$		-7.72	± 0.84
Control	$p = 0.001$	$p \leq 0.001$	$p \leq 0.001$	$p \leq 0.001$	$p = 0.018$		

Notes: P-values are provided for treatments with significant results. Dash lines indicate no significant differences between the treatments and redundant cells are grayed out.

Mean volume change results for each treatment were graphed along the hydrograph to illustrate how inflow rates influenced volume reduction cumulatively over the entire event and at individual time-steps (Figure 4.2). When observed at each 30-minute time-step, most soil treatments retained water along the rising limb of the hydrograph with the exception of two individual mesocosms (one 25% sand replicate and one gravel replicate). A one-way ANOVA analysis of volume changes among media treatments indicated differences among treatments occurred during different time-steps along the hydrograph. A post hoc Tukey test suggested the 100% sand mixture retained significantly more water than the 25% sand mixture during the 60- ($p = 0.027$) and 90- ($p \leq 0.001$) minute time-steps. At the 60-minute time-step the 100% sand and 75% sand treatments retained greater volume amounts than the control ($p = 0.004$ and $p = 0.026$, respectively). Additionally at the 90-minute time-step, the 100% sand retained a significantly greater volume amount then the 50% sand ($p = 0.002$), 75% sand ($p = 0.001$) and the gravel treatments ($p \leq 0.001$). During the peak (120-minute) of the hydrograph, the 25% and 100% sand treatments retained significantly more water than the controls ($p = 0.015$, $p = 0.009$, respectively), however no other significant values for volume were found between treatments after the peak of the event.

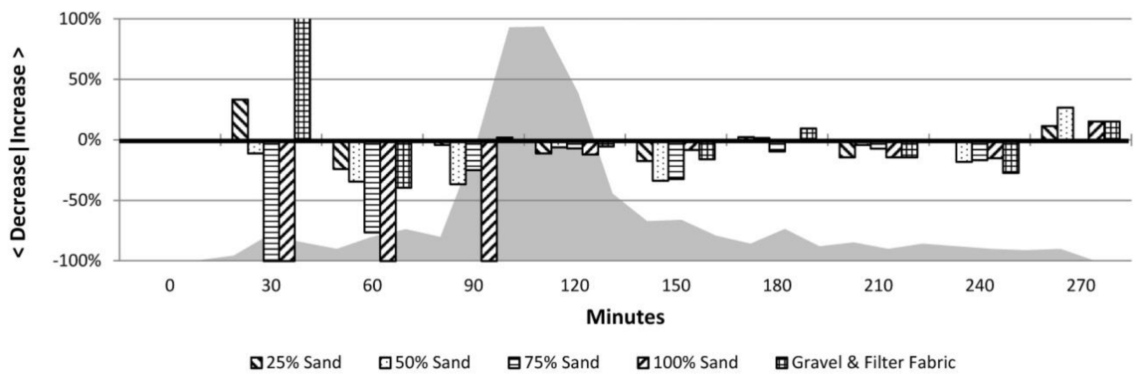


Figure 4.2 Mean volume percentage changes at individual time-steps along the hydrograph for all treatments

Notes: Each treatment's mean cumulative volume is show for each 30-minute time-step along the inflow hydrograph. The 0% line represents the control treatment.

While there were significant differences among treatments at individual time-steps, an overall pattern was difficult to observe. Thus, the hydrograph was reorganized into two parts consisting of a rising limb (30-90 minutes) and a falling limb (120-270 minutes) (Figure 4.3). All treatment mean volume values were grouped into these two categories, thus increasing sample sizes. An independent samples t-test comparing volume reduction before and after the peak indicated the 100% sand mixture retained significantly more water volume during the rising limb than during the falling limb (t = -5.711, p = 0.001).

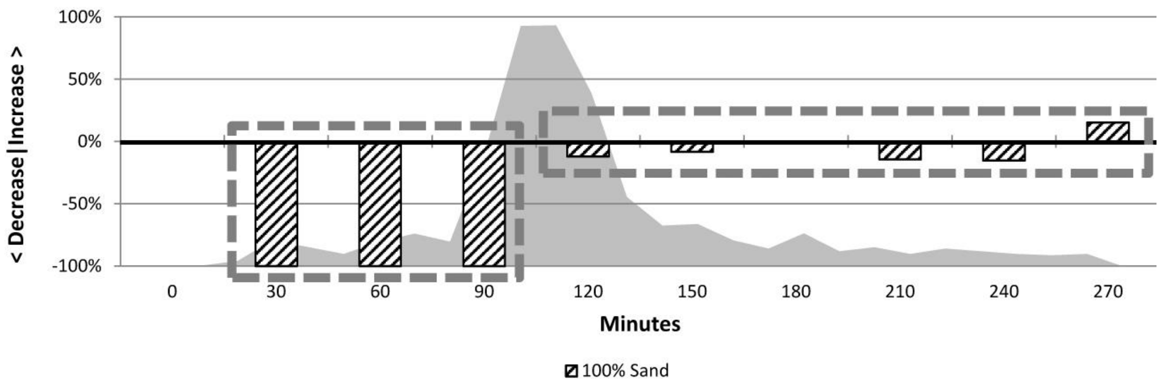


Figure 4.3 Grouped time-steps for mean volume percentage changes along the hydrograph for 100% sand treatment

Notes: The 30-90 minute group compared to the 120-270 minute group for 100% sand.

4.3 Peak Flow Attenuation

All treatments reached their peak flow rate during the 120-minute time-step (Table 4.3). Peak flow rates varied minimally across all treatments with the 100% sand treatment showing the highest outflow rate at the peak. However, a Kruskal-Wallis test comparing peak flows among treatments indicated there were no significant differences (H = 3.829, p = 0.574). Additionally, only a small amount of ponding (0.635 cm (0.25 in.)) was observed in the mesocosms during the 120-minute time-step.

Table 4.3 Mean peak flow for all treatments

Treatments	Peak (min.)	Mean Peak Flow (mL/min.)
25% Sand	120	591.67
50% Sand	120	596.67
75% Sand	120	600.00
100% Sand	120	613.33
Gravel	120	550.00
Control	120	591.67

Notes: Peak flow occurred at the 120- minute time-step for each treatment. Overall H = 3.829, p = 0.574.

4.4 Phosphate Change

Cumulative PO_4^{3-} concentration percentages ranged from a removal of -19 ± 22% to a loading of 21 ± 11% (Figure 4.4). Mean PO_4^{3-} concentration removals appeared to be greatest in treatments with increased amounts of organic matter or those with less sand content (Table 4.4). However, a Kruskal-Wallis test comparing mean PO_4^{3-} concentration change percentages indicated no significant differences across treatments (H = 6.266, p = 0.281).

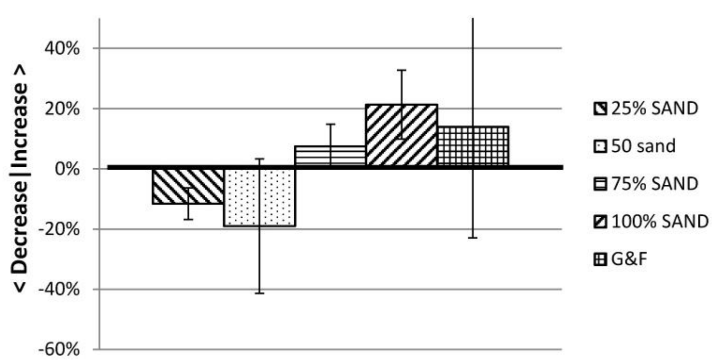


Figure 4.4 Mean cumulative PO_4^{3-} concentration percentage changes for each treatment

Notes: Mean PO_4^{3-} concentration increases and decreases for each treatment are shown with standard error bars and are being compared to the controls, which are represented by the 0% line.

Table 4.4 Mean cumulative PO_4^{3-} concentration percentage changes for all treatments

Treatment	PO_4^{3-} Mean Change (%)	Standard Error (%)
25% Sand	-11.59	± 5.26
50% Sand	-19.04	± 22.33
75% Sand	7.49	± 7.35
100% Sand	21.32	± 11.43
Gravel	13.95	± 36.89
Control	-------	-------

Notes: One-way ANOVA tests showed no significant difference. Overall (two-tailed), F = - 6.266, $p = 0.281$.

Phosphate mean load changes ranged from a removal of -41 ± 12% to a loading of 13 ± 19% (Figure 4.5). The PO_4^{3-} load mean percentages were compared with a one-way ANOVA which indicated no significant differences among treatments (F = 2.677, p = 0.75) (Table 4.5).

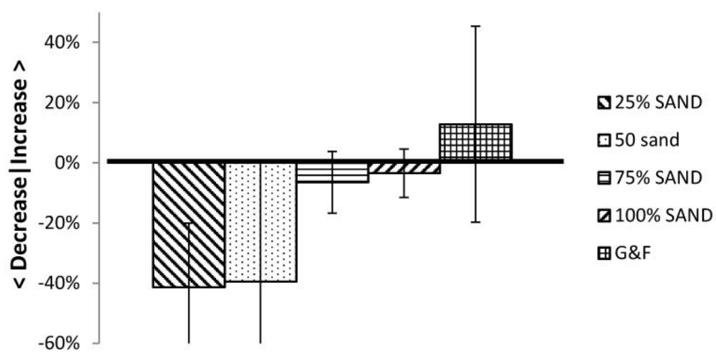


Figure 4.5 Mean cumulative PO_4^{3-} load percentage changes for each treatment

Notes: Mean PO_4^{3-} load increases and decreases for each treatment are shown with standard error bars and are being compared to the controls, which are represented by the 0% line.

Table 4.5 Mean cumulative PO_4^{3-} load percentage changes for each treatment

Treatments	PO_4^{3-} Mean Change (%)	Standard Error (%)
25% Sand	-41.34	± 12.30
50% Sand	-39.45	± 23.38
75% Sand	-6.49	± 5.91
100% Sand	-3.44	± 4.63
Gravel	12.80	± 18.80
Control	-------	-------

Notes: One-way ANOVA tests showed no significant difference. Overall (two-tailed), F = 2.677, p = 0.75.

A one-way ANOVA comparison was used to determine if PO_4^{3-} load and concentration mean percentages were significantly different from one another among treatments. This analysis showed that there were no significant differences among treatment's concentration and load values.

Mean PO_4^{3-} loads for all treatments at individual time-steps were graphed along the hydrograph, illustrating possible increases and decreases in load values during varied inflow rates (Figure 4.6). A Kruskal-Wallis test with pair wise comparisons comparing PO_4^{3-} loads between treatments at individual time-steps indicated differences at the 60-minute, 90-minute, and 120-minute time-steps. The 60-minute time-step indicated differences between the 100% sand and control (H = 3.351, p = 0.001), the 100% sand and gravel (H = -1.964, p = 0.049), and the 75% sand treatment and control (H = 2.195, p = 0.028). Differences were indicated at the 90-minute time-step between the 100% sand and control (H = 3.225, p = 0.001) and the 100% sand and gravel treatments (H= -2.534, p = 0.011).

Significant differences in PO_4^{3-} loads were also observed at the 120- minute (peak of storm event) time-step between the 25% and 100% sand (H = -2.032, p = 0.042), the

25% sand and gravel (H = -2.952, p = 0.003), and the 50% sand and gravel (H = -2.722, p = 0.006) (Table 4.6). Compared with the cumulative mean percentages, these results suggest 25% sand treatment removed significantly more PO_4^{3-} than the 100% sand and gravel treatments but it did not differ significantly from the 50% and 75% sand treatments.

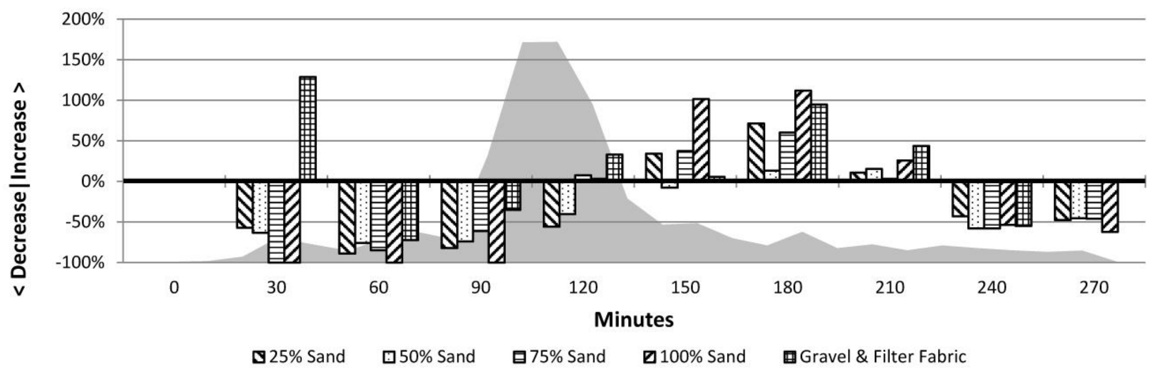


Figure 4.6 Mean PO_4^{3-} percentage changes at individual time-steps along the hydrograph for all treatments

Notes: The PO_4^{3-} changes are graphed along the inflow hydrograph compared to the control for each time-step. The control is represented on the graph along the 0% line.

Table 4.6 Kruskal-Wallis table for mean PO_4^{3-} load percentage changes at peak of the hydrograph (120-minute time-step)

Treatments	25% Sand	50% Sand	75% Sand	100% Sand	Gravel	PO_4^{3-} Mean Change (%)	Standard Error (%)
25% Sand						-55.67	± 24.59
50% Sand	------					-40.31	± 26.97
75% Sand	------	------				7.57	± 12.41
100% Sand	p = 0.042 H = -2.032	------	------			3.27	± 5.26
Gravel	p = 0.003 H = -2.952	p = 0.006 H = -2.722	------	------		33.17	± 14.50
Control	------	------	------	------	------		

Notes: P-values are provided for treatments with significant results. Dash lines indicate no significant differences between the treatments and redundant cells are grayed out.

Individual treatments were tested at all time-steps to determine if treatments performed differently in terms of PO_4^{3-} over the course of the hydrograph. There were some significant differences in individual treatment's performance among time-steps however, a clear pattern was not observed. The hydrograph was then broken down into the following categories: 30-90 minute, 120 minute, 150-180 minute, 210 minute, and the 240-270 minute time-steps based observed trends. This break down grouped similar results and increased analysis sample size. While all treatments showed some significant differences among the different time-step categories, the 25% and 100% sand treatments showed the clearest patterns in PO_4^{3-} reductions and increases in load and were chosen for comparison (Figure 4.7).

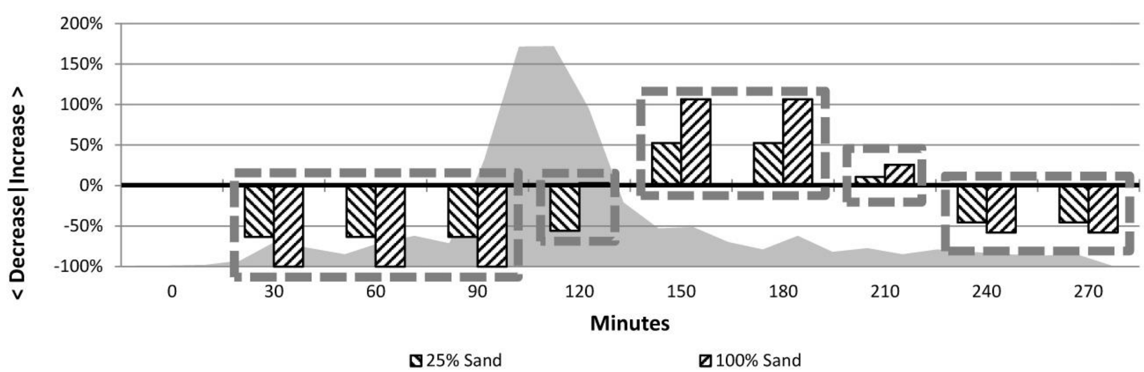


Figure 4.7 Grouped time-steps for mean PO_4^{3-} percentage changes featuring 25% and 100% sand treatments.

Notes: Grouped time-step categories included the 30-90 time-steps, 120 time-step, 150-180 time-steps, 210 time-step, and 240-270 time-steps.

A Kruskal-Wallis analysis of the grouped time-steps indicated significant differences in PO_4^{3-} for the 25% sand mixture between the 30-90 and 150-180 time-steps, 30-90 and 210 time-steps, 120 and 210 time-steps, and the 150-180 and 240-270 time-steps (Table 4.7). The same analysis was performed with the 100% sand mixture and results indicated significant differences between the following grouped time-steps: 30-90 and 120 time-steps, 30-90 and 150-180 time-steps, 30-90 and 210 time-steps, 120 and 150-180 time-steps, and the 150-180 and 240-270 time-steps (Table 4.8). No significant differences were observed for the 25% and 100% sand treatments at the 30-90 and 240-270 time-steps.

Table 4.7 Kruskal-Wallis test table for mean PO_4^{3-} grouped time-steps results featuring 25% sand treatment

Grouped Time-steps (min)	30-90	120	150-180	210
30-90				
120	------			
150-180	$p \leq 0.001$ H = -3.559	$p = 0.022$ H = -2.316		
210	$p = 0.046$ H = -1.995	------	------	
240-270	------	------	$p = 0.006$ H = 2.728	------

Notes: P-values are provided for treatments with significant results. Dash lines indicate no significant differences between the treatments and redundant cells are grayed out.

Table 4.8 Kruskal-Wallis test table for mean PO_4^{3-} grouped time-step results featuring 100% sand treatment

Grouped Time-steps (min)	30-90	120	150-180	210
30-90				
120	$p = 0.12$ H = -2.503			
150-180	$p \leq 0.001$ H = -4.709	$p = 0.022$ H = -2.316		
210	$p = 0.006$ H = -2.760	------	------	
240-270	------	------	$p = 0.009$ H = 2.631	------

Notes: P-values are provided for treatments with significant results. Dash lines indicate no significant differences between the treatments and redundant cells are grayed out.

4.5 Nitrate Change

Changes in mean concentration percentages of NO_3^--N across treatments ranged from 13 ± 4% in the gravel treatment to 43 ± 16% in the 25% sand treatment. Loading of NO_3^--N occurred in treatments with greater amounts of organic matter and less sand (Figure 4.8). A Kruskal-Wallis analysis indicated significant differences in NO_3^--N concentration changes among treatments (H = 11.410, $p = 0.044$). A pair wise comparison of the treatments indicated that the 25%, 50%, and 75% sand treatments loaded significantly more NO_3^--N than the control but no significant differences were found among the different soil treatments or between the 100% sand treatment and the control (Table 4.9).

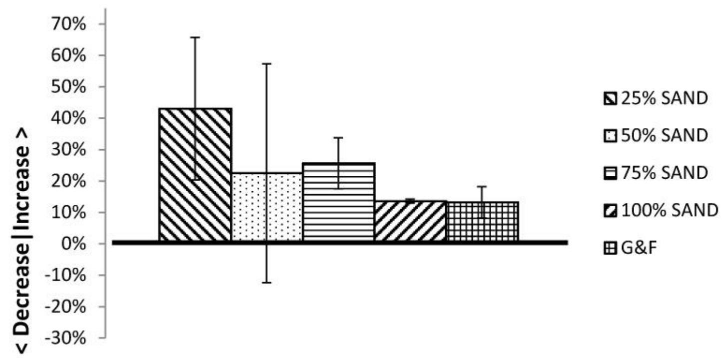


Figure 4.8 Mean cumulative NO_3^--N concentration percentage changes for each treatment

Notes: Total average concentration changes and total average load changes for treatments for NO_3^--N. Graphs show the average decreases or increases of NO_3^--N for each treatment compared to the control. The control is represented on the graph along the 0% line.

Table 4.9 Kruskal-Wallis test table for mean NO_3^--N concentration percentage changes

Notes: P-values are provided for treatments with significant results. Dash lines indicate no significant differences between the treatments and redundant cells are grayed out.

Cumulative NO_3^--N load change values ranged from -7 ± 25% removal to 53 ± 26% loading (Figure 4.9). A Kruskal-Wallis analysis comparing mean load percentages among treatments indicated that the 25% sand treatment differed significantly from the control (H = -2.069, p = 0.039), the 75% sand treatment (H = 2.376, p = 0.018), and the 100% sand treatment (H = 3.142, p = 0.002). These results suggested the 25% sand treatment loaded more NO_3^--N than the controls, 75% sand and 100% sand treatments (Table 4.10). High variability within the 25% and 50% sand treatments was observed.

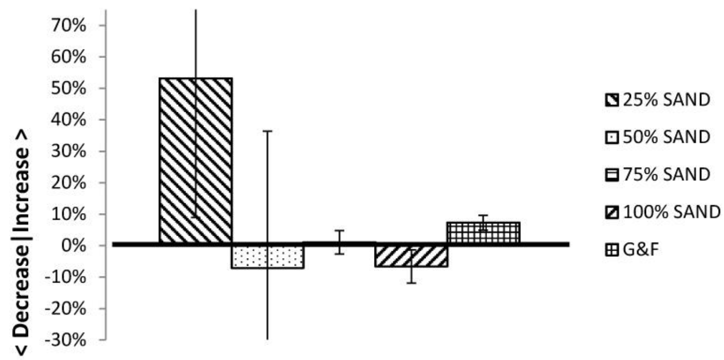


Figure 4.9 Mean cumulative NO_3^--N load percentage changes for each treatment

Treatments	25% Sand	50% Sand	75% Sand	100% Sand	Gravel	NO_3^--N Mean Change (%)	Standard Error (%)
25% Sand						43.05	± 15.88
50% Sand	------					22.49	± 10.77
75% Sand	------	------				25.66	± 3.81
100% Sand	------	------	------			13.59	± 0.48
Gravel	------	------	------	------		13.20	± 4.28
Control	$p = 0.004$ H = -2.912	$p = 0.032$ H = -2.146	$p = 0.009$ H = -2.605	------	------	------	---------

Notes: Total mean load percentage changes for treatments for NO_3^--N. Graphs show the average decreases or increases of NO_3^--N for each treatment compared to the control. The control is represented on the graph along the 0% line.

Table 4.10 Kruskal-Wallis test table for mean NO_3^--N load percentage changes

Treatments	25% Sand	50% Sand	75% Sand	100% Sand	Gravel	NO_3^--N Mean Change (%)	Standard Error (%)
25% Sand						53.14	± 25.50
50% Sand	------					-7.15	± 25.12
75% Sand	$p = 0.018$ H = 2.376	------				1.04	± 2.15
100% Sand	$p = 0.002$ H = 3.142	------	------			-6.62	± 3.07
Gravel	------	------	------	$p = 0.046$ H = -1.992		7.29	± 1.35
Control	$p = 0.039$ H = -2.069	------	------	------	------	------	---------

Notes: P-values are provided for treatments with significant results. Dash lines indicate no significant differences between the treatments and redundant cells are grayed out. Overall, H = 11.739, $p = 0.039$ (two-tailed).

A one-way ANOVA analysis was used to determine if mean NO_3^--N load and concentration values differed significantly across treatments. Analysis results indicated that the 75% sand mixture treatment (F = 31.741, p = 0.005) and 100% sand treatment (F = 42.430, p = 0.003) concentrations and loads differed significantly between each other. However, there were no other significant differences among treatment concentration and load values.

Nitrate load values were graphed along the hydrograph, illustrating changes that occurred during varied applied inflow rates (Figure 4.10). A Kruskal-Wallis analysis looking at changes in NO_3^--N loads across treatments at each time-step indicated differences at the 60-minute time-step between the 100% sand treatment and controls (H = 2.889, p = 0.004), the 100% sand and 25% sand treatments (H = 2.118, p = 0.034), the 100% sand and 50% sand treatments (H = 2.349, p = 0.019) and the 75% sand treatment and control (H = 2.118, p = 0.034). A one-way ANOVA analysis indicated significant differences in NO_3^--N loads at the 90-minute time-step between the 100% sand treatment and control (p = 0.012), the 100% sand and 25% sand treatments (p = 0.009), and the 100% sand and gravel treatments (p = 0.023). Additionally, the 180-minute time-step indicated the 25% sand treatment loaded significantly more NO_3^--N than the control (p = 0.012). No significant differences were found for NO_3^--N loads among treatments at the 120- minute (peak of event) time-step (H = 9.930, p = 0.077) (Table 4.11).

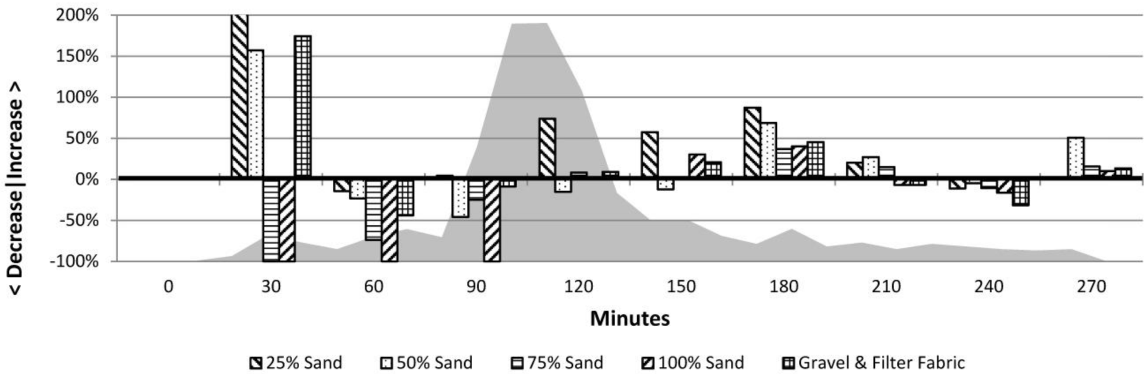


Figure 4.10 Mean NO_3^--N percentage changes at individual time-steps along the hydrograph for all treatments

Notes: Mean NO_3^--N Percent changes for treatments at each 30-minute time-step graphed against the outflow hydrograph of the controls. The control is represented on the graph along the 0% line. Overall, H = 9.930, p = 0.077(two-tailed).

Table 4.11 Kruskal-Wallis test table for mean NO_3^--N load percentage changes for all treatments at event peak (120-minute time-step)

Treatments	NO_3-N Mean Change (%)	Standard Error (%)
25% Sand	74.06	± 46.45
50% Sand	-15.07	± 38.26
75% Sand	8.31	± 1.42
100% Sand	0.91	± 3.82
Gravel	9.11	± 3.52
Control	------	------

Notes: Kruskal Wallis Test indicated no significant difference at the 120-minute time-step. Overall (two-tailed), H = 9.930, p = 0.077.

Individual treatments were tested at all time-steps to determine if treatments performed differently for NO_3^--N removal over the course of the hydrograph. Like PO_4^{3-}, there were several significant differences in individual treatment's performance between time-steps however, a clear pattern was not observed. Therefore, the hydrograph for NO_3^--N load change was broken down into two time-step categories: 30-90 minutes and the 120-270 minutes. Time-steps were grouped into these categories in order to group similar results and increase the analysis sample size. While all treatments showed some significant differences between the two time-step groups, the 100% sand treatment showed the clearest pattern and was chosen for comparison.

As shown in Figure 4.11, the hydrograph was grouped into two parts with the rising limb encompassing 30-90 minutes and the falling limb covering 120-270 minutes. A Mann-Whitney U Test indicated a significant difference in NO_3^--N values for 100% sand treatment between the rising limb and the falling limb ($p \leq 0.000$). No other significant differences between the rising limb and falling limb were observed with the other treatments.

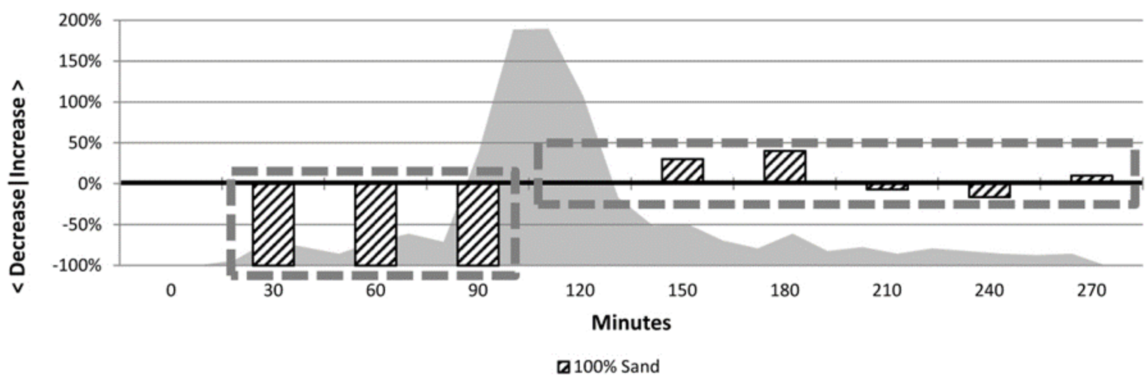


Figure 4.11 Grouped time-steps for mean NO_3^--N change featuring 100% sand treatment

Notes: Grouped time-step categories included the 30-90 time-steps and 120-270 time-steps

CHAPTER V

DISCUSSION

5.1 Volume Change

Results from volume analyses indicated that all soil treatments reduced water volume relative to the control. Treatments with the largest percentage of sand (75% and 100%) retained a greater amount of water than the 25% and 50% sand. The treatments with the lowest sand content most likely did not perform well for volume reduction due to their high compost content, which created a coarser overall soil texture. Coarser soil particles contain larger void spaces which have lesser water holding capabilities (O'Leary et al. 2002). Void space in the lower sand content treatments may decrease as compost breaks down over time or if finer texture compost is utilized in the mixture.

Differences and patterns between treatments can also be observed over the course of the hydrograph that are not apparent in the cumulative results. Specifically, the differences between the rising and falling limbs and at the 120-minute time-step offer insights into the overall performance of the treatments. Results from testing volume at individual time-steps indicated the 100% sand treatment retained a greater amount of water during the rising limb of the hydrograph when compared to the 25% sand treatment. However, the 100% sand treatment outperformed the 25% sand treatment until the peak of the event, indicating the greater adsorption potential of the 100% sand treatment. Additionally, because there were no significant differences after the peak, the volume reductions at the beginning of the simulated event were the primary contributor to the overall volume reductions seen in the treatments. Once the treatments were saturated (*i.e.* reached their field capacity), they lost the ability to retain additional inflow. The dry condition of the mesocosms was likely responsible for the reductions at the 30 minute time-step for all treatments.

5.2 Peak Flow Attenuation

No significant peak flow reductions occurred among any of the treatments. A lack of peak flow reduction is likely related to the thin soil layer used in the experiment, as well as observed preferential flow paths through the soil located directly under the inflow point. Preferential flow paths were observed in an experiment by Hsieh and Davis (2005) with deeper bioretention systems (up to 80 cm (31.5 in.)) and were stated as the reason peak flow reduction was not observed. Specific to flow-through configurations, higher than expected infiltration rates and preferential flow paths in soil media were also noted in existing flow-through facilities in the Pacific Northwest (BES 2010). However, other studies have concluded that increased depth of soil media improves volume retention and has the ability to delay the peak flow (Davis 2008; Hunt et al. 2008).

The single inflow point in this study, which reflects real-world applications currently used in practice, may have played a prominent role in the inability to find significant reductions in peak flows among treatments. As the water entered each system at a single point, contact between the runoff and the soil was limited. Runoff was observed to mostly utilize the soil directly beneath the inflow point and quickly reached the underdrain pipe where it was carried out of the system. Elimination of a single inflow point by distributing the inflow over more of the soil surface could increase contact between runoff and the soil, which might lead to improved results.

5.3 Phosphate Change

Results from both cumulative concentration and load changes indicated no differences among the soil mixtures and their ability to reduce PO_4^{3-}. This may have been in part due to the low number of replicates and the high variability in some of the treatments as well as observed preferential flow path which caused the system to 'short circuit'. However, when comparing concentration to load for individual treatments the concentration for 75% sand and gravel treatments differed significantly from their load. These findings hint toward the differences that occur when reductions in outflow volume are accounted for when calculating cumulative nutrient changes.

Although no significant differences in PO_4^{3-} were found among treatments cumulatively over the entire simulated storm event, there were differences at the 120-minute time-step as the 25% sand treatment had a greater PO_4^{3-} reduction rate than the 100% sand treatment. This result indicates that reductions in PO_4^{3-} may have improved with lower sand content (or higher organic matter) which was also found in a mesocosm study by Lucas and Greenway (2008). Significant reductions of PO_4^{3-} at the 120-minute time-step are also important because nearly half of the total volume (and pollutant load) of the simulated storm event enters the system at this time-step (Table 4.3). Therefore, the 120-minute time-step is critical in determining the overall performance of each treatment.

Phosphate binds to smaller silt and clay soil particles present in topsoil and compost which is why sandy soils have a much lower PO_4^{3-} adsorption capacity (Brady and Weil 1996; Ige et al. 2005). Adsorption of PO_4^{3-} onto silt and clay minerals was also noted as the dominant uptake mechanism in lab and field experiments by Davis et al. (2006) and Lucas and Greenway (2008). The greatest amount of clay and silt in my study was found in the topsoil while the sand contained considerably less silt. Greater silt and clay content in the 25% and 50% sand treatments might account for the reductions in PO_4^{3-} seen at the 120-minute time-step and further explain why treatments with lower sand content removed a greater amount of PO_4^{3-}.

Levels of pH in the soils may have also played a role in aiding PO_4^{3-} adsorption to soil particles in the treatments with less sand. The native topsoil and compost used in mixtures in this experiment were more acidic (4.7 and 5.5, respectively) than the sand which had a pH level of 6.4. More acidic soils have been shown to increase PO_4^{3-} adsorption to small soil particles (Busman et al. 2002). In acidic conditions however, PO_4^{3-} becomes less available to plants (Busman et al. 2002), which are a standard component of bioretention BMPs. A lack of plant materials in this study is a possible reason less PO_4^{3-} was retained than expected, but, Lucas and Greenway (2008) concluded that plant uptake only accounts for minimal P assimilation and that geochemical processes such as adsorption to smaller particles are the primary mechanism for which PO_4^{3-} is retained in bioretention systems. The results from this study as well as Lucas and Greenway (2008), highlight the importance of optimizing soil media and accounting for local conditions when targeting PO_4^{3-} removal in bioretention.

Differences among treatments over time provide additional insight into the removal potential of each treatment. Flow rate through soil media has been related to PO_4^{3-} removal efficiency of bioretention soil media (Hsieh et al. 2007b). Studies that investigated different flow rates determined nutrient removal was more successful during lower flows due to increased contact time between the runoff and soil media (Davis et al. 2006; DeBusk and Wynn 2011). Removal of PO_4^{3-} in this study occurred in the greatest amounts during low flow rates at the tails of the hydrograph.

Results for PO_4^{3-} from the grouped time-step category analysis, indicated patterns in removal for both 25% and 100% sand treatments were similar in that they removed and loaded PO_4^{3-} at the same time-steps along the hydrograph. Interestingly, no differences were found for PO_4^{3-} between the beginning and the end of the hydrograph for either treatment, indicating that PO_4^{3-} was retained equally as well at the end of the event as it was at the beginning. A lag between the peak of the event and the peak of PO_4^{3-} load which occurred later was noted, with the highest increases occurring at the 150-180 grouped time-step. This lag in increased PO_4^{3-} leaving the systems resulted in differences between the 150-180 and the beginning and end of the hydrograph, indicating that the 25% and 100% sand treatments loaded PO_4^{3-} for one hour after the peak before being able to reduce PO_4^{3-} again. Lag time between the peak of the event and peak of a pollutant leaving the system has not been observed in literature before due to the lack of application of a simulated storm event. The observed peak in PO_4^{3-} is important to note because it is when the greatest amount of pollutant is leaving the system.

5.4 Nitrate Change

Results indicate that treatments with greater amounts of sand had significantly less NO_3^--N load reduction than those with greater organic matter. However, there were no differences between the treatments and the controls. Variability in NO_3^--N load reduction in bioretention experiments is not uncommon and leaching from the soil media is often times observed (Hsieh and Davis 2005; Davis et al. 2006; Hunt et al. 2006; Hsieh et al. 2007a). Nitrate does not adsorb to soil particles the way PO_4^{3-} does, but instead is highly mobile in soil profiles and prone to leaching (Brady and Weil 1996). These results were the opposite of those found for PO_4^{3-} which may indicate difficulties in managing for both PO_4^{3-} and NO_3^--N in the same facility unless modifications are made to target both.

Poor NO_3^--N reduction in bioretention facilities has also been attributed to limited contact time with soils and preferential flow paths (Hsieh and Davis 2005; Li and Davis 2009; DeBusk and Wynn 2011). Limited contact time between the runoff and soil media is often due to increased flow rates through the soil profile or thin soil layer (Li and Davis 2009). The thin layer of soil in this experiment and observed preferential flow patterns, led to decreased contact time between the storm water solution and the soil which was likely the reasons NO_3^--N removal among treatments was not achieved. Successful NO_3^--N removal results have been observed with deeper soil media systems and when a saturated (i.e. anaerobic) layer was added below the soil layer (Dietz and Clausen 2006; Hunt et al. 2006; Brown and Hunt 2010; DeBusk and Wynn 2011). Soils with adequate organic matter and an anaerobic zone are expected to perform well for NO_3^--N removal through microbial transformation processes (O'Leary et al. 2002). Additionally, plant material, which assimilates N in the form of NO_3^--N, could have improved NO_3^--N reduction in this study.

When comparing concentration to load for individual treatments, NO_3^--N concentrations differed significantly from the loads for the 75% and 100% sand treatments, suggesting either volume reduction or flow rate influenced the load results.

Results from comparing treatments at individual time-steps indicated the 100% sand significantly differed from other treatments before the peak. Additionally, results indicated that the 100% sand treatment removed a significantly greater amount of NO_3^--N during the 30-90 minute time-step group when compared to the 120-270 time-step groups. However, unlike PO_4^{3-}, the NO_3^--N reductions were not observed at the end tail of the hydrograph indicating a decreasing capacity through time among all treatments to remove NO_3^--N. The pattern of NO_3^--N reduction in the 100% sand mixture mirrors the volume reduction pattern of the 100% sand mixture throughout the course of the simulated storm event. This may indicate that the reduction of NO_3^--N was directly related to volume reduction, which results indicate happened during the first 90 minutes of the experiment. Brown and Hunt (2010) have noted that volume reduction was one of the primary factors in improving nutrient removal in field applications. While volume reductions were significantly reduced in my experiment, the thin layer of soil and preferential flow patterns could have reduced the potential for the mesocosms to further reduce volume and thus NO_3^--N.

CHAPTER VI

CONCLUSIONS

Findings of this study are promising, as no studies have explored the changes that occur along a simulated storm event for runoff volume, peak flow, and nutrients in bioretention facilities. By simulating a rainfall event and observing these changes over the length of the hydrograph, patterns were found that would not otherwise be observable. This chapter offers general conclusions, thoughts on the application of flow-through stormwater planters by landscape architects, limitations of the study, and future research recommendations.

6.1 Experiment Conclusions

Volume reduction in the flow-through mesocosms was successful in all of the treatments. Despite only 11.43 cm (4.5 in.) of media, a reduction in volume of up to 20% occurred. This indicates that flow-through facilities have the potential to retain a substantial percentage of rainfall even though they lack exfiltration to surrounding soils. Since flow-through facilities rely solely on the soil available in the cell, it is important to optimize the soil for runoff volume retention. While treatments with the most sand content retained the greatest volume in this experiment, this may have been due to the particle size of the incorporated compost. Treatments high in compost retained the least amount of runoff, indicating that the size of the compost particles used for this experiment may be reducing the volume retention potential of the media. It is important to specify particle size in bioretention soil mixes to avoid large pieces of compost which do not promote volume reduction.

Peak flow attenuation has been a difficult goal to achieve in bioretention studies. A lack of peak flow reduction is believed to be due to large compost size and the single inflow point which both promoted preferential flow paths through the mesocosms. While infiltration rates were not measured, observations noted during the experiment showed that runoff was not coming in contact with all the soil in the facility. This was due to preferential flow paths, which were in part due to the use of a single inflow point.

Changes in PO_4^{3-} loads were not conclusive in the overall results but several conclusions can be drawn from observations made over the course of the hydrograph. Soil treatments reduced PO_4^{3-} at the beginning of the event, primarily when volume reductions occurred and when flow rates were low. At the peak of the hydrograph (120-minute time-step) differences between the 25% sand and 100% sand treatments indicated treatments with greater clay and silt content reduced PO_4^{3-} more effectively than treatments with greater sand content. Treatment performance at the 120-minute time-step was found to be important due to the fact that roughly half of the synthetic runoff was delivered over this 30 minute time period. A lag time occurred for an hour after the peak of the event, where PO_4^{3-} was loading even though lower flow rates were occurring. After the lag period, treatment performance improved to the same level observed at the beginning of event, indicating PO_4^{3-} removal efficiency is not directly tied to volume reduction.

The overall removal performance of NO_3^--N in the flow-through mesocosms was inconclusive; however, patterns were observed between the rising and falling limbs of the hydrograph. Observations along the hydrograph revealed that NO_3^--N changes are directly related to volume changes in the mesocosms. Removal was observed at the beginning of the hydrograph when volume reduction was greatest, however after the treatments were saturated, they did not perform well for NO_3^--N removal. As other research suggests, successful NO_3^--N improvements could occur in deeper systems unlike the scaled mesocosms used in this experiment. Deeper systems increase residence time and create conditions favorable for denitrification processes to take place.

Observations indicate there may be a design flaw with a single, concentrated inflow point into a flow-through facility, which is common in real-world applications. Single inflow points created preferential flow paths which prevented runoff from contacting all of the soil and therefore the soil media was underutilized. This potential design flaw in flow-through planters could lead to lower nutrient reductions and limited peak flow reductions. One solution to this dilemma may be a dispersed inflow system which has the potential to be more effective than a single inflow point due to increased contact time between runoff solution and the soil media.

6.2 Application for Landscape Architects

6.2.1 In Research

Interdisciplinary research provided in this project is noteworthy. Stormwater runoff modeling and experimental methodologies given by the Department of Landscape Architecture were complimented by the water quality and analyses expertise provided by the Department of Wildlife, Fisheries & Aquaculture. As further research on urban stormwater BMPs is conducted, it is critical that the process continues to incorporate knowledge from appropriate fields. Structural design, modeling, and application are strengths of landscape architects; however, this research requires a much broader base of knowledge that includes horticulture, soil science and wetland science in order to understand the specific biogeochemical processes, which occur in these systems.

6.2.2 In Practice

With further research and proper modifications, flow-through planters can be a practical solution for runoff quantity and quality concerns in the urban environment. Potential for flow-through planter application is promising although designers should promote exfiltration into *in situ* soils as the preferred option, which literature has shown to be the most effective at retaining volume, mitigating peak flows, and reducing nutrients. Design professionals such as landscape architects and civil engineers should encourage the adaptation of guidelines and specifications to ensure proper application of small-scale stormwater BMPs, especially as new research becomes available.

As flow-through planters and other bioretention facilities are considered for application, it is important to consider which pollutants are being targeted, which type of storm event is being managed, and how to adjust the structural components of a flow-through planter to achieve quantity and quality goals. While an optimal soil for the removal of both PO_4^{3-} and NO_3^--N has not yet been pinpointed, commonalities in research are beginning to emerge that narrow down the critical variables. As illustrated in this research, soils with higher organic content perform better for PO_4^{3-} removal while NO_3^--N reduction was more related to volume reduction. Additional variables which have been found in literature to contribute to removing these nutrients are deeper soil, a subsurface reservoir to promote denitrification, plant material for assimilation, and a mulch layer to promote adsorption, filtering and some dispersion.

Different bioretention facilities are utilized based on location and management goals. Flow-through planters are lined and are therefore highly dependent on the engineered soil mixture. If the mixture is deeper with greater volume capacity then it could mitigate both PO_4^{3-} and NO_3^--N, given the hydraulic residence time is long enough and the configuration does not allow for nitrification or preferential flow paths to occur. Bioretention should also incorporate plants that are efficient at PO_4^{3-} assimilation. P-index of the native soil used should be examined to determine the existing amount. This could help determine how much additional P can be adsorbed by the soil.

6.3 Limitations of the Study

There are several limitations to this study that should be recognized. First, this was a scaled experiment. While the results should be somewhat indicative of what can be expected in real-world applications, the results are only relative between the mesocosms in the experiment. Second, this experiment only simulated a Type II storm event and water quantity and quality results may differ during storm types from other regions of the United States. Third, this experiment only tested one concentration of two nutrients. Flow-through mesocosms may yield different results with varied concentrations or other types of urban pollutants. Lastly, the low number of replicates may have limited the number of statistically significant findings.

6.4 Recommendations for Future Research

Further research is needed to determine how to eliminate preferential flow paths in flow-through planters. Preferential flow paths undermine the purpose of the bioretention soil which is to increase contact time and encourage water to absorb into the soil media. This could be accomplished by first eliminating the single inflow point and dispersing inflow water evenly across the soil surface. Preferential flow paths may also be reduced by eliminating excessive void space created by coarse compost. Further research should investigate whether or not compost is necessary or if finer compost should be specified by bioretention recommendations.

This research project was unique in that it used varied flow rate pumps to simulate a storm event in an experimental setting. Other types of storm events could be simulated with this setup to determine how the varied soil mixtures perform in different areas of the country. Exploring other types of storm events may suggest in which regions of the country flow-through planters could be the most beneficial for quality and quantity improvements, and may additionally reveal which structural components need to be modified for those regions.

Optimizing soil depth in flow-through planters for quantity and quality improvements is also warranted. Increased soil media depths will encourage increased volume reductions and therefore may provide desired nutrient reductions and peak flow reductions. Greater depth and volume of soil has been shown to contribute to greater NO_3^--N reductions, however, space in urban environments may limit this component. Deeper soil media may also help reduce the potential for preferential flow paths.

Additional research is encouraged to investigate additional pollutants removal potential in flow-through stormwater planters. While further research for nutrients is needed, common urban pollutants such as hydrocarbons and heavy metals could be targeted with flow-through facilities. As results from scaled mesocosms become available, field scale experiments can be implemented to provide additional insight into how these types of facilities perform in the field.

REFERENCES

Arnold Jr, C. L., and C. J. Gibbons. 1996. "Impervious Surface Coverage: The Emergence of a Key Environmental Indicator." *Journal of the American Planning Association* 62 (2): 243-258.

Bannerman, R. T., D. W. Owens, R. B. Dodds, and N. J. Hornewer. 1993. "Sources of Pollutants in Wisconsin Stormwater." *Water Science & Technology* 28 (3): 241-259.

BES (Bureau of Environmental Services) City of Portland. 2006. *Flow-Through Planters (November 2006)*. Accessed July 15, 2012. http://www.portlandoregon.gov/bes/article/127475.

———. 2008. *Stormwater Management Manual Revision 4.*

———. 2010. *2010 Stormwater Management Facility Monitoring Report Summary.* Sustainable Stormwater Management Program. Accessed April 23, 2013. http://www.portlandoregon.gov/bes/article/343463.

Bohn, H. L., R. A. Myer, and G. A. O'Connor. 2002. *Soil Chemistry*. John Wiley and Sons.

Boller, M. 2004. "Towards Sustainable Urban Stormwater Management." *Water Science and Technology-Water Supply* 4 (1): 55-66.

Booth, D. B., and C. R. Jackson. 1997. "Urbanization of Aquatic Systems: Degradation Thresholds, Stormwater Detections, and the Limits of Mitigation 1." *Journal of the American Water Resources Association* 33 (5): 1077-1090.

Booth, D. B., and L. E. Reinelt. 1993. "Consequences of Urbanization on Aquatic Systems-Measured Effects, Degradation Thresholds, and Corrective Strategies." In *Proceedings of the Watershed '93 Conference*: 545-550.

Brady, N. C., and R. R. Weil. 1996. *The Nature and Properties of Soils*. 11th ed. Prentice-Hall.

Brown, R. A., and W. F. Hunt. 2010. "Impacts of Media Depth on Effluent Water Quality and Hydrologic Performance of Undersized Bioretention Cells." *Journal of Irrigation and Drainage Engineering* 137 (3): 132-143.

Busman, L., J. Lamb., G. Randall, G. Rehm, and M. Schmitt. 2002. *The Nature of Phosphorus in Soils*. Minnesota Extension Service, University of Minnesota. Accessed June 1, 2013. http://www.extension.umn.edu/distribution/cropsystems/dc6795.html.

Carpenter, S. R., N. F. Caraco, D. L. Correll, R. W. Howarth, A. N. Sharpley, and V. H. Smith. 1998. "Nonpoint Pollution of Surface Waters with Phosphorus and Nitrogen." *Ecological applications* 8 (3): 559-568.

Carson, R. 1962. *Silent Spring*. Cambridge, Mass, Houghton Mifflin.

CWP (Center for Watershed Protection). *Stormwater Management*. Accessed February 10, 2013. http://www.cwp.org/2013-04-05-16-15-03/stormwater-management

Davis, A. P. 2007. "Field Performance of Bioretention: Water Quality."*Environmental Engineering Science* 24 (8): 1048-1064.

———. 2008. "Field Performance of Bioretention: Hydrology Impacts." *Journal of Hydrologic Engineering* 13 (2): 90-95.

Davis, A. P., W. F. Hunt, R. G. Traver, and M. Clar. 2009. "Bioretention Technology: Overview of Current Practice and Future Needs." *Journal of Environmental Engineering* 135 (3): 109-117.

Davis, A. P., and R. H. McCuen. 2005. *Stormwater Management for Smart Growth*. Springer.

Davis, A. P., M. Shokouhian, H. Sharma, and C. Minami. 2006. "Water Quality Improvement Through Bioretention Media: Nitrogen and Phosphorus Removal." *Water Environment Research* 78 (3): 284-293.

DeBusk, K. M., and T. M. Wynn. 2011. "Storm-Water Bioretention for Runoff Quality and Quantity Mitigation." *Journal of Environmental Engineering* 137 (9): 800-808.

Debo, T. N., and A. Reese. 2003. *Municipal Stormwater Management*. 2nd ed. Boca Raton, FL: Lewis Publishers, CRC Press.

Dietz, M. E., and J. C. Clausen. 2006. "Saturation to Improve Pollutant Retention in a Rain Garden." *Environmental science & technology* 40 (4): 1335-1340.

Echols, S., and E. Pennypacker. 2008. "From Stormwater Management to Artful Rainwater Design." *Landscape Journal* 27 (2): 268-290.

EPA (Environmental Protection Agency). 1996. *Managing Urban Runoff*. EPA-841-F-96-004G. Washington D.C.

———. 1999a. *Preliminary Data Summary of Urban Storm Water Best Management Practices*. EPA-821-R-99-012. Washington, D.C.

———. 1999b. *Storm Water Technology Fact Sheet: Bioretention*. EPA-832-F-99-012. Office of Water, Washington, DC.

———. 2000a. *Stormwater Phase II Final Rule: An Overview. Fact Sheet 1.0*. EPA-833-F-00-001. Office of Water, Washington, D.C.

———. 2000b. *Low Impact Development (LID): A Literature Review (October 2000)*. EPA-841-B-00-005. Prepared by the Low Impact Development Center. Office of Water, Washington, D.C.

———. 2003. *After the Storm: A Citizen's Guide to Understanding Stormwater*. EPA-833-B-03-002. Washington, D.C.

———. 2005. *Storm Water Phase II Final Rule: Small MS4 Storm Water Program Overview. Fact Sheet 2.0*. EPA-833-F00-002. Washington, D.C.

Erickson, A. J., J. S. Gulliver, and P.T. Weiss. 2007. "Enhanced Sand Filtration for Storm Water Phosphorus Removal." *Journal of Environmental Engineering* 133 (5): 485-497.

Gallo, C., A. Moore, and J. Wywrot. 2012 "Comparing the Adaptability of Infiltration Based Bmps to Various US Regions." *Landscape and Urban Planning* 106 (4): 326-335.

Goldberg, S, and G. Sposito. 1985. "On the Mechanism of Specific Phosphate Adsorption by Hydroxylated Mineral Surfaces: A Review."*Communications in Soil Science & Plant Analysis* 16 (8): 801-821.

Hatt, B. E., T. D. Fletcher, and A. Deletic. 2008. "Hydraulic and Pollutant Removal Performance of Fine Media Stormwater Filtration systems." *Environmental science & technology* 42 (7): 2535-2541.

Hinman, C. 2009. *Bioretention Soil Mix Review and Recommendations for Western Washington (January 2009).* Tacoma, WA: Puget Sound Partnership and Washing State University Pierce County Extension.

Holman-Dodds, J. K. 2007. "Towards Greener Stormwater Management." *Journal of Green Building* 2 (1): 68-96.

Horner, R. R. 1994. "Fundamentals of Urban Runoff Management: Technical and Institutional Issues." Terrene Institute, Washington, D.C.

Hsieh, C., and A. P. Davis. 2005. "Evaluation and Optimization of Bioretention Media for Treatment of Urban Storm Water Runoff." *Journal of Environmental Engineering* 131 (11): 1521-1531.

Hsieh, C., A. P. Davis, and B. A. Needelman. 2007a. "Nitrogen Removal from Urban Stormwater Runoff through Layered Bioretention Columns." *Water Environment Research* 79 (12): 2404-2411.

———. 2007b. "Bioretention Column Studies of Phosphorus Removal from Urban Stormwater Runoff." *Water Environment Research* 79 (2): 177-184.

Hunt, W. F., A. P. Davis, and R. G. Traver. 2011. "Meeting Hydrologic and Water Quality Goals through Targeted Bioretention Design." *Journal of Environmental Engineering* 138 (6): 698-707.

Hunt, W. F., J. T. Smith, S. J. Jadlocki, J. M. Hathaway, and P. R. Eubanks. 2008. "Pollutant Removal and Peak Flow Mitigation by a Bioretention Cell in Urban Charlotte, NC." *Journal of Environmental Engineering* 134 (5): 403-408.

Hunt, W. F., A. R. Jarrett, J. T. Smith, and L. J. Sharkey. 2006. "Evaluating Bioretention Hydrology and Nutrient Removal at Three Field Sites in North Carolina." *Journal of Irrigation and Drainage Engineering* 132 (6): 600-608.

Ige, D. V., O. O. Akinremi, and D. N. Flaten. 2007. "Direct and Indirect Effects of Soil Properties on Phosphorus Retention Capacity." *Soil Science Society of America Journal* 71 (1): 95-100.

Kim, H., E. A. Seagren, and A. P. Davis. 2003. "Engineered Bioretention for Removal of Nitrate from Stormwater Runoff." *Water Environment Research* 75 (4): 355-367.

Lee, J. H., and K.W. Bang. 2000. "Characterization of Urban Stormwater Runoff." *Water Research* 34 (6): 1773-1780.

Lee, J. H., K.W. Bang, L. H. Ketchum Jr, J. S. Choe, and M. J. Yu. 2002. "First Flush Analysis of Urban Storm Runoff." *Science of the Total Environment* 293 (1): 163-175.

Li, H., L. J. Sharkey, W. F. Hunt, and A. P. Davis. 2009. "Mitigation of Impervious Surface Hydrology Using Bioretention in North Carolina and Maryland." *Journal of Hydrologic Engineering* 14 (4): 407-415.

Li, H., and A. P. Davis. 2009. "Water Quality Improvement through Reductions of Pollutant Loads Using Bioretention." *Journal of Environmental Engineering* 135 (8): 567-576.

LID (Low Impact Development Center). 2003. *Drainage - Bioretention Specification*. Accessed June 7, 2012. http://www.lowimpactdevelopment.org/epa03/biospec.htm.

Lucas, W. C., and M. Greenway. 2008. "Nutrient Retention in Vegetated and Nonvegetated Bioretention Mesocosms." *Journal of Irrigation and Drainage Engineering* 134 (5): 613-623.

O'Leary, M., G. Rehm, and M. Schmitt. 2002. "Understanding Nitrogen in Soils." *Minnesota Extension Service Folders*. Accessed February 20, 2013. http://www.extension.umn.edu/distribution/cropsystems/dc3770.html.

Passeport, E., W. F. Hunt, D. E. Line, R. A. Smith, and R. A. Brown. 2009. "Field Study of the Ability of Two Grassed Bioretention Cells to Reduce Storm-Water Runoff Pollution." *Journal of Irrigation and Drainage Engineering*135 (4): 505-510.

PGC (Prince George's County) Department of Environmental Resources. 1999. *Low-Impact Development Design Manual*. Prince George's County, MD.

———. 2007. *Bioretention Manual*. Prince George's County, MD.

Reddy, K. R., and R. D. DeLaune. 2004. *Biogeochemistry of Wetlands: Science and Applications*. Boca Raton, FL, CRC Press.

Roy-Poirier, A., P. Champagne, and Y. Filion. 2010. "Review of Bioretention System Research and Design: Past, Present, and Future." *Journal of Environmental Engineering* 136 (9): 878-889.

Schueler, T. R. "The Importance of Imperviousness. 1994." *Watershed protection techniques* 1 (3): 100-111.

USDA (United States Department of Agriculture) Soil Conservation Service. 1986. *Urban Hydrology for Small Watersheds: Technical Release 55 (June 1986)*. Prepared by the Natural Resources Conservation Service, Conservation Engineering Division. Washington, D.C.

Wise, S. 2008. "Green Infrastructure Rising." *Planning* 74 (8): 14-19.