An evaluation of engineered media for phosphorus removal from greenroof stormwater runoff

by

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A thesis

presented to the University of Waterloo

in fulfillment of the

thesis requirement for the degree of

Master of Science

in

Geography

Waterloo, Ontario, Canada, 2011

AUTHOR'S DECLARATION

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Abstract

Greenroofs are increasingly being recognized as an effective site level best management practice (BMP) to reduce the volume of stormwater runoff in urban environments. For some water quality constituents, greenroofs can improve runoff water quality but recent studies demonstrate greenroofs are sources rather than sinks of phosphorus (P). Accordingly, further research is required to evaluate treatment technologies that improve the performance of these BMPs. This study examined the use of two engineered media types to reduce phosphorus loadings from a greenroof located on the Archetype Sustainable House at Kortright in Vaughan, Ontario.

A treatment system was installed to capture and remove P in stormwater runoff using sorptive properties of an engineered media. A mass balance approach was used to evaluate pre and post-treatment water quality. Pre and post-treatment water samples were collected for 25 rainfall events from July 11, 2009 to August 22, 2010 and analyzed for soluble reactive phosphorus (SRP), total phosphorus (TP), suspended solids (SS) and total dissolved solids (TDS). Storm events ranged in return frequencies from < 2 years to 35 year periods. The results show that the greenroof was a consistent source of P. The volume weighted mean concentrations were 0.769 mg/L and 0.630 mg/L for 2009 and 2010 events, respectively. The media used in 2009 reduced SRP loadings by 32.0% and TP loadings by 25.4%. The media evaluated in 2010, reduced SRP loadings by 82.4% and TP loadings by 86.6%. The greater P removal demonstrated by the 2010 media is attributed to a higher specific surface area and increased P sorptive capacity. Results of this study will help inform the use of sorptive materials in greenroof applications and a wider range of best management practices for stormwater quality treatment.

Acknowledgements

I would like to express my sincerest gratitude to my supervisor, Mike Stone. Thank you for your help, guidance and advice, for matters in academia and in life. Your unwavering positive encouragement was always greatly appreciated.

I would also like to thank Monica Emelko for her patient assistance with several key concepts related to this research. Thank you to my examining committee members Tim Van Seters and Merrin Maccrae for offering your time and comments to this study.

I am indebted to my partners in this study, the Toronto and Region Conservation Authority and Imbrium Systems Inc. Thank you to Glenn MacMillan, Tim Van Seters, and Alex Waters for helping initiate this study and the continued support and advice at the study site. A huge thank you goes to Christy Somerville, Matt Derro, and Paul Greck for their extremely patient help with the organization and collection of data and general problem solving. This study would have been much more difficult without your help. Thank you to Brian Lee, Stephen Braun, and Reagan Davidson for providing technical information and materials used in this study. I would also like to acknowledge the financial support of the MITACS Accelerate Internship program.

Thank you to all my labmates who shared the space in the Sediment and Water Quality Lab. In the frustration of labwork my labmates were often a beacon of sanity.

Lastly, and most importantly, I am thankful for all the support and love from my family and friends throughout this experience. My time at the University of Waterloo has been truly incredible and I have all the amazing people in my life to thank for that.

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Chapter 1: Introduction

1.1: Background

There is increasing concern at the global scale regarding the impacts of urbanization on water quantity and quality. In urban areas vegetation is typically removed and replaced with impervious surfaces that dramatically alter natural hydrologic processes (Bernhardt and Palmer, 2007). Impervious surface cover such as paved roadways and rooftops is a commonly used index of urban intensity and surrogate measure of aquatic health (Arnold and Gibbons, 1996; Schueler, 1994; Booth et al., 2005). Impervious surfaces decrease infiltration rates and subsequently increase the rates and magnitude of surface runoff (Dunne and Leopold, 1978). Accordingly the increase in stormwater runoff can result in flooding (OME, 2003b; USEPA, 1999), contamination of drinking water supplies (Marsalek and Rochfort, 2004), alteration of the hydrology (Booth and Jackson, 1997), stream morphology (Paul and Meyer, 2001) and ecology (Walsh et al., 2005b) of receiving waters. Stormwater has traditionally been viewed from a flood management perspective (Roy et al., 2008) but there is increasing recognition of its role in the transport of nonpoint pollution (Novotny and Olem, 1994; Carpenter et al., 1998). Urban stormwater runoff often contains elevated levels of pollutants such as metals, nutrients, salts, sediments and organic contaminants such as pesticides (Schueler, 1987; Grapentine et al., 2004). Furthermore, the increased peak flow of stormwater runoff can result in combined sewer overflows (CSOs) that release untreated sewage into the receiving waters (Bernhardt and Palmer, 2007).

To mitigate the effects of impervious surfaces and stormwater runoff, stormwater management (SWM) programs have been implemented in many urban areas (OME, 2003b). Early approaches to stormwater management were to direct runoff quickly and efficiently to rivers and streams (Roy et al., 2008). The subsequent increase in stream flow resulted in geomorphic changes such as bank erosion and destruction of pool and riffle zones which impact aquatic ecosystems (Bradford and Gharabaghi, 2004). Stormwater management has more recently evolved to include consideration of both the quantity and quality of stormwater. Recently the concept of sustainable urban drainage systems (SUDS) or low impact development (LID) which use a treatment train approach has been incorporated in the design of urban infrastructure to treat stormwater. This approach employs a range of source, conveyance and end of pipe treatment controls to reduce the volume of stormwater runoff while improving the water quality (Dietz, 2007; Marsalek and Chocat, 2002). These systems attempt to restore predevelopment hydrology by increasing pervious surface coverage and minimizing the connection between stormwater drainage and receiving waters (Dietz and Clausen, 2008; Walsh, 2005b).

A variety of structural and non-structural controls, or best management practices (BMPs) are used to mitigate the effects of stormwater runoff (Marsalek and Chocat, 2002). One source control BMP that is quickly gaining recognition within North America is the use of greenroofs. Greenroofs often consist of a growing media to support plant communities underlain with drainage materials which are incorporated into the roof membrane (Carter and Fowler, 2008). Properly designed greenroofs can reduce stormwater runoff (Liu, 2003; VanWoert et al., 2005; TRCA, 2006; Carter and Rasmussen, 2006; DeNardo et al., 2005). Additional benefits of greenroofs include energy conservation through reduced heating and cooling of buildings (Del Barrio, 1998; Niachou et al., 2001), potential reduction of the urban heat island (Oberndorfer et al., 2007) and increased habitat (Lundholm, 2006). Additionally, they offer a potential retrofit implementation in heavily urbanized areas which may not be able to incorporate other BMPs such as stormwater ponds due to available land constraints (Carter and Jackson, 2007).

Germany has been at the forefront of greenroof technology, policy and research for several decades (Ngan, 2004). These developments have helped promote European standards of greenroof use in stormwater management. The adoption of this technology as a stormwater source control in North America has been slowed by barriers such as lack of quantifiable research, technical expertise, public awareness, financial incentives and industry standardization (Velazquez, 2005b; Getter and Rowe, 2006, Carter and Fowler, 2008). In addition to reducing the volume of stormwater runoff, greenroofs have the potential to improve stormwater quality (Johnston and Newton, 1996; Peck et al., 1999). German studies (Köhler et al., 2002) report an improvement of water quality from greenroof runoff. However results of recent studies in North American indicate that greenroofs act as a source of phosphorus (P) and are an ineffective technology for metals removal from stormwater (Van Seters et al., 2009; Vander Linden and Stone, 2009; Moran, 2004; Berndtsson et al., 2006). The export of P is previously attributed to composition of the greenroof growing media, and the application of fertilizers (Emilsson et al., 2007).

The leaching of nutrients such as P from greenroofs is problematic. Because P is the limiting nutrient in fresh water bodies, the transfer of P in stormwater to surface waters may cause eutrophication, resulting in algal blooms and related ecological impacts (Carpenter et al., 1998). Various P removal technologies have been developed for use in treating both wastewater and stormwater. The mechanisms for P removal include adsorption, ion exchange and precipitation reactions (Zhang et al., 2008). Chemical precipitation using alum, lime, and ferric chloride is a widespread removal method, however the cost and production of sludge by-products make this unsuitable for stormwater treatment (Li et al., 2006) Several studies have explored the use of low-cost sorptive materials such as limestone, bauxite, zeolite, fly ash, shale, steel wool, gas concrete, red mud and cement for P removal (Drizo et al., 1999; Agyei et al., 2002, Erickson et al., 2007). In recent years, the use of engineered media such as aluminum-oxide and iron-oxide coated media has been investigated for P removal (Ma and Sansalone, 2007; Boujelben et al., 2008) and these engineered media show promise as a technology for the treatment of stormwater quality. However, most studies of engineered media for P removal have been conducted at the laboratory scale and there is a need to further examine these technologies at the field scale. Such information is required to measure the performance of engineered media to remove P in runoff from greenroofs and determine its utility for stormwater management programs.

1.2: Study Objectives:

The specific objectives of this study are to:

1. Conduct a field scale study to investigate the water quantity (runoff retention, lag time) and water quality characteristics (soluble reactive phosphorus, total phosphorus, pH, total dissolved solids, suspended solids, grain size) of stormwater from greenroof runoff and evaluate the efficiency of a treatment system with engineered media to remove phosphorus.

2: Determine the phosphorus sorption capacity and predicted lifespan of the engineered media in a series of laboratory experiments.

3. Recommend design and logistical considerations for future implementation of engineered media for P removal in greenroof runoff.

1.3: Literature Review

1.3.1: Stormwater Runoff Impacts

Urbanization alters the natural hydrologic cycle by increasing the magnitude of surface runoff and flow velocities (Walsh et al., 2005b). Areas with high impervious surface cover (ISC) are often susceptible to flooding when the design capacity of drainage infrastructure is exceeded. This occurs when rainfall rates exceed a critical value (typically the design storm) (OME, 2003b; Schueler, 1987). In addition to flooding, storm runoff in urban systems can change stream geomorphology (Paul and Meyer, 2001) and destabilize stream banks that often lead to a disconnection between riparian areas and streams (Booth and Jackson, 1997).

Stormwater has increasingly been recognized as a major source of nonpoint pollution (Novotny and Olem, 1994). Stormwater transports a wide range of pollutants and pathogens into receiving waters (Chocat et al., 2007; Tsihrintzis and Hamid, 1997). The natural pollutant filtration processes of vegetation and soil are typically decreased because of the widespread use of asphalt and concrete (TRCA, 2007). Compared to pre-development levels, urban runoff typically contains increased levels of suspended solids (SS), nutrients, bacteria, oils, metals, organic contaminants and chlorides (Schueler, 1987). Suspended solids are an important water quality indicator as they are the primary vector for many of the other pollutants such as nutrients and metals which adhere to these particles (Arnold and Gibbons, 1996). The degraded water quality and resulting impacts on storm flow on stream geomorphology can severely impact the ecology of aquatic systems (Finkenbine et al., 2000; Walsh et al., 2005a).

1.3.2: Stormwater Management in Urban Systems

The wide range of problems associated with stormwater runoff has necessitated the development of stormwater management (SWM) programs to effectively manage these problems (OME, 2003b). Traditionally stormwater was viewed as a flood risk. Consequently, runoff was managed by a rapid removal from the land through sewer systems, curbs, and gutters and into receiving streams and rivers (Roy et al., 2008). Many streams were lined with concrete and channelized, effectively transforming streams into extended gutter systems (Dunne and Leopold, 1978). By the 1970s, Ontario had begun to incorporate stormwater ponds as a means of controlling runoff volume. While ponds did reduce flooding and infrastructure requirements, they did not fully address the water quality issues of stormwater runoff (Watt et al., 2003). By the 1990s SWM had shifted towards mitigating water quality impacts through the use of best management practices (BMPs). BMPs are defined by USEPA (1999) as a device, practice or method for removing, reducing, or preventing targeted storm water runoff quantity, constituents, pollutants and contaminants from reaching receiving waters. BMPs can be classified into non-structural and structural categories. Non-structural BMPs include policies, education and public participation, urban planning and development and behavioral modifications that reduce stormwater impacts (Marsalek, 2005). Structural BMPs include many of the physical measures used to manage stormwater including ponds, catchbasins, oil/grit separators, porous pavements and grassed swales (Cameron et al., 1999). Currently Ontario's stormwater management recommends a treatment train approach as described in the *Stormwater Management and Planning Design Manual* (OME, 2003b). The treatment train approach uses BMPs ordered in a series of source (lot level), conveyance and end-of-pipe controls (OME, 2003b). This approach directs stormwater through various structures, allowing for cumulative water quality improvement and an overall mitigation of peak flows (Bernhardt and Palmer, 2007). While the benefits of a treatment train approach are recognized, there continues to be an over-reliance on stormwater ponds and end-of-pipe measures (Bradford and Gharabaghi, 2004). The ponds are generally effective in controlling stormwater quantities, however at large volumes, overflow can still occur which reduces the effectiveness of some treatment processes such as sedimentation (Bäckström et al., 2002). Additionally, ponds can be costly to implement and maintain and require large areas of land (Weiss et al., 2007).

1.3.3: Use of Low Impact Development Techniques

The use of low impact development (LID) techniques represents a shift from reliance on end-of-pipe structures. LID is a recent stormwater practice which emphasizes better site design by minimizing impervious cover, reducing EIA and protecting natural vegetation and landscapes (Dietz, 2007; Walsh et al., 2005b). Originating in the 1990s from the State of Maryland, LID is a movement towards a sustainable form of stormwater management (TRCA, 2007). The use of small-scale controls in a uniformly decentralized manner attempts to restore the pre-development hydrology by using site level processes of infiltration, storage, evaporation and detention (USEPA, 2000). Additionally, stormwater quality is improved through filtration, biodegradation and other natural processes (Marsalek, 2005). Graham et al. (2004) mentions the various benefits of LID including the protection of downstream geomorphology, improved water quality and ecology, reduced flooding and combined sewer overflow risks, and a reduction in the requirements for traditional end-of-pipe controls. It should be noted that the use of end-of-pipe controls is still needed to capture runoff not treated by source controls and they also provide a means of flood protection. The various advantages that LID techniques provide over traditional stormwater management appear to make them an attractive option for implementation into stormwater plans. However, Roy et al. (2008) list several barriers to LID use. These include performance uncertainties and cost, insufficient engineering standards, fragmented responsibilities, lack of institutional capacity and legislative mandate, a lack of market incentives and an overall resistance to change. The authors offer several solutions to these impediments with emphasis on increased research on performance and more wide-spread education to both professionals and the public.

1.3.4: Greenroof Definition

Greenroofs are becoming increasingly popular source control processes within LID projects. Greenroofs involve the creation of a green space on top of an urban structure (Peck et al., 1999), allowing for infiltration and storage of stormwater. Often referred to by a variety of names including vegetated roofs, ecoroofs, or rooftop gardens, these systems can be broadly defined as a specialized roof system which supports vegetation (Liu and Baskaran, 2005). While the use of greenroofs is not a recent innovation, there has been increasing recognition of their potential application in stormwater management as well as various other benefits. Modern greenroofs use various components (Fig. 1) which have been designed to increase the stormwater function (Scholz-Barth, 2001). Collectively these systems are often referred to as greenroof technology (GRT).

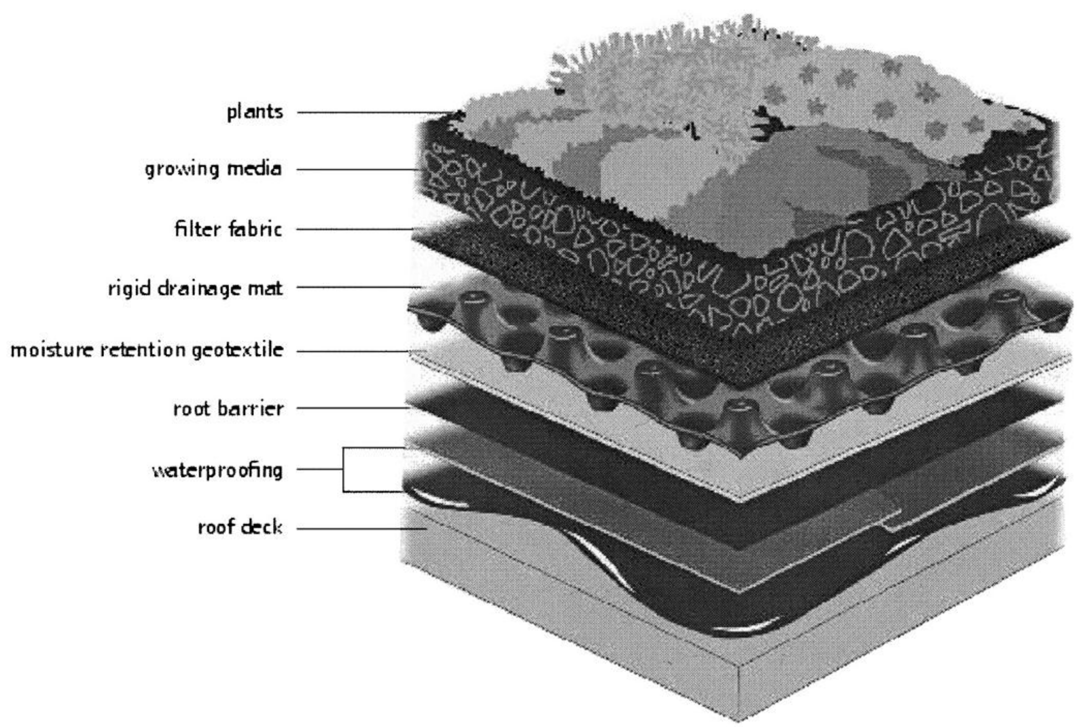


Figure 1: Extensive roof structural components (Carter and Rasmussen, 2006).

1.3.5: Greenroof Application

The modern use of greenroofs can be traced to their adoption in Germany, Switzerland, Austria and Scandinavia in the 1960s. In the following decades, a considerable amount of research was conducted to advance knowledge and use of components such as vegetation, growing media and drainage technology (Peck et al., 1999). By the 1980s, Germany was at the forefront of greenroof technology and policy. The greenroof market was expanding by 15-20% annual growth, becoming a large financial opportunity (Peck et al., 1999). By 2001, approximately 13.5 million m^2 of greenroofs had been installed, which is equivalent to 14% of the total roof area in Germany (Ngan, 2004). Comparatively, greenroof use in North America has been limited to individual building owners promoting construction efforts (Banting et al., 2005). Several cities such as Toronto, Portland and Chicago have promoted research through demonstration projects and developed policies to increase their use (City of Toronto, 2008). In 2008, the city of Chicago instituted a stormwater ordinance which requires new developments and redevelopments of a specific size to retain the first 1.27 cm of stormwater on site (City of Chicago, 2007). The city Toronto has recently introduced a by-law requiring the implementation of a greenroof on new developments, with varying size criteria for different building types such as industrial, commercial and residential (City of Toronto, 2008).

1.3.6: Greenroofs: Classification, Components, Functions

Greenroofs are most commonly classified as either intensive or extensive (Mentens et al., 2006). These classifications are largely based on growing media depth, substrate type and plant type. Intensive greenroofs have a wide range of media depths but generally contain at least 15 cm of media. The greater media depth supports larger vegetation such as shrubs and trees. Intensive greenroofs are often used for public access to create a natural aesthetic environment for urban areas (Getter and Rowe, 2006). The use of a deep substrate and heavier vegetation creates a greater structural load on the building. Retrofit projects are often required to upgrade the roof structure resulting in increased costs (Lawlor, 2006). Intensive roofs provide a range of advantages such as greater biodiversity, accessibility and the largest capacity for stormwater retention and insulation properties. However, their significant cost, maintenance and structural requirements often restrict widespread application. Extensive roofs are much more commonly used in greenroof research and implementation. These roofs use a substrate depth which ranges between 5 and 15 cm and typically weigh between 72.6 and 169.4 kg/m^2 (Lawlor, 2006). These roofs are lightweight and cost-effective, resulting in their use over much larger areas. Generally, vegetation biodiversity is low and the roofs inaccessible to the public. However, this also lowers the maintenance requirements (Carter and Fowler, 2008). The typical greenroof consists of vegetation, growing medium, filter layer, drainage layer and a waterproof membrane (Liu and Baskaran, 2005; Peck et al., 1999).

A variety of mosses, grasses and herbaceous plants are used with extensive greenroofs. Succulent plants such as *Sedum spp* (Stone Crop) are the most commonly used vegetation. *Sedum spp* are drought and frost tolerant, making them suitable for a North American climate. The plants are appropriate for greenroof use due to their shallow-rooting system, ability to store water in the succulent portions above the soil and exhibit CAM photosynthesis which helps in drought conditions (Wolf and Lundholm, 2008). Monterusso et al. (2005) determined that a variety of *Sedum spp* was suitable for greenroof use in the Midwestern United States. The typical green roof using *Sedum spp* can withstand drought periods for several months, although watering should occur approximately every 28 days (VanWoert et al., 2005b). Furthermore, Sedum *spp* can survive in substrates containing minimal organic matter and reduced fertilizer applications (Rowe et al., 2006). The composition of the growing media depends on the vegetation (Liu and Baskaran, 2005). A small amount of organic material may be mixed with lightweight aggregates such as perlite, to reduce structural loads (FLL, 2002). The filter layer is composed of a geotextile material which prevents fine particles originating from the growing medium from clogging the drainage layer (Liu and Baskaran, 2005). The drainage layer is composed of a lightweight composite material such as foam panels or a highly porous polymeric mat which allows excess water to drain from the growing medium. The waterproof membrane acts a barrier to water and prevents root penetration into the building structure (Liu and Baskaran, 2005; Ngan, 2004).

The components are engineered to provide a variety of benefits to greenroof use. The use of greenroofs for stormwater management has generated recent interest in LID literature (Dietz, 2007; Mentens et al., 2006; USEPA, 2000). The reduction of stormwater runoff volume is a strategy for reducing the polluting effects associated with urban stormwater (VanWoert et al., 2005; Carter and Rasmussen, 2006). In addition to managing stormwater quantities, greenroofs have been promoted for their use in improving water quality (Köhler et al., 2002; Berndtsson et al., 2006; Long et al., 2006). Additional benefits include decreased heating and cooling costs (Del Barrio, 1998; Kosareo, 2007; Niachou et al., 2001; Liu, 2004; Wong et al., 2003a), reduction in urban heat island (Getter and Rowe, 2006; Oberndorfer et al., 2007), urban amenity through green space creation (Velazquez, 2005b; Pincetl and Gearin, 2005), extension of roof life (Peck et al., 1999; Wong et al., 2003b) and creation of habitat to improve biodiversity (Baumann, 2006; Grant, 2006; Köhler, 2006). Only within the past five years have any North American research papers been published examining greenroofs performance as a stormwater management technology. Previously, greenroof research was primarily focused on roof energy budget studies (Carter and Rasmussen, 2006). Currently, there is a limited body of published research examining greenroof stormwater performance. Continued research within this field is required to further the adoption of this technology into urban stormwater management policies.

1.3.7: Stormwater Volume Control Using Greenroofs

One of the most important aspects of stormwater management is controlling the quantity of stormwater runoff. Many urban centres have combined sewer systems that convey both stormwater and wastewater (Semadeni-Davies et al., 2006). The infrastructure capacity is often overwhelmed by large stormwater volumes moving at a high flow rate, resulting in combined sewer overflows (CSOs) which discharge untreated sewage into waterways (Marsalek, 2005). Greenroofs represent a technology which decreases stormwater volumes and peak flow rates, lessening the harmful impacts of runoff and reducing the risk of CSOs. Furthermore, greenroofs may be implemented through retrofit projects into heavily urbanized areas which are restricted by land constraints (Carter and Rasmussen, 2006; Marsalek and Chocat, 2002).

The retention capacity of a greenroof is affected by several variables. Various studies have examined the influence of storm size, storm frequency, seasonality, roof slope, substrate depth as well as vegetation type and coverage (Hathaway et al., 2008; Villarreal and Bengtsson, 2005; VanWoert et al., 2005). Storm size has been an important variable in greenroof research. Carter and Rasmussen (2006) determined stormwater retention by greenroofs decreased with greater precipitation depth. For storms under 25.4 mm, retention was approximately 90%. Between 25.4 mm and 76.2 mm, retention was 54%. Beyond 76.2 mm, retention dropped to less than 48%. Liu and Minor (2005) had 100% retention for storms less than 15 mm. Similarly sized storm events of 15 mm demonstrated a much smaller percent reduction (> 20%) in a study by Bliss et al. (2009). VanWoert et al. (2005) examined the performance of various substrate depths and roof slopes compared to different storm sizes. For a 2% slope, 4.0 cm substrate depth roof had the highest mean percentage for all storm sizes. The roof retained 97.1% for light storms (< 2 mm), 85.5% for medium storms (2-6 mm) and 65.1% for heavy storms (> 6 mm). The study also showed that vegetation did not have a large effect on retention, thus attributing the largest influence on the media. The results show that media depth did not change the retention on the steepest slope of 6.5° but did at the most gradual slope of 2°. The results were not statistically significant to draw any conclusions regarding the influence of media depth.

When storm events occur, the greenroof substrate will eventually become saturated, reducing the retention abilities. In soil, water retention is controlled by osmotic and capillary forces. As saturation occurs, the force of gravity overcomes these adhesive forces, draining water from the soil. The maximum amount of water that a freely drained soil can store is referred to as the field capacity of the soil (Dunne and Leopold, 1978). The aforementioned studies quantify the field capacity of the greenroof substrate. Upon reaching field capacity, greenroofs exhibit largely reduced retention ability. The field capacity of greenroof substrates has been examined through increased substrate depth, roof age and slope. Jarrett et al. (2006) reported that a 3 mm substrate depth could retain 20-45% of annual rainfall and that increasing substrate depth did not improve retention. Liu and Minor (2005) examined two greenroofs with depths of 75 mm and 100 mm, yet reported no difference with both roofs demonstrating a 57% average annual reduction. Similarly, Hathaway et al. (2008) tested two roofs with depths of 75 mm and 100 mm and reported an equal retention of 64%; however, these two roofs were at separate locations and subjected to differing hydrologic conditions. The effects of roof age and slope have shown conflicting results (Villarreal and Bengtsson, 2005; VanWoert et al., 2005). A review by Mentens et al. (2006) reported that greenroof age and slope angle were not significantly correlated with the yearly runoff. In contrast, a study by MHW Americas Inc., conducted for the City of Chicago (2006), concluded that a three year old greenroof absorbed approximately 40-50% of storm events, whereas a newly installed greenroof with identical characteristics absorbed 10-30%. The authors attribute the establishment of the sedum root systems for the increase in retention. A study conducted at Michigan State University demonstrated significant relationships between both roof age and slope and total retention. A five year old substrate had almost twice the field capacity as compared to a new substrate (Getter et al., 2007). The authors attribute increased porosity within the mature substrate for a higher field capacity. This may be due to root growth and burrowing insects. Roof slopes of 2°, 7°, 15°, and 25° were examined over various storm sizes. Slope was significant when comparing 2° and 15°, as well as 2° and 25°. Differing methodology and antecedent moisture conditions may have led to the conflicting results (Getter et al., 2007).

The frequency of storm events is an important variable for the field capacity and moisture conditions of a greenroof. Hathaway et al. (2008) demonstrated the importance of an inter-event dry period on retention percentages. As several rain events occurred over consecutive days, the percent precipitation retained quickly dropped to only 53%. Bliss et al. (2009) also noted the effect of moisture saturation within the roof substrate. The authors reported that when the roof was near field capacity (30% water by volume before a storm event) only 0.1 cm of rain was absorbed. Reducing the water content to approximately 15% increased the absorption to 0.5 cm. Liu and Minor (2005) reported that the two greenroofs used in their study retained 100% of a 15 mm storm event when preceded by six days of dry weather. During warm seasons, evapotranspiration rates increase which regenerates the field capacity more rapidly than during cool seasons (Mentens et al., 2006). Hathaway et al. (2008) noted the role of evapotranspiration in precipitation retention. During the months of September, October and December, there was approximately 85 mm of rainfall. During the first two months 78% retention occurred but in December it dropped to 62%. The potential evapotranspiration rates (PET) were calculated as 96, 51, and 6.9 mm / month matching the decrease in retention. The winter performance of greenroofs has been addressed in only a few studies (VanWoert et al., 2005; Teemusk and Mander, 2007). While retention percentages may be less than during summer months, greenroofs still act to reduce the runoff rates.

The reduction of peak flows and extension of time until runoff (lag time), has important implications for stormwater management that has been widely reported in the literature. The reduction in peak flow can be influenced by storm size and frequency. TRCA (2006) reported an average peak flow reduction of 87% for storm events between 10 and 29 mm and a 50% reduction for storm events greater than 40 mm. Conversely, Vander Linden and Stone (2009) reported that storm size only affected the control roof peak flow, but not the greenroof rate. Several papers have shown peak flow rate reductions of 50-70% (Hathaway et al., 2008; Liu and Minor, 2005; Bliss et al., 2009; Vander Linden and Stone, 2009). The reduction in peak flow rates can be partially attributed to the increase in lag time. Carter and Rasmussen (2006) note that runoff lag times increased from 17.0 min from the control roof to 34.9 min for the greenroof. Liu and Minor (2005) reported an increase in lag time of 20-40 min, while DeNardo et al. (2005) found that greenroof runoff for all rain events was delayed an average of 5.7 hours. Bliss et al. (2009) also reported a runoff delay of several hours on the greenroof; however after the substrate became saturated, further precipitation would runoff at the same rate as the control roof (Fig. 2).

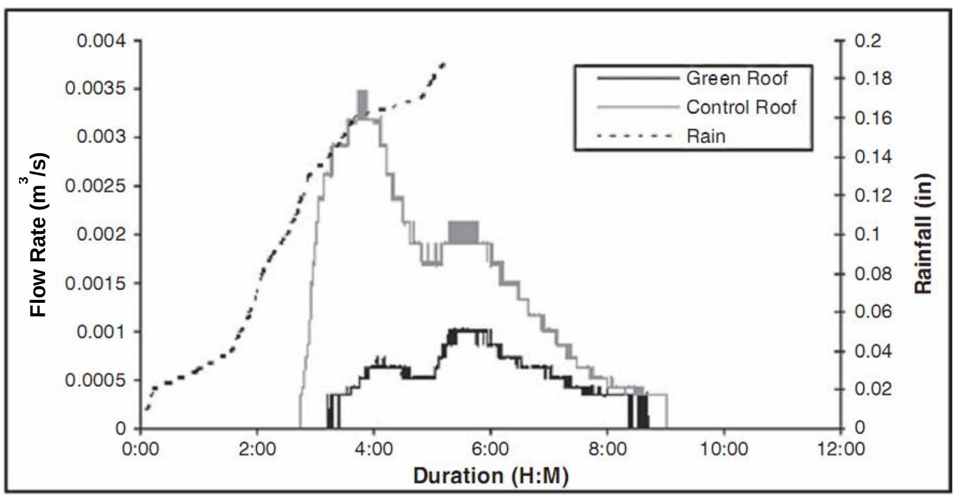


Figure 2: Comparison of peak flow rates and lag time for greenroof and control roof (Bliss et al., 2009).

Following a rainfall event, greenroofs may slowly release runoff over an extended period of time. Getter et al. (2007) showed runoff extending from 4 h 20 min for light rain events (<2.0 mm) to 13 h 45 min for heavy rain events (>10.0 mm).

The use of greenroofs in stormwater management may represent a significant reduction in the volumes and flow rates of urban runoff. Applying the wet weather performance to a larger scale emphasizes the possible benefits. Getter et al. (2007) estimate that if the 1.1 km^2 of flat roof surfaces at Michigan State University were converted to greenroofs, based on a mean retention of 80.2%, 377 041 m^3 of rainfall would have been retained during the course of 2005. Using hydrologic simulations, the TRCA (2006) reported that if 100% of the roofs within the Highland Creek watershed were greened, there would be an overall 4% reduction in annual runoff volumes and 15% peak flow reduction for events between 20 and 30 mm. A study conducted in Toronto by Banting et al. (2005) estimated the potential monetary savings associated with implementing greenroofs on a city-wide scale. Based on 4 984 hectares of available roof area, the initial savings from complete coverage was estimated at $100,000,000 for stormwater costs alone. This estimate included savings associated with pollution and erosion control as well as avoidance of other costly BMP use. The reduction of CSOs was estimated to have an additional savings of over $46,000,000.

1.3.8: Greenroof Influence on Stormwater Quality

Greenroofs have been thought to improve urban runoff water quality by absorbing pollutants of wet and dry atmospheric deposition (Berndtsson et al., 2006). Through plant uptake or binding within the growing medium, contaminants such as metals, nitrogen and P are filtered out of the rainwater. Earlier studies in Germany reported decreased concentrations of metals and nutrients from vegetated roofs (Köhler et al., 2002). There are a limited number of studies which have investigated the water quality characteristics of greenroof runoff. Conventional roofs have been shown to act as a source of pollutants during rain events (Zobrist et al., 2000; Schueler, 2000; Mason et al., 1999). During a storm event, the initial concentrations of pollutants are elevated as accumulated pollutants are washed off the roof surface. This “first flush” effect has been observed in various greenroof studies (Berndtsson et al., 2006; Van Seters et al., 2009).

There have been conflicting results regarding greenroofs acting as a source of nitrogen. Moran (2004) recorded significantly higher concentrations than both the rainfall and control roof runoff. However, Van Seters et al. (2009) noted that mean nitrogen levels were lower from the greenroof compared to the control roof. In studies examining the effects of fertilization practices and use of slow release fertilizers, Monterusso et al. (2005) found that nitrogen levels were greater in greenroof runoff. Emilsson et al. (2007) showed that vegetated growing media retained more nitrogen than unvegetated substrates, likely due to uptake by the vegetation. Several studies have noted 10 to 20 fold increases in P concentrations in greenroof runoff compared to rainfall and control roof runoff (Van Seters et al., 2009; Bliss et al., 2009; Vander Linden and Stone, 2009; Moran 2004; Berndtsson et al., 2006; Hutchinson et al., 2003). High rates of nutrient runoff are linked to roof age, substrate composition and fertilizer applications (Emilsson et al., 2007, Long et al., 2006). As vegetated roofs mature in age, nutrient export tends to decrease (TRCA, 2006; Köhler et al., 2002).

1.3.9: Phosphorus: Freshwater Impacts, Forms and Sorption Mechanisms

P export from greenroofs may be problematic in the context of surface water quality. Ontario's *Provincial Water Quality Objectives* recommend total phosphorus (TP) levels <0.03 mg/L to avoid problematic plant growth in streams and rivers. The concentrations of P in greenroof can be orders of magnitude higher than this guideline. Runoff leaves the rooftops it is transported through a sewer system where it is eventually discharged into a receiving water body. The loss of P to surface waters can result in environmental problems such as eutrophication (Correll, 1998). As P is often the limiting nutrient in lentic bodies of water such as lakes and ponds, an excess input of P increases the productivity of the system (Schindler, 1977). This increased productivity results in the rapid growth of algae as they consume bioavailable P. P levels as low as 0.01 to 0.05 mg/L may result in eutrophication (Smil, 2000). Algal blooms can be problematic because they decrease light penetration in the water and lower oxygen levels. Bacterial decomposition of dead algae removes oxygen from the water column resulting in hypoxic conditions that adversely affecting aquatic fauna (Carpenter et al., 1998). Certain algal species can release noxious metabolites which are harmful to humans and produce taste and odour problems in drinking water (Watson et al., 2008).

P is most commonly separated into operationally defined forms as either particulate phosphorus (PP) or dissolved (DP). DP is defined as the P fraction passing through a 0.45 µm filter (Logan, 1982). Total phosphorus (TP) is the sum of DP and PP (Reddy et al., 1999). Both DP and PP have inorganic and organic fractions. The inorganic fraction of DP includes the orthophosphate forms $H_2PO_4^-$, HPO_4^{2-}, PO_4^{3-}, all of which are considered bioavailable. Bioavailable P is defined as the sum of immediately available P and P that can be transformed into an available form by naturally occurring processes (Boström et al., 1988). Dissolved inorganic phosphate is immediately available for algal uptake. Certain forms of PP are also considered bioavailable. Inorganic P is highly immobile and adsorbs onto amorphous and crystalline forms of Fe, Al, Ca and other cationic elements. These forms are labile and become available through dissolution or desorption processes (Logan, 1982).

The phosphorus sorption capacity (PSC) of sediments is dependent on several factors such as clay content, organic matter content, pH, concentrations of Al, Fe, and Ca, reaction time and temperature. (Tisdale et al., 1985; Del Bubba et al., 2003; Reddy and DeLaune, 2008). The anionic properties of soluble P result in electrostatic attraction to positively charged soil surfaces. The soluble P forms are also attracted to the hydrous oxide surfaces of Al and Fe. These surfaces carry a net positive charge in acidic soils due to their protonated surface sites and therefore have a high anion exchange capacity (vanLoon and Duffy, 2005).

The surfaces of particulate matter may also retain P through specific binding mechanisms which involve covalent bonding (vanLoon and Duffy, 2005). Amorphous precipitates are formed as the P anions form inner sphere ligand complexes with the metal ions (Hedley, 2008; Reddy and DeLaune, 2008). Ligands are anions, cations, or neutral molecules which bind to a central metal atom or ion. Inner sphere complexes involve direct sharing of electrons between the metal and the ligand (Kasprzyk-Hordern, 2004). The formation of these precipitates is often due to high concentrations of either phosphate or metal cations. In calcareous soils orthophosphates are adsorbed to the surface of $CaCO_3$. With high concentrations of phosphate ions the calcium phosphate complex will form an insoluble precipitate (Reddy and DeLaune, 2008). When phosphates are precipitated to Fe, Al, or Ca ions the structure is initially amorphous. Over time the precipitate will become crystalline in nature, increasing the stability and reducing the potential for P desorption. Increased temperature is a large factor in promoting the crystallization of the precipitate and is therefore a factor in P sorption (Reddy and DeLaune, 2008).

1.3.10: Phosphorus Removal Techniques and Technologies

The high level of P in urban stormwater has generated interest in developing P removal technologies to mitigate this problem. P removal technologies have been developed and used in the wastewater industry for several decades, and this knowledge and experience is quickly being used to manage the stormwater field. Conventional wastewater treatment technology uses a combination of biological (activated sludge processes) and physiochemical mechanisms (including chemical precipitation, ion exchange and membrane processes) (Ayoub and Kalinian, 2006). Physiochemical mechanisms of treatment are favoured for stormwater, as the use of P storing bacteria requires various conditions such as anaerobic zones and the presence of biodegradable soluble organics (Zhang et al., 2008).

In stormwater runoff P is typically in both dissolved and particulate forms, removal technologies use a combination of chemical removal techniques such as adsorption and ion exchange and physical processes such as settling and filtration (Hatt et al., 2008; Hsieh et al., 2007). P is preferentially bound to finer materials (<63 µm) but can associate with the entire size range of particulate matter that is transported through urban stormwater (1 – 10 000 µm) complicating removal procedures (Ma and Sansalone, 2007). Particles can be operationally defined as sediment (>75µm), settleable (75-25 µm) and suspended (<25 µm) based on the 1 hour Imhoff settling test (Kim et al., 2008). Many of the BMPs used in stormwater control and water treatment rely on settling processes to remove pollutants associated with larger particles. However, ponds, basins and constructed wetlands may not effectively remove suspended and colloidal fractions which fail to settle, necessitating the use of sorption processes (Genç-Fuhrman et al., 2007).

Several studies have investigated the PSC of various media. Most of the research has been focused on the use of different media in wastewater treatment, yet there remains a limited set of literature focused on media for stormwater application (Kandasamy et al., 2008; Erickson et al., 2007; Hatt et al., 2008; Davis et al., 2001; Liu et al., 2001; Lucas and Greenway, 2008). Several studies have explored the use of low-cost sorptive materials such as limestone, bauxite, zeolite, fly ash, slag, shale, steel wool, gas concrete, red mud and cement for P removal (Drizo et al., 1999; Johansson and Gustafsson, 1999; Agyei et al., 2002; Erickson et al., 2007). Low-cost sorptive materials are an attractive option for financially constrained stormwater projects. Furthermore, the use of industrial by-products such as fly ash and steel slags is a way in which to recycle waste products into a useful function. However, the industrial by-products are naturally variable and there remains a concern for metal leaching (Li et al., 2006). In recent years the use of engineered media such as aluminum-oxide and iron-oxide coated media has been investigated for their efficiency in P removal (Ma and Sansalone, 2007; Boujelben et al., 2008). Engineered media are designed to maximize sorption potential by using characteristics such as large specific surface area (Liu et al., 2001), increased porosity (Khadhraoui et al., 2002) and materials with a high cation exchange capacity (>10 meq) (Hunt et al., 2006; Kasprzyk-Hordern, 2004; Mengel, 1982).

Sorption processes can be categorized into electrostatic forces, physical forces and chemical bonding (Minton, 2002). Ion exchange is a form of electrostatic forces. As solution passes through the media, a pollutant ion (preferred ion) may be replaced with an ion from the media (less preferred ion). The media is exhausted when all the least preferred ions are exchanged for preferred ions (Minton, 2002). Adsorption is the process where matter (eg. molecules, ions, particles, polymers, colloids) is dispersed in solution and accumulates on the surface of an adsorbant (Kasprzyk-Hordern, 2004). Adsorption may occur through weaker physisosorption processes and stronger chemisorption through the formation of bonds between the adsorbate and adsorbent. The physical forces are due to van der Waal interactions as partial charges of the absorbate are attracted to the electrostatic charges of the absorbent (Minton, 2002). Chemisorption is a much stronger interaction but requires an available sorption site on the adsorbent. Once chemical bonds are formed, desorption becomes more difficult. Generally physical adsoption has rapid kinetics, followed by a slower diffusion of the adsorbed particle into the matrix of the adsorbent (Kim et al., 2008).

Metal oxides are ideal for engineered media due to their high affinity towards phosphate molecules. Through ligand bonds the phosphate molecule (anion) form inner-sphere surface complexes with the metal ion (cation) (Fig. 3).

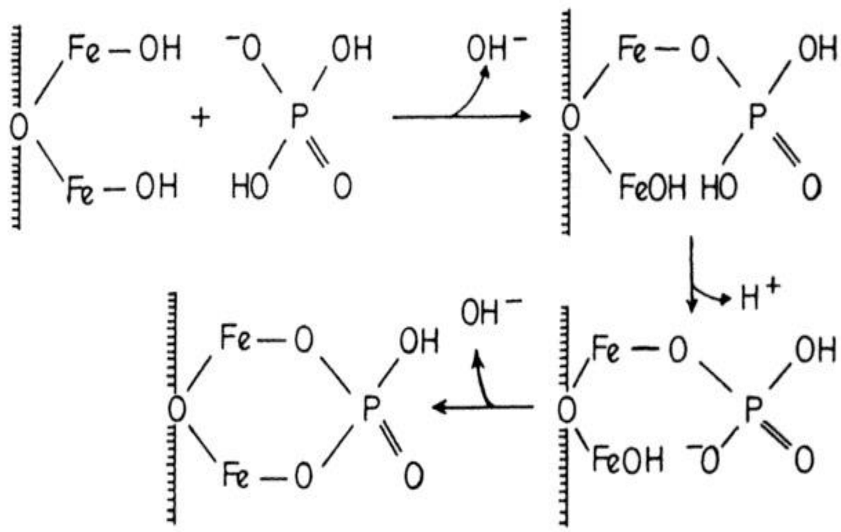


Figure 3: Ligand exchange model illustrating an inner-sphere surface complex. (Adapted from Mengel, 1982).

Due to the surface interactions involved within adsorption processes, a higher surface area enhances sorption capacity and rates of removal as more sorption sites are available (Sansalone, 1999; Minton, 2002). Sorption reactions are affected by competing species which may impede the removal of pollutants. Changes in temperature can also affect sorption processes. Higher temperatures increase the vibration frequencies of sorbed molecules, making desorption more likely (Minton, 2002). Georgantas and Grigoropoulou (2007) reported that as an aluminum hydroxide media aged the P removal performance was altered. After six months the performance decreased 10-15%, which is attributed to the hydroxide undergoing polymerization leading to an increase in crystal size. As the size increases, P sorption sites are lost due to structural bridging.

The largest influence on sorption reactions is pH of the solution and this is best exemplified when considering the surface chemistry of metal oxides. In the presence of water, the metal oxides are surrounded by hydroxyl groups, protons and coordinated water molecules (Liu et al., 2001b). The metal oxides are amphoteric, allowing them to act as either an acid or base. This property is influenced by the surface charge and therefore pH of the surrounding solution. When a certain pH is attained and the surface charge of the metal oxide is zero, this is referred to as the point of zero net charge (PZC). If the pH increases, the surface becomes negatively charged due to the increased prevalence of hydroxide ions (OH^-) in solution. Conversely, a decrease in pH results in a positively charged surface as more hydrogen ions (H^+) are in solution (Fig. 4). Engineered media are designed to perform within a specific pH range. For example, an engineered media with a PZC of 11 has a high positive charge within the typical stormwater pH range of 6-8 (Liu et al., 2001a) and therefore would effectively bind anionic species, such as phosphates.

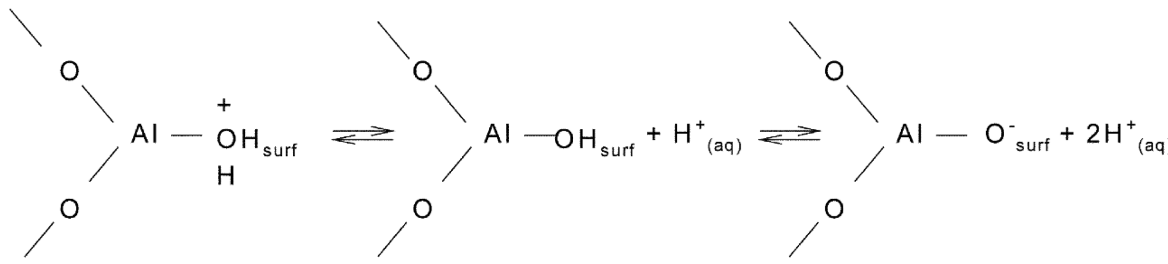


Figure 4: Effect of pH on aluminum hydroxide surface chemistry (Kasprzyk-Hordern, 2004).

1.3.11: Optimization of Sorptive Media

Much of the literature examining the removal of pollutants from stormwater has focused on heavy metals (Seelsaen et al., 2006; Genç-Fuhrman et al., 2007; Hatt et al., 2008; Liu et al., 2001a,b; Liu et al., 2004; Liu et al., 2005; Sansalone and Teng, 2004). Field studies examining P removal have focused primarily on the filtration technologies for the removal of particulate P. These studies include a wide range of BMP structures such as: bioretention cells (Dietz and Clausen, 2006; Hunt et al., 2006; Hsieh et al., 2007; Lucas and Greenway, 2008), storage basins (Kandasamy et al., 2008), sedimentation tanks (Sonstrom et al., 2002), wet ponds (Wang et al., 2002), partial exfiltration reactors (Sansalone and Teng, 2004) and constructed wetlands (Lüderitz and Gerlach, 2002). Several lab studies have examined the sorption capacities of media for DP removal. Erickson et al. (2007) examined the performance of sand amended with steel wool, calcareous sand and limestone enhancements. Each enhancement significantly increased performance; however column tests made the issue of clogging apparent. Often very fine sized media will have high P removal capacities due to its large surface area however clogging may result when particles become trapped on the surface of the filter. An effective filter needs to maintain hydraulic conductivity, allowing for solution to move through the filter, yet have high sorption capacity to achieve removal objectives. Hsieh et al. (2007) noted that media with large pore sizes are effective at preventing clogging from suspended solid inputs. Maintenance and regeneration is an important consideration in the life-cycle analysis of filter systems. As clogging of fine media filters occurs at the top of the filter, Hatt et al. (2008) recommend simply scraping off the clogged layer to regain removal performance but eventually the filter reaches a point where the desired effluent concentrations are not being maintained.

This concept is referred to as the operating capacity. Accordingly, as the filter becomes saturated with pollutants, the performance reaches a breakpoint and effluent concentrations rapidly increase (Minton, 2002).

1.3.12: Summary

There remains a need to examine the use of adsorptive filter media for application in a variety of stormwater BMPs. Urban stormwater runoff has a multitude of adverse effects including flooding, erosion, water quality impairment and ecological degradation (Marsalek and Chocat, 2002). Modern approaches to stormwater management use a treatment train approach that incorporates a sequence of BMPs to manage water quantities and improve water quality. A review of available literature demonstrates the potential for greenroof implementation within LID projects. Greenroofs provide a site-level control of stormwater by increasing storage, thereby reducing runoff volumes and peak flow rates. However, the literature also indicates that vegetated roofs are a source of pollutants, particularly P. Due to the risks associated with increased P loading to freshwaters, there has been increasing research activity developing technologies for P removal. Many of these technologies act to filter PP, without removing the bioavailable dissolved forms. The use of adsorptive media, such as engineered metal oxides, provides a means of attaining very low concentrations of P in stormwater runoff. There are limited BMP field studies which have examined the use of an engineered media in pollutant removal performance. Currently, no study has integrated an engineered media component to treat the runoff of a greenroof system. This study addresses this gap in knowledge and will provide valuable information for the management and implementation of greenroof systems.

Chapter 2: Methods

2.1: Experimental Design

The purpose of this research is to examine the utility of engineered media for P removal from greenroof runoff. To achieve this goal the research was conducted at two scales; field scale (to assess the effectiveness of the engineered media to remove P in runoff) and laboratory scale (to evaluate the P sorption capacity of the engineered media). For the field scale study, a mass balance approach was used to assess the P removal performance of an engineered media treating stormwater runoff from a greenroof. Runoff was directed from the roof surface into a treatment system containing engineered media. Water samples were collected at the inlet and outlet of the treatment system for twenty five rainfall events from July 11, 2009 to October 28, 2009 and May 7, 2010 to August 22, 2010. The samples were analyzed for soluble reactive phosphorus (SRP), total phosphorus (TP), suspended solids (SS), total dissolved solids (TDS) and grain size (GS). SRP is operationally defined as dissolved P in the form PO_4^{3-} (passing through a 0.45 µm filter).

The second objective was to assess the PSC of engineered media through a series of equilibrium batch tests. Engineered media was exposed to a series of working solutions of varying P concentrations (0 – 100 mg/L). The adsorption data were fitted to the Langmuir equation to determine the adsorption capacity of the media under specific chemistry conditions such as temperature and pH. Isotherm tests also permit the media's functional lifetime expectancy to be estimated.

To assess the ability of the greenroof to reduce the quantity of stormwater runoff, hydrologic data was collected for 68 rainfall events from June 11, 2009 to October 31, 2009 and May 7, 2010 to August 22, 2010. The flow data was used to examine the greenroof's stormwater retention capacity and calculate mass loadings of pollutants. The influence of the hydrological parameters such as storm size, intensity and antecedent dry period were examined with regard to their influence on overall stormwater retention.

2.2: Study Limitations

The design and location of the greenroof presented several technical and logistical limitations for the study. Throughout the first six storm events, half of the Archetype House roof area was being drained onto the greenroof through a downspout (Figure 5). This increased the total drainage area to approximately 37.13 m^2. During the July 11, 2009 event, the volume of runoff exceeded the treatment system capacity due to the increased drainage area. The downspout was eventually rerouted to decrease the excess runoff volumes.

A variety of equipment problems occurred during the study. Power outage during storm events resulted in the ISCO 6712 Automated Samplers® failing to complete sampling programs on several occasions. Additionally, the samplers experienced equipment malfunctions. On July 25th 2009, the post-treatment sampler failed to trigger due to a malfunctioning in the distribution arm. Continued problems with the distribution arm resulted in further loss of samples, and necessitated the replacement of the sampler. The tipping bucket rain gauge malfunctioned from June 1 to June 16, 2010. During this time precipitation data was used from a meteorological station located approximately 450 m from the greenroof.

Due to the distance between the study site and the University of Waterloo, logistical constraints and equipment malfunction prevented 12 events from being monitored for water quality. At times storm events would follow in quick succession. The first storm would be sampled but the second storm would follow before the site could be visited and the programs reset. Additionally, access to the greenroof was limited to Kortright's hours of operation resulting in further logistical problems. Although there were technical and logistical problems associated with the study, hydrological data (precipitation input and greenroof runoff) was collected for 68 rainfall events from June 11th, 2009 to October 31st, 2009 and May 7, 2010 to August 22, 2010. Water quality monitoring was completed on 25 of 37 runoff producing storm events.

Figure 5: Downspout connecting the Archetype House roof to the greenroof.

2.3: Study Site

2.3.1: Greenroof Description

The field study was conducted in Vaughan, Ontario on the greenroof of the Archetype Sustainable House at the Living Campus Centre at Kortright (-79.59° N, 43.83° E). The Archetype Sustainable House is a proof-of-concept showcase of sustainable technologies and innovations for residential housing. The greenroof on this building was designed to demonstrate the concept of low impact development and provide aesthetic appeal to visitors. The roof was built in the summer of 2008 and is located on top of the garage attached to the house (Figure 6). The extensive greenroof was designed with a slope of less than 5° and has a drainage area of 19 m^2. Runoff drains through two downspouts located at opposite corners of the garage. In an effort to simplify sampling procedures, the downspout which drained the majority of the runoff was monitored. The area of the greenroof that drains to the monitored downspout is 13.9 m^2, which represents approximately 73% of the total greenroof area. The area draining to the unmonitored downspout contains a section of the greenroof and a walkway for roof access. Runoff from the unmonitored section of the roof area drained to an area adjacent to the garage.

Figure 6: Archetype House at Kortright, Vaughan, Ontario.

2.3.2: Greenroof Components

The greenroof consists of a vegetation layer, growth medium, root barrier, water retention layer and drainage panel (Figure 7). The vegetation layer is composed of a variety of native plant species selected by a Toronto and Region Conservation Authority botanist for their drought tolerance and ability to grow in thin soils. A complete description of the plants is presented in Appendix C. The vegetation was planted in the spring of 2009 and was slow to reach maximum growth and coverage.

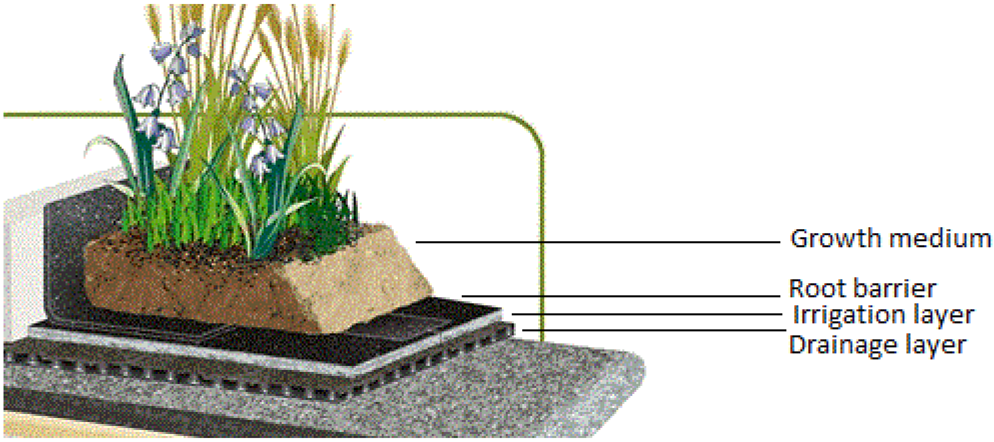


Figure 7: Components of the study greenroof (Soprema Inc.).

The growth medium is 180 mm deep in a layer of SOPRAFLOR I®. This media is designed to support wild flowers, perennials and grasses and was therefore recommended for the plants selected for the vegetation layer. The media consists of mineral aggregates, blond peat, perlite, sand and vegetable based compost (Soprema Inc. Technical data sheet 087021CAN2E, 2009). Beneath the growth medium is a Microfab® root barrier which is composed of a woven polyethylene fabric with micro-perforations that allows the flow of water and air while preventing root penetration (Soprema Inc. Technical data sheet 081219CAN1E, 2009). Beneath this layer is the Aquamat Jardin® irrigation layer. This component can be used for integrated irrigation systems. However, due to the drought tolerance of the vegetation layer no irrigation system was installed. Therefore, the irrigation layer's primary function for this greenroof is a reservoir, holding 11.6 L/m^2 of water (Soprema Inc. Technical data sheet 050428CAN1E, 2009). The final component is the Sopradrain Eco-5® drainage panel to allow excess water to drain at a maximum flow rate of 109 L/min·m (Soprema Inc. Technical data sheet 080208CAN2E).

2.3.3: Treatment System Description

Two different engineered media were examined during the study. Both media were designed using a lightweight pumice substrate and modified with an oxide-coating to improve sorption capacity for dissolved P. The media used during the 2009 monitoring season had a D_{50} of 1.18 ~ 2.00 mm, D_{90} 2.36 ~ 4.75 mm and a specific surface area of 75 – 100 m^2/g. The media used in 2010 had a D_{50} of 0.85 ~ 1.00 mm, D_{90} of 1.0 mm ~ 1.18 mm and an approximate specific surface area of 300 m^2/g (Figure 8).

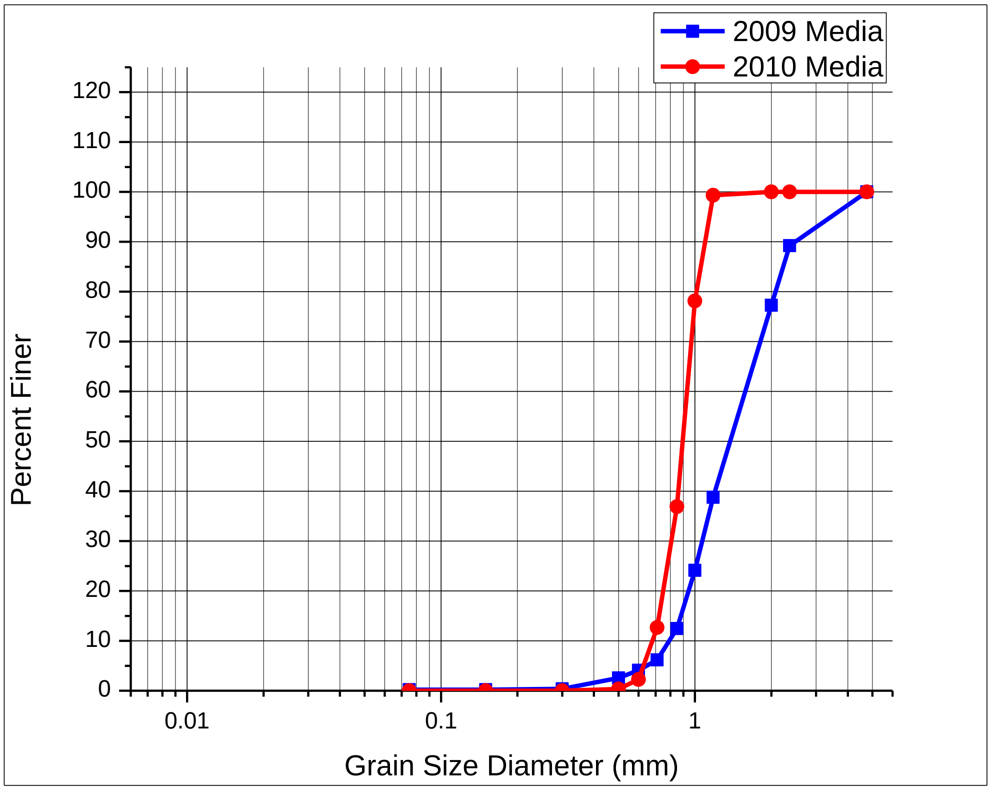


Figure 8: Grain size distribution of engineered media.

The media is housed within a cartridge structure (Figure 9) which is 0.558 m in height and 0.457 m in diameter. The cartridge has an outer surface area of 0.80 m^2. During 2009, the media in the cartridge was filled to a depth of 0.330 m, which represents a volume of 0.051 m^3. In 2010, the depth of media was increased to 0.550 m thus increasing the volume to 0.079 m^3. The entire cartridge structure is contained within a rain barrel 0.914 m high and 0.558 m in diameter (volume of 0.224 m^3). Subtracting the volume of the media, the total volume available for stormwater collection is approximately 0.173 m^3 and 0.145 m^3 for 2009 and 2010, respectively. The cartridge was designed to treat a maximum flow capacity of 1.14 L/s and a surface loading rate of 1.42 $L/m^2 \cdot s$.

Runoff from the greenroof was routed through a tipping bucket rain gauge then into the treatment system through radial flow (Figure 9). The stormwater is treated by the engineered media and exits the cartridge with gravitational flow through a center tube. The treated runoff moves through 30.5 cm of ABS piping before exiting through a 3 mm drain orifice. A 50 mm diameter overflow pipe was installed at the same height of the media to manage for flows exceeding the rate of draindown. Two HOBO® U20

Water Level Loggers were used to measure the runoff volumes within the rain barrel. One logger was placed inside the barrel to record the absolute pressure. The absolute pressure includes both water head and atmospheric pressure. To avoid errors in water level measurements due to barometric variations a second logger was placed outside the barrel. Data from the two loggers were analyzed using HOBOware Pro® version 2.7.3.1 software. Accuracy of the water level measurement is +/- 0.3 cm. The transducers were programmed to take measurements every 30 seconds.

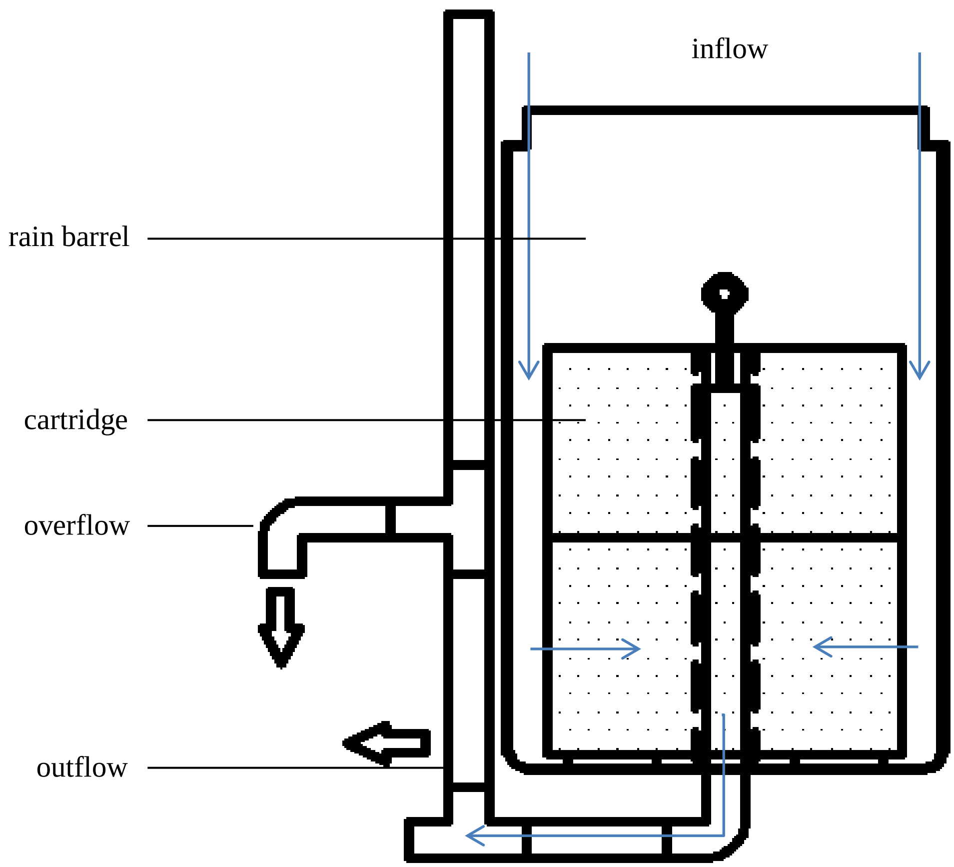


Figure 9: Flow pathways in the treatment system containing engineered media.

2.4: Water Quantity

2.4.1: Runoff

Greenroof runoff drained into a downspout and was then directed through piping into a storage hut containing the cartridge system and flow monitoring equipment (Figure 10). Flows were measured by a VKWA 2000 tipping bucket flow meter with the data being recorded by an attached ISCO 6712 Automated Sampler®. The volume of the tipping bucket is 2.0 L with a maximum flow rate of 24 L/min.

Flow rates were recorded at 1 minute intervals. Flow measurements were downloaded and analyzed with ISCO Flowlink® 5.10.101 software.

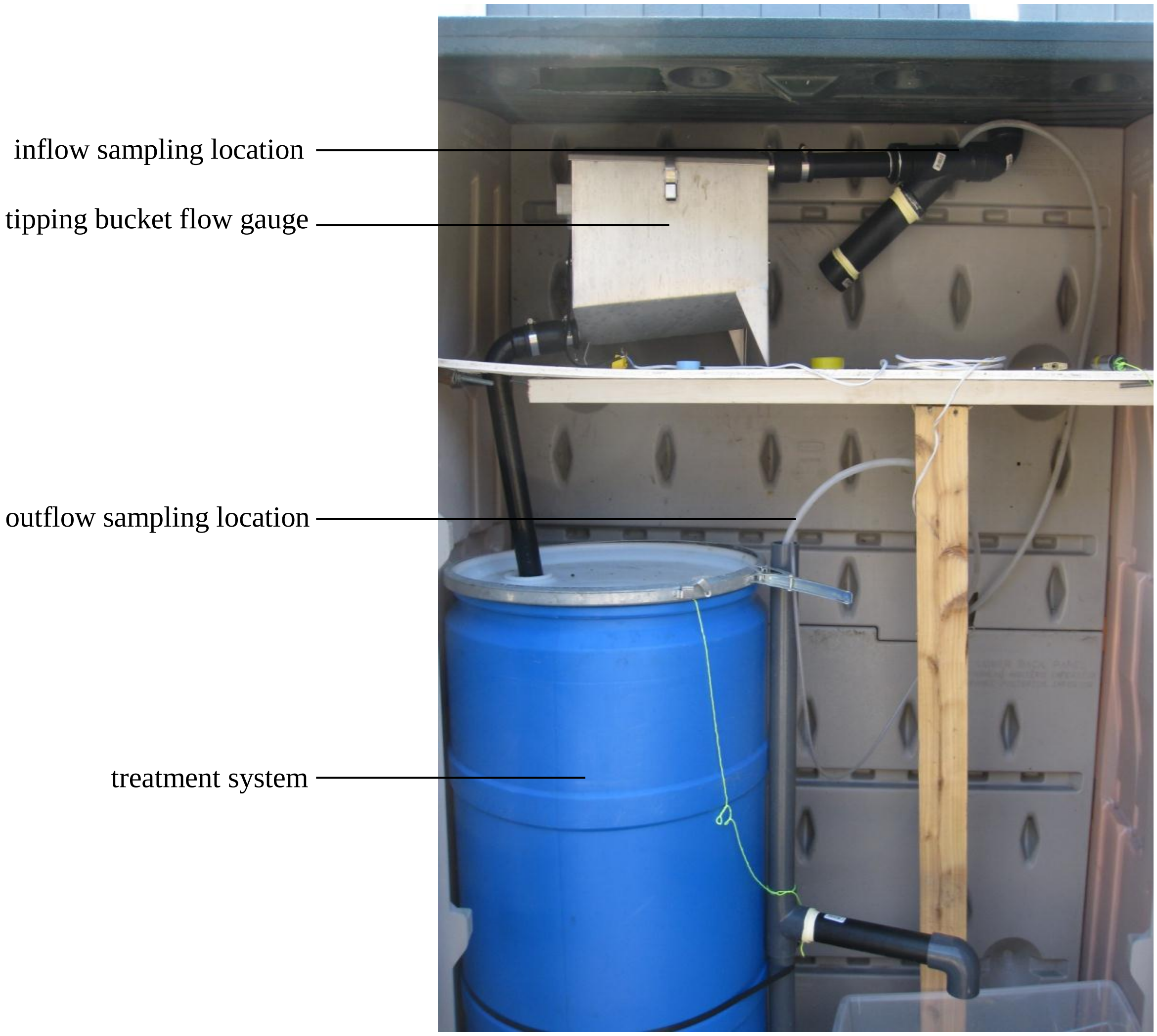


Figure 10: Tipping bucket flow meter and rain barrel containing treatment system.

2.4.2: Precipitation

Precipitation was recorded on the greenroof using a Hydrological Services tipping bucket rain gauge in 0.2 mm increments (Figure 11). Due to potential shadowing effects by the surrounding second story of the Archetype House, the rain gauge was positioned in a way to best represent the precipitation reaching the green roof surface. Data from the rain gauge was recorded using an Onset Computer Data Logger at 5 minute intervals and downloaded with HOBOware Pro® version 2.7.3.1 software.

Total rainfall volume was calculated using the following equation:

$$V_p = R \times D \qquad (1)$$

where: V_p = Volume of precipitation falling on the greenroof (L)

R = Rainfall depth (mm)

D = Drainage area (m^2)

The percent runoff retention of the greenroof relative to precipitation ($\%R_p$) was calculated as:

$$\%R_p = \frac{V_p - V_r}{V_p} \times 100 \qquad (2)$$

Where: V_p = Volume of precipitation falling on the greenroof (L)

V_r = Volume of runoff measured from the greenroof (L)

Figure 11: Tipping bucket rain gauge located on the greenroof.

Composite precipitation samples were collected in an acid washed triple rinsed 10 L carboy bottle connected to a rain gauge at the meteorological station located on site (Figure 12).

Figure 12: Meteorological station containing rain gauge located 450 m from greenroof.

2.4.3: Sample Collection

Two ISCO 6712® automatic samplers were used to collect runoff at the inlet and outlet of the treatment cartridge at specified intervals during each rainfall event. The samplers were housed inside the garage which provided security and a constant power source. Pre-treatment runoff was collected from a pipe elbow that is approximately 350 mL in volume. Initiation of pre-treatment sample collection was triggered by the first tip of the flow meter. The pre-treatment sampler took twenty four 350 mL samples at regular time intervals throughout the storm event. The post-treatment sampler was triggered using an ISCO 730 Bubble Module® that measured when water level in the rain barrel reached the point of contact with the media. The module malfunctioned after the September 28th, 2009 event; for subsequent events a connection cable was used to trigger the post-treatment sampler after the pre-treatment sampler had collected its first sample. The post-treatment sampler collected twenty four 500 mL samples from the drainage pipe at the bottom of the cartridge structure. The post-treatment sampler was programmed to collect at the same time intervals as the pre-treatment sampler. Samples were collected on 15 minute intervals for the first storm for a total 6 hour sampling time. The sampling interval was shortened following this storm to provide a better representation of the hydrograph, most notably during the early stages of the runoff.

Stormwater sampling is ideally flow paced which allows for a larger percentage of samples during high flow, thus providing better representation of the pollutograph (Davis and McCuen, 2005). Based on the greenroof design and limited use of space in which to store the treatment system, a tipping bucket flow meter was selected to record flow measurements. However, to initiate sampling the tipping bucket flow meter needed to be configured to act as a rain gauge. Each tip of the flow gauge would send one pulse which would be recorded by the ISCO sampler, with the first pulse triggering the sampling program. These pulses were then back calculated to determine flow rates. Without the use of a weir, or area velocity flow logger, the sampling could not be flow paced. Uniform time pacing was selected for the sampling program. The uniform time pacing provided an overall representation of the hydrograph. Due to the greenroof's largely uniform release of pollutants, the loss of information gained with flow paced sampling was considered minimal. A summary of the sampling time intervals for each monitored storm is provided in Table 1.

Following each storm event, the samples were collected from the site. Data from the samplers was downloaded and the sampling program reset for the next storm event. The bottles were replaced for each event with acid washed triple rinsed bottles. From the 68 rain events monitored, 26 produced no runoff, 25 were monitored and 5 occurred before the treatment system was installed.

Table 1: Monitoring period sampling times.

Sample Date	Start Time		End Time		Sampling Interval
	Influent	Effluent	Influent	Effluent	
11-Jul-09	9:42	9:44	14:12	14:14	15 min
23-Jul-09	11:11	11:13	15:01	15:03	10 min
29-Jul-09	9:27	9:27	13:17	11:17	10 min
9-Aug-09	20:27	20:32	0:17	0:22	10 min
11-Aug-09	15:12	15:16	19:02	19:06	10 min
20-Aug-09	17:40	17:46	21:30	21:36	10 min
29-Aug-09	2:02	2:34	5:52	6:24	10 min
28-Sep-09	18:42	18:42	20:37	20:37	10 min
29-Sep-09	2:19	2:19	4:14	4:14	5 min
2-Oct-09	13:09	13:09	15:04	15:04	5min
9-Oct-09	2:25	2:25	6:15	6:15	10 min
23-Oct-09	19:13	19:17	0:43	0:47	10 min*
28-Oct-09	3:55	4:00	9:26	9:30	10 min*
7-May-10	21:24	21:26	1:13	1:15	10 min
13-May-10	19:45	19:47	23:35	23:37	10 min
2-Jun-10	19:38	19:40	23:28	23:30	10 min
3-Jun-10	13:50	13:52	17:40	17:42	10 min
12-Jun-10	10:25	10:27	14:15	14:17	10 min
16-Jun-10	9:25	9:27	13:15	13:17	10 min
22-Jun-10	14:14	14:16	18:04	18:04	10 min
24-Jun-10	4:11	4:13	8:01	8:03	10 min
26-Jun-10	11:26	11:28	15:16	15:18	10 min
24-Jul-10	15:07	15:09	18:57	18:59	10 min
15-Aug-10	8:47	8:49	0:37	0:39	10 min
22-Aug-10	3:56	3:58	7:46	7:48	10 min

* 10 min (1-12 samples) 20 min (13-24 samples)

2.5: Water Quality Analysis

Upon collection from the study site, water samples were returned to the University of Waterloo for analysis in the Sediment and Water Quality Lab of the Department of Geography and Environmental Management.

2.5.1: Conductivity and pH

Conductivity and temperature were measured with a regularly calibrated Orion 105A+ Conductivity Meter following Standard Method 2510B. The pH was measured using a calibrated Orion 250A pH meter (±0.02) following Standard Method 4500-H^+ (Eaton, 1995).

2.5.2: Suspended Solids and Total Dissolved Solids

Suspended solids concentrations were determined by filtering the samples through pre-weighed 0.45 µm glass microfiber filters. The filters were then dried at 100°C for 24 hours (Standard Method 2540 D) and weighed. The suspended solids concentrations (mg/L) were calculated with the following equation:

$$TSS = \frac{(F_i - F_d) \times 1000}{V} \tag{3}$$

where:

F_i = initial filter mass (g)

F_d = dried filter mass (g)

V = sample volume (L)

Total dissolved solids concentrations (mg/L) were calculated with the formula:

$$TDS = \left(\frac{C}{1 + \left(0.02 \times (T - 25)\right)}\right) \times 0.666 \tag{4}$$

where:

C is conductivity (µS/cm)

T (°C) is the sample temperature.

2.5.3: Phosphorus

A 20 mL aliquot of each water sample was filtered through a 0.45 µm filter into glass scintillation vials then stored at 4°C for subsequent analysis of SRP (Standard Method 4500 P A). Samples were preserved for TP analysis by adding 1 mL of 20% sulfuric acid (H_2SO_4) to 100 mL of sample. TP samples were digested using a potassium persulfate method prior to analysis. A Technicon Autoanalyzer II® and NAP analysis software were used to measure SRP and TP concentrations according to the stannous chloride ammonium molybdate colorimetric method (Environment Canada, 1979). The detection limit of this method is 1 µg/L. The colorimeter and analysis software were configured for a range of P concentrations from 0 – 400 µg/L. Samples with higher concentrations were diluted to a measurable range using de-ionized water.

2.5.4: Grain Size Analysis

In order to characterize the nature of particulate matter collected pre and post treatment, suspended solids were collected and evaluated with an image analysis system. For selected rainfall events, the grain size distribution of suspended solids were measured and compared. A 20 mL aliquot was pipetted from each sample bottle and filtered through a 0.45 µm Millapore HA nitrocellulose filter. Each bottle was gently inverted (turned end to end) to ensure complete resuspension of the solids. Solids on the filters were examined using a Zeiss Axiovert S100 microscope and Northern Eclipse Image software. The filters were rendered semi-transparent using low viscosity microscope immersion oil (Richard-Allen Scientific). A minimum of 2500 particles for each sample were measured for statistical significance. Images of representative particles were collected.

2.5.5: Water Quality Data Analysis

Summary statistics including number of samples, mean, min, max, and standard deviation (SD), are reported for all water quality parameters. Event mean concentrations (EMC), unit area loadings (UAL), volume weighted mean concentration (MC_{vw}) and extrapolated unit area loading (UAL_{ex}) were calculated for measured SRP, TP and TDS.

The EMC is a flow-weighted average of the pollutant concentration for a storm event and provides a more representative estimate of the pollutant concentration than averaging the concentrations of multiple discrete samples (Davis and McCuen, 2005). The equation used to calculate EMC is:

$$\mathrm{EMC} = \frac{\sum_{i=1}^{n}(C_i \times Q_i \times \Delta t)}{\sum_{i=1}^{n}(Q_i \times \Delta t)} \qquad (5)$$

where:

C_i= concentration of discrete sample

Q_i = discharge at the time of discrete sample

Δt = time interval between samples (Davis and McCuen, 2005).

The unit area load (mg/m^2) was calculated using the following equation:

$$\mathrm{UAL} = \frac{\sum_{i=1}^{n}(V_i \times \mathrm{EMC}_i)}{A} \qquad (6)$$

where:

V_i = Total volume of runoff measured for event *i* (L)

EMC_i = Event mean concentration of the pollutant for event *i* (mg/L)

A = Area of greenroof sampled (m^2)

Volume-weighted mean concentration (MC_{vw}) was calculated using:

$$MC_{vw} = \frac{\sum_{i=1}^{n} (V_i \times EMC_i)}{\sum_{i=1}^{n} (V_i)} \tag{7}$$

To account for storm events where water quality monitoring was not completed, an extrapolated load was calculated using the MC_{vw} and the total volume of runoff measured. The extrapolated load value is based on using the volume-weighted mean concentration as a best possible estimate for influent concentrations. Additionally, the extrapolated effluent load requires the assumption that all unmonitored effluent is treated with similar performance to monitored effluent. The extrapolated load was calculated using:

$$UAL_{ex} = \frac{MC_{vw} \times V_t}{A} \tag{8}$$

where:

V_t = Total volume of greenroof runoff measured during the monitoring season.

Treatment efficiency was calculated for both the UAL and UAL_{ex} with the following equation:

$$TE = \left(\frac{UAL_{pre} - UAL_{post}}{UAL_{pre}}\right) \times 100 \tag{9}$$

where:

UAL_{pre} = Unit area load (mg/m^2) of greenroof runoff

UAL_{post} = Unit area load (mg/m^2) of runoff after treatment (TRCA, 2006)

2.6: Sorption Experiments

A series of batch experiments were conducted on the two types of engineered media to determine the maximum sorptive capacity. Triplicate samples of the media were weighed individually into 50 mL centrifuge tubes. Each sample was coned and quartered in an attempt to minimize variation in particle size. Twenty five mL aliquots of 0.00, 0.05, 0.100, 0.250, 0.500, 1.00, 2.00, 3.00, 4.00, 5.00, 10.0, 25.0 and 50.0 mg P/L were added to each tube. An additional 100 mg P/L aliquot was tested for the 2010 media. Although this concentration is much higher than the range of reported greenroof runoff P concentrations (0.6-3.0 mg/L TP), a high concentration was needed in order to saturate the media and establish a sorption isotherm (Zhu et al., 1997). Ionic control was maintained by adding 0.25 mL of 1.0M KCl to each centrifuge tube. The centrifuge tubes were capped and shaken at approximately 50 rpm for 20 hours on a heavy-duty shaker (Eberbach 6000). Twenty hours was considered sufficient time to allow the solution to reach equilibrium, after which the samples were filtered through 0.45 µm filter (Del Bubba et al., 2003). Temperature remained constant at normal room temperature of 22ºC. P concentrations were measured using methods previously described. The mass of P sorbed per mass of media (q) was calculated using the following:

$$q = \frac{(C_0 - C_E) \times V}{M} \tag{10}$$

where:

q = mass of P sorbed per mass of media (mg/g)

C_0 = initial concentration of P in solution (mg/L)

C_E = concentration of P in solution after equilibrium (mg/L)

V = volume of P aliquot (L)

M = mass of media (g)

The Langmuir isotherm was used to estimate the maximum sorption capacity. The Langmuir isotherm assumes sorption has reached equilibrium as well as a monolayer of coverage on a uniform surface with no interaction between the adsorbed molecules (Islam et al., 2004). The Langmuir isotherm is as follows:

$$q = \frac{q_{max} \times b \times C_E}{1 + (b \times C_E)} \tag{11}$$

where:
q_{max} = the maximum adsorption capacity of the media (mg/g)
b = Langmuir equilibrium constant (unitless)

Following the sorption experiments, isotherm curves were generated to examine the sorptive capabilities of the media. The Freundlich isotherm is an empirical model which does not require assumptions on the nature of the sorption such as mono-layer coverage of the media. The Freundlich isotherm is as follows:

$$q = KCe^{1/n} \tag{12}$$

where:
K, n = Freundlich constants related to the strength of binding between the absorbant and absorbent material.
The n coefficient describes the adsorptive capacity of the material. Values close to 1 indicate a higher adsorptive capacity at high equilibrium concentrations. If values are greater than 1, the binding affinity decreases (Heal et al., 2004). The isotherm can be plotted linearly by log transforming the data and plotting logq vs log$C_{E.}$ The experimental data was fit to the Langmuir model using non-linear regression analysis with the software program OriginPro 8.0.

2.7: Quality Control

During chemical analysis at least 5% of the samples were run in duplicate to measure analytical precision. Duplicate analysis reported precision of within 5% for measured values. Reagent blanks were inserted after every three samples during analysis. Prior to analysis, five P standards were used to generate a standard calibration curve. Only if the software reported high correlation between the measured and expected values would analysis proceed. To test for P contamination, laboratory blanks were prepared from equipment used in the sampling and analysis. This included: 1L ISCO bottles, 25 mL glass bottles, and 20 mL filtering syringes. Many samples required dilution to bring P concentrations into a detectable range. Incomplete mixing of solutions can introduce dilution error during sample preparation. Samples subject to dilution were tested at different dilution factors to ensure between at least 10% agreement.

2.8: Statistical Analysis

Water quality and water quantity data analysis was completed with OriginPro 8.0. The normality of the data was determined using one sample Kolmogorov-Smirnov tests. The water quality and water quantity data were not normally distributed. To determine significant differences in water quality parameters between influent and effluent samples the Mann-Whitney U test was used. The Mann-Whitney U test compares two independent samples with non-normal distributions. Due to the inherent time delay in the sampling procedure the samples are considered independent, making the Mann-Whitney test more appropriate than a paired t-test. Linear regression was used to compare the treatment efficiency of the engineered media over the monitoring period. Bivariate correlation analysis was used to determine significant relationships between water quantity parameters such as storm size, antecedent dry period and lagtime. Spearman rank-order (r_s) correlation was used as a non-parametric alternative to Pearson product-moment (r) correlation. Boxplots are presented for water quality parameters SRP, TP, Conductivity, TDS and SS. The boxplots were created using SPSS Statistics 17.0 software. The software defined outlier values (°) as between 1.5 interquartile ranges and 3 interquartile ranges of the data. Extreme values (*) were defined as more than 3 interquartile ranges of the data.

Chapter 3: Results

3.1: Introduction

The following chapter addresses limitations of the study and presents sections on the greenroof hydrological response and water quality data for the field study. Precipitation data collected on the greenroof is compared to data collected at the Buttonville Airport weather station and to long term average values compiled by Environment Canada. Following the field study sections, lab data for the sorption experiments is reported.

3.2: Precipitation Data

Monthly total precipitation values for the study period are compared to a 30 year climate average in Table 2. Monthly precipitation data was collected from the Buttonville Airport, located approximately 18 km away from the study site. Long-term averages are available from the AES station located at Pearson International Airport. The Buttonville airport precipitation data demonstrate that 2009 had much higher rainfall amounts than the historical norms. During 2009 July, August and October received higher than average precipitation amounts, while September was a dryer month than average. During the 2010 monitoring season May was the only month with lower than average precipitation levels. July and August had slightly higher amounts, while June received three times the average rainfall. With the exception of June, the 2010 monitoring period had lower monthly rainfall amounts than in 2009. Excluding August 2009 and July 2010, the greenroof rain gauge recorded less rainfall than the Buttonville airport. The lower total rainfall volumes can be attributed to the shadowing effect caused by the Archetype House that partially blocked the eastern side of the greenroof.

Table 2: Comparison of total monthly precipitation measured during the monitoring period.

Precipitation (mm)					
Month	Pearson Airport Climate Normals 1971-2000	Buttonville Airport 2009	Archetype House Greenroof 2009	Buttonville Airport 2010	Archetype House Greenroof 2010
May	72.5			63.2	25
Jun	74.2	70.9	46.6	228.6	212.2
Jul	74.4	110.2	56.2	76.6	83.4
Aug	79.6	107.6	149.8	91	79
Sep	77.5	47	39.2		
Oct	64.1	78.2	55.2		
Total	**442.3**	**413.9**	**347**	**459.4**	**399.6**

Source: Environment Canada, 2010

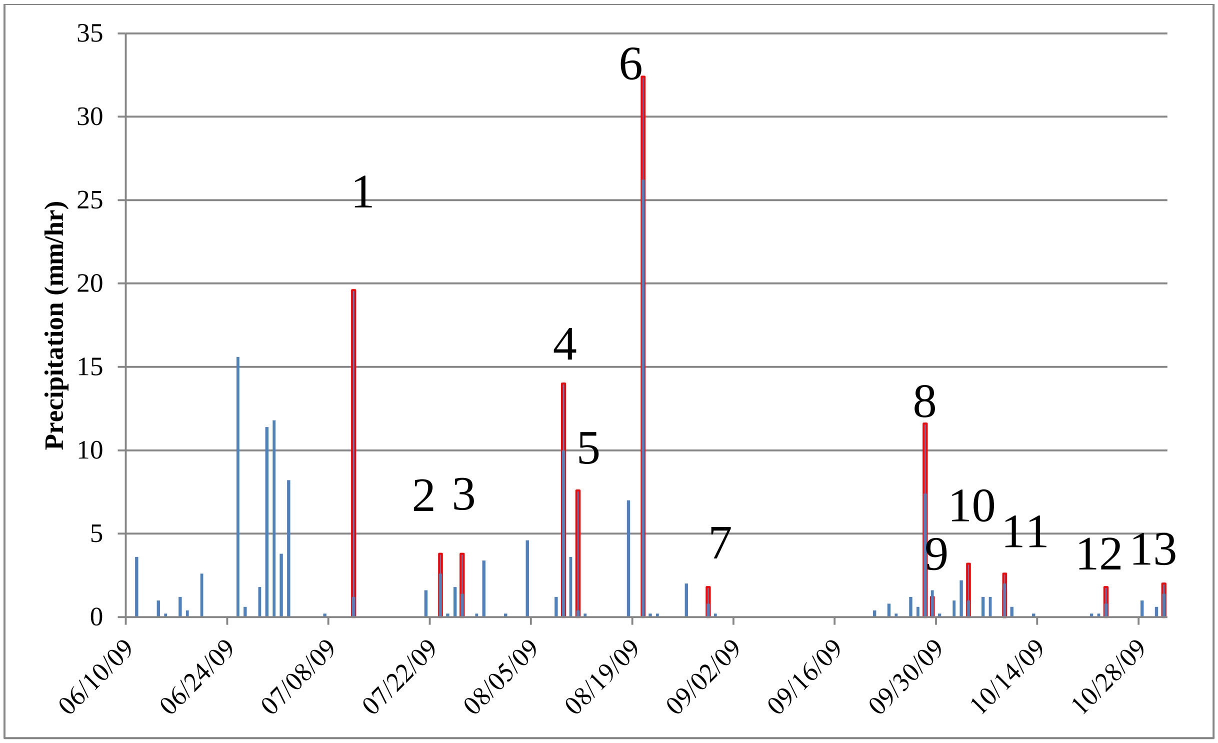


Figure 13: Temporal distribution of precipitation events on the greenroof during 2009. The numbers 1 through 13 indicate storm events sampled for water quality analysis.

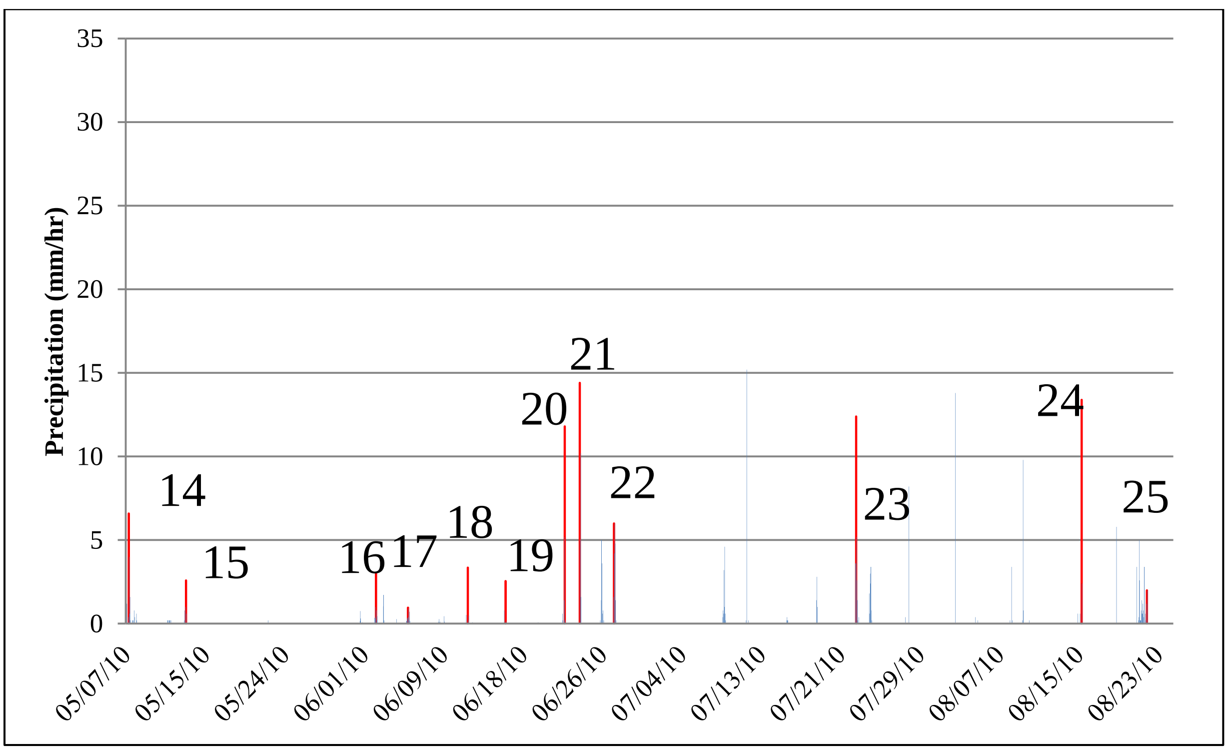


Figure 14: Temporal distribution of precipitation events on the greenroof during 2010 monitoring season. The numbers 14 through 25 indicate storm events sampled for water quality analysis.

3.3: Characteristics of Storm Events

Over the entire monitoring period, the greenroof received rainfall events that ranged from 1.0 to 83.2 mm. The 83.2 mm event occurred on August 20, 2009. Due to the intensity of the storm this measurement is expected to be an overestimation. It is probable that the rain gauge on the greenroof recorded water running off the surrounding roof of the Archetype House. The rain gauge at the meteorological station recorded a rainfall volume of 56.8 mm which is a more realistic volume, and therefore this value was used to calculate the storm intensity and return period for this event. However, for the calculation of the greenroof stormwater retention, the use of 83.2 mm is maintained. Twenty five storm events were sampled throughout 2009 and 2010. Based on storm intensity and storm duration, return periods were calculated from an Intensity Duration Frequency curve supplied by Environment Canada (Table 3). The August 20, 2009 event had the highest storm intensity of 26.22 mm/hr and a return period of approximately 35 years. The July 11, 2009 event had a return period of between 2 and 5 years. The June 24, 2010 event had a 2 year return period, and the remaining events had return periods of < 2 years. It is important to have control practices that are effective at treating small storms due to their frequency and critical contribution to water quality problems (Pitt and Clark, 2008). Summaries of hydrological parameters for each rainfall event including total rainfall (mm), total runoff outflow (L), mean flow (L/min), peak flow (L/min), rainfall intensity (mm/hr), previous dry hours (hrs) and rainfall duration (min) are found in Appendix B.

The greenroof runoff volumes ranged from 2 to 1028 L per storm event and a total of 6140 L was generated over the 2009 and 2010 monitoring periods. Peak runoff flow rates varied between 0.1 and 16 L/min and the mean runoff flow rates between 0.02 to 4.4 L/min. The highest peak flows and highest mean flow was observed during the August 20, 2009 event. Previous dry hours spanned from 3 hours to 605 hours. Several days were considered to have multiple individual storm events resulting in lower previous dry hour values (August 4, 2009, August 9, 2009, September 28, 2009, September 29, 2009 and June 9, 2010).

Table 3: Rainfall amount, intensity and return period of water quality monitored events.

Date	Rainfall (mm)	Intensity (mm/hr)	Return Period (yrs)
11-Jul-09	20.8	22.7	2--5
23-Jul-09	14.2	1.16	<2
29-Jul-09	5.4	2.59	<2
09-Aug-09	17.6	9.60	<2
11-Aug-09	8.4	4.38	<2
20-Aug-09	56.8	26.2	~35
29-Aug-09	4.2	0.55	<2
28-Sep-09	6.2	2.76	<2
29-Sep-09	3.8	1.09	<2
02-Oct-09	1.6	0.16	<2
09-Oct-09	23.2	1.02	<2
23-Oct-09	5	0.32	<2
28-Oct-09	3.8	0.45	<2
07-May-10	18.4	0.84	<2
13-May-10	5.2	0.80	<2
02-Jun-10	33	3.44	<2
03-Jun-10	14.8	8.46	<2
12-Jun-10	23.6	3.45	<2
16-Jun-10	17.8	14.2	<2
22-Jun-10	19.4	2.74	<2
24-Jun-10	31.2	8.14	2
26-Jun-10	11.8	1.46	<2
24-Jul-10	12.2	2.19	<2
15-Aug-10	17	1.62	<2
22-Aug-10	27.8	1.01	<2

3.4: Water Quantity

3.4.1: Runoff Retention Rates

The percent runoff retention relative to precipitation ($\%R_p$) data for all monitored events is presented in Figures 15 and 16. A complete summary of runoff retention data is found in Appendix B Table 1. During the 2009 monitoring period, the greenroof retained approximately 41.5% (144 mm of 347 mm) of the precipitation. For the 2010 monitoring period the precipitation retained by the greenroof was 53.3% (213 mm of 400 mm). Retention rates ranged from 0 – 100%. A total of 27 storm event (<15.6 mm) produced no runoff. The largest storm event that was completely retained was 15.6 mm on July 11th, 2010. The largest overall storage of runoff occurred on June 25th, 2009 with the roof retaining approximately 437 L of runoff (11.8 mm of 22.8 mm precipitation). During the period with the smaller drainage area, the largest overall storage on June 2, 2010 was approximately 393 L (28.3 mm of 33 mm of precipitation).

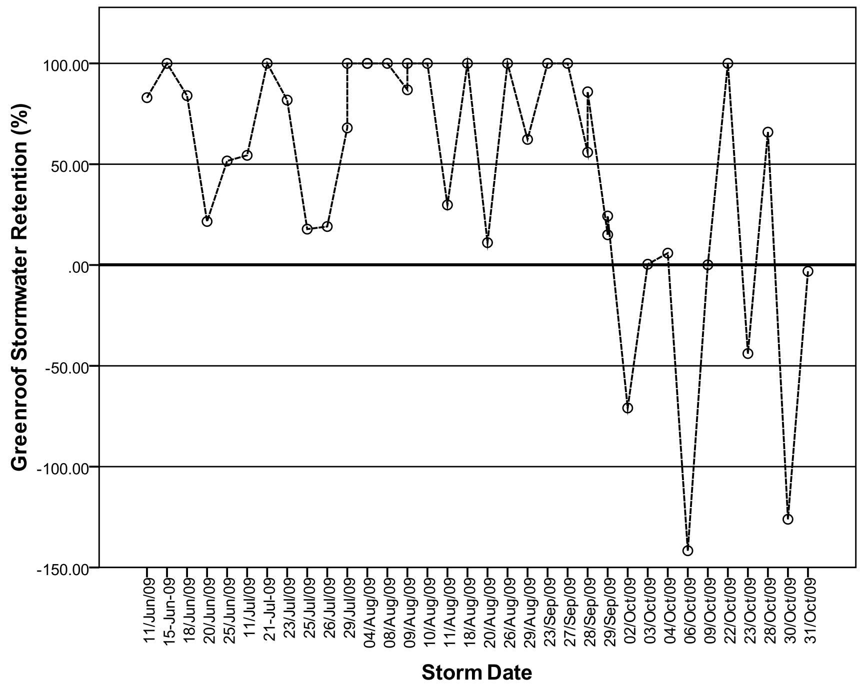


Figure 15: Greenroof stormwater retention (%) relative to precipitation – 2009 period.

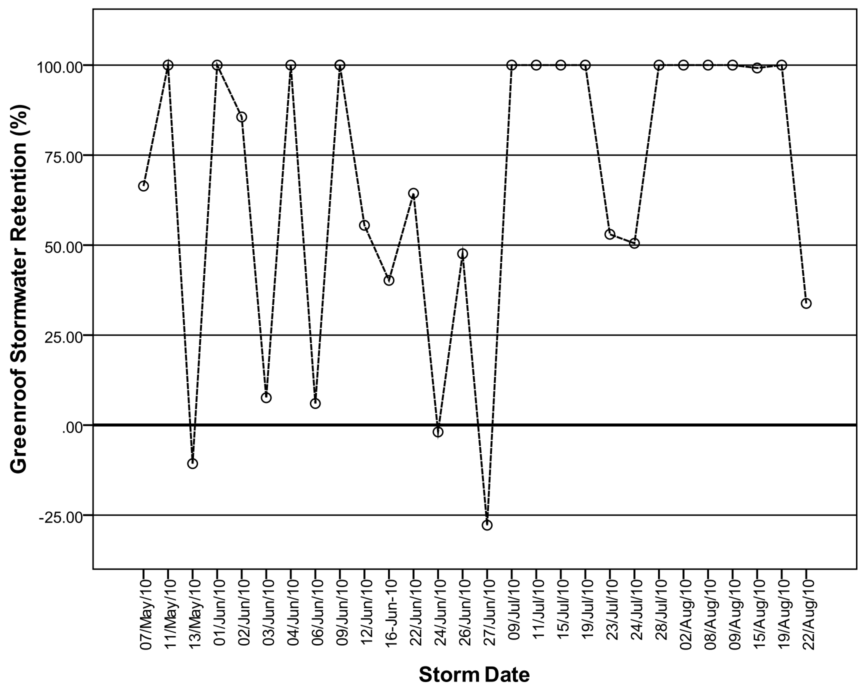


Figure 16: Greenroof stormwater retention (%) relative to precipitation – 2010 period.

The monthly retention percentages shown in Figure 17 demonstrate that retention rates were high during the summer months, before decreasing to a net negative value for October 2009. Excluding the retention rates for October, the average retention is 71% for the summer months. Negative values denote a larger volume of runoff outflow than precipitation inflow. August 2009 and June 2010 had lower retention percentages than all other months, excluding October 2009. During these months storm events were more numerous than other months with 11 and 13 events for August 2009 and June 2010 respectively. The increased frequency of storm events resulted in larger volumes of water falling on the greenroof during these months, and smaller antecedent dry periods.

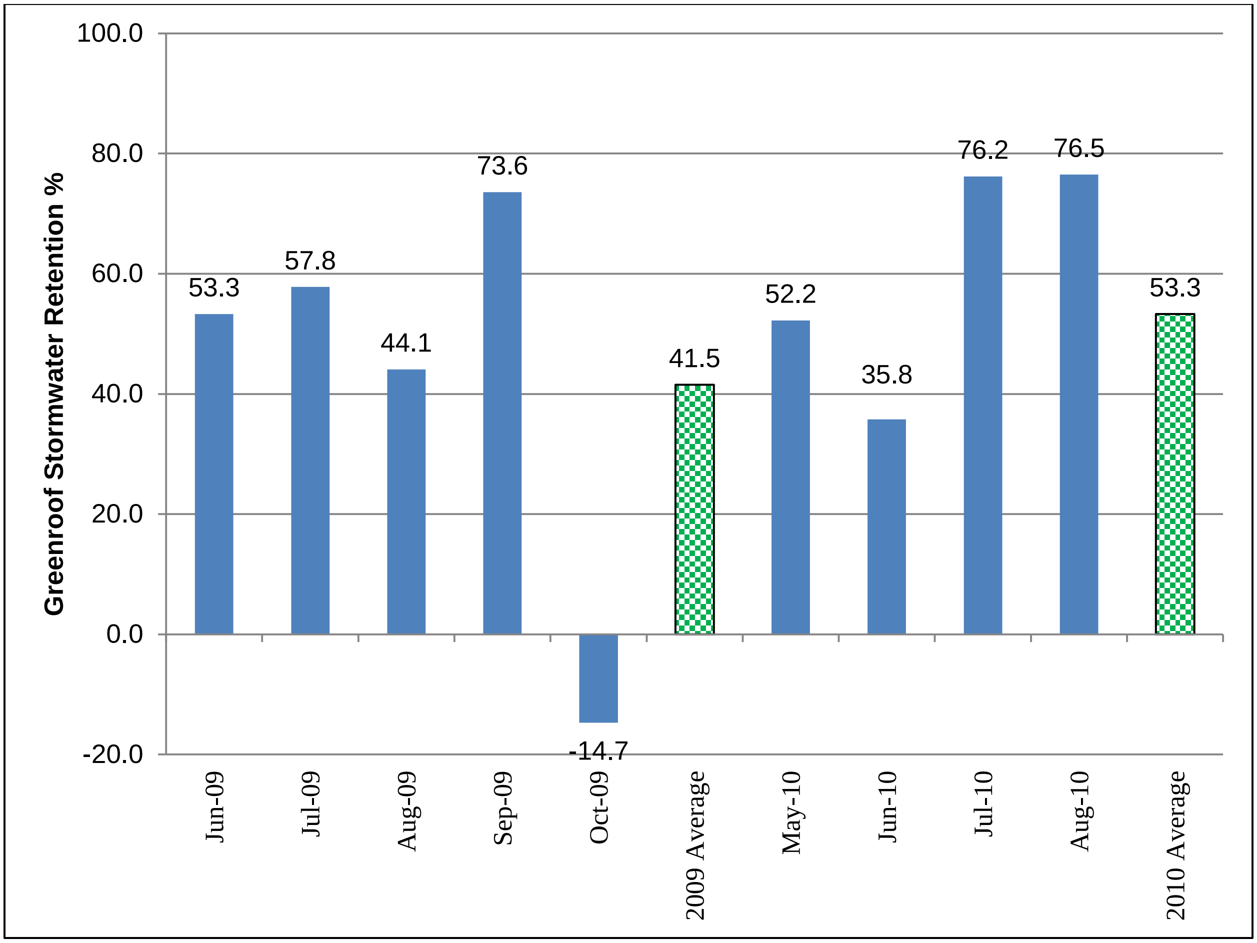


Figure 17: Greenroof stormwater retention (%) for each month of monitoring period.

Table 4: Summary of monthly runoff retention (mm).

Storm Event	Total Rain (mm)	Greenroof Runoff (mm)	Greenroof Retention (mm)	% Runoff retention (relative to precipitation)
June 2009	46.6	21.8	24.8	53.3
July 2009	56.2	23.7	32.5	57.8
August 2009	150	83.7	66.1	44.1
September 2009	39.2	10.4	28.8	73.6
October 2009	55.2	63.3	0	-14.7
Total 2009	**347**	**203**	**144**	**41.5**
May 2010	25	11.9	13.1	52.2
June 2010	212	136	75.9	35.8
July 2010	83.4	19.9	63.5	76.2
August 2010	79	18.6	60.4	76.5
Total 2010	**400**	**187**	**213**	**53.3**

3.4.2: Rainfall Size Influence on Retention

The retention values vary considerably with differing storm sizes and are strongly affected by factors such as antecedent dry periods, storm intensity and duration. The mean percent retention of storms (<5 mm) was 54.0%, with a mean absolute retention of 1.7 mm. Storms >5 mm had a mean percent retention of 57.3% with a mean absolute retention of 7.1 mm. The scatterplot in Figure 18 displays the relationship between rain depth and the percent retention by the roof. The data suggest a slight negative trend as the smaller events are clustered towards the higher percentages. There is considerable spread within the data, indicating that the relationship between rain depth and runoff retention is complicated by other factors. Figure 19 demonstrates a strong positive relationship between the absolute retention and the size of the rainfall event.

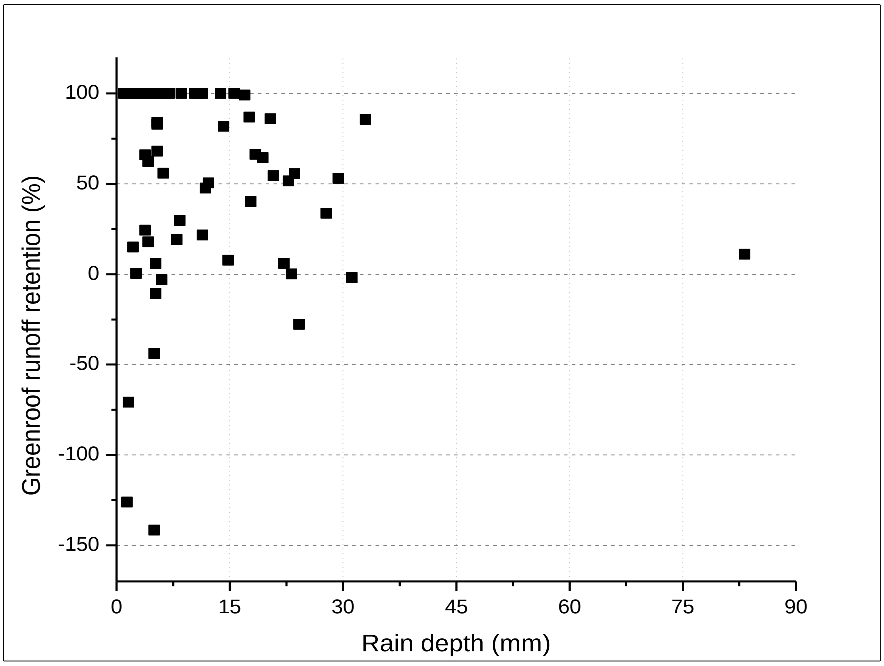


Figure 18: Influence of rainfall size on greenroof stormwater retention %.

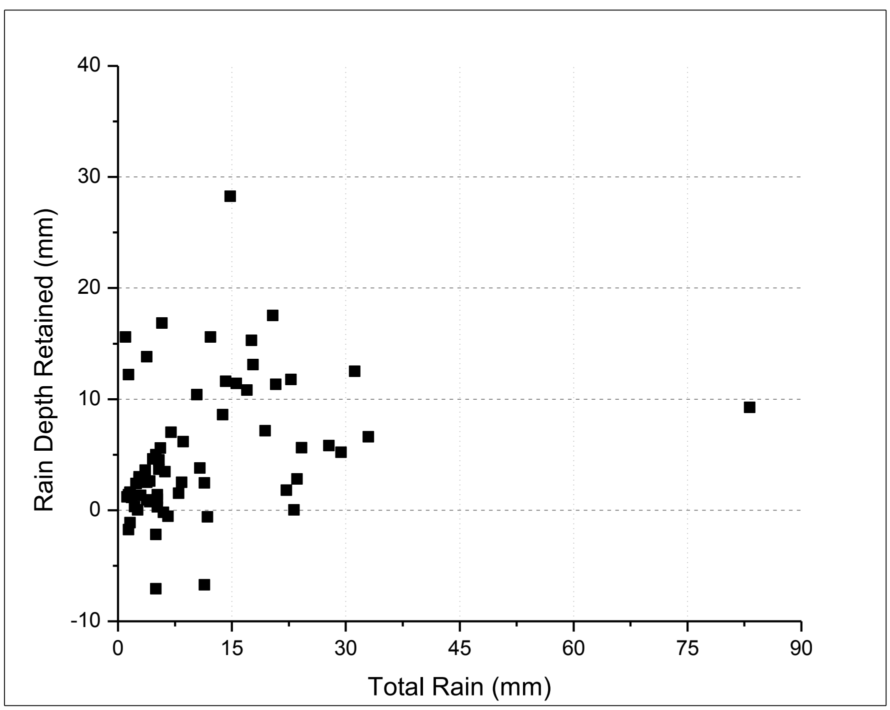


Figure 19: Influence of rainfall size on greenroof stormwater retention (mm).

3.4.3: Influence of Antecedent Dry Period on Retention

The antecedent dry period or previous dry hours (PDH) were measured for each storm event (Appendix B Table 2). This parameter was calculated as the time between the last rain tipping bucket measurement and the start of the storm event. The events were grouped into periods of less than 48 PDH and greater than 48 PDH. For the entire monitoring period, events with less than or equal to 48 PDH had a mean retention of 34.0%. Events with greater than 48 PDH had a mean retention of 77.7%, a 43.7% increase. Excluding the month of October, the retention performance increase for both categories. Events with less than 48 PDH have a mean retention of 52.0%, and events with equal or greater have 84.2%.

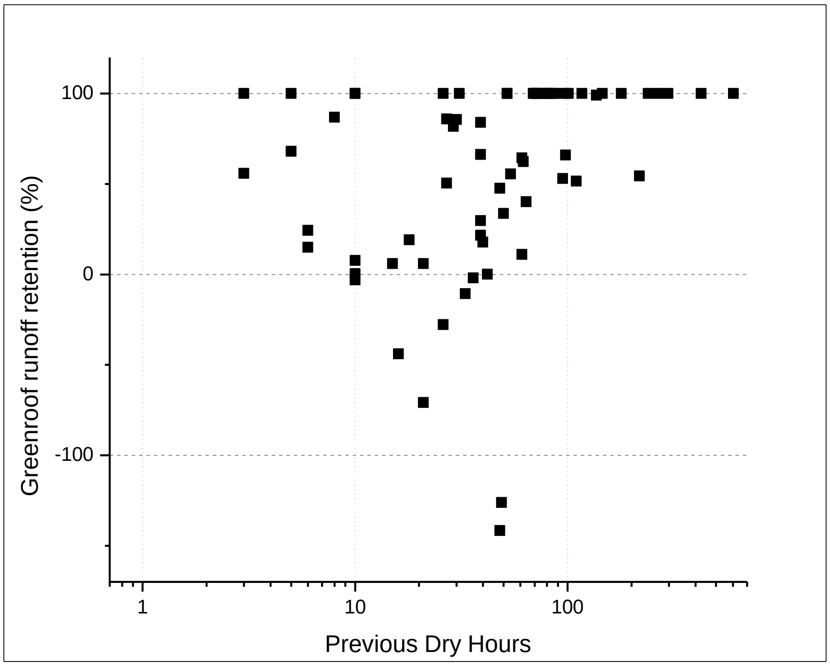


Figure 20: Influence of antecedent dry period on greenroof stormwater retention %.

3.4.4: Lag Time Response

The lag time is defined as the time between the peak rainfall intensity and the peak runoff flow. No control roof was monitored as part of this study. Accordingly, the results are reported without an on-site comparison. The monthly average lag times are presented in Table 5. The lag time for each storm event is presented in Appendix B, Table 2.

The overall average lag time for all events monitored was 78.1 minutes. The lowest monthly average lag time occurred in June 2010. The high volume of total rain likely saturated the greenroof for the duration of the month, resulting in shortened runoff lag times for the individual events. Higher average lag times were measured in October 2009 and August 2010. While the greenroof experienced lower evapotranspiration and overall stormwater retention during October, the low intensity rainfall events could have resulted in a longer delay to peak runoff. Two events in August 2010 produced runoff; one which demonstrated no lag time and one which had a very long delay to peak runoff (Figure 21). The data becomes biased by storms which had very long lag times, such as those found on October 28, 2009 and August 22, 2010 (Appendix B, Table 2). The median value of 35.0 minutes is a better representation of lag time performance. The low number of runoff producing events throughout the monitoring period, make it difficult to discern a lag time trend.

Table 5: Average monthly runoff lag times.

Period	**Total Rain (mm)**	**Average runoff lag time (min)**
2009		
June	46.6	40
July	56.2	48.6
August	150	48.5
September	39.2	37.5
October	55.2	167
2010		
May	25	31
June	212	10.2
July	83.4	89
August	79	354.5
Monitoring Period	**746.6**	**91.8**

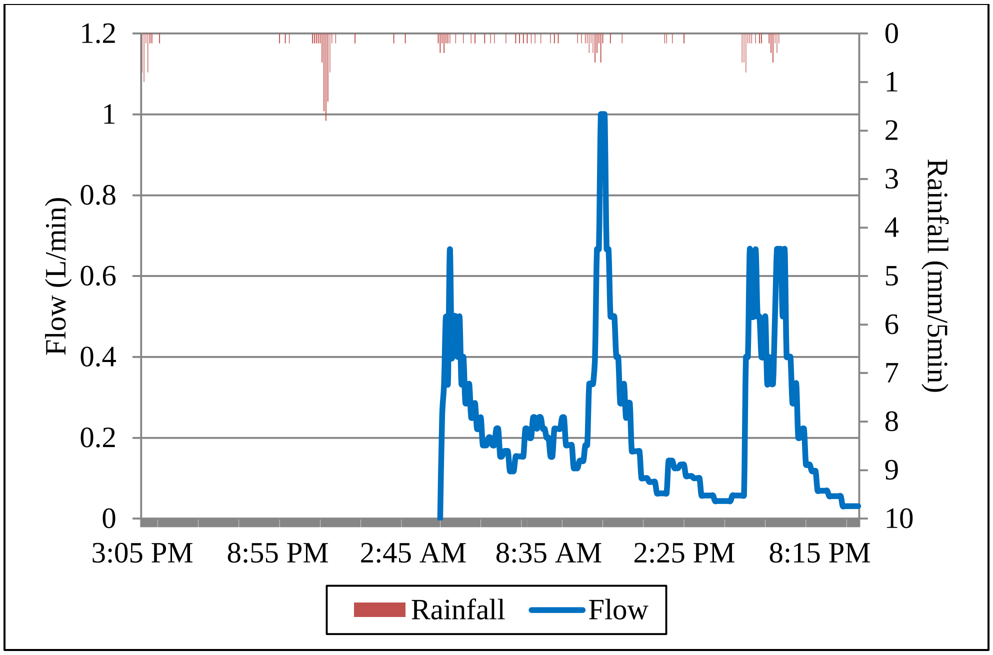


Figure 21: Greenroof lag time response of August 22, 2010 event.

There is a large range of lag time responses when compared to rainfall intensity (mm/hr) (Figure 22). Three events on October 9, 2009, October 28, 2009 and August 22, 2010 have much higher lag times of 431 minutes, 482 minutes and 709 minutes, respectively. The rainfall durations for these events were higher than the average storm duration of 385 minutes at 1360 minutes, 510 minutes and 1645 minutes, respectively. Longer, low intensity storms produced an in increased lag time response. When the storm events are grouped into categories of intensity, the lag time response becomes clearer. During low intensity storms (0-2.0 mm/hr) the greenroof had a mean lag time of 119.0 minutes. During moderate intensity storms (2.1-5.0 mm/hr) mean lag time decreased to 31.3 minutes. At high intensity (>5.0 mm/hr), the mean lag time was lowest at 23.3 minutes.

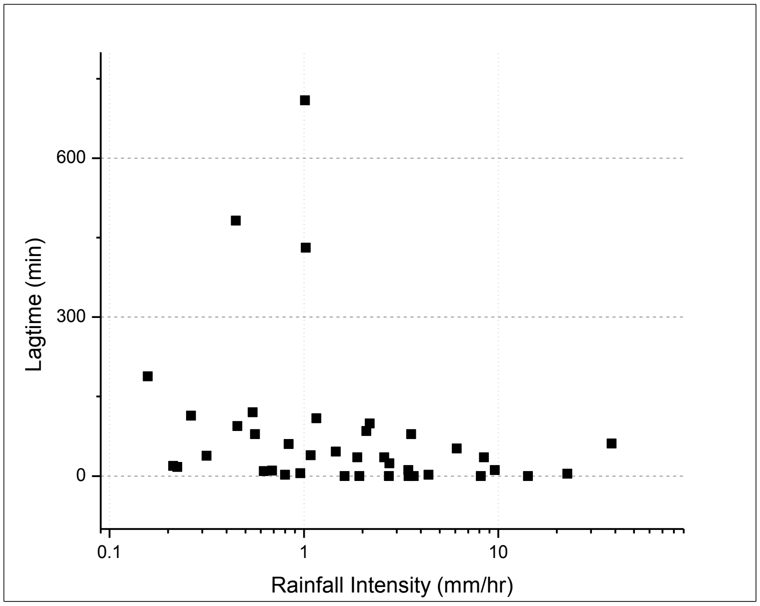


Figure 22: Greenroof lag time response to rainfall intensity.

3.5: Water Quality

Water quality results for 25 monitored storms from June 11, 2009 to October 28, 2009 and May 7, 2010 to August 22, 2010 are presented in this chapter. Data and summary statistics for individual events are presented in Appendix A.

3.5.1: Precipitation

During the monitoring period precipitation samples were collected for water quality analysis. Due to the logistical constraints related to visiting the site immediately after each rainfall event, it was difficult to sample each precipitation event discretely. Therefore some were composite of multiple events. The P concentrations in the precipitation are much lower than the concentrations observed in the greenroof runoff and constitute only a small fraction of the greenroof P export. SRP concentrations ranged from 0.002 – 0.065 mg/L and TP concentrations ranged from 0.004 – 0.103 mg/L. The mean pH was 5.26 and mean conductivity was 162 µS/cm.

3.5.2: Soluble Reactive Phosphorus

During the 2009 monitoring season the SRP concentrations in runoff from the greenroof ranged from 0.244 to 1.43 mg/L (Figure 23). The volume-weighted mean concentration was 0.769 mg/L. The total unit area loading was 118 mg/m^2 with a mean UAL of 9.08 mg/m^2 for each storm. The UAL_{ex} was calculated as 196 mg/m^2 based on total runoff volume of 3546 L. After treatment, the SRP concentrations ranged from 0.005 to 0.816 mg/L and the volume-weighted mean concentration was 0.523 mg/L. The treatment reduced the UAL to 80.2 mg/m^2 with a mean UAL of 6.17 mg/m^2 for each storm. The UAL_{ex} was calculated as 133.3 mg/m^2 based on similar treatment efficiency for the unmonitored volumes. Overall, the SRP TE_{2009} was equal to 32.0%.

During the 2010 monitoring season the SRP concentrations in the greenroof runoff ranged from 0.211 to 6.16 mg/L (Figure 23). Several events during the 2010 season contained outlier and extreme concentrations much higher than the 2009 events. There was a decrease in influent SRP concentrations over the two monitoring seasons with median concentrations of 0.786 mg/L and 0.628 mg/L in 2009 and 2010 respectively. For 2010, the volume-weighted mean concentration was 0.630 mg/L. The total UAL was 76.1 mg/m^2 with a mean UAL of 6.92 mg/m^2 for each storm. The UAL_{ex} was calculated as 117 mg/m^2 based on total runoff volume of 2592 L. Following treatment by the media, the SRP concentrations ranged from 0.013 to 0.318 mg/L and the volume-weighted mean concentration was 0.110 mg/L. The UAL was reduced to 13.4 mg/m^2 and a mean of 1.21 mg/m^2 for each storm. The UAL_{ex} was calculated as 20.1 mg/m^2. SRP TE_{2010} was equal to 82.4%.

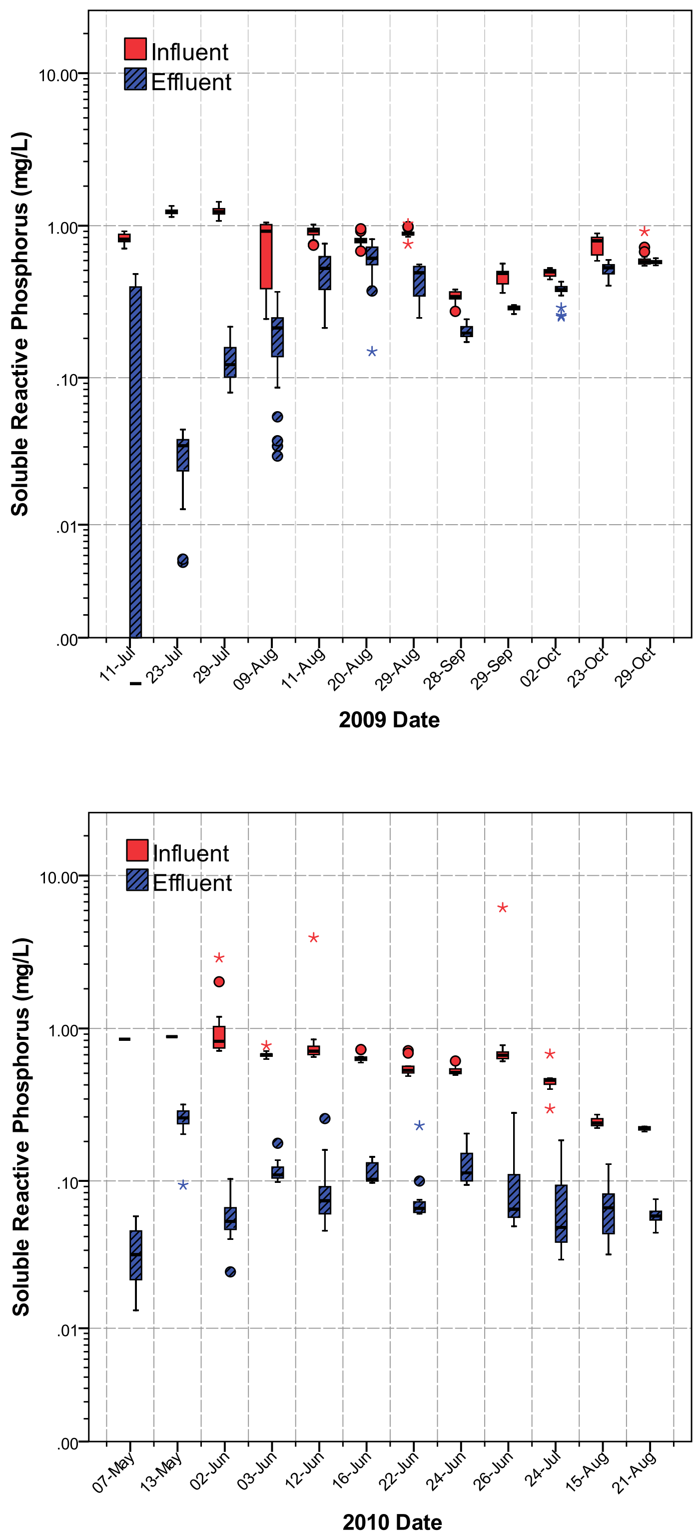


Figure 23: Distribution of SRP concentrations for influent and effluent samples (*=extreme value; °=outlier).

3.5.3: Total Phosphorus

In 2009, influent TP concentrations ranged from 0.511 to 2.89 mg/L (Figure 24). The volume-weighted mean concentration was 1.30 mg/L. The total unit area loading was 189 mg/m^2 with a mean UAL of 17.2 mg/m^2 for each storm. The UAL_{ex} was calculated as 333 mg/m^2. After treatment, the TP concentrations ranged from 0.021 to 1.90 mg/L and the volume-weighted mean concentration was 0.973 mg/L. The treatment reduced the UAL to 141 mg/m^2 with and a mean UAL of 12.8 mg/m^2 for each storm. The calculated UAL_{ex} was 248 mg/m^2 and TP TE_{2009} was 25.4%.

For the 2010 monitoring season the influent TP concentrations ranged from 0.229 to 10.2 mg/L. The volume-weighted mean concentration was 1.18 mg/L. The total UAL was 142 mg/m^2 with a mean UAL of 13.0 mg/m^2 per storm. The UAL_{ex} was calculated as 220 mg/m^2. The effluent concentrations ranged from 0.048 to 0.373 mg/L with a volume-weighted mean concentration of 0.158 mg/L. The total UAL was 19.1 mg/m^2 with a mean 1.73 mg/m^2 per storm. The calculated UAL_{ex} was 29.4 mg/m^2 and TP TE_{2010} was 86.6%.

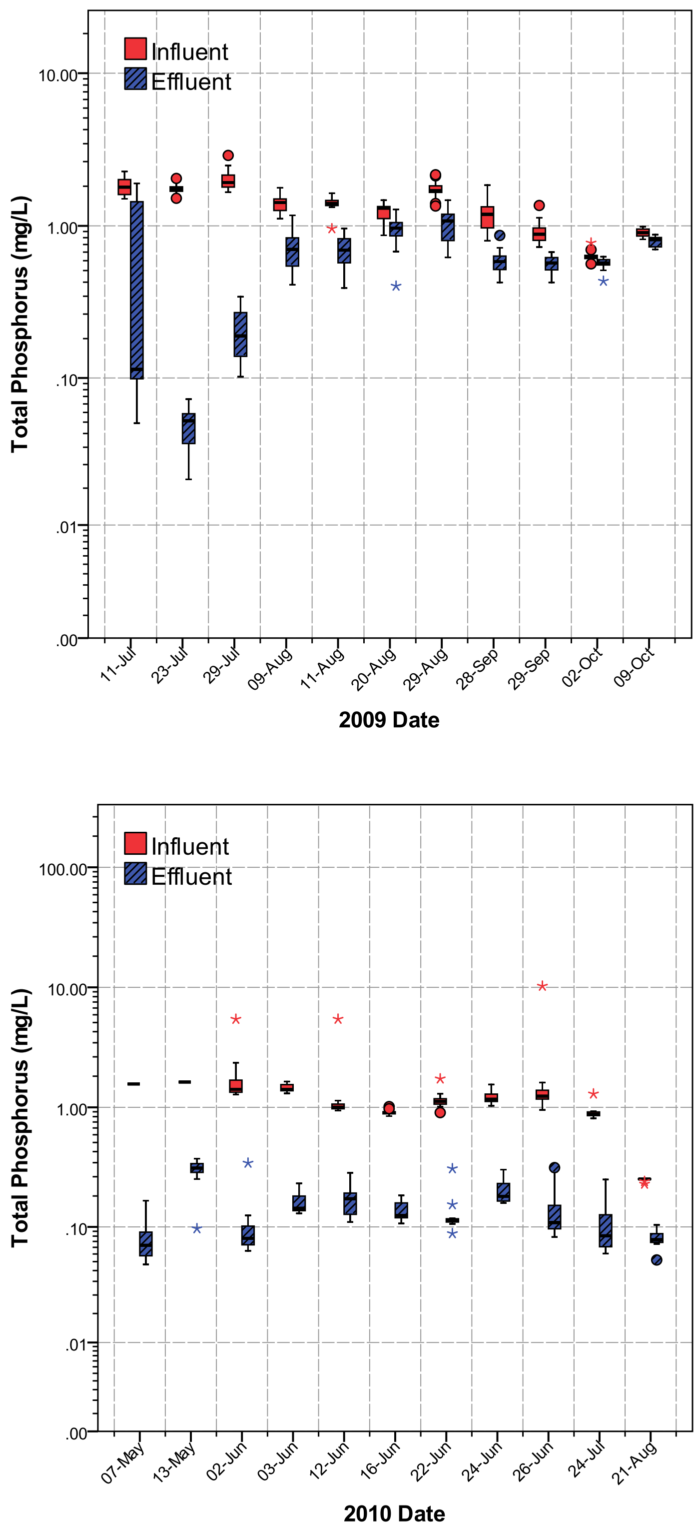


Figure 24: Distribution of TP concentrations for influent and effluent samples (*=extreme value; °=outlier).

3.5.4: Temporal Variation in Phosphorus Concentrations

P concentrations varied during storm events for both the influent and effluent samples. Generally the P concentrations remained consistent during the event. Compared to SRP concentrations through the hydrograph, TP concentrations were often more variable between samples. For some events the concentrations slightly increased through the sampling program (Figure 25), while others demonstrated a large decrease in concentrations over the sampling time, indicating the possibility of the first flush phenomenon (Figure 26).

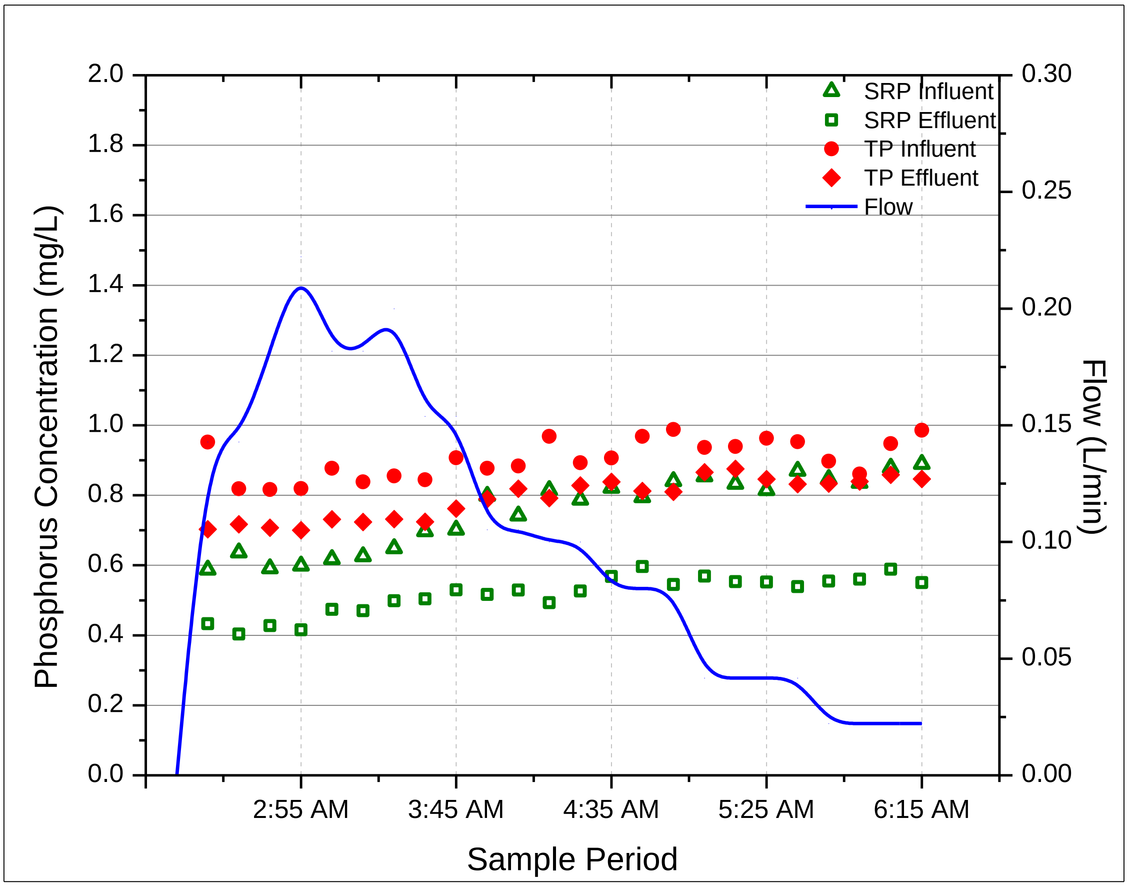


Figure 25: Temporal variability of P concentrations for influent and effluent on October 9, 2009.

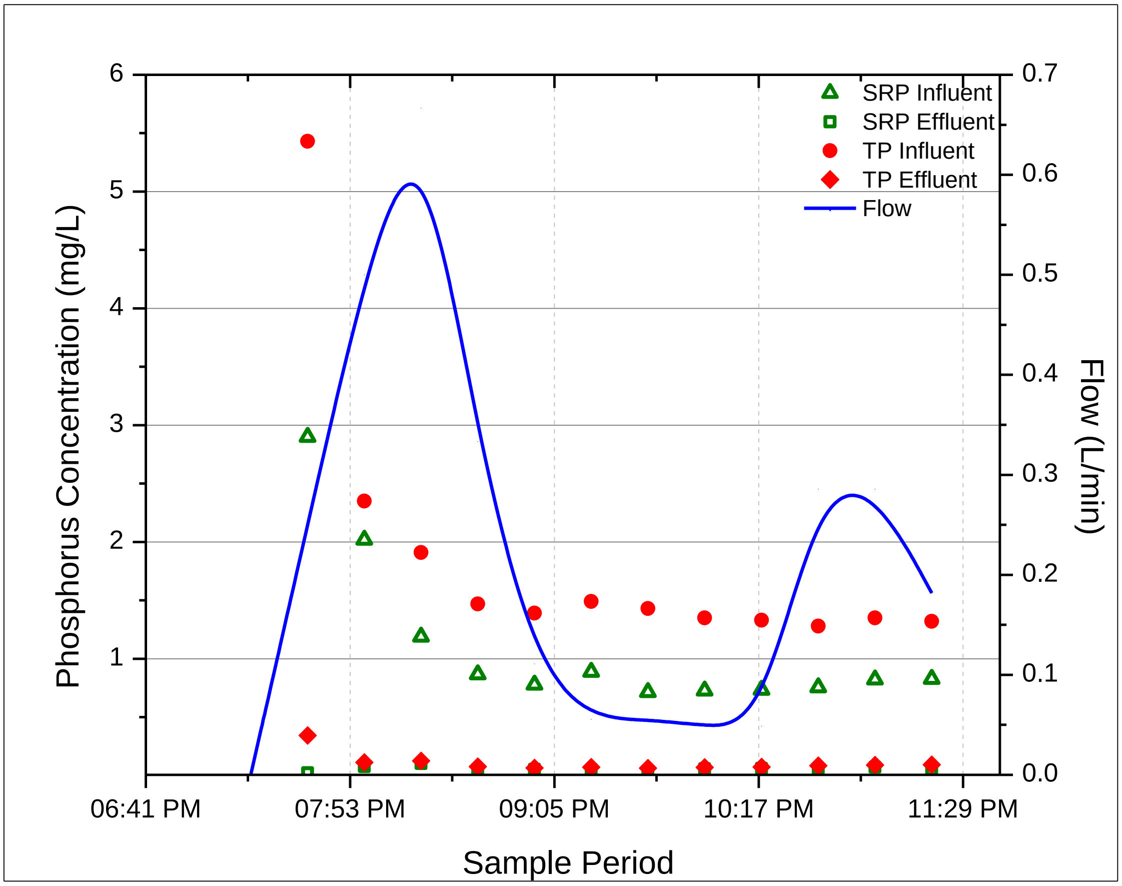


Figure 26: Temporal variability of P concentrations for influent and effluent on June 2, 2010.

Over the two year monitoring period, SRP concentrations decreased in the greenroof runoff (Figure 27). The highest SRP concentration of 1.44 mg/L occurred on the July 29, 2009 event. The lowest SRP concentration of 0.19 mg/L was measured in runoff from an event on September 28, 2010. During 2009, the SRP concentration decreased to a low of 0.27 mg/L, before increasing towards the end of the monitoring season. The start of the 2010 monitoring season had concentrations comparable to the end of the 2009 season. In 2010, influent concentrations consistently decreased. The final influent samples may indicate that the greenroof had leached the majority of the P from the growth substrate. In Figure 27, four extreme values (2.9 mg/L, 2.0 mg/L, 3.9 mg/L and 6.1 mg/L) were removed from events

16, 18, and 22 to allow for a better visual comparison of influent concentrations.

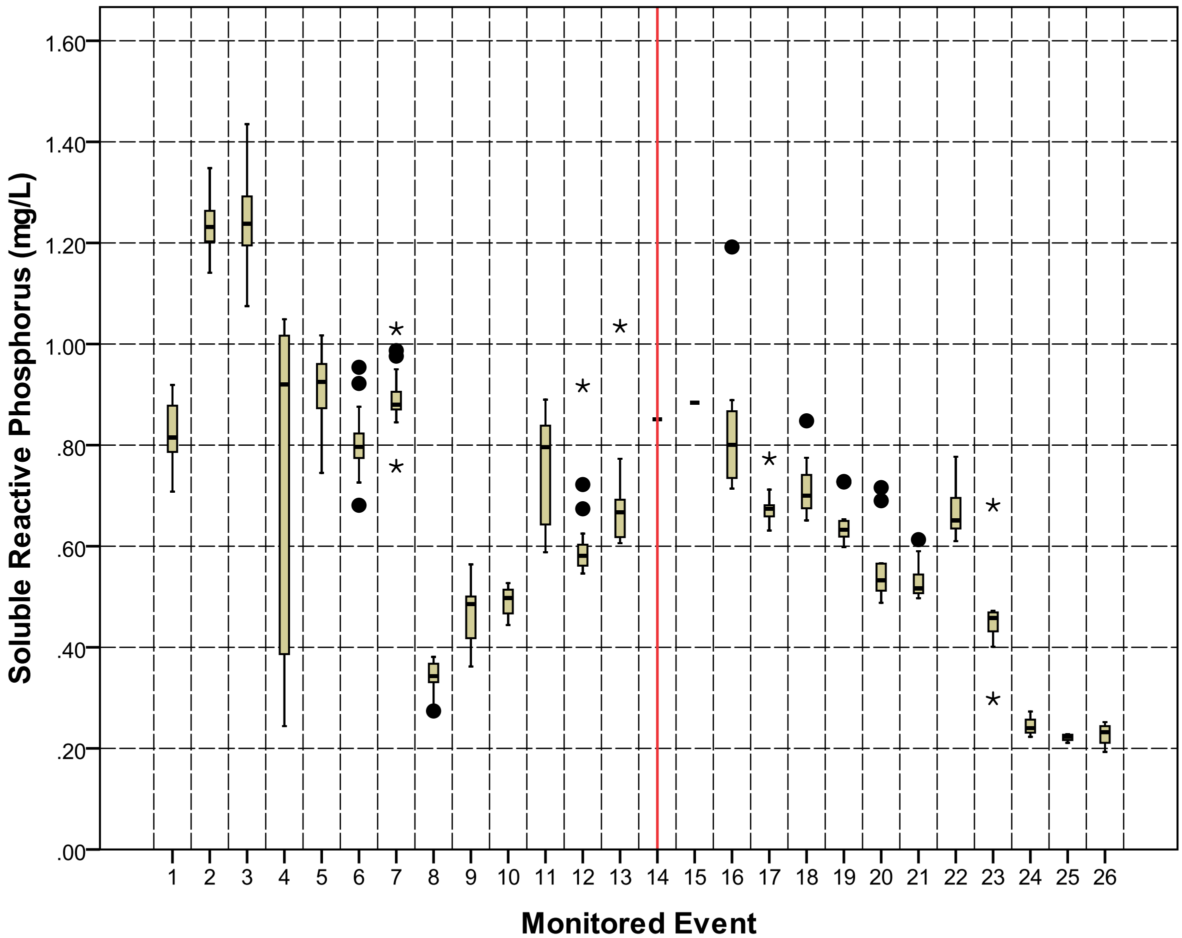


Figure 27: SRP influent concentrations measured over two monitoring periods.

*Red line divides 2009 and 2010 measurements

3.5.5: Phosphorus Treatment Efficiency

The P treatment efficiency was initially very high for the 2009 media (Figure 28). The lower percent removal shown in the first storm event monitored on July 11th was likely due to 'short-circuiting' of water (reduced contact time) that occurred within the system. The first 9 post-treatment samples had a mean SRP concentration of 0.375 mg/L but the following 13 samples had concentrations below the method detection limit. Due to the high intensity of the July 11th storm the rain barrel filled to capacity very quickly. Runoff may have entered the centre drainage tube without contacting the media. Following this storm a plug was installed within this centre tube to prevent short-circuiting.

For the next storm on July 23rd much higher removal rates for both SRP and TP were achieved. SRP concentrations were reduced by 98.2%, to a mean concentration of 0.022 mg/L. Similarly, TP concentrations were reduced by 97.8%, attaining a mean concentration of 0.038 mg/L. With the exception of the August 20th event, the percent P removal decreased linearly (R^2 = 0.947, y = -11.23x + 111.9). Based on a local rainfall Intensity Duration Frequency (IDF) curve, the August 20th event was approximately a 35 year event. The volume of runoff produced vastly exceeded the system design, making it difficult to characterize the low performance previously demonstrated. The larger storms had lower percent removal in concentration, but removed larger loads (Figure 30). The largest SRP load removed occurred during the first event on July 11th. Although the treatment efficiency was lower due to the short-circuiting issues, the high runoff volume resulted in a load removal of 11.3 mg/m^2. In comparison, the July 23rd event, while achieving over 98% percent removal, only removed 3.13 mg/m^2 SRP load due to a 10 times smaller runoff volume. Similarly, the largest TP load of 16.3 mg/m^2 was removed on August 20th. Overall percent removal was only 17.5%. However, the total runoff volume was over 1000 L, three times greater than any other monitored storm event. By the end of the 2009 monitoring season the percent removal rates were fell below 10%, demonstrating evidence of almost complete loss of P removal capacity.

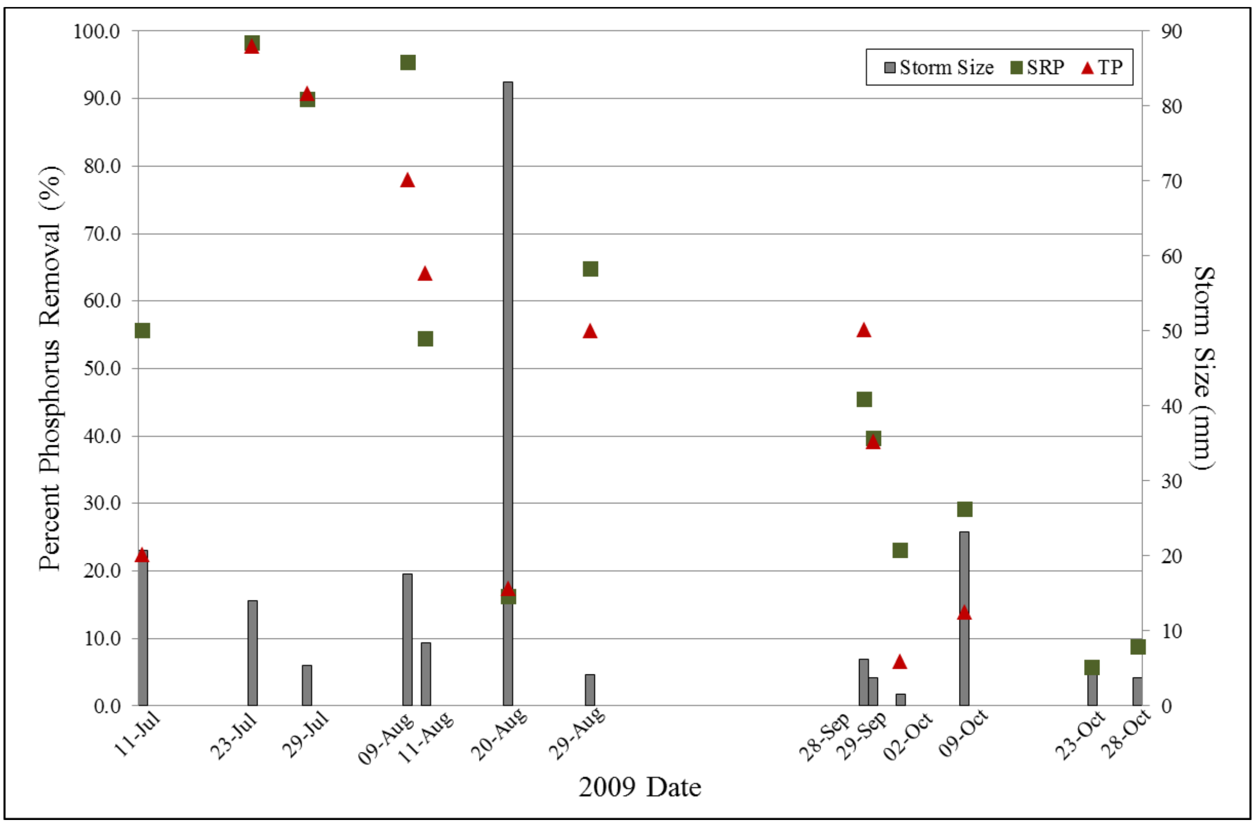


Figure 28: Percent P removal for each 2009 storm event.

The 2010 monitoring season started with the 17.6 mm storm event on May 7. Treatment efficiency was very high, achieving removal percentages of 95.7 and 94.8% for SRP and TP, respectively (Figure 29). The lowest mean effluent concentrations of the 2010 monitoring season were recorded on this date measuring 0.036 mg/L and 0.082 mg/L for SRP and TP, respectively. For the events on May 7th and May 13th, a problem occurred with the influent collection. Sufficient volumes were collected for SRP analysis; however TP analysis could not be completed. An estimate for TP concentrations was calculated based on the average SRP fractionation of 54.4% found in the other 2010 monitored events. For example, an SRP concentration of 0.884 mg/L collected on May 7th was equivalent to a TP concentration of 1.62 mg/L. Consequently, the TP loading values for these two storm events are based on calculated estimations.

The largest load removed for both SRP and TP occurred on June 24th with 12.3 mg/m^2 and 32.5 mg/m^2, respectively (Figure 31). Over the course of the monitoring period removal percentages remained high, with only three events falling below 80% removal for SRP and only one event for TP. The 2010 media did not appear to be exhibiting any evidence of loss of sorptive capacity which was apparent in the 2009 media.

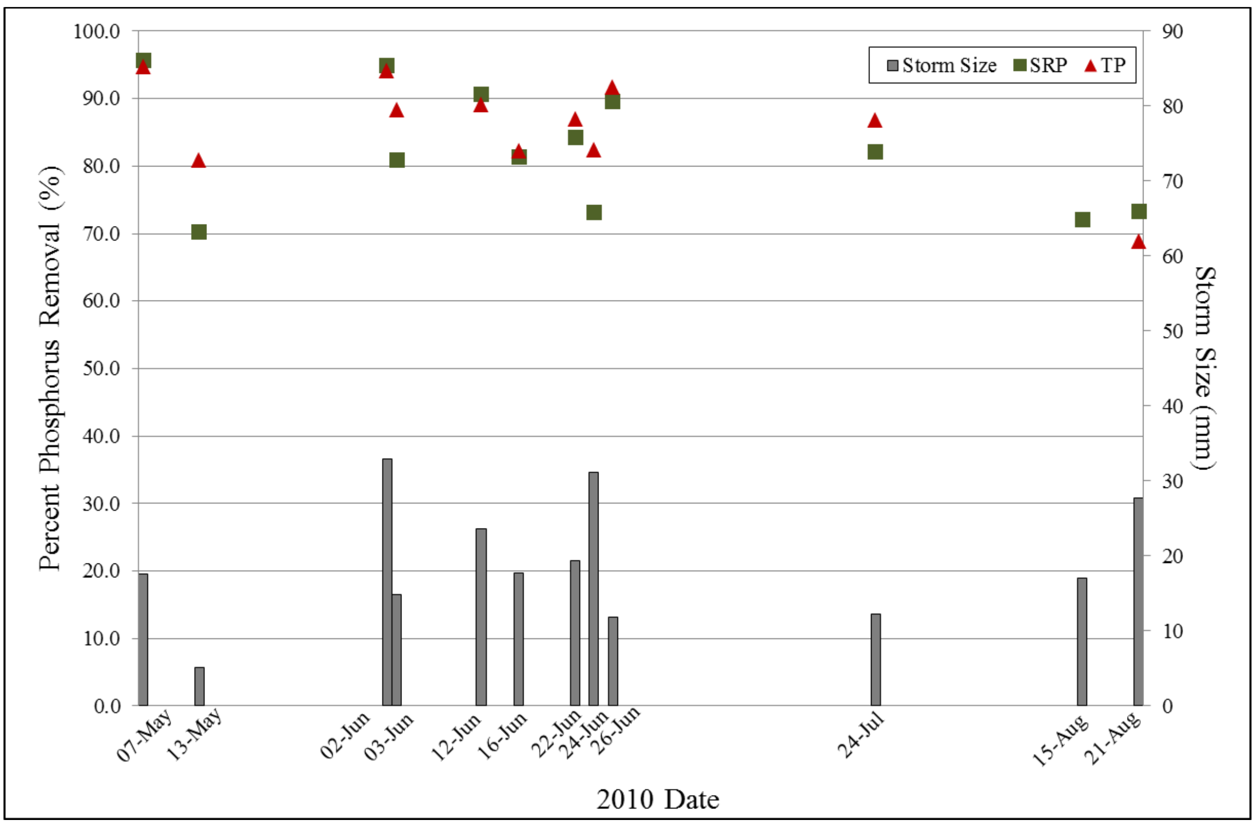


Figure 29: Percent P removal for each 2010 storm event.

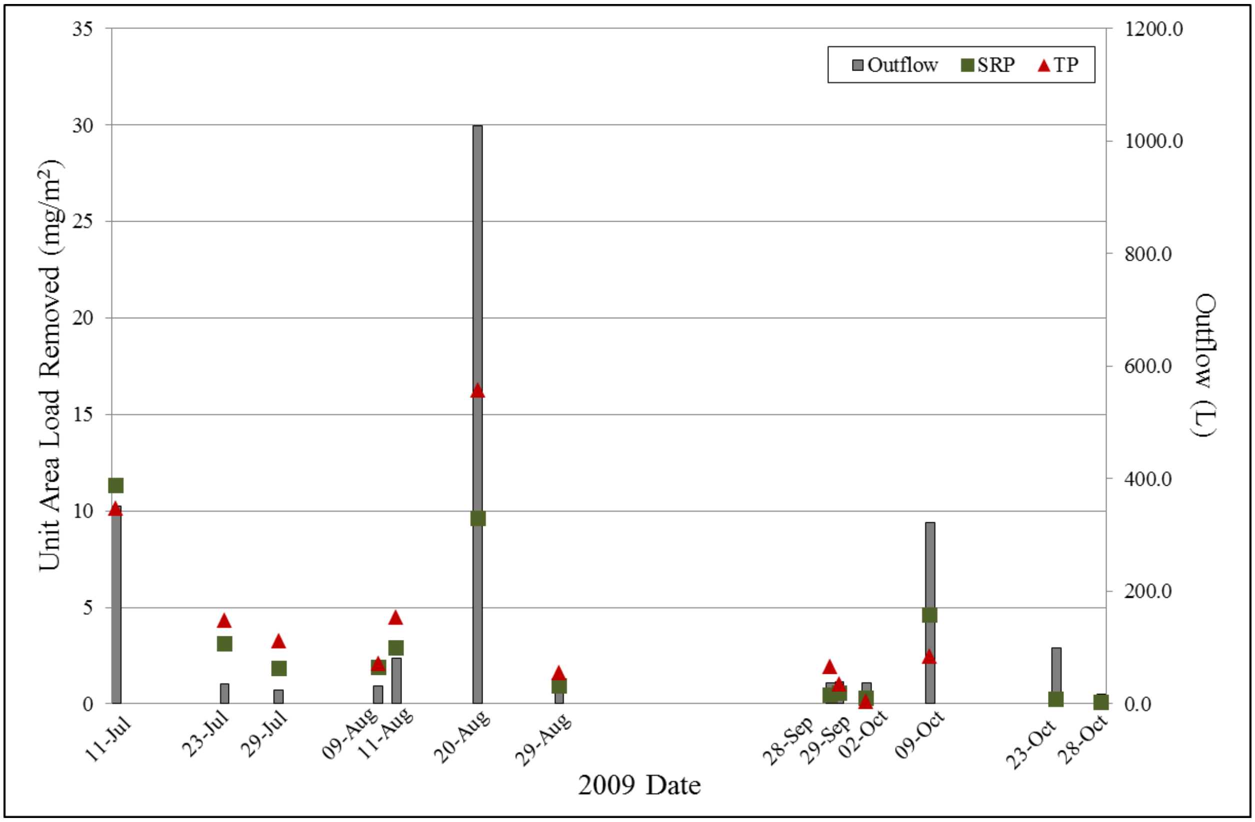


Figure 30: 2009 P mass removed by treatment system.

Note: July 11th, 2009 event had larger drainage area and runoff volumes.

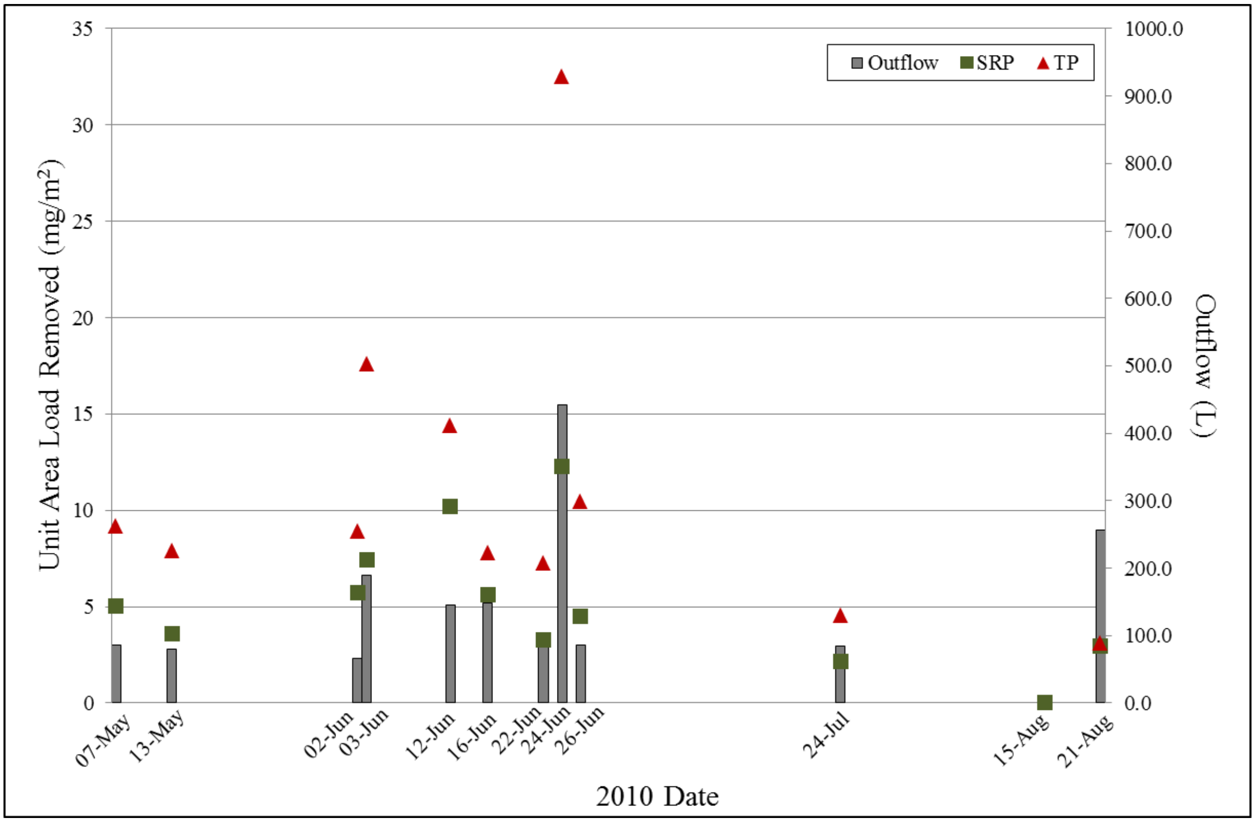


Figure 31: 2010 P mass removed by treatment system.

By the end of the 2009 monitoring season the sorption capacity of the media had decreased markedly. Low removal performance may be as a result of diminishing sorptive capacity within sections of the treatment system. The top of the media is located at 0.330 m of the total 0.914 m height of the barrel. Water level data from the HOBO® level loggers demonstrate that during certain storm events the volumes and rate of runoff were too low to fully saturate the entire depth of media (Figure 32). Water level data for each storm is presented in Table B3 in Appendix B. Accordingly, the media located at the bottom of the filter cartridge is more frequently exposed to higher volumes of runoff over the monitoring period. Runoff from low-intensity storms (October 23rd and 28th) was likely treated by media which had lost the majority of its sorptive capacity. The full depth of the media was utilized for events on July 11th, August 9th, August 11th, August 20th, October 6th, and October 9th.

Due to a lack of barometric readings for the August 29th event, and a lack of available memory for the September 28th and 29th events, water level readings are not available. However, comparing similar peak flow values of other storms, a reasonable estimate of the runoff stage can be made. All three events were unlikely to utilize the entire depth of media based on the runoff volumes of 22, 38, 40 L respectively.

During the 2010 monitoring season problems with the HOBO level loggers persisted. Data corruption prevented several storm events from being recorded. Runoff stage data was recorded for 8 storm events. The monitored events in 2010 had a higher average peak stage of 0.465 m, than the 2009 events which had an average peak stage of 0.339 m. The 2010 value would likely decrease if the monitoring period extended later into the year and included more low-intensity storms. As the system design remained the same between monitoring periods, the 2010 media would have experienced similar exhaustion in the lower depth of the cartridge. However, the increased sorptive capacity of this media appeared to counteract this effect for the time in which the study occurred.

3.5.6: pH

During 2009 the greenroof runoff pH ranged from 6.61 to 9.02 and had a mean pH of 7.86. The effluent ranged from 6.32 to 8.93 and had a mean pH of 7.79. For 2010 the influent pH ranged from 5.82 to 9.02 with a mean of 8.22. The effluent ranged from 4.65 to 8.63 with a mean of 7.56. The greenroof runoff changed the pH of the acidic precipitation to a more neutral value. Overall the greenroof runoff pH was slightly decreased after passing through the treatment system.

3.5.7: Conductivity

During 2009, the greenroof runoff conductivity ranged from 175 to 1164 µS/cm with a mean value of 619 µS/cm. After treatment the conductivity ranged from 232 to 982 with a mean value of 516 µS/cm. For the 2010 monitoring season the influent conductivity ranged from 210 to 792 µS/cm, with a mean value of 539 µS/cm (Figure 32). The effluent concentrations ranged from 363 to 20 200 µS/cm, with a mean value of 1618 µS/cm. The first 2010 samples displayed very high conductivity readings before decreasing to a consistent level. Particulates bound to the media from the manufacturing process were likely washed off during the first storm events accounting for the higher initial conductivity readings (Figure 34). The median conductivity value for the 2010 effluent samples was 615 µS/cm, a value similar to the mean effluent concentration in 2009. Conductivity readings per storm event are presented in Appendix A.

Generally, both the pre influent and effluent samples demonstrated a slightly increasing conductivity measurement through the storm event (Figure 33).

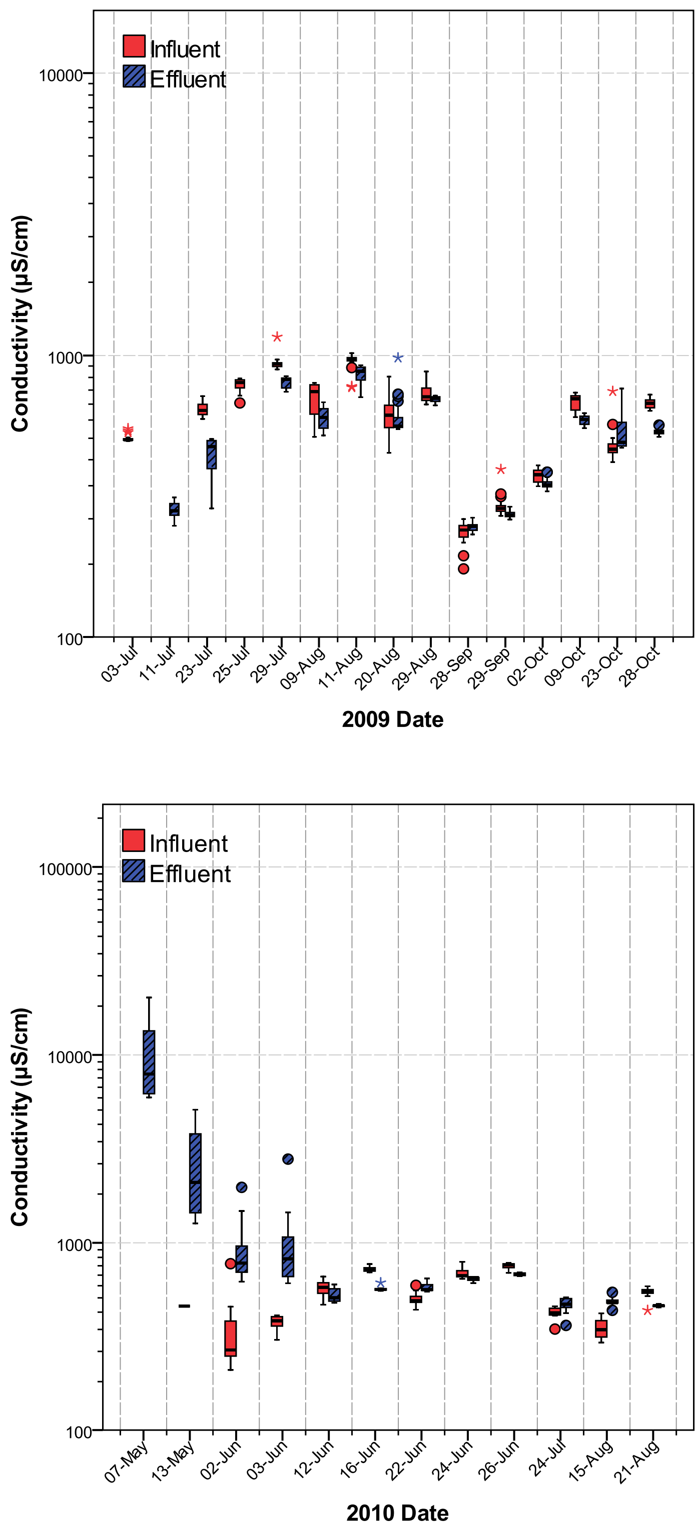


Figure 32: Distribution of conductivity measurements during 2009 and 2010 monitoring period (*=extreme value; °=outlier).

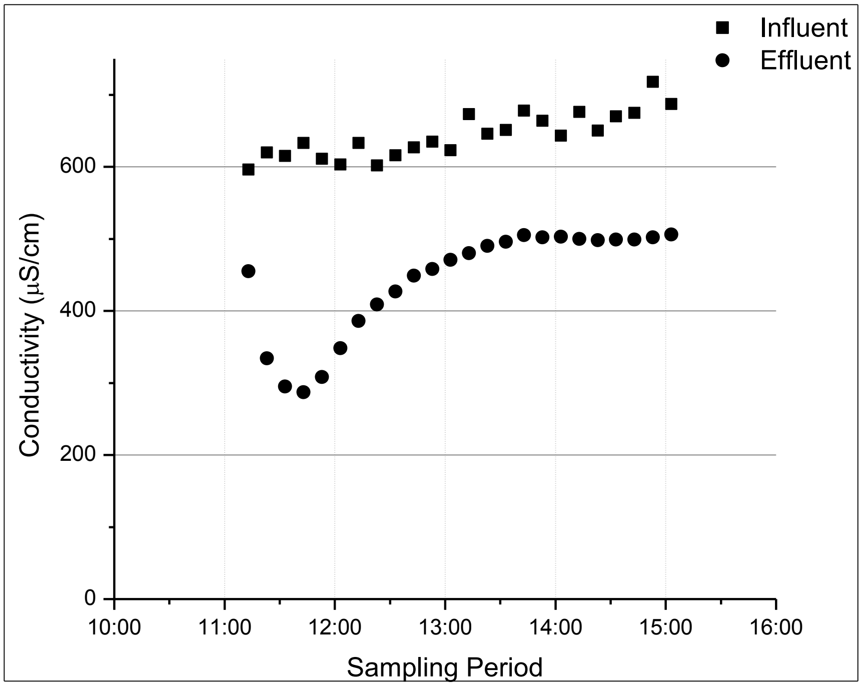


Figure 33: Conductivity measurements for the storm event on July 23, 2009.

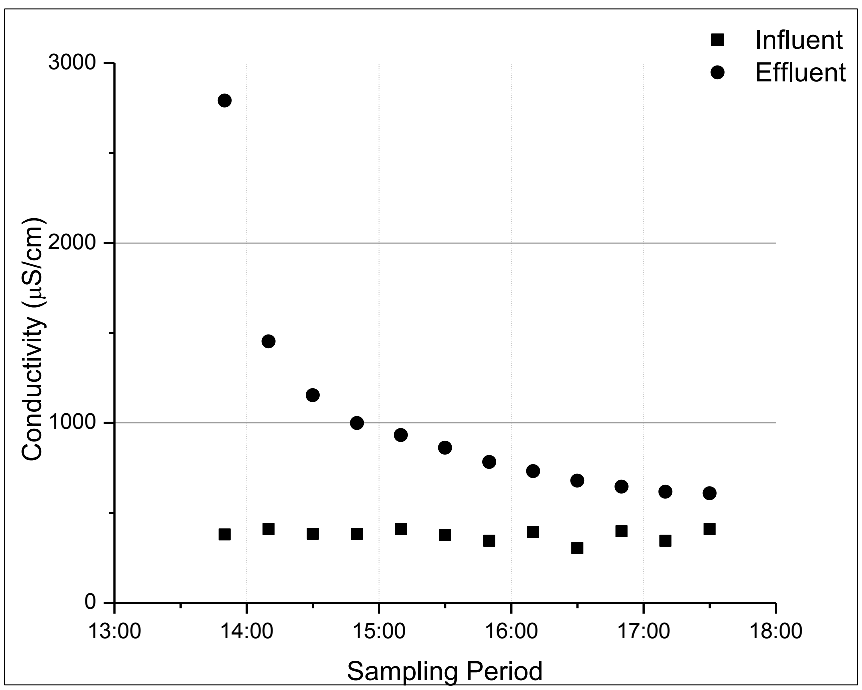
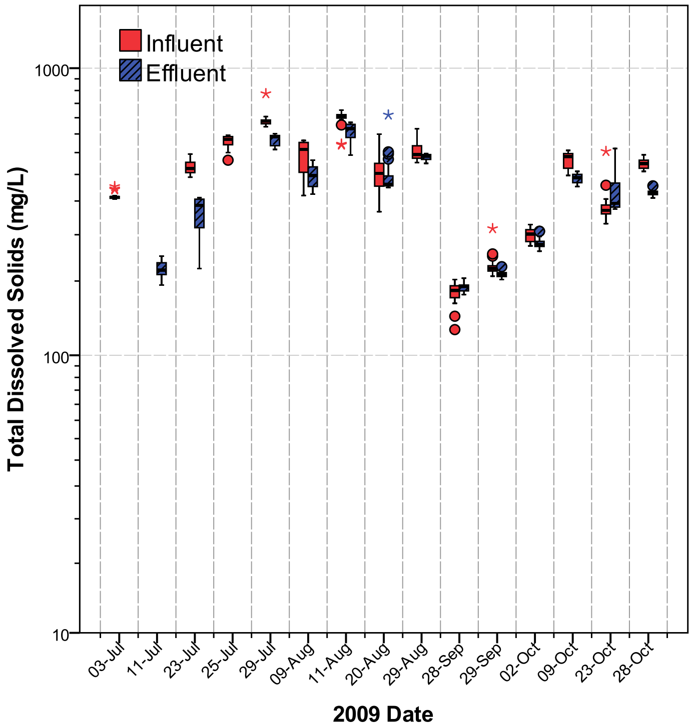


Figure 34: Conductivity measurements for the storm event on June 3, 2010.

3.5.8: Total Dissolved Solids

During 2009, TDS in the greenroof runoff ranged from 123 to 816 mg/ L with a mean of 408 mg/L. The post-treatment samples ranged from 162 to 648 mg/L and had a mean of 371 mg/L (Figure 35). The UAL values are essentially equal at 53 800 mg/m^2 due to the volume-weighting of a larger storm where TDS concentrations were higher in the effluent samples. The volume-weighted mean concentration is 420 mg/L for both the influent and effluent samples. The UAL_{ex} was 107 000 mg/m^2. For 2010 the TDS influent concentrations ranged from 149 to 548 mg/L and had a mean value of 374 mg/L. The effluent concentrations ranged from 248 to 14 100 mg/L and had a mean value of 11 300 mg/L. The volume weighted mean concentration was 389 and 857 mg/L for the influent and effluent respectively. The UAL_{ex} was 72 500 and 160 000 mg/L for the influent and effluent respectively.



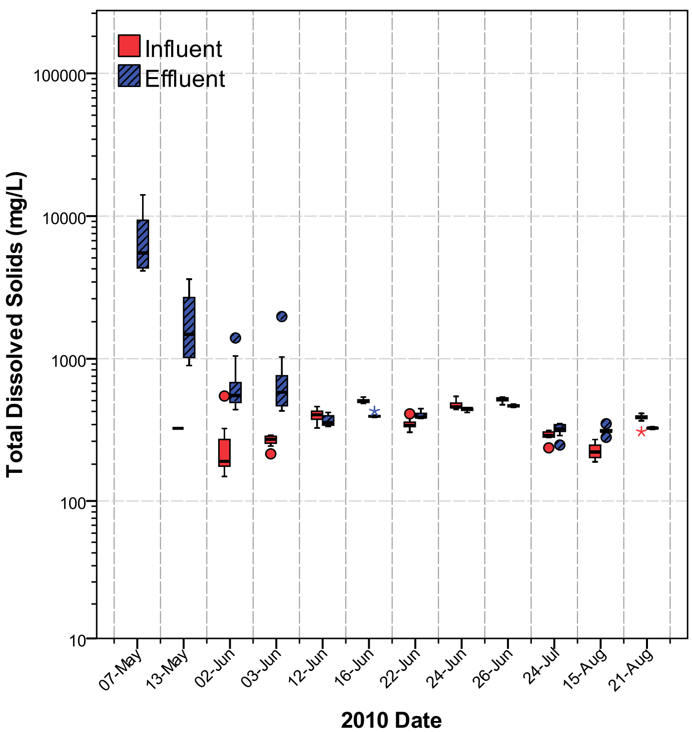


Figure 35: Distribution of total dissolved solids concentrations during 2009 and 2010 monitoring periods (*=extreme value; °=outlier).

3.5.9: Suspended Solids Concentrations

The greenroof runoff contained low concentrations of SS. Generally the first sample had the highest concentration and the quickly decreased in subsequent samples (Figure 36). The range of SS was 0.0 to 42.4 mg/L with a mean value of 7.6 mg/L (Figure 37). After the runoff passed through the treatment system concentrations decreased, ranging from 0.0 to 27.4 mg/L with a mean of 3.5 mg/L. SS measurements were only collected for the 2009 monitoring season.

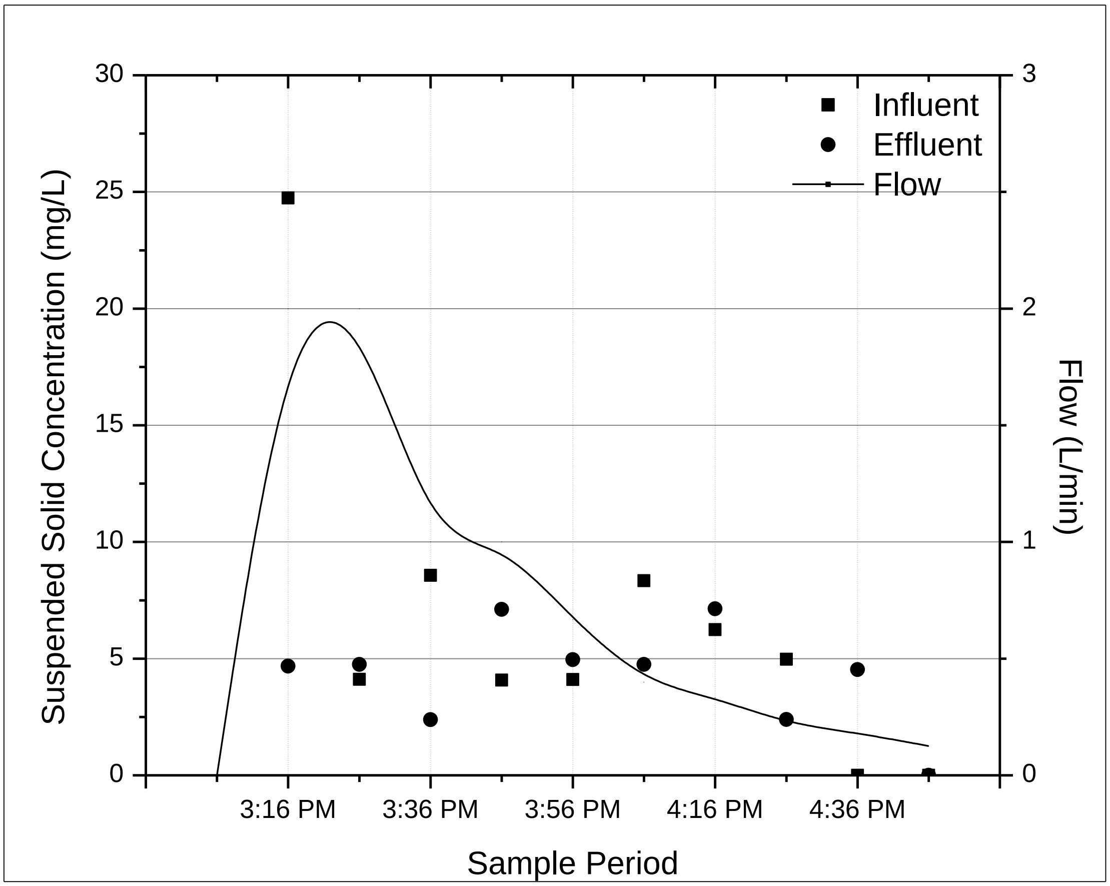


Figure 36: Suspended solids concentrations for storm event on August 11, 2009.

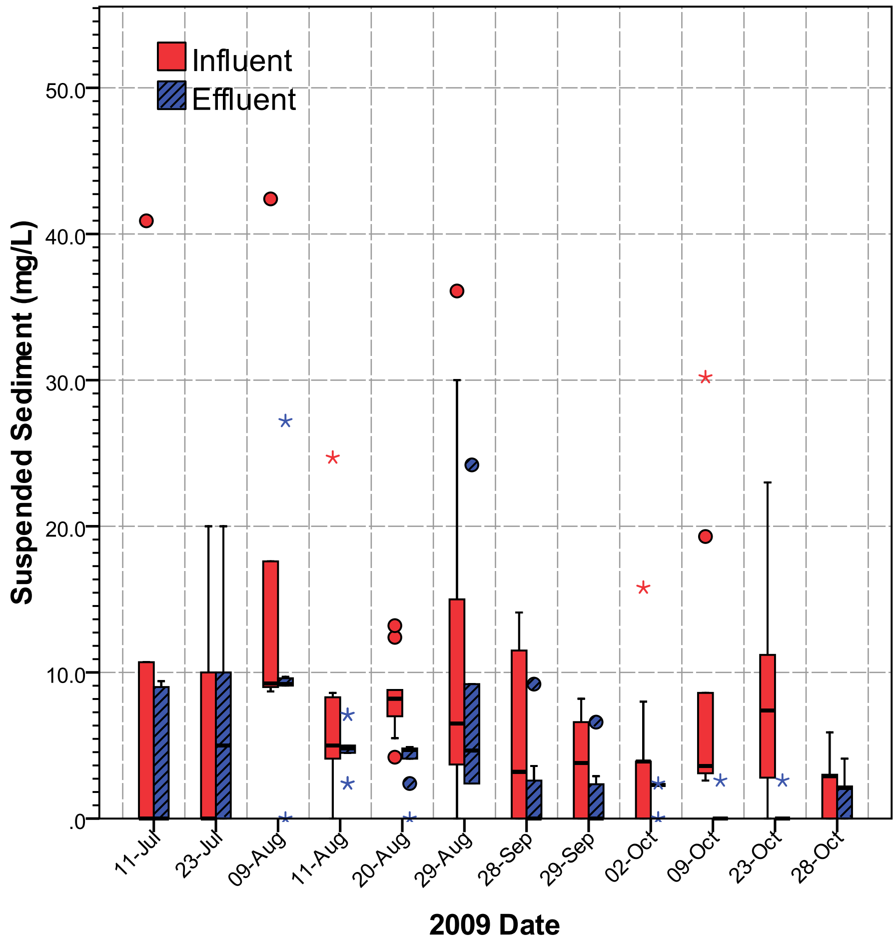


Figure 37: Range of suspended solids concentrations during 2009 monitoring period (*=extreme value; °=outlier).

3.5.10: Grain Size Distribution

Grain size distribution of SS was determined for 4 storm events. The size distribution of suspended solids for a storm event on September 28, 2009 is presented in Figure 40. GS distributions for events on August 11, August 20 and August 29, 2009 are found in Appendix A. For each event, grain size distribution was analyzed for each sample to examine temporal variation over the hydrograph. No temporal variation in GS distribution was evident (Figure 40). The D_{90}, D_{50} and D_{10} which represent the median grain size at the 90th, 50th and 10th percentiles of the grain size distribution were calculated for each event. For the storms of August 11, August 20 and September 28, 2009, the D_{90} was slightly higher in the post-treatment samples (Table 6). Photomicrographs show evidence of flocculation which likely accounted for the observed increase in particle size (Figure 38).

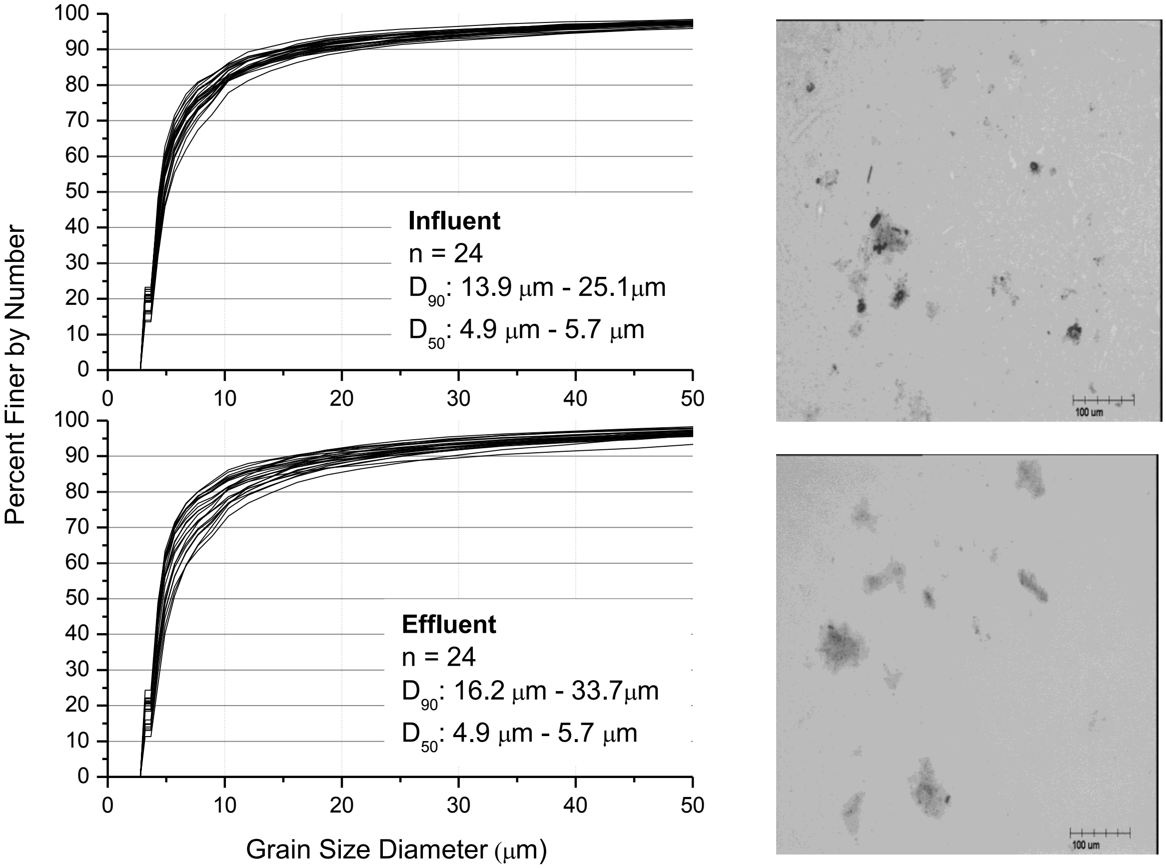


Figure 38: Grain size distribution of suspended sediment in influent and effluent samples of storm event on September 28, 2009. Representative micrograph of particles with scale = 100 microns.

As the runoff passed through the treatment system the largest particles became trapped. Figure 39 demonstrates the largest particles were measured in the influent samples. For September 28, the largest particle size in the influent was 662 µm, constituting 3.0% of the total volume of suspended sediment. The largest particle in the effluent was 362.7 µm which was 1.5% of the total volume. The particle size class of 94.8 µm represented the largest percentage of volume in the influent samples with 9.6%. In the effluent samples the particle size class of 127.6 µm constituted the majority of the total volume with 11.2%.

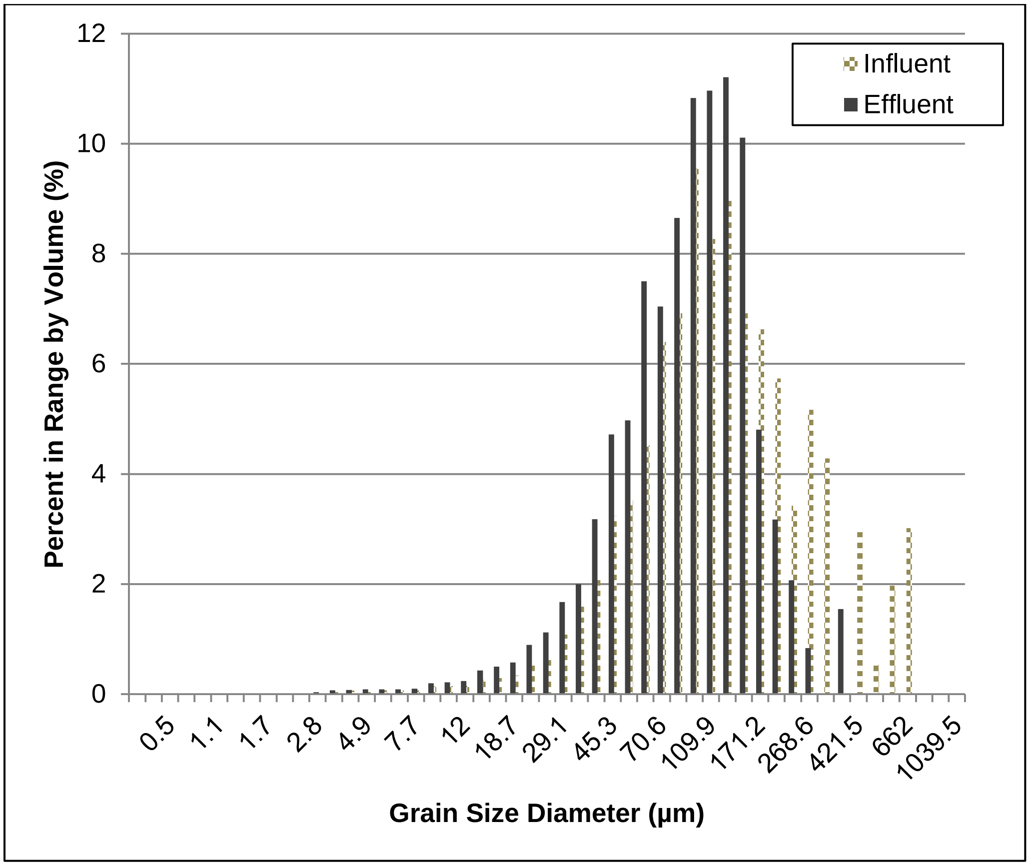


Figure 39: Volumetric distribution of particle sizes measured in samples for storm event on September 28, 2009.

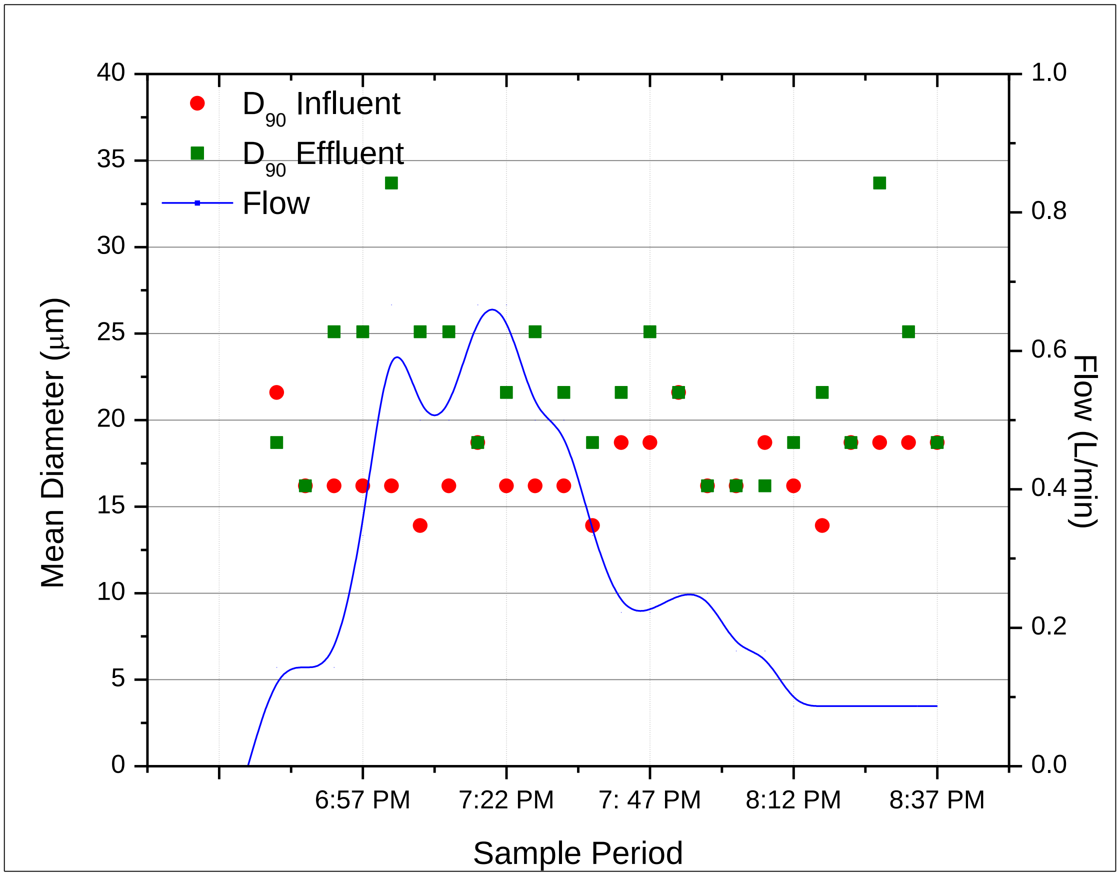


Figure 40: D_{90} variability throughout September 28, 2009 storm event.

Table 6: Grain size distribution results for 4 sampled storm events.

	Influent			**Effluent**		
2009 Date	**D_{90}**	**D_{50}**	**D_{10}**	**D_{90}**	**D_{50}**	**D_{10}**
Aug. 11	33.7	5.7	3.2	39.1	5.7	3.2
Aug. 20	33.7	4.9	3.2	39.1	4.9	3.2
Aug. 29	33.7	4.9	3.2	33.7	4.9	3.2
Sep.28	16.2	4.9	3.2	21.6	4.9	3.2
Overall Mean	**29.3**	**5.1**	**3.2**	**33.4**	**5.1**	**3.2**

3.6: Isotherm Test for Engineered Media

Batch tests were conducted to evaluate the sorptive behavior of the engineered media. The sorption data are presented in Tables 7 and 8 and Figures 41, 42, 43 and 44. The 2009 media was a better fit to the Freundlich plot, however the higher n values indicate a lower adsorption affinity. The 2010 media had n values indicating high adsorption affinity. The K coefficient confidence intervals ranged in value from 2.937 to 21.38. This range was much larger than the 2009 media, due to the lower R^2 value that increased the range of confidence intervals. The Freundlich coefficients support the observation of higher treatment efficiency in the 2010 media. During the batch experiments the 2010 media was demonstrating almost complete P removal for most initial P solutions. The initial P solutions had to be increased from 50 mg/L and 100 mg/L to fully determine the adsorptive capacity of the media. In comparison, the 2009 media demonstrated signs of complete adsorption at the 25 mg/L initial concentration.

Table 7: Freundlich isotherm data for the 2009 media.

Media	**Species**	**Isotherm Equation for Freundlich**	**R^2**	**1/n**	**n**	**LogK**	**K (mg/g)**
2009 Media	orthophosphate	y=0.494x-1.055	0.898	0.494	2.024	-1.055	0.088
UCL				0.551	1.815	-0.975	0.106
LCL				0.437	2.288	-1.135	0.073

Table 8: Freundlich isotherm data for the 2010 media.

Media	**Species**	**Isotherm Equation for Freundlich**	**R^2**	**1/n**	**n**	**LogK**	**K (mg/g)**
2010 Media	orthophosphate	y=1.011x+0.899	0.672	1.011	0.989	0.899	7.925
UCL				1.259	0.794	1.330	21.38
LCL				0.762	1.312	0.468	2.937

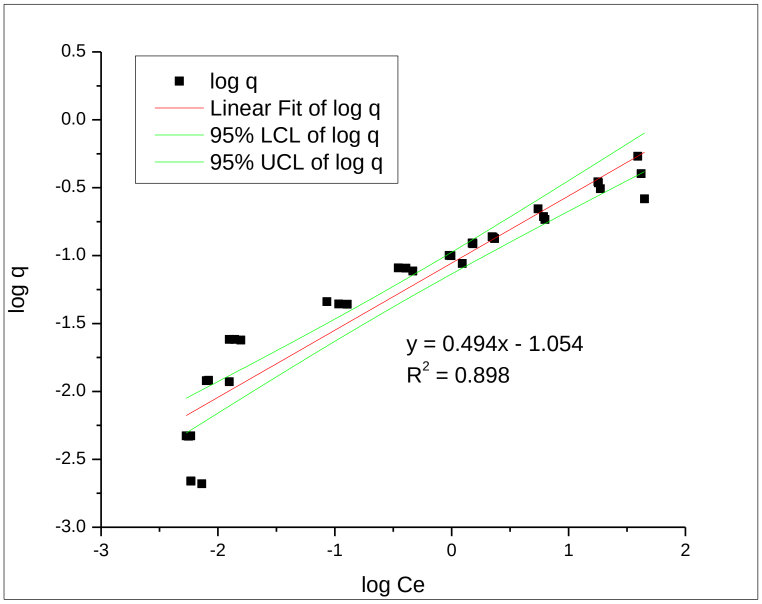


Figure 41: Freundlich equilibrium isotherm for 2009 media.

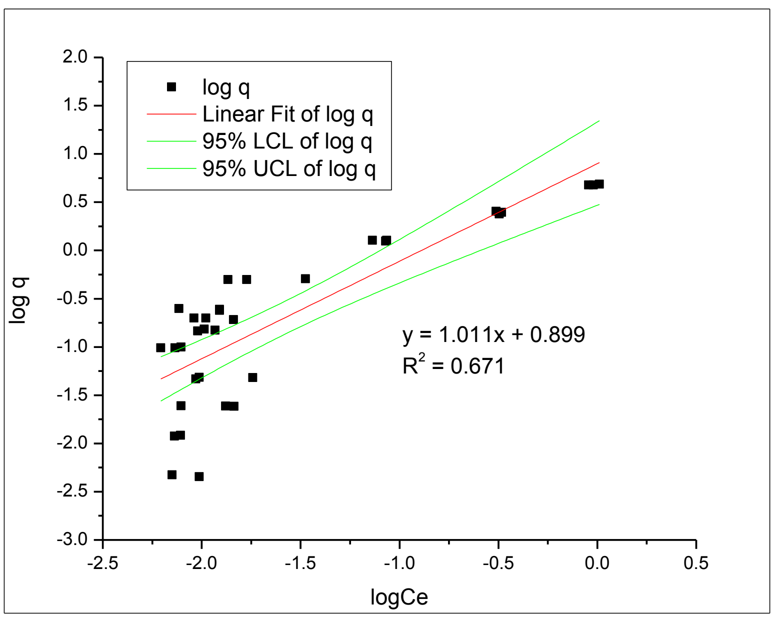


Figure 42: Freundlich equilibrium isotherm for 2010 media.

To obtain a theoretical maximum sorption capacity, the batch experiment data was fit to the Langmuir isotherm. The 2009 media had a q_{max}value of 0.420 ± 0.031 mg/g, with an UCL of 0.483 mg/g and a LCL of 0.356 mg/g (Figure 43). The 2010 media had a q_{max}value of 7.71 ± 0.51 mg/g. The UCL was 8.75 mg/g and the LCL 6.66 mg/g (Figure 44). The isotherms demonstrate that with increased P concentrations applied in solution, the mass adsorbed to the engineered media also increases. As the media becomes saturated the Langmuir line fit becomes more horizontal. The point in which the line becomes horizontal is the theoretical q_{max} value, indicating the maximum sorption capacity of the media. For the 2009 media the saturation is apparent as the curve noticeably flattens. The 2010 media displayed much higher sorption capacity with the Langmuir plot demonstrating only a gradual saturation curve at higher q values.

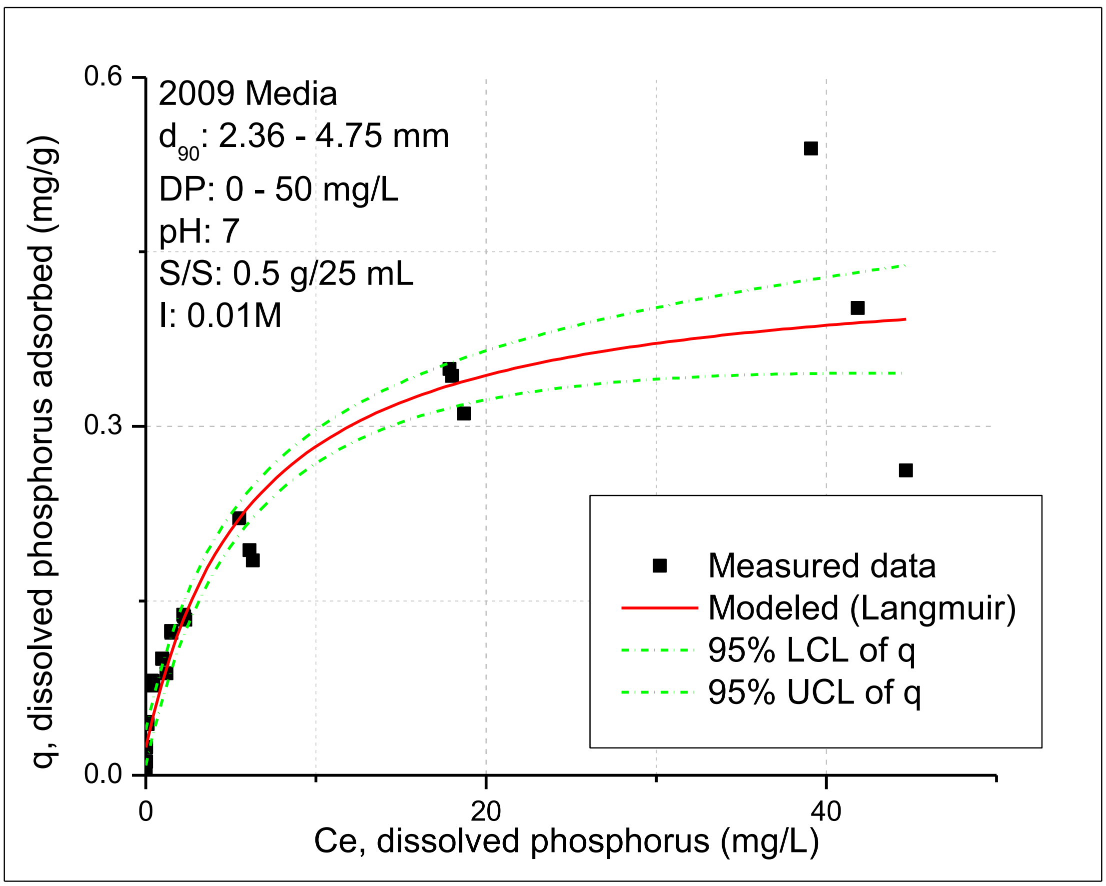


Figure 43: Langmuir equilibrium isotherm for 2009 media.

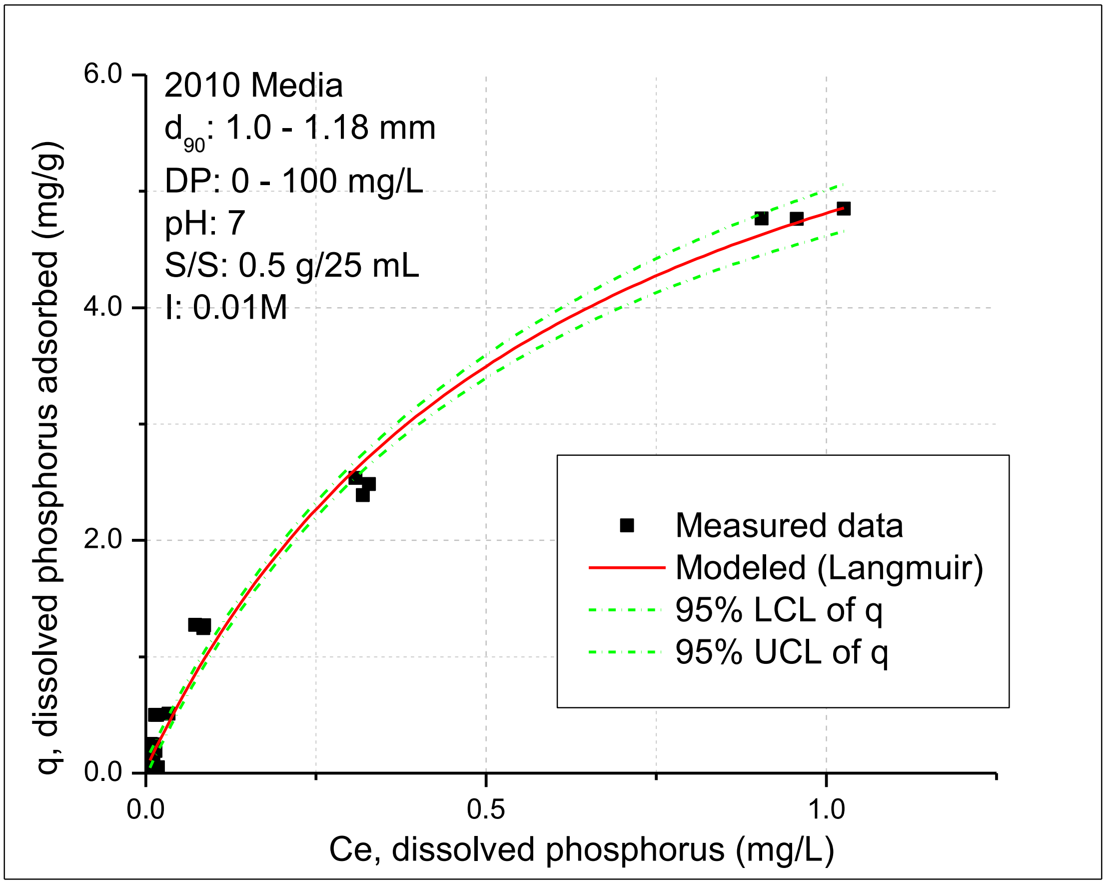


Figure 44: Langmuir equilibrium isotherm for 2010 media.

Chapter 4: Discussion

4.1: Introduction

Stormwater is recognized as a significant nonpoint source of pollution (Pitt and Clark, 2008; Carpenter et al., 1998). Urban stormwater management has evolved from a management of water quantity perspective to also addressing the quality of the stormwater. Stormwater has been shown to be a vector of various pollutants, including nutrients, metals, suspended solids and toxicants such as polyaromatic hydrocarbons (Scheueler, 1987). Berghage et al. (2009) estimated that the cost of wet weather flow pollution control in the United States is in the tens of billions of dollars. The recognition of this pollutant source has led to the incorporation of water treatment principles of filtration, coagulation and flocculation, ion exchange and adsorption into stormwater BMPs. As the field of stormwater treatment continues to evolve, researchers are investigating ways to optimize treatment applications that achieve the highest performance of stormwater treatment (Chang et al., 2010a). The use of sorptive materials incorporated into LID and stormwater BMPs is one strategy to further reduce nutrient loadings and subsequent freshwater eutrophication.

Sorptive materials are being examined for use in various disciplines of engineering, including stormwater, wastewater, groundwater and drinking water treatment (Chang et al., 2010a). A wide range of materials have been studied for the removal of nutrients using physical and chemical mechanisms such as sorption, sedimentation, filtration and precipitation (Hsieh and Davis, 2005). These contact adsorption systems have been used extensively in wastewater treatment (McKay, 1996). The materials used for sorptive materials are classified into three groups: natural materials, industrial by-products and manufactured products (Westholm, 2006). The physical and chemical properties of these sorptive materials affect the ability of these materials to remove pollutants (Cucarella and Renman, 2009). Physical characteristics include surface area and porosity, while chemical characteristics are related to the content of metal ions such as Ca, Fe and Al. The combination of the physical and chemical properties of these materials determines the abundance of sorption site and the affinity for which these sites will bind pollutants (Cucarella and Renman, 2009).

The use of greenroofs as effective stormwater source controls are well documented (DeNardo et al., 2005; Van Seters et al., 2009; Mentens et al., 2006). Greenroofs provide an opportunity for retrofitting space-constrained urban areas and partially restoring the natural hydrologic processes of infiltration and decreased stormwater runoff during storm events. Greenroofs have been reported to help improve water quality through the reduction of nutrients and metals (Peck et al., 1999; Köhler et al., 2002**)**. Recent studies, however demonstrate that treatment of nutrients in greenroof runoff are more problematic (Vander Linden and Stone, 2009; Van Seters et al., 2009; Moran, 2004; Berndtsson et al., 2006; Hutchinson et al., 2003). Consequently there has been a lack of studies addressing the treatment of greenroof runoff. Berndtsson (2010) states that greenroof runoff quality is influenced by several factors including: substrate type, media depth, vegetation, maintenance and application of fertilizers, drainage, dynamics of precipitation and wind direction influencing dry deposition of pollutants. Characterizing greenroof runoff quality is difficult based on the various influencing factors that can change per individual greenroof. The incorporation of sorptive materials is a potential improvement to the general use of greenroofs as stormwater quality BMPs. However, before sorptive materials can be incorporated into greenroof design, operation and maintenance procedures, it is necessary to rigorously test and review the potential benefits and limitations of greenroofs. The results of the present study are compared to previous literature in the following section.

4.2: Water Quantity

4.2.1: Retention Rates of the Greenroof

Quantification of the hydrologic characteristics of a greenroof system is critical in its design and implementation for water treatment application. This section examines the greenroof literature to help understand the hydrological responses to a range of environmental conditions. The retention rate of the greenroof in the present study is compared to previous greenroof studies in Table 9.

Table 9: Summary of literature on extensive vegetated roof stormwater retention performance.

Study	Location	Area (m^2)	Monitoring Period	Events	Media depth (mm)	Slope (°)	Retention (%)	Notes
Camm, 2011	Vaughan, ON	13.9	Jun. – Oct. 2009 May – Aug. 2010	39 29	180	<2	41.5 53.3	
Van Seters et al., 2009	Toronto, ON	241	Apr. 2003-Aug. 2005 excluding winters	-	140	10	65.3	Summer retention rates were 78-85% and 39 – 64% in the spring

Vander Linden and Stone, 2009	Waterloo, ON	450.5	Jun. – Oct. 2006	19	20	<2	41.4	
Berndtsson et al., 2006	Augustenborg, Sweden	9500[1]	Aug. 2001 – Aug. 2002	-	30	1.5	49	Nov. – Dec. 30-40% retention. Apr. – Jun. 80% retention
Bliss et al., 2009	Pittsburgh, PN	330	Aug. 2006-Jan. 2007	13	140	-	Up to 70	Peak flows lower by 5-70%
Carter and Rasmussen, 2006	Athens, GA	42.64	Nov. 2003-Nov. 2004	31	76.2	<2	78	
DeNardo et al., 2005	Rock Springs, PN	4.65	Oct.-Nov. 2002	7	89	<2	45	
Liu, 2003; Liu and Baskaran, 2005	Ottawa, ON	72	Nov. 2000-Sept. 2002	-	150	2	54[2]	
VanWoert et al., 2005	Lansing, MI	5.95	Aug. 2002-Nov. 2003	83	40	2	87	
Liu and Minor, 2005	Toronto, ON	200	Mar. 2003-Nov. 2004 (excluding winters)	-	100 75	<2	57[2]	
Moran, 2004	Kingston, NC	27	Jul.-Aug. & Nov.-Dec. 2003	11	102	3	63	
	Goldsboro, NC	70	Apr. – Dec. 2003	39	51 & 102	<2	62	
Teemusk and Mander, 2007	Tartu, Estonia	120	Jun. 2004-Apr. 2005	3 rain events / snowmelt measured for 2	100	<2	85.7[3]	Side of roof with less plant coverage released more

				weeks				runoff
Hathaway et al., 2008	Goldsboro, NC	70	Apr. 2003-Jun. 2004	-	100	3	64	

[1]Multiple roofs

[2] Relative to control roof runoff, not precipitation

[3] Highest retention for light rain event (2.1mm) and initial dry conditions. Retention decreased with more intense rain events

Based on a review of literature, greenroofs have been found to retain 41 to 85% of storm runoff. The current study found retention rates to be similar to those previously reported (Vander Linden and Stone, 2009; Berndtsson et al., 2006; DeNardo et al., 2005). During the 2009 monitoring period, the absolute retention rate was among the lowest found in the literature. During 2010, the absolute retention rate increased, although the monitoring period was shorter in length. This retention rate may have decreased if the monitoring had continued into the fall months, as in the previous year. The majority of the greenroof studies had absolute retention rates greater than either of the current study's findings (Van Seters et al., 2009; Bliss et al., 2009; Carter and Rasmussen, 2006; VanWoert et al., 2005; Moran, 2004; Teemusk and Mander, 2007; Hathaway et al., 2008). The lower retention rates are unexpected considering the greenroof growth medium depth in the current study was greater than all other reported studies (Table 10). Previous research has indicated that increased media depth results in greater retention rates (VanWoert et al., 2005). A study by Buccola and Spolek (2010) compared pilot scale plots of greenroof materials. Under medium rainfall conditions (3.0 cm/h), a 5 cm depth media retained approximately 36% of the applied rainfall, while a 14 cm depth media retained 64%. During heavy rainfall conditions (34 cm/h), the 5 cm depth media retained 20% and the 14 cm depth media retained 56%. The authors state that it is reasonable to expect a 55-65% retention rate with a media depth of approximately 14 cm. The media in the current study is slightly deeper but the retention rates are similar to this predicted percentage. The lower than expected retention rates may be due to a potential underestimation of precipitation volumes by the monitoring equipment. Due to the design and location of the greenroof, additional rainfall could have been directed onto the greenroof by the surrounding walls of the Archetype house. This additional precipitation would not be recorded by the rain gauge situated on the greenroof.

The retention rates during 2009 were lower than the 2010 monitoring period. In 2009, the greenroof retained 41.5% compared to the 53.3% in 2010. The difference between the two sampling periods is attributed to the seasonal performance of the greenroof. In 2010, 399.6 mm of rainfall was recorded on the greenroof. Although the 2009 monitoring period was longer, only 347 mm of rainfall was measured. The increased retention rate during 2010 can be attributed to the majority of the rainfall

(212 mm) occurring during the month of June when evapotranspiration rates were higher. Generally the summer period has short, high-intensity storms followed by longer dry periods. The 2010 retention rates may also be higher without the inclusion of fall rainfall events.

During October 2009, four events had negative retention rates. The net export of runoff is indicative of low evapotranspiration rates and a consistent low-intensity rainfall throughout the month. Under these conditions the stormwater from previous events would remain in the greenroof substrate creating a larger runoff volume in subsequent storms. During the 2010 monitoring season, three events had negative retention rates. The events on May 13th and Jun 27th were following large storms in previous days and the greenroof would have still contained water from these storms. Previous greenroof studies have reported similar lowered stormwater retention percentages through cooler months (Bengtsson et al., 2005; Vander Linden and Stone, 2009; Van Seters et al., 2009; Carter and Rasmussen, 2006). Vander Linden and Stone (2009) reported negative stormwater retention percentages occurring in October on a greenroof in similar climate conditions. The negative retention rates demonstrated in this study may be due to a combination of physical processes and an underestimation of rainfall volumes by monitoring equipment as previously mentioned. The four events with negative retention rates in October 2009 had a measured rain depth of 5 mm or less. Any errors in measurement may account for the large relative differences in retention rates during these smaller events.

4.2.2: Rainfall Size, Intensity, and Antecedent Dry Period Influence on Runoff Retention

The rainfall size and runoff retention data were not normally distributed. Therefore the non-parametric Spearman (r_s) rank-order correlation was used to determine the relative strength of the linear relationship between rainfall size and retention rates. Rainfall size and retention percentage are inversely related. Throughout the monitoring periods of 2009 and 2010, there was a weak but significant negative association (r_s= -0.300, $p<0.05$) between rainfall size and retention percentage. This negative relationship has been reported in other greenroof studies (Bliss et al., 2009; Carter and Rasmussen, 2006; Getter et al., 2007; VanWoert et al., 2005). In the current study a very significant relationship existed between rainfall size and rain depth retained (r_s= 0.443, $p<0.01$). Van Seters et al. (2009) noted that the greenroof runoff coefficient (percentage of rainfall converted to runoff) generally increased with event size. Smaller events produced no runoff. Seasonality was also shown to influence this trend, as smaller events during cooler months elicited lower runoff coefficients due to high soil moisture levels. Bliss et al. (2009) and Carter and Rasmussen (2006) found no relationship between runoff retention and rainfall intensity (r_s= 0.0494, p = 0.689).

In the present study the antecedent dry period appeared to have more influence than event magnitude. The distribution of previous dry hours between storm events was also non-normal. There was a very significant positive correlation (r_s= 0.449, $p<0.01$) between the length of time between storm events and the stormwater retention performance of the greenroof. Differing periods between storms would vary the soil moisture conditions and influence the adsorptive properties of the greenroof (Carter and Rasmussen, 2006).

4.2.3: Lag Time Response

In general the greenroof was effective in delaying the peak flow of runoff. Over the two years of monitoring, 8 of 68 total events had no lag time. During these events the peak runoff flow rate was measured at the same time as the maximum intensity of rainfall. It is difficult to detect any seasonal trend occurring with lag time response due to the lack of rainfall events during certain months. Several factors such as storm size, antecedent dry period and evapotranspiration combine to influence the greenroof response for each storm event (Van Seters et al., 2009). The intensity of the rainfall had the largest effect on the lag time. The maximum rainfall intensity (mm/5 min) was inversely correlated with lag time r_s=-0.582 ($p<0.01$). This relationship is apparent in the rainfall events during the period of June 12th – June 24th, 2010 (Appendix B, Table B2). Four successive events had no lag time effect. The max rainfall intensities for these events were 6.4, 2.8, 5.0 and 4.2 mm/5 min; values much higher than the median value of 1.0 mm/5 min for all events. The high volumes of rainfall delivered in a short period of time quickly exceed the greenroof's field capacity thus minimizing the lag time effect.

The transverse dimension (distance that the runoff must travel to the drain) of a greenroof influences lag time responses. With increased greenroof area, the distance the runoff travels to the drain increases, creating longer lag times (Buccocla and Spolek, 2010). In the current study, the greenroof area was very small, likely decreasing the potential lag time effect. The median lag time value of 35.0 minutes is comparable to a study completed by Carter and Rasmussen (2006). This study demonstrated that a greenroof had a median lag time of 23.1 minutes for 31 rainfall events. In this study the greenroof was an area of 42.64 m^2, so it would be expected that the lag time would also be greater due to a larger transverse dimension. However, the growing media depth was thinner at 76 mm, compared to the 180 mm substrate in the present study. Buccocla and Spolek (2010) noted increased lag times with greater substrate depth in their pilot-scale studies. Studies by Van Seters et al. (2009) and Vander Linden and Stone (2009), found average lag times of 29.8 and 69.0 min, respectively. Both of these studies defined the lag time as the difference between the start of rainfall and the initiation of greenroof runoff, thereby making direct comparisons to this study impossible.

4.3: Water Quality

4.3.1: Phosphorus in Greenroof Runoff

Throughout the study period the greenroof was a net source of P. A summary of greenroof literature indicates the results of the current study are similar to other studies. Concentrations were higher than those reported in some studies (Van Seters et al., 2009; Vander Linden and Stone, 2009; Berndtsson et al., 2006; Teemusk and Mander, 2007) but lower than those reported by Moran (2004) and Bliss et al. (2009) (Table 10).

Table 10: Summary of literature on greenroof P export.

Study	Location	Media type	P form	Mean P concentration in runoff		% difference	Notes
				Control (mg/L)	Greenroof (mg/L)		
Camm, 2011	Vaughan, ON	Mineral aggregates, blond peat, perlite, sand and vegetable based compost	TP	--	1.24[1]	--	
			SRP	--	0.700[1]	--	
Van Seters et al., 2009	Toronto, ON	Composite of crushed volcanic rock, compost, blonde peat, cooked clay, washed sand	TP	0.071	0.629[2]	322	Values in unit area loads. P levels in runoff significantly decreased over the 2 year study
			SRP	0.033	0.539[2]	675	
Vander Linden and Stone, 2009	Waterloo, ON	Composite of inert crushed brick, pumice, expanded slate, fine washed sand, organic compost, dolomite	TP	0.0154	0.0998	548	
			SRP	0.0038	0.040	953	

Berndtsson et al., 2006	Augustenborg, Sweden	Crushed lava, calcareous soil, clay, shredded peat	TP	--	0.6 – 1.2	--	Simulated rainfall events ranging from 10 – 20 mm
			SRP	--	0.3-0.5	--	
Bliss et al., 2009	Pittsburgh, PN	Expanded shale, perlite, coconut husk	TP	0.05	2.0-3.0	3500	
Moran, 2004	Kingston, NC	55% Perma Till (expanded slate), 30% sand, 15% compost	TP	0.0583	1.277	2090	
			SRP	0.048	1.046	2079	
	Goldsboro, NC	55% Perma Till (expanded slate), 30% sand, 15% compost	TP	0.050	1.036	1980	
			SRP	0.018	0.890	4844	
Teemusk and Mander, 2007	Tartu, Estonia	66% Lightweight aggregates (P, K, Ca, Mg, organic), 30% humus, 4% clay	TP	1.041	0.036	-188	

[1]Average of 2009 and 2010 volume-weighted mean concentrations
[2]Volume-weighted mean concentrations

Total phosphorus and soluble reactive phosphorus concentrations in the greenroof runoff decreased over the course of the two year study. The water quality monitoring was initiated shortly after the construction of the greenroof. The P contained in the growing media substrate quickly leached into the rainwater. This trend has also been reported in other greenroof studies (Van Seters et al., 2009). Many greenroofs are fertilized after construction, and intermittently thereafter. No fertilizers were applied to the study greenroof during construction or the monitoring period. A study by Emilsson et al. (2007) examined the use of conventional fertilizers and controlled release fertilizers (CRF) on greenroof substrates. The authors note that as expected high concentrations of nutrients are found in greenroof runoff after fertilizer application. They also reported that sustained elevated levels were measured after the course of the study. Conventional fertilizers supply more nutrients than the total exchange capacity of the substrate and uptake rate of the plants, leading to elevated nutrient levels in the stormwater runoff (Emilsson et al., 2007). CRF applications had lower nutrient concentrations in the runoff and were recommended for maintaining nutrient levels for plant growth over an extended period of time, yet reducing nutrient losses to surface water. The use of CRF lowers nutrient leaching problems, but does not eliminate them completely. A pollution control layer may be required to reduce nutrient levels to recommended levels (TP< 0.03 mg/L).

The current study did not demonstrate a consistent first-flush release of P occurring throughout storm events. Only 9 of 26 monitored events had the highest concentrations of SRP and TP in the initial samples. This finding is consistent with the variability of P export reported in other greenroof studies (Vander Linden and Stone 2009; Bliss et al., 2009). Berndtsson et al. (2006) reported a first flush of TP but not phosphate which was consistent in concentration through the sampled event.

4.3.2: Phosphorus Dynamics in Treatment System

During 2009 and 2010, the treatment system significantly decreased the concentrations of SRP ($p<0.001$) and TP ($p<0.001$) from the greenroof runoff. The media achieved very high removal rates however the effluent concentrations still exceeded the MOE Provincial Water Quality Objective (OMEE, 1994) value of 0.03 mg/L for TP. Achieving this water quality objective would require a more efficient use of the media. Optimizing the sorption processes with increased contact time may achieve these effluent concentrations, however holding runoff for an extended period of time may not be possible based on design constraints and hydrologic responses of the greenroof.

A linear regression model was used to analyze the TE of the media over the monitoring period. The null hypothesis of the model was that there was no change in TE over time. This is reported as a coefficient (B), representing slope that is modifying the independent variable of time. No trend (B = 0) would be represented as a straight line. First examining the SRP TE, the model reported a downward trend for percent removal for both the 2009 and 2010 periods. The 2009 data had a significant downward trend ($p<0.001$, B = -8.205) with a fit $R^2 = 0.570$. The 2010 data was initially a very poor fit ($R^2 = 0.089$) due to outlier and extreme values measured on June 2, June 12 and June 26. The very high P concentrations may be due to biofilm formation in the tubing equipment which is sloughed during sampling. Without these values included in the analysis, the fit increased to $R^2 = 0.772$ and a significant downward trend was also reported ($p<0.001$, B = -5.633). The 2010 percent removal data had a smaller slope indicating the TE was decreasing at a slower rate than during 2009. However, examining the influent and effluent data individually shows that the downward trend reported during 2010 is due to decreasing influent concentrations. Analyzing the 2010 effluent data shows a slight downward trend ($p=0.001$, B = -0.602). A positive trend would be expected for effluent concentrations, as the TE of the media decreases over time. However, the small downward trend is due to the influence of high effluent concentrations measured during the second event of the season. The influent concentrations demonstrate a significant downward trend ($p<0.001$, B = -6.385), $R^2 = 0.804$. Comparing the 2009 effluent concentrations, there is a significant increasing trend ($p<0.001$, B = 3.201). Influent concentrations also demonstrated a significant downward trend ($p<0.001$, B = -5.473), however this effect was not as severe in 2009. The linear regression model demonstrates that the sorptive capacity of the 2010 media was greater than in the 2009 media.

Few studies have examined the use of sorptive materials for P removal in a stormwater field application (Table 11). Results are highly variable and the use of different materials and treatment systems make comparisons difficult. However, these studies may help inform better design for a variety of sorptive media applications.

Table 11: Field results of sorptive media applied to stormwater treatment facilities.

Study	Sorptive material	System	Treatment Efficiency
Camm, 2010	Oxide-coated pumice (version 1) Oxide-coated pumice (version 2)	Gravity flow filter cartridge treating greenroof runoff	2009 media: 32.0% SRP, 25.4% TP 2010 media: 82.4% SRP, 86.6% TP
DeBusk et al., 1997	Sand	Sand filters treating a wet detention pond	57% TP summer 1995 34% TP winter 1996 25% TP summer 1996
Hsieh and Davis, 2005	100% sand	6 bioretention sites	37 – 99 %
Ádám et al., 2005	Filtralite P®	Simulated constructed wetland	23-53%

Ádám et al. (2005) simulated a constructed wetland, applying daily volumes of 1.25 L, 2.5 L, and 5 L at a range of concentrations to each experimental setup. They demonstrated that after a period without P loading, the media exhibited signs of sorptive capacity regeneration. Media which was 70-90% saturated decreased P concentrations by 22-28% after 44 days of rest. Media which was only 10% saturated did not improve noticeably. An increase in the specific surface area from calcium phosphates was attributed for the increased sorption capacity after the resting period. Regeneration of sorptive materials complicates lifespan estimates for field scale applications. It is difficult to detect any recovery of sorptive capacity in the present study. For example, the time between May 13th, 2010 and June 2nd, 2010 appears to improve sorption as SRP TE from 70.2 % to 94.9%. However, between June 26th, 2010 and July 24th, 2010 the SRP TE decreases from 89.5% to 82.1%. The hydrological characteristics of each rainfall event and variability in loadings to the treatment system prevent any clear evidence of regeneration of sorptive capacity. Further laboratory testing would be required to characterize any potential regeneration effect within the media.

P removal capacities are often correlated to the chemical properties of sorptive media. High Al, Fe and Ca content create larger CEC which favours anion bonding. Debusk et al. (1997) tested a quartz sand media that had low mineral content and achieved only 38% P removal in a sand filter application. Comparative batch testing demonstrated that the same sand removed only 41% compared to Wollastonite, a media high in Fe and Ca content which removed 98%. In contrast, Hsieh and Davis (2005) found no relationship between media with high CEC and P removal. The lack of correlation was attributed to preferential flow paths and dynamic runoff processes preventing P complexation and removal.

4.3.3: Phosphorus Treatment Efficiency

At the conclusion of the 2009 monitoring season, samples of used treatment media were collected with cores to evaluate differences in the vertical treatment efficiency of the media in the cartridge profile. A series of batch experiments were completed to examine the P sorption capacity of the used media. These sorption experiments clearly demonstrate that the media's sorption capacity had not been completely exhausted. A full profile of the used media was removed, however compaction of the media was difficult to avoid while using the coring tubes. The media samples which were included in the sorption tests are the closest representation of the 'top' and 'bottom' of the media profile. Comparing the inner core to the outer core, the outer core had lower sorption capacity (Figures 45 and 46). This result is expected as the media around the outside was exposed to runoff with the highest P concentrations because the runoff enters the perimeter of the cartridge. As the media adsorbs P, the interior media will be subject to lower concentrations. Figures 45 and 46 also show that samples from the bottom of the cartridge (0-3 cm depth) had a lower P sorption capacity than samples taken from the top (9-12 cm depth). This difference is an indication that the full volume of media was not being fully utilized. The HOBO level logger data shows that during smaller runoff volumes, the stage within the rain barrel was not sufficient to completely saturate the media (Appendix B). Therefore the frequency of smaller events resulted in the media at the bottom of the cartridge being exhausted at an increased rate. The low treatment efficiency observed at the end of the 2009 season is a result of the smaller rainfall events which were only treated by the most exhausted portion of the media. To ensure more effective treatment, the full volume of the media needs to be used for each runoff event. The scale of the greenroof may have been too small to produce the runoff needed to full test the design capacity of the system. The maximum flow capacity of the cartridge is 1.14 L/s which is greater than the flows experienced in this study. Designing a treatment system for a greenroof system becomes more complicated due to the difficulties predicting the runoff volumes for each event.

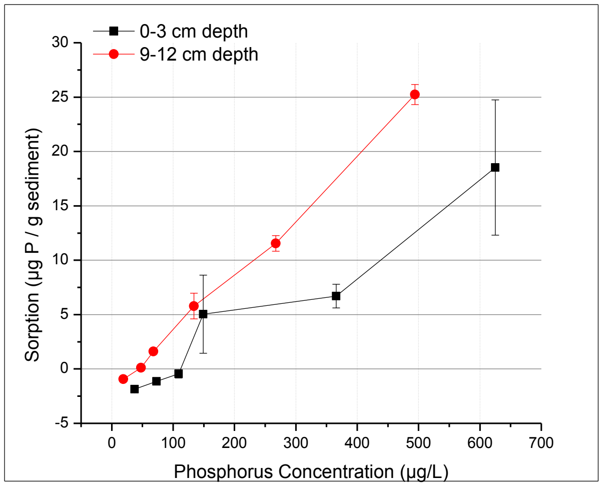


Figure 45: Used engineered media inner core sorption test. Error bars indicate standard deviation.

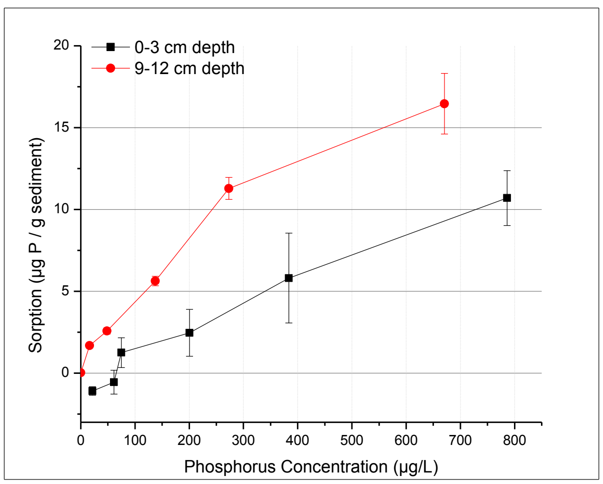


Figure 46: Used engineered media outer core sorption test. Error bars indicate standard deviation.

4.3.4: Treatment System Influence on Additional Water Quality Parameters

The mean pH for the greenroof runoff was significantly higher than the rainfall pH for both 2009 and 2010 ($p<0.001$). The buffering of acidic rainfall by greenroof media has been reported in other studies (Long et al., 2006; Van Seters et al., 2009; Vander Linden and Stone, 2009). Berghage (2009) considers the pH buffering effect of greenroofs one of the most consistent benefits to water quality. Neutralizing acidic precipitation through the widespread use of greenroofs could help improve surface water quality. However, beneficial buffering effects by greenroofs are finite in nature as the media eventually loses its buffering capacity. A study conducted by Berghage et al. (2007) examined the pH buffering capacities of two greenroof media through an accelerated aging test. Daily sulfuric acid treatments were applied to a slate-based media and clay-based media and the pH response curves were measured. The test indicated that the buffering capacity could last approximately 13 and 19 years for the slate based and clay based media respectively. A greenroof management and maintenance plan would be required for long-term acid rain mitigation. The occasional application of lime to a greenroof may be required for recovery of pH buffering effects. During 2009 the pH of the runoff measured after passing through the treatment system was not significantly lower than the influent ($p = 0.74$). During 2010 the post-treatment water had a significantly lower pH ($p<0.001$). The engineered media is acidic in nature but did not lower the pH to a harmful level.

Conductivity measurements for both influent and effluent samples were significantly higher than the rainfall ($p<0.001$). Measurements from other greenroof studies have also reported higher conductivity levels in greenroof runoff (Berghage, 2009; Van Seters et al., 2009; Vander Linden and Stone, 2009). The conductivity reading is a measure of inorganic dissolved solids in the runoff. Conductivity readings are a general water quality measure, and can indicate the presence of anions such as nitrates, chloride and P, as well as cations such as metals. The composition of the greenroof media includes mineral aggregates and organic matter which contribute to these elevated levels. The greenroof runoff decreased in conductivity levels from 2009 to 2010. As the roof ages the inorganic dissolved solids leaching from the roof should decrease, an effect also reported in the Berghage (2009) study. The conductivity levels were significantly decreased after passing through the treatment system during the 2009 season ($p<0.001$). During 2010, the very high initial conductivity readings in the effluent, resulted in a significant increase ($p<0.001$) overall. Further water quality analysis on the initial samples would have elucidated the nature of the particulates washed from the media during this time.

Total dissolved solids (TDS) were significantly higher in both the influent and effluent measurements than measured in rainfall ($p<0.001$). As with the conductivity measurements, TDS levels were significantly decreased by the treatment system during 2009, and significantly increased during 2010 ($p<0.001$). The TDS levels were approximately three times higher than in the study by Vander Linden and Stone (2009). This study also measured much lower concentrations of P, which could partially explain the difference in TDS concentrations. Greenroof runoff will vary in TDS concentrations based on individual greenroof composition and fertilization practices.

The concentrations of SS that were measured in the effluent samples were significantly different than the influent concentrations ($p<0.001$). However, the median values were only 4.3 and 2.3 mg/L for the influent and effluent samples, respectively. Often the SS concentrations were at the method detection limit of the lab analysis, making thereby introducing possible uncertainty in the results. The treatment system was primarily designed for the removal of dissolved fractions of P. Treatment of areas with high SS concentrations, such as highway runoff, a filter component may be required to remove the particles before reaching the engineered media. In addition to acting as a pre-treatment, a filter would help prevent clogging, a common concern in water treatment systems (Hatt et al., 2006). With high concentrations of SS clogging an adsorbent bed, backwashing procedures would need to be performed. These procedures add complexity to a water treatment system and disrupt the adsorptive processes in the system (Thomas and Crittenden, 1998). The SS concentrations were very low in the runoff due to the use of components in the greenroof such as the root barrier. Low concentrations of SS were also measured in the studies by Van Seters et al. (2009) and Vander Linden and Stone (2009). These studies were conducted on much larger roofs, demonstrating the issue of clogging within a treatment system would be unlikely if integrated with a larger greenroof.

During the storm events of August 29, September 28 and September 29, 2009 further water quality testing was completed for metals. The tests were completed to characterize if the engineered media was releasing any metal by-products. Aluminum was the only metal to consistently increase in all post-treatment samples. The mean concentration increased from 26 µg/L in the greenroof runoff to 96 µg/L after passing through the treatment system. The post treatment aluminum concentrations did exceed the PWQO of 75 µg/L. The sample size was small so conclusions should be cautioned; however future use of this engineered media should be aware of potential metal release.

4.4: Isotherm Test for Engineered Media

The isotherm tests were completed to characterize the adsorptive capacity of the media. An isotherm test is conducted under ideal conditions for sorption processes and may not accurately represent the complexities of a field-scale operation. For example the isotherm test uses ultrapure water and is often completed at a time scale which may not represent the treatment times experienced within the field. Additionally, competing ions will be present in the stormwater, resulting in lowered P removal by the media. The greenroof runoff had a very noticeable yellow colour. This colour is evidence of inorganic and humic substances from the compost found in the greenroof media (Berghage, 2009). After passing through the treatment system, the water was consistently lighter and clearer in colour. Competitive adsorption may have been occurring with the engineered media adsorbing the negatively charged humic and fulvic acids. Selectivity of contaminant removal is difficult when treating natural water systems due to the range of contaminants (Faust and Aly, 1987).

Sorptive processes are strongly influenced by pH values. Several studies have reported increases in P removal as the pH becomes more acidic (Agyei et al., 2002; Arias et al., 2006; Hossain et al., 2010). With more acidic pH values positive charges accumulate upon the adsorbent surface, thereby favouring the adsorption of negatively charged phosphate ions. The slightly alkaline nature of the greenroof runoff may reduce this effect. A field-scale application also introduces the complicating factor of bacterial growth. Biofilm formation on the sorptive media may interfere with sorption processes (Chang et al., 2010b). Abiotic testing in batch experiments can determine whether nutrients are being removed via physicochemical processes rather than microbial processes (Hossain et al., 2010). However, field-scale applications are more difficult to control and there may be a microbial influence that contributes to nutrient uptake or release.

Estimate of P sorptive capacity becomes further complicated by differences in substrate characteristics such as grain size, shape, packing and porosity. These influences are amplified as the quantities of media are increased to field-scale (Faust and Aly, 1987). Furthermore, differential flow pathways are avoided in batch tests as the media is shaken allowing for potential maximum adsorbant coverage (Drizo et al., 1999; 2002). Differences in the grain size of the media may have resulted in large variation in sorption results. The media was coned and quartered in an attempt to minimize the variation in grain size, however the differences could not be completely controlled (Figure 47). The 2009 media had more variation in grain size than the 2010 media which complicates predicting the maximum sorption capacity.

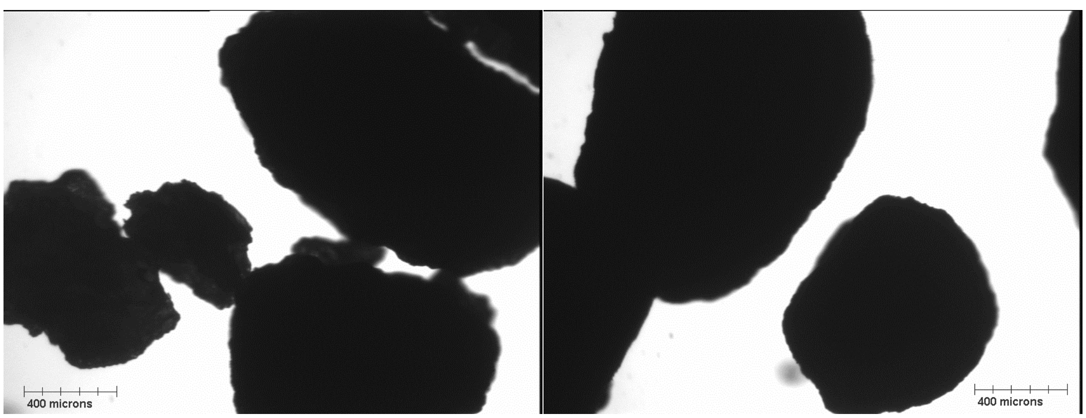


Figure 47: Representative photomicrographs of the engineered media. 2009 (left), 2010 (right).

The isotherm tests showed that the PSC for the media used in 2009 and 2010 were approximately 0.420 mg/g and 7.71 mg/g, respectively. These results are compared to P sorption studies from the literature in Table 12. Due to the wide range of materials and lack of standard batch experiment procedures, comparisons of results are difficult, however some general observations can be made. Cucarella and Renman (2009) note that several parameters of batch experiments can affect the PSC including: concentration of P solution, particle size of adsorbent, mass of adsorbent, material-to-solution ratio, pH of solution, length of contact time, degree of agitation and temperature. Caution should be used when applying batch experiment results to design estimates. Jenssen and Krogstad (2003) used a conservative design capacity for a sorptive lightweight aggregate that was 50% of the batch experiment results.

Table 12: P sorption capacities of various sorptive materials tested in batch experiments.

Study	Sorptive material	P sorption capacity (mg/g)	Notes
Camm, 2011	Oxide-coated pumice (version 1) Oxide-coated pumice (version 2)	0.420 7.71	
Boujelben et al., 2008	Synthetic iron coated sand Naturally iron oxide coated sand Iron oxide coated crushed brick	1.50 0.88 1.75	Max sorption at pH 5 Lower specific surface area
Mann and Bavor, 1993	Blast furnace slag Fly ash Granulated slag Gravel	0.42 0.26 0.16 0.03-0.05	BFS considered for use in a constructed wetland system
Del Bubba et al. 2003	13 sands ranging in Ca and Mg, grain size, porosity, bulk density	0.014 – 0.266	Sands with high Ca content (>60 mg/g) had greatly increased sorption
Chen et al., 2007	15 Chinese fly ashes	5.51-42.55	Positively correlated to Ca & Fe content, but negatively correlated to total Si and Al.
Cheung et al. 1994	Black oxide Fly ash (I/II) Red mud gypsum	0.89 1.19/3.08 5.07	Alkaline fly ash has better hydraulic conductivity making it more desirable for high infiltration rates
Liu et al., 2008	Zirconium oxide	29.71	Strong bonding indicated by lack of desorption
Ádám et al., 2007	Filtralite P® (expanded clay) Shell sand	2.50 9.60	High Ca concentration in shell sand
Xu et al., 2006	Bentonite Fly ash Furnace slag Sand (I-IV)	0.93 8.81 8.89 0.13-0.29	Furnace slag considered the best substrate for constructed wetland
Drizo et al., 1999	Bauxite Shale Burnt oil shale Limestone Zeolite LECA Fly ash	0.61 0.65 0.58 0.68 0.46 0.42 0.86	
Zhu et al., 1997	UTELITE™ Lehigh Cement VA LECA (I/II) Filtralite P®	3.46 2.91 0.16/0.57 1.39/2.21	Total metal content was correlated highest to P sorption capacity Ca was the metal ion with highest correlation to P sorption capacity

	Sand (I/II)	0.43/0.44	
Golder et al., 2006	Chrome hydroxide sludge	23.30	Low pH (3.0) favours adsorption
Khadhraoui et al., 2002	Ca-based sorbent (bentonite, calcium hydroxide, Yalloun coal)	10 - 17	Increased sorption with greater pore volumes
Lu et al., 2009	Fly ash	90-107.53	Removal associated with calcium content of fly ash
Sakadevan and Bavor, 1998	Blast furnace slag Steel furnace slag Zeolite	44.25 1.43 2.15	Adsorption had higher correlation with Al than Fe
Tanada et al., 2003	Aluminum oxide hydroxide	0.76	
Drizo et al., 2002	Electric arc furnace steel slag	3.93 (batch exp.) 1.35-2.35 (column exp.)	The material regenerated P sorption capacity after resting for 4 weeks
Hossain et al., 2010	50% sand, 20% limestone, 15% sawdust, 15% tire crumb	7.70 SRP 7.30 TP	
Zhou and Li, 2001	Calcareous soils	0.59-5.55	
Kostura et al., 2005	Amorphous slag (I/II) Crystalline slag	6.47/8.50 18.94	

4.5: Life Expectancy of Media

A variety of physical, chemical and biological factors complicate the extrapolation of batch experiment data to a field-scale application. The life expectancy calculations are based on ideal lab conditions and may vary according to fluctuations in pollutant concentrations and loads, residence times, treatment system design and hydrological conditions (Chang et al., 2010b). Furthermore, caution must be used when applying Langmuir estimates to field studies due to bias created in the model and experimental data (Drizo et al., 2002).

Based on the batch experiments, the q_{max} for the 2009 media was approximately 0.420 mg/g. Using the bulk density and volume of media, the total mass of media was 38 760 g. Therefore the theoretical maximum P load the system could adsorb was 16.28 g (0.420 mg/g x 38 760 g). The volume weighted mean concentration (VW_{MC}) of SRP in the greenroof runoff during 2009 was 0.769 mg/L. Throughout the entire monitoring period the average runoff producing event exported 160 L of stormwater. Assuming that during an entire year, 30 runoff events occur, then the total amount of SRP export is 3.69 g/yr ((30 x 160 x 0.769)/1000). Therefore, the life expectancy for SRP removal can be estimated as 4.4 yr (16.28/3.69). Substituting the UCL and LCL maximum sorption values, the lifespan is calculated to be 5.07 yr and 3.74 yr, respectively. Using the 2010 q_{max} of 7.71 mg/g and a mass of 60 040 g, the theoretical max P load was 463.9 g. Using the VW_{MC} for 2010 of 0.630 mg/L and for the above conditions the SRP export is 3.02 g/yr. The theoretical max life expectancy is then calculated to be 153 yr, with an UCL of 173 yr and LCL of 132 yr. These figures are based on the maximum theoretical adsorption and therefore an overestimation for field application. As was demonstrated by the field results, complete nutrient removal was not accomplished during the monitoring period, indicating the problems inherent in these predictions.

4.6: Implications to Stormwater Management

Stormwater management during the 1970s was primarily concerned with flood control, but has recently evolved to incorporate water quality treatment, protection of stream channels, maintenance of groundwater flow and the protection of aquatic habitat (Bradford and Gharabaghi, 2004). Over the past decade the treatment train approach has become favoured for stormwater management plans. The treatment train approach emphasizes a move from strict end-of-pipe controls, to treating stormwater from source, conveyance and finally at the end-of-pipe (OME, 2003b, Bradford and Gharabaghi, 2004). This shift to a treatment train approach reflects the increased desire to incorporate LID and sustainable urban drainage design.

Greenroofs are considered an important stormwater technology that can be used in the treatment train at the source control level. The widespread adaptation of greenroofs in North America has been previously limited by a lack of technical information on how to incorporate this practice into stormwater planning (Getter and Rowe, 2006; Banting et al., 2005). The recent research and prevalence of greenroofs has resulted in increased public awareness and a movement towards their increased use in urban development.

This study adds to the body of knowledge for greenroof research. As reported in this research and several other studies greenroofs are an important source control BMP for the reduction of stormwater volumes, the delay of peak flow rates and increase of lag times (VanderLinden and Stone, 2009; Van Seters et al., 2009; Carter and Rasmussen, 2006; VanWoert et al., 2005; Bliss et al., 2009; Moran, 2004). Urban areas are often space limited, making the implementation of SW BMPs very difficult. Greenroofs may be a more realistic retrofit option as they can be built onto existing buildings which have sufficient structural integrity. The use of greenroofs in urban areas will decrease the stormwater runoff volumes and reduce the overwhelming of sewer facility capacities.

Recent studies have shown greenroofs to be a source of P (VanderLinden and Stone, 2009; Van Seters et al., 2009; Bliss et al., 2009). The export of P from the roofs represents an issue to freshwater quality (Berndtsson et al., 2006). While the OME focuses on treatment of stormwater runoff for suspended solids, the sediment associated P may be removed but not the SRP load (OME, 2003b). The SRP has been considered more bioavailable and possibly presents a larger impact to freshwaters. The current study investigated the use of an adsorptive media designed to remove the SRP from the greenroof. In an EPA greenroof report it was recommended that any greenroof runoff be directed to another LID system such as a vegetated filter strip, rain garden or stormwater collection system (Berghage, 2009). The authors recommend that greenroof runoff not be directly discharged directly to any receiving waters without first being treated. For suburban or agricultural areas the greenroof runoff may simply be directed to a vegetated area or collected in rain barrels for reuse. Hardin and Wanielista (2007) also recommend that greenroof runoff is collected in cisterns for reuse. The authors suggest the use of a pollution control media to be placed underneath the growth media. New greenroof developments may benefit from the addition of an adsorptive media layer in the construction stage. While this would increase the structural load of the greenroof, the implementation may be simpler than a post-runoff system used in the current study. The use of adsorptive media within the greenroof matrix would prevent replacement of the media, however as the majority of P tends to leach from the growth substrate during the first years after construction, continual replacement may not be required (Camm, 2010; Van Seters et al., 2009).

The use of an additional layer of adsorptive media will increase the costs associated with greenroof construction. The cost of greenroof implementation has been a continued barrier to implementation. Carter and Keeler (2008) suggest greenroofs becoming more economically attractive with rising energy costs and increased public awareness of stormwater issues. A detailed cost-benefit analysis needs to be completed to assess the cost of implementing treatment systems to further treat greenroof runoff. Stormwater financing has been used in the past to help develop and maintain stormwater facilities (Cameron et al., 1999).

Emilsson et al. (2007) note the lack of regulations or guidelines for P content of greenroof substrates or fertilization in the industry standard German greenroof practices. For nitrogen, 5g N/year/m^2 is recommended as a nutrient requirement. The P demand of greenroof vegetation is less well understood and no guidelines have been set. In North America, the ASTM has created a Green Roof Task Force to create greenroof standards; however these standards are concerned with structural requirements, retention and water capture, and selection of vegetation.

The focus of this study was removing P from greenroof runoff using an adsorptive media. Greenroofs represent only a small fraction of the technologies employed for stormwater BMPs. The use of this media could be extended into a much larger stormwater context. Opportunities for improved water quality exist as stormwater moves through the treatment train. BMPs such as rain gardens, road swales and stormwater ponds could easily be optimized to improve P removal with the addition of a sorptive media (Chang et al., 2010a). A study by the Centre for Watershed Protection in Maryland compiled results from various studies on conventional stormwater treatment methods, to create a National Pollutant Removal Performance Database (CWP, 2007). The findings for TP and SRP are presented in Table 13. Most of these practices exhibited very low removal rates for SRP, with dry ponds, bioretention and open channels demonstrating a net export of SRP. There represents a clear need to address these water quality issues for this range of SWBMPs. The use of sorptive media is one possibility for the improvement of these practices.

Table 13: Range of BMP median P removal performances (CWP, 2007).

Median Treatment Performance **n = number of studies**	**Dry Ponds**	**Wet Ponds**	**Wetland**	**Filtering Practices**[1]	**Bioretention**	**Infiltration Practices**[2]	**Open Channels**[3]
TP %	20 n = 10	52 n = 45	48 n = 37	59 n = 17	5 n = 10	65 n = 8	24 n = 16
SRP %	-3 n = 6	64 n = 28	25 n = 26	3 n = 7	-9 n = 5	84 n = 4	-38 n = 14

1. organic filters and sand filters
2. infiltration trenches and porous pavement
3. grass channels and dry swales

Chapter 5: Conclusions and Further Research

The main objective of this thesis was to investigate the treatment efficiency of a sorptive material for the removal of P from greenroof runoff. The study results are intended to provide information that will assist the planning and management of greenroof policy, construction and optimization of this important source control BMP for water quantity and quality control. Conclusions and recommendations for further research are presented in the following sections.

5.1: Water Quantity

- 68 rainfall events were monitored from June 2009 – October 2009, and May 2010 – August 2010.
- Rainfall events ranged from 1.0 mm to 56.8 mm. The majority of the events had <2 year return periods and the largest event had a return period of approximately 35 years.
- The greenroof retained 41.5% (144 mm of 347 mm) of runoff volumes for 39 rainfall events during the 2009 monitoring period. During 2010, 53.3% (213 of 400 mm) of precipitation was retained.
- The highest monthly retention rates of 76.5% occurred in August, 2010 and the lowest of -14.7% occurred in October, 2009. The net export of runoff during October is attributed to low evapotranspiration rates and an underestimation of rainfall depths.
- The mean retention for storms <5 mm was 54.0%, with a mean absolute retention of 1.7 mm. Storms >5 mm had a mean retention of 57.3% with a mean absolute retention of 7.1 mm.
- For the entire monitoring period, events with less than 48 PDH had a mean retention of 34.0%. For events with greater than or equal to 48 PDH had a mean retention of 77.7%.
- The greenroof was effective at extending the lag time. The median lag time for all events was 35.0 minutes. The lag time effect of the greenroof was strongly influenced by the intensity of the rainfall event. High intensity events resulted in the largest decrease in lag time.

5.2: Water Quality

2009 Media

- The treatment system reduced the SRP volume-weighted mean concentration from 0.769 mg/L to 0.523 mg/L. Total UAL of 118 mg/m^2 was reduced to 80.2 mg/m^2 (32.0% load reduction).
- The treatment system reduced the TP volume-weighted mean concentration from 1.30 mg/L to 0.973 mg/L. Total UAL of 189 mg/m^2 was reduced to 141 mg/m^2 (25.4% load reduction).
- There was a downward trend for treatment efficiency over the monitoring period.

- The mean pH of the greenroof runoff was 7.90, after passing through the treatment system the pH decreased to 7.82.

- The mean conductivity of the greenroof runoff was 611 µS/cm, after passing through the treatment system the conductivity decreased 11% to 544 µS/cm.

- The volume-weighted mean concentration of TDS was is 420 mg/L for both pre and post-treatment samples.

- The suspended solids concentrations for the greenroof runoff averaged 7.6 mg/L and 3.5 mg/L after passing through the treatment system.

- The grain size distribution did not significantly change between the pre and post treatment samples for the 4 storm events analyzed. There was also no indication of a change in the D_{90} throughout the hydrograph.

2010 Media

- The treatment system reduced the SRP volume-weighted mean concentration from 0.630 mg/L to 0.110 mg/L. Total UAL of 76.1 mg/m^2 was reduced to 13.4 mg/m^2 (82.4% load reduction).

- The treatment system reduced the TP volume-weighted mean concentration from 1.18 mg/L to 0.158 mg/L. Total UAL of 142 mg/m^2 was reduced 19.1 mg/m^2 (86.6% load reduction).

- The treatment efficiency of the engineered media did not show a decreasing trend in treatment efficiency over the monitoring period

- The mean pH of the greenroof runoff during 2010 was 8.22 and 7.56 after passing through the treatment system.

- The mean conductivity of the greenroof runoff was 539 µS/cm, after passing through the treatment system the conductivity increased to a mean value of 1618 µS/cm. The new media initially produced high conductivity readings, as particulates were washed off the media. The conductivity readings rapidly decreased after the first few storms.

- The volume-weighted mean concentration of TDS was 389 mg/L and 857 mg/L for influent and effluent samples, respectively.

5.3: Further research

The present study focused on characterizing the water quality of greenroof runoff and the use of an engineered media to reduce P export. The study raised a number of questions concerning the design and application of the treatment system. The issue of greenroof stormwater quality requires further research. The following recommendations are based on findings in the literature and the current study.

- Previous studies have reported initially high loads of P export followed by decreased loading as the greenroof ages. The monitoring for this study occurred shortly after construction of the greenroof and planting. Further research needs to be completed on greenroofs that have aged for several years. Decreasing P export would have implications for the utility of treatment systems in older greenroof systems.

- The influence of greenroof vegetation on P behavior needs to be investigated. How much P can be expected to be used by the vegetation during growing season and consequently is there a release of this P during senescence? Understanding this behavior will help control P export from the greenroof.

- The P removal performance of the engineered media decreased over the monitoring period. The variable stage depth within the rain barrel for each storm event makes it difficult to discern how effectively the entire volume of media was utilized. The design of the treatment system needs to ensure entire usage of the media for each runoff event.

- Performance metrics need to be developed to identify the cost per unit mass of P removed. Practical and economic considerations need to be given to the installation, potential maintenance and removal of a treatment system.

- Pilot-scale studies using sorptive media incorporated into the greenroof matrix could examine the treatment efficiency under a range of controlled hydrological conditions. The lifespan of the sorptive media could be tested quickly using rainfall simulations.

- Further isotherm testing of engineered media would benefit from a closer representation of field-scale conditions. Simulating the composition of greenroof runoff for the batch experiments would help characterize the adsorptive behavior with effects of pH and competing ions.

- Key parameters affecting P removal need to be standardized for each batch of media to allow for treatment predictability. These parameters include: grain size, surface chemistry, bulk density, hydraulic conductivity and porosity.

- Microbial influence on nutrient removal needs to be characterized for future treatment systems. Batch testing of sorptive materials with bacterial dosing could characterize any microbial influence on nutrient removal.

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Appendix A: Water Quality

Table A1: Soluble reactive phosphorus concentrations for the 2009 monitoring period.

	Influent (mg/L)						**Effluent (mg/L)**					
Date	**N total**	**Mean**	**Minimum**	**Median**	**Maximum**	**EMC**	**N total**	**Mean**	**Minimum**	**Median**	**Maximum**	**EMC**
11-Jul	19	**0.824**	0.708	0.815	0.919	0.803	22	**0.154**	0	0	0.482	0.356
23-Jul	24	**1.23**	1.14	1.23	1.35	1.23	24	**0.0303**	0.00524	0.0352	0.0452	0.0216
29-Jul	24	**1.25**	1.08	1.24	1.44	1.18	9	**0.134**	0.0797	0.122	0.217	0.119
09-Aug	23	**0.769**	0.244	0.92	1.05	0.868	22	**0.196**	0.0300	0.213	0.368	0.0405
11-Aug	23	**0.91**	0.745	0.925	1.02	0.896	23	**0.503**	0.213	0.525	0.763	0.408
20-Aug	16	**0.805**	0.681	0.797	0.954	0.799	17	**0.598**	0.149	0.611	0.816	0.669
29-Aug	23	**0.893**	0.758	0.88	1.03	0.919	23	**0.444**	0.248	0.491	0.557	0.324
28-Sep	24	**0.344**	0.274	0.343	0.381	0.354	24	**0.202**	0.172	0.197	0.243	0.193
29-Sep	24	**0.466**	0.362	0.486	0.564	0.476	24	**0.286**	0.263	0.287	0.301	0.287
02-Oct	24	**0.491**	0.444	0.498	0.527	0.485	24	**0.372**	0.252	0.381	0.429	0.373
09-Oct	24	**0.755**	0.588	0.796	0.890	0.679	24	**0.517**	0.404	0.530	0.596	0.481
23-Oct	23	**0.603**	0.546	0.581	0.917	0.601	23	**0.577**	0.549	0.575	0.612	0.567
28-Oct	21	**0.675**	0.606	0.667	1.04	0.683	24	**0.614**	0.526	0.619	0.69	0.623

Table A2: Soluble reactive phosphorus summary statistics for 2009 monitoring period.

	Influent (mg/L)						
Date	**N total**	**Mean**	**Standard Deviation**	**SE of mean**	**Lower 95% CI of Mean**	**Upper 95% CI of Mean**	**Median**
11-Jul	19	0.824	0.06	0.01	0.794	0.853	0.815
23-Jul	24	1.23	0.04	0.009	1.22	1.25	1.23
29-Jul	24	1.25	0.08	0.02	1.21	1.28	1.24
09-Aug	23	0.769	0.3	0.07	0.632	0.906	0.92
11-Aug	23	0.910	0.08	0.02	0.876	0.944	0.925
20-Aug	16	0.805	0.07	0.02	0.769	0.841	0.797
29-Aug	23	0.893	0.05	0.01	0.869	0.917	0.88
28-Sep	24	0.344	0.03	0.006	0.333	0.356	0.343
29-Sep	24	0.466	0.06	0.01	0.439	0.493	0.486
02-Oct	24	0.491	0.02	0.005	0.480	0.501	0.498
09-Oct	24	0.755	0.1	0.02	0.712	0.799	0.796
23-Oct	23	0.603	0.08	0.02	0.569	0.638	0.581
28-Oct	21	0.675	0.09	0.02	0.633	0.717	0.667

	Effluent (mg/L)						
Date	**N total**	**Mean**	**Standard Deviation**	**SE of mean**	**Lower 95% CI of Mean**	**Upper 95% CI of Mean**	**Median**
11-Jul	19	0.154	0.2	0.04	0.0658	0.242	0
23-Jul	24	0.0303	0.01	0.002	0.0252	0.0353	0.0352
29-Jul	24	0.101	0.07	0.02	0.0539	0.147	0.104
09-Aug	23	0.196	0.1	0.02	0.151	0.241	0.213
11-Aug	23	0.503	0.2	0.03	0.431	0.575	0.525
20-Aug	16	0.598	0.2	0.04	0.507	0.689	0.611
29-Aug	23	0.444	0.1	0.02	0.395	0.493	0.491
28-Sep	24	0.202	0.02	0.004	0.193	0.210	0.197
29-Sep	24	0.286	0.01	0.002	0.282	0.291	0.287
02-Oct	24	0.372	0.04	0.009	0.353	0.390	0.381
09-Oct	24	0.517	0.05	0.01	0.494	0.540	0.530
23-Oct	23	0.577	0.01	0.003	0.570	0.584	0.575
28-Oct	21	0.614	0.04	0.009	0.595	0.632	0.619

Table A3: Total phosphorus concentrations for the 2009 monitoring period.

	Influent (mg/L)						Effluent (mg/L)					
Date	**N total**	**Mean**	**Minimum**	**Median**	**Maximum**	**EMC**	**N total**	**Mean**	**Minimum**	**Median**	**Maximum**	**EMC**
11-Jul	18	**1.82**	1.50	1.79	2.27	1.81	22	**0.708**	0.0501	0.114	1.90	1.41
23-Jul	24	**1.74**	1.52	1.74	2.05	1.72	24	**0.0484**	0.0209	0.052	0.0724	0.0375
29-Jul	23	**2.01**	1.66	1.92	2.89	2.09	12	**0.208**	0.102	0.19	0.343	0.193
09-Aug	22	**1.39**	1.12	1.42	1.78	1.37	22	**0.717**	0.411	0.701	1.17	0.301
11-Aug	23	**1.41**	0.959	1.40	1.64	1.36	23	**0.693**	0.391	0.693	0.961	0.487
20-Aug	16	**1.24**	0.867	1.31	1.47	1.26	17	**0.939**	0.403	0.966	1.28	1.04
29-Aug	23	**1.73**	1.35	1.70	2.16	1.83	18	**1.03**	0.621	1.08	1.47	0.813
28-Sep	24	**1.21**	0.799	1.19	1.85	1.27	24	**0.586**	0.425	0.583	0.865	0.562
29-Sep	23	**0.910**	0.726	0.881	1.36	0.908	23	**0.564**	0.425	0.571	0.673	0.552
02-Oct	20	**0.630**	0.563	0.627	0.773	0.614	20	**0.569**	0.434	0.571	0.628	0.573
09-Oct	24	**0.904**	0.816	0.902	0.988	0.876	24	**0.791**	0.700	0.811	0.876	0.754

Table 4: Total phosphorus summary statistics for the 2009 monitoring period.

	Influent (mg/L)						
Date	**N total**	**Mean**	**Standard Deviation**	**SE of mean**	**Lower 95% CI of Mean**	**Upper 95% CI of Mean**	**Median**
11-Jul	18	1.82	0.2	0.06	1.70	1.94	1.79
23-Jul	24	1.74	0.1	0.02	1.69	1.79	1.74
29-Jul	23	2.01	0.3	0.06	1.88	2.15	1.92
09-Aug	22	1.39	0.2	0.03	1.32	1.47	1.42
11-Aug	23	1.41	0.1	0.03	1.36	1.47	1.40
20-Aug	16	1.24	0.2	0.04	1.15	1.33	1.31
29-Aug	23	1.73	0.2	0.05	1.64	1.83	1.70
28-Sep	24	1.21	0.3	0.05	1.09	1.32	1.19
29-Sep	23	0.910	0.1	0.03	0.848	0.972	0.881
02-Oct	20	0.630	0.05	0.01	0.608	0.653	0.627
09-Oct	24	0.904	0.05	0.01	0.881	0.927	0.902

	Effluent (mg/L)						
Date	**N total**	**Mean**	**Standard Deviation**	**SE of mean**	**Lower 95% CI of Mean**	**Upper 95% CI of Mean**	**Median**
11-Jul	22	0.708	0.7	0.1	0.397	1.02	0.114
23-Jul	24	0.0484	0.01	0.003	0.0425	0.0543	0.052
29-Jul	12	0.208	0.08	0.02	0.156	0.260	0.190
09-Aug	22	0.717	0.2	0.05	0.618	0.817	0.701
11-Aug	23	0.693	0.2	0.04	0.619	0.767	0.693
20-Aug	17	0.939	0.2	0.05	0.828	1.05	0.966
29-Aug	23	1.03	0.2	0.05	0.915	1.15	1.08
28-Sep	24	0.586	0.09	0.02	0.546	0.626	0.583
29-Sep	24	0.564	0.07	0.01	0.536	0.593	0.571
02-Oct	24	0.569	0.04	0.01	0.549	0.590	0.571
09-Oct	24	0.791	0.06	0.01	0.766	0.816	0.811

Table A5: Soluble reactive phosphorus concentrations for the 2010 monitoring period.

	Influent (mg/L)						Effluent (mg/L)					
Date	**N total**	**Mean**	**Minimum**	**Median**	**Maximum**	**EMC**	**N total**	**Mean**	**Minimum**	**Median**	**Maximum**	**EMC**
07-May	1	**0.851**	0.851	0.851	0.851	0.851	12	**0.0337**	0.0134	0.0323	0.0583	0.0364
13-May	1	**0.884**	0.884	0.884	0.884	0.884	12	**0.251**	0.0939	0.260	0.318	0.263
02-Jun	12	**1.10**	0.714	0.825	2.90	1.27	12	**0.0576**	0.0247	0.0539	0.103	0.0647
03-Jun	12	**0.678**	0.631	0.674	0.773	0.672	12	**0.117**	0.0983	0.110	0.177	0.128
12-Jun	12	**0.985**	0.651	0.711	3.93	1.07	12	**0.0933**	0.0467	0.0745	0.257	0.101
16-Jun	12	**0.644**	0.598	0.633	0.728	0.649	12	**0.113**	0.0971	0.103	0.144	0.121
22-Jun	12	**0.557**	0.488	0.533	0.716	0.566	12	**0.082**	0.0606	0.0659	0.231	0.0892
24-Jun	12	**0.531**	0.497	0.517	0.613	0.528	12	**0.128**	0.0941	0.113	0.205	0.142
26-Jun	12	**1.13**	0.610	0.669	6.16	0.814	12	**0.103**	0.0500	0.0651	0.280	0.0854
24-Jul	12	**0.457**	0.298	0.458	0.681	0.438	12	**0.0752**	0.0299	0.0493	0.185	0.0783
15-Aug	4	**0.244**	0.223	0.240	0.273	-	12	**0.0682**	0.0324	0.0664	0.129	-
21-Aug	12	**0.222**	0.211	0.223	0.228	0.218	12	**0.0598**	0.0453	0.0585	0.0757	0.0582

Table A6: Soluble reactive phosphorus summary statistics for 2010 monitoring period.

	Influent (mg/L)						
Date	**N total**	**Mean**	**Standard Deviation**	**SE of mean**	**Lower 95% CI of Mean**	**Upper 95% CI of Mean**	**Median**
07-May	1	0.851	--	--	--	--	0.851
13-May	1	0.884	--	--	--	--	0.884
02-Jun	12	1.10	0.7	0.2	0.675	1.53	0.825
03-Jun	12	0.678	0.04	0.01	0.655	0.701	0.674
12-Jun	12	0.985	0.9	0.3	0.395	1.57	0.711
16-Jun	12	0.644	0.04	0.01	0.617	0.671	0.633
22-Jun	12	0.557	0.07	0.02	0.511	0.603	0.533
24-Jun	12	0.531	0.04	0.01	0.507	0.554	0.517
26-Jun	12	1.13	2	0.5	0.118	2.14	0.669
24-Jul	12	0.457	0.08	0.02	0.403	0.511	0.458
15-Aug	4	0.244	0.02	0.01	0.211	0.277	0.240
21-Aug	12	0.222	0.006	0.002	0.218	0.225	0.223

	Effluent (mg/L)						
Date	**N total**	**Mean**	**Standard Deviation**	**SE of mean**	**Lower 95% CI of Mean**	**Upper 95% CI of Mean**	**Median**
07-May	12	0.0337	0.01	0.004	0.0249	0.0425	0.0323
13-May	12	0.251	0.06	0.02	0.213	0.289	0.260
02-Jun	12	0.0576	0.02	0.006	0.0451	0.070	0.0539
03-Jun	12	0.117	0.02	0.006	0.103	0.131	0.110
12-Jun	12	0.0933	0.06	0.02	0.0556	0.131	0.0745
16-Jun	12	0.113	0.02	0.005	0.101	0.125	0.103
22-Jun	12	0.082	0.05	0.01	0.0515	0.113	0.0659
24-Jun	12	0.128	0.04	0.01	0.104	0.153	0.113
26-Jun	12	0.103	0.08	0.02	0.0541	0.152	0.0651
24-Jul	12	0.0752	0.05	0.01	0.042	0.108	0.0493
15-Aug	12	0.0682	0.03	0.009	0.0492	0.0872	0.0664
21-Aug	12	0.0598	0.008	0.002	0.0545	0.0652	0.0585

Table A7: Total phosphorus concentrations for the 2010 monitoring period.

	Influent (mg/L)						Effluent (mg/L)					
Date	**N total**	**Mean**	**Minimum**	**Median**	**Maximum**	**EMC**	**N total**	**Mean**	**Minimum**	**Median**	**Maximum**	**EMC**
07-May	1	**1.57**	1.57	1.57	1.57	1.57	12	**0.0804**	0.0482	0.0700	0.166	0.0815
13-May	1	**1.63**	1.63	1.63	1.63	1.63	12	**0.298**	0.0968	0.312	0.373	0.311
02-Jun	12	**1.84**	1.28	1.41	5.43	1.99	12	**0.105**	0.0628	0.0802	0.343	0.116
03-Jun	12	**1.45**	1.31	1.41	1.64	1.46	12	**0.159**	0.130	0.144	0.232	0.171
12-Jun	12	**1.38**	0.940	1.00	5.44	1.56	12	**0.171**	0.110	0.173	0.284	0.169
16-Jun	12	**0.909**	0.845	0.902	1.01	0.892	12	**0.137**	0.107	0.125	0.184	0.158
22-Jun	12	**1.16**	0.904	1.12	1.73	1.21	12	**0.130**	0.0880	0.113	0.308	0.157
24-Jun	12	**1.21**	1.03	1.17	1.55	1.24	12	**0.201**	0.159	0.181	0.302	0.218
26-Jun	12	**2.00**	0.952	1.24	10.2	1.85	12	**0.143**	0.0823	0.109	0.314	0.155
24-Jul	12	**0.909**	0.810	0.876	1.30	0.864	12	**0.112**	0.0596	0.0842	0.250	0.114
21-Aug	12	**0.250**	0.229	0.252	0.255	0.244	12	**0.0802**	0.0526	0.0781	0.104	0.076

Table A8: Total phosphorus summary statistics for the 2010 monitoring period.

	Influent (mg/L)						
Date	**N total**	**Mean**	**Standard Deviation**	**SE of mean**	**Lower 95% CI of Mean**	**Upper 95% CI of Mean**	**Median**
07-May	1	1.57	--	--	--	--	1.57
13-May	1	1.63	--	--	--	--	1.63
02-Jun	12	1.84	1	0.3	1.10	2.59	1.41
03-Jun	12	1.45	0.1	0.03	1.38	1.53	1.41
12-Jun	12	1.38	1	0.4	0.568	2.19	1.00
16-Jun	12	0.909	0.04	0.01	0.881	0.938	0.902
22-Jun	12	1.16	0.2	0.06	1.03	1.29	1.12
24-Jun	12	1.21	0.1	0.04	1.11	1.30	1.17
26-Jun	12	2.00	2	0.7	0.349	3.66	1.24
24-Jul	12	0.909	0.1	0.04	0.828	0.990	0.876
21-Aug	12	0.250	0.007	0.002	0.245	0.254	0.252

	Effluent (mg/L)						
Date	**N total**	**Mean**	**Standard Deviation**	**SE of mean**	**Lower 95% CI of Mean**	**Upper 95% CI of Mean**	**Median**
07-May	12	0.0804	0.03	0.01	0.0586	0.102	0.0700
13-May	12	0.298	0.07	0.02	0.252	0.345	0.312
02-Jun	12	0.105	0.08	0.02	0.0563	0.154	0.0802
03-Jun	12	0.159	0.03	0.009	0.139	0.179	0.144
12-Jun	12	0.171	0.05	0.01	0.138	0.204	0.173
16-Jun	12	0.137	0.02	0.007	0.121	0.153	0.125
22-Jun	12	0.130	0.06	0.02	0.0935	0.167	0.113
24-Jun	12	0.201	0.05	0.01	0.170	0.233	0.181
26-Jun	12	0.143	0.08	0.02	0.0934	0.193	0.109
24-Jul	12	0.112	0.06	0.02	0.0716	0.153	0.0842
21-Aug	12	0.0802	0.01	0.004	0.0722	0.0882	0.0781

Table A9: Conductivity measurements for the 2009 and 2010 monitoring periods.

	Influent (µS/cm)						Effluent (µS/cm)					
2009 Date	**N total**	**Mean**	**Std. Dev.**	**Minimum**	**Median**	**Maximum**	**N total**	**Mean**	**Std. Dev.**	**Minimum**	**Median**	**Maximum**
11-Jul	24	**644**	31	596	639	718	24	**442**	75	287	476	506
23-Jul	19	**942**	57	893	934	1160	12	**802**	37	744	824	845
29-Jul	23	**706**	101	515	745	800	22	**603**	56	521	604	683
09-Aug	24	**955**	60	771	971	1020	24	**855**	66	712	880	924
11-Aug	24	**609**	86	452	615	842	24	**605**	97	549	563	982
20-Aug	23	**732**	53	671	714	878	22	**700**	18	666	705	721
29-Aug	24	**236**	21	175	241	263	24	**246**	8.1	232	248	266
28-Sep	23	**293**	26	270	287	395	23	**273**	7.6	262	272	291
29-Sep	23	**375**	19	344	378	408	24	**351**	14	330	348	386
02-Oct	21	**681**	50	605	703	739	24	**590**	22	553	594	625
09-Oct	24	**479**	65	419	466	747	24	**547**	101	471	493	764
23-Oct	22	**678**	26	637	678	727	16	**537**	14	515	538	567

	Influent (µS/cm)						Effluent (µS/cm)					
2010 Date	**N total**	**Mean**	**Std. Dev.**	**Minimum**	**Median**	**Maximum**	**N total**	**Mean**	**Std. Dev.**	**Minimum**	**Median**	**Maximum**
07-May	**--**	**--**	--	--	--	--	12	**10200**	5200	5940	7920	20200
13-May	**1**	**460**	--	460	460	460	12	**2640**	1400	1270	2100	5110
02-Jun	**12**	**337**	160	210	269	774	12	**943**	400	621	780	1970
03-Jun	**12**	**378**	32	304	384	410	12	**1020**	610	609	823	2790
12-Jun	**12**	**574**	55	468	579	661	12	**529**	46	479	510	600
16-Jun	**12**	**724**	21	696	722	771	12	**568**	15	555	564	613
22-Jun	**12**	**503**	41	440	491	594	12	**579**	33	550	563	646
24-Jun	**12**	**686**	43	643	668	792	12	**643**	18	610	652	657
26-Jun	**12**	**751**	29	692	763	782	12	**680**	11	664	680	697
24-Jul	**12**	**424**	30	347	424	459	12	**469**	44	363	471	512
21-Aug	**12**	**543**	38	437	549	587	12	**462**	5.4	453	463	472

Table A10: Total dissolved solids measurements for the 2009 and 2010 monitoring periods.

	Influent (mg/L)						Effluent (mg/L)					
2009 Date	**N total**	**Mean**	**Std. Dev.**	**Minimum**	**Median**	**Maximum**	**N total**	**Mean**	**Std. Dev.**	**Minimum**	**Median**	**Maximum**
11-Jul	24	**451**	22	418	448	503	24	**310**	52	201	334	355
23-Jul	19	**661**	40	626	655	816	12	**562**	26	522	578	592
29-Jul	23	**495**	71	361	522	561	22	**423**	39	365	424	479
09-Aug	24	**670**	42	541	681	715	24	**599**	47	499	617	648
11-Aug	24	**427**	61	317	431	590	24	**424**	68	385	395	688
20-Aug	23	**513**	37	470	501	616	22	**491**	13	467	494	505
29-Aug	24	**165**	14	123	169	184	24	**172**	5.6	163	174	186
28-Sep	23	**206**	18	189	201	277	23	**192**	5.3	184	191	204
29-Sep	23	**263**	14	241	265	286	24	**246**	10	231	244	271
02-Oct	21	**477**	35	424	493	518	24	**414**	15	388	417	438
09-Oct	24	**330**	45	288	321	514	24	**377**	69	324	339	526
23-Oct	22	**467**	18	438	466	500	16	**369**	9.8	354	370	390

	Influent (mg/L)						Effluent (mg/L)					
2010 Date	**N total**	**Mean**	**Std. Dev.**	**Minimum**	**Median**	**Maximum**	**N total**	**Mean**	**Std. Dev.**	**Minimum**	**Median**	**Maximum**
07-May	0	--	--	--	--	--	12	**7110**	3600	4140	5520	14100
13-May	1	**325**	--	325	325	325	12	**1870**	1020	897	1490	3610
02-Jun	12	**239**	110	149	191	548	12	**668**	280	440	553	1400
03-Jun	12	**268**	23	215	272	290	12	**723**	430	431	583	1980
12-Jun	12	**402**	39	327	405	462	12	**370**	32	335	357	420
16-Jun	12	**507**	15	487	505	539	12	**398**	11	388	395	429
22-Jun	12	**348**	28	305	340	411	12	**401**	23	381	390	447
24-Jun	12	**472**	29	442	459	545	12	**442**	12	420	449	452
26-Jun	12	**517**	20	476	525	538	12	**468**	7.7	457	468	480
24-Jul	12	**290**	21	237	290	314	12	**320**	30	248	322	350
21-Aug	12	**383**	27	308	387	414	12	**326**	3.7	320	327	333

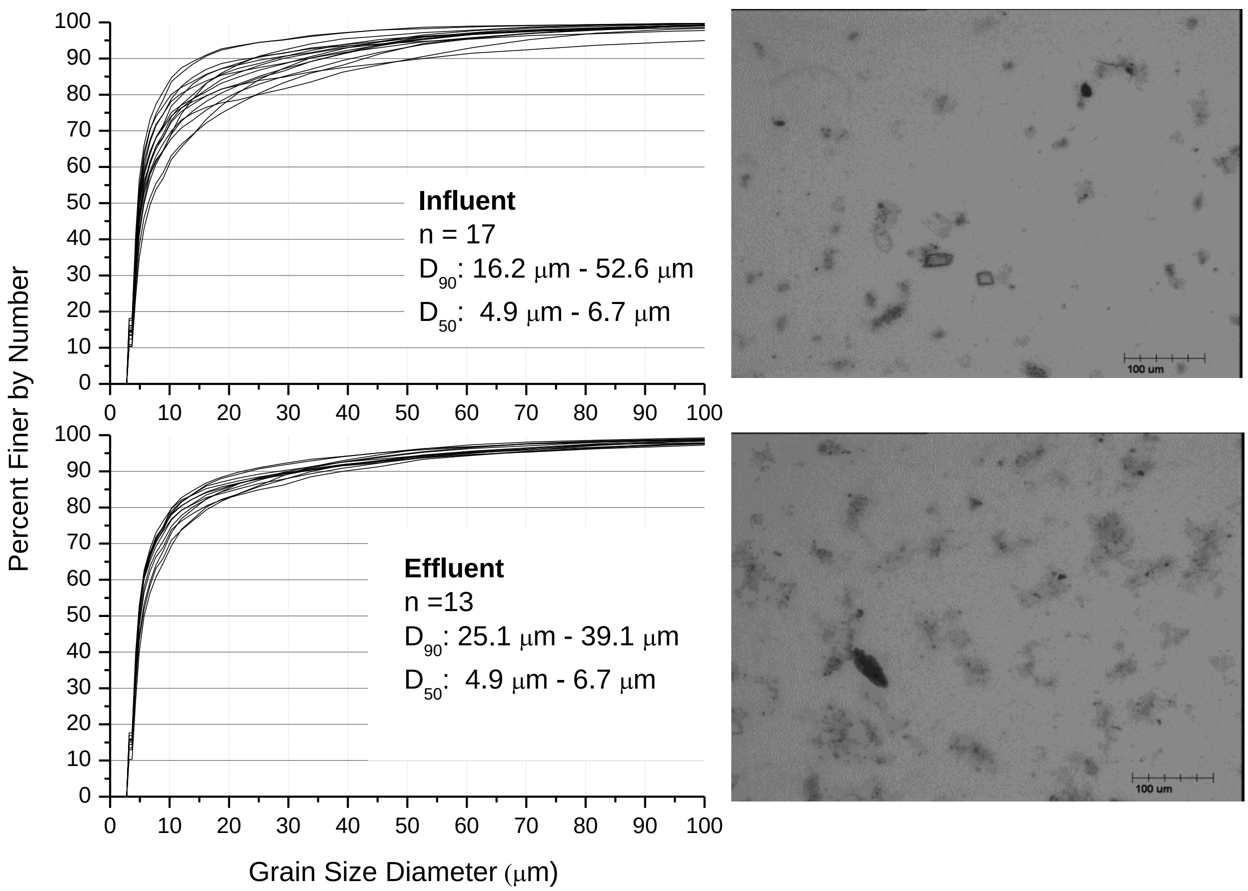
Table A12: Suspended sediment concentrations for the 2009 monitoring period.

	Influent (mg/L)						Effluent (mg/L)					
2009 Date	**N total**	**Mean**	**Std. Dev.**	**Minimum**	**Median**	**Maximum**	**N total**	**Mean**	**Std. Dev.**	**Minimum**	**Median**	**Maximum**
11-Jul	5	**10.3**	18	0	0	40.9	5	**3.68**	5.0	0	0	9.40
23-Jul	8	**5.00**	7.6	0	0	20.0	8	**6.25**	7.4	0	5	20.0
29-Jul	6	**16.0**	13	8.70	9.25	42.4	7	**10.5**	8.1	0	9.20	27.2
09-Aug	9	**7.23**	7.0	0	5.00	24.7	9	**4.76**	1.7	2.40	4.80	7.10
11-Aug	9	**8.34**	2.9	4.20	8.20	13.2	9	**3.89**	1.6	0	4.70	4.90
20-Aug	9	**12.1**	13	0	6.50	36.1	6	**7.92**	8.4	2.40	4.65	24.2
29-Aug	9	**4.98**	5.8	0	3.20	14.1	9	**1.71**	3.1	0	0	9.20
28-Sep	11	**3.64**	3.3	0	3.80	8.20	11	**1.50**	2.1	0	0	6.60
29-Sep	9	**4.37**	5.0	0	3.90	15.8	9	**1.80**	1.0	0	2.30	2.40
02-Oct	9	**8.60**	9.7	2.60	3.60	30.2	9	**0.289**	0.87	0	0	2.60
09-Oct	9	**8.11**	7.3	0	7.40	23.0	9	**0.289**	0.87	0	0	2.60
23-Oct	9	**2.52**	2.2	0	2.90	5.90	9	**1.68**	1.4	0	0	4.10

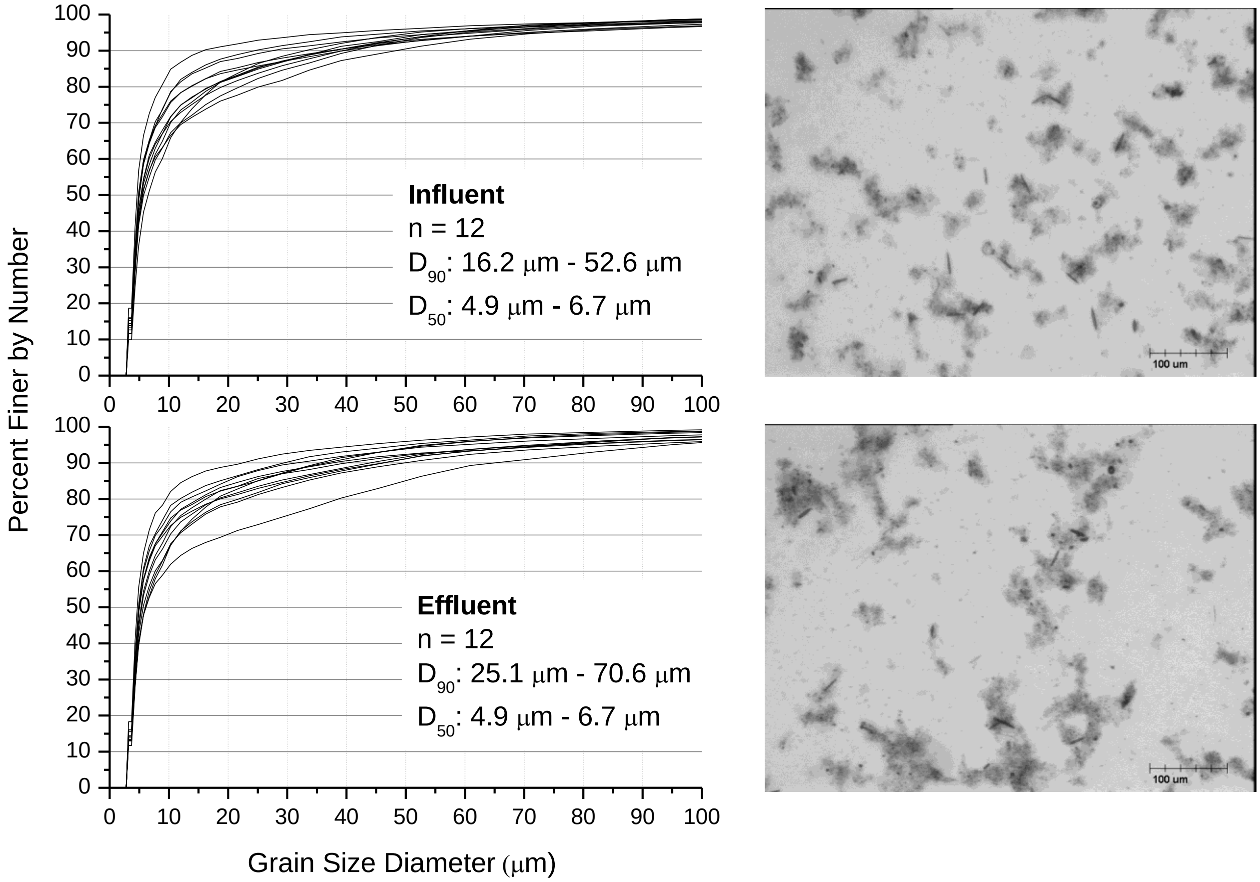
Table A13: pH measurements for the 2009 and 2010 monitoring periods.

	Influent						Effluent					
2009 Date	**N total**	**Mean**	**Std. Dev.**	**Minimum**	**Median**	**Maximum**	**N total**	**Mean**	**Std. Dev.**	**Minimum**	**Median**	**Maximum**
11-Jul	24	**8.02**	0.19	7.51	8.05	8.27	24	**7.54**	0.11	7.37	7.53	7.73
23-Jul	19	**8.23**	0.17	7.90	8.20	8.59	12	**8.41**	0.22	8.10	8.38	8.93
29-Jul	24	**7.84**	0.12	7.64	7.87	8.04	24	**7.56**	0.15	7.23	7.57	7.93
09-Aug	24	**8.16**	0.16	7.54	8.19	8.31	24	**8.13**	0.071	7.95	8.16	8.23
11-Aug	24	**7.81**	0.13	7.50	7.83	8.06	24	**7.73**	0.22	7.44	7.68	8.42
20-Aug	16	**7.99**	0.19	7.35	8.06	8.13	16	**7.91**	0.092	7.68	7.92	8.06
29-Aug	24	**6.98**	0.16	6.61	6.95	7.27	23	**7.42**	0.10	7.25	7.42	7.63
28-Sep	24	**7.29**	0.20	6.91	7.29	7.81	23	**7.10**	0.24	6.32	7.08	7.57
29-Sep	23	**7.43**	0.24	7.05	7.44	7.80	22	**7.55**	0.21	7.13	7.58	7.86
02-Oct	21	**7.89**	0.24	6.94	7.92	8.10	24	**7.90**	0.20	7.38	7.95	8.32
09-Oct	24	**8.07**	0.19	7.77	8.03	8.41	24	**8.23**	0.21	7.77	8.33	8.49
23-Oct	23	**8.75**	0.17	8.48	8.68	9.02	18	**8.41**	0.19	7.97	8.40	8.79

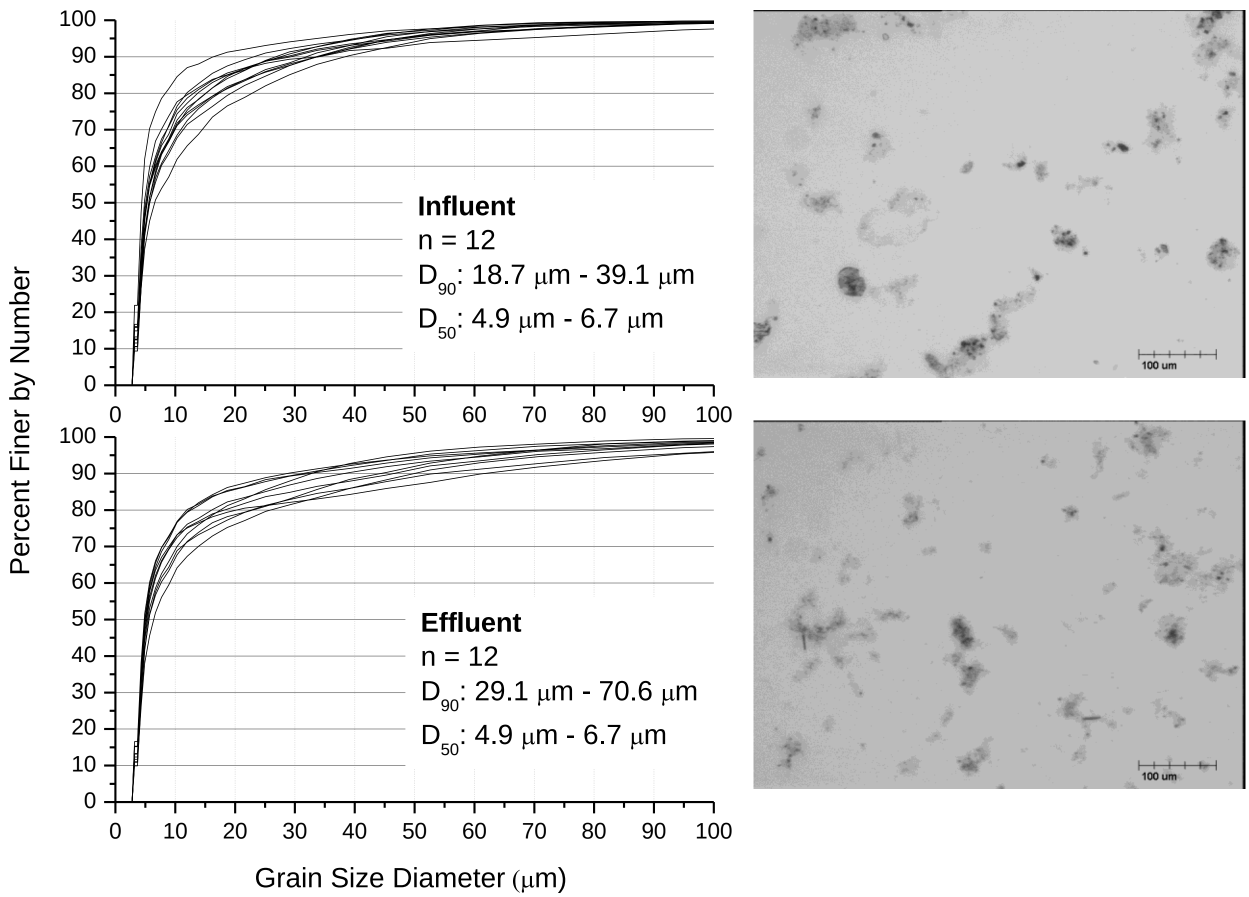
	Influent						Effluent					
2010 Date	**N total**	**Mean**	**Std. Dev.**	**Minimum**	**Median**	**Maximum**	**N total**	**Mean**	**Std. Dev.**	**Minimum**	**Median**	**Maximum**
07-May	0	--	--	--	--	--	11	**6.74**	1.0	4.65	7.07	7.57
13-May	1	**7.51**	--	7.51	7.51	7.51	12	**7.59**	0.19	7.14	7.63	7.84
02-Jun	10	**6.84**	0.59	5.82	6.94	7.54	7	**5.82**	0.78	5.11	5.54	7.13
03-Jun	12	**8.14**	0.45	6.74	8.30	8.40	12	**7.24**	0.20	6.72	7.28	7.55
12-Jun	12	**8.77**	0.18	8.31	8.78	9.02	12	**8.24**	0.16	8.04	8.21	8.63
16-Jun	12	**8.45**	0.13	8.25	8.45	8.67	12	**8.03**	0.15	7.63	8.04	8.25
22-Jun	12	**8.15**	0.16	7.89	8.16	8.38	12	**7.65**	0.30	7.22	7.60	8.50
24-Jun	12	**7.97**	0.30	7.61	7.88	8.51	12	**7.68**	0.13	7.54	7.65	7.99
26-Jun	12	**8.55**	0.12	8.21	8.57	8.66	12	**7.72**	0.33	7.43	7.59	8.47
24-Jul	12	**8.75**	0.14	8.39	8.76	8.98	12	**7.68**	0.19	7.29	7.64	7.95
21-Aug	12	**8.19**	0.16	7.72	8.22	8.33	12	**7.95**	0.21	7.58	7.99	8.21



Grain size distribution of suspended sediment in influent and effluent samples of storm event on August 29, 2009. Representative micrograph of particles with scale = 100 microns.



Grain size distribution of suspended sediment in influent and effluent samples of storm event on August 20, 2009. Representative micrograph of particles with scale = 100 microns.



Grain size distribution of suspended sediment in influent and effluent samples of storm event on August 11, 2009. Representative micrograph of particles with scale = 100 microns.

Appendix B: Water Quantity

Table B1: Summary of runoff retention based on calculated precipitation inflows and observed runoff volumes.

Storm Event	Total Rain	Inflow	Outflow	Outflow per unit area	% Runoff retention (relative to precipitation)	Average runoff coefficient
	mm	L	L	L/m^2		
11-Jun-09	5.4	200	34	0.92	83.0	0.170
15-Jun-09	1.6	59.4	0	0.00	100	0.000
18-Jun-09	5.4	200	32	0.86	84.0	0.160
20-Jun-09	11.4	423	332	8.94	21.6	0.784
25-Jun-09	22.8	847	410	11.0	51.6	0.484
11-Jul-09	20.8	772	352	9.48	54.4	0.456
21-Jul-09	2.4	33.4	0	0.00	100	0.000
23-Jul-09	14.2	197	36	2.59	81.8	0.182
25-Jul-09	4.2	58.4	48	3.45	17.8	0.822
26-Jul-09	8.0	111	90	6.47	19.1	0.809
29-Jul-09	1.2	16.7	0	0.00	100	0.000
29-Jul-09	5.4	75.1	24	1.73	68.0	0.320
04-Aug-09	4.6	63.9	0	0.00	100	0.000
04-Aug-09	2.4	33.4	0	0.00	100	0.000
08-Aug-09	3.6	50.0	0	0.00	100	0.000
09-Aug-09	10.4	145	0	0.00	100	0.000
09-Aug-09	17.6	245	32	2.30	86.9	0.131
10-Aug-09	5.6	77.8	0	0.00	100	0.000
11-Aug-09	8.4	117	82	5.90	29.8	0.702
18-Aug-09	7.0	97.3	0	0.00	100	0.000
20-Aug-09	83.2	1160	1030	74.0	11.1	0.889
26-Aug-09	2.8	38.9	0	0.00	100	0.000
29-Aug-09	4.2	58.4	22	1.58	62.3	0.377
23-Sep-09	1.6	22.2	0	0.00	100	0.000
27-Sep-09	5.0	69.5	0	0.00	100	0.000
28-Sep-09	20.4	284	40	2.88	85.9	0.141
28-Sep-09	6.2	86.2	38	2.73	55.9	0.441
29-Sep-09	3.8	52.8	40	2.88	24.3	0.757
29-Sep-09	2.2	30.6	26	1.87	15.0	0.850
02-Oct-09	1.6	22.2	38	2.73	-70.9	1.71
03-Oct-09	2.6	36.1	36	2.59	0.400	0.996
04-Oct-09	5.2	72.3	68	4.89	5.90	0.941

06-Oct-09	5.0	69.5	168	12.09	-141.7	2.417
09-Oct-09	23.2	322	322	23.17	0.000	1.00
22-Oct-09	1.4	19.5	0	0.00	100.0	0.000
23-Oct-09	5.0	69.5	100	7.19	-43.9	1.439
28-Oct-09	3.8	52.8	18	1.29	65.9	0.341
30-Oct-09	1.4	19.5	44	3.17	-126.1	2.261
31-Oct-09	6.0	83.4	86	6.19	-3.10	1.031
Total 2009	**365**	**7040**	**3550**	**5.2 (Mean)**	**47.2 (Mean)**	**0.562 (Mean)**
07-May-10	18.4	256	86	2.32	66.4	0.336
11-May-10	1.4	19.5	0	0.00	100.0	0.000
13-May-10	5.2	72.3	80	2.15	-10.7	1.11
Jun-01-10	6.6	91.7	0	0.00	100.0	0.000
02-Jun-10	33.0	459	66	4.75	85.6	0.144
03-Jun-10	14.8	206	190	13.7	7.60	0.924
04-Jun-10	1.8	25.0	0	0.00	100.0	0.000
06-Jun-10	22.2	309	290	20.9	6.0	0.940
09-Jun-10	3.0	41.7	0	0.00	100.0	0.000
09-Jun-10	2.8	38.9	0	0.00	100.0	0.000
12-Jun-10	23.6	328	146	10.5	55.5	0.445
16-Jun-10	17.8	247	148	10.6	40.2	0.598
22-Jun-10	19.4	270	96	6.91	64.4	0.356
24-Jun-10	31.2	434	442	31.8	-1.90	1.02
26-Jun-10	11.8	164	86	6.19	47.6	0.524
27-Jun-10	24.2	336	430	30.9	-27.8	1.28
09-Jul-10	11.4	158	0	0.00	100.0	0.000
11-Jul-10	15.6	217	0	0.00	100.0	0.000
15-Jul-10	1.0	13.9	0	0.00	100.0	0.000
19-Jul-10	5.2	72.3	0	0.00	100.0	0.000
23-Jul-10	29.4	409	192	13.8	53.0	0.470
24-Jul-10	12.2	170	84	6.04	50.5	0.495
28-Jul-10	8.6	120	0	0.00	100.0	0.000
02-Aug-10	13.8	192	0	0.00	100.0	0.000
08-Aug-10	3.8	52.8	0	0.00	100.0	0.000
09-Aug-10	10.8	150	0	0.00	100.0	0.000
15-Aug-10	17.0	236	2	0.14	99.2	0.008
19-Aug-10	5.8	80.6	0	0.00	100.0	0.000
22-Aug-10	27.8	386	256	18.4	33.8	0.662
Total 2010	**400**	**5550**	**2590**	**6.2 (Mean)**	**67.9 (Mean)**	**0.321 (Mean)**

Table B2: Summary of flow, antecedent dry period, and rainfall characteristics

Storm Event	Total Rain	Mean Flow	Peak Flow	Lag time	Previous Dry Hours	Max Intensity	Mean Intensity	Rainfall Duration
	mm	L/min	L/min	min	hrs	mm/5 min	mm/hr	min
11-Jun-09	5.4	0.208	0.667	20	--	0.80	1.20	270
15-Jun-09*	1.6	--	--	--	70	0.6	0.22	430
18-Jun-09	5.4	0.041	0.5	17	39	0.2	0.26	1235
20-Jun-09	11.4	0.227	4	114	39	1.4	0.62	1100
25-Jun-09	22.8	0.644	10	9	110	6.4	3.42	400
11-Jul-09	20.8	1.54	12	4	218	10.8	22.7	55
21-Jul-09*	2.4	--	--	--	240	1.2	0.25	570
23-Jul-09	14.2	0.157	0.5	109	29	0.8	1.16	735
25-Jul-09	4.2	0.226	0.667	85	40	0.6	2.10	120
26-Jul-09	8.0	0.086	0.667	10	18	1.2	0.69	700
29-Jul-09*	1.2	--	--	--	52	0.6	3.60	20
29-Jul-09	5.4	0.103	0.4	35	5	0.6	2.59	125
4-Aug-09*	4.6	--	--	--	146	1.4	11.0	25
4-Aug-09*	2.4	--	--	--	3	1.8	14.4	10
8-Aug-09*	3.6	--	--	--	101	0.4	0.54	400
9-Aug-09*	10.4	--	--	--	10	3.8	13.9	45
9-Aug-09	17.6	0.151	2	11	8	4.6	9.60	110
10-Aug-09*	5.6	--	--	--	5	1.0	3.20	105
11-Aug-09	8.4	0.358	2	2	39	2.8	4.38	115
18-Aug-09*	7.0	--	--	--	179	2.0	21.0	20
20-Aug-09	83.2	4.38	16	61	61	11.2	38.4	130
26-Aug-09*	2.8	--	--	--	80	0.6	0.84	200
29-Aug-09	4.2	0.076	0.333	120	62	0.4	0.55	462
23-Sep-09*	1.6	--	--	--	605	0.6	0.24	405
27-Sep-09*	5.0	--	--	--	72	0.4	0.39	760
28-Sep-09	20.4	0.349	1	52	27	2.4	6.12	200
28-Sep-09	6.2	0.314	1	24	3	0.6	2.76	135
29-Sep-09	3.8	0.150	0.667	39	6	0.4	1.09	210
29-Sep-09	2.2	0.197	0.333	35	6	0.4	1.89	70
2-Oct-09	1.6	0.196	1	188	21	0.4	0.16	610
3-Oct-09	2.6	0.132	0.5	19	10	0.8	0.21	735
4-Oct-09	5.2	0.121	1	5	15	2.4	0.96	325
6-Oct-09	5.0	0.178	2	94	48	0.4	0.45	660
9-Oct-09	23.2	0.202	1	431	42	0.4	1.02	1360
22-Oct-09*	1.4	--	--	--	297	0.2	0.07	1280
23-Oct-09	5.0	0.105	1	38	16	0.4	0.32	950
28-Oct-09	3.8	0.015	0.1	482	98	0.2	0.45	510
30-Oct-09	1.4	0.108	0.286	79	49	0.2	0.56	150
31-Oct-09	6.0	0.165	0.5	0	10	0.6	0.77	470

07-May-10	18.4	0.136	0.667	60	39	1.4	0.84	1320
11-May-10*	1.4	--	--	--	77	0.2	0.13	628
13-May-10	5.2	0.149	2	2	33	1.0	0.80	390
Jun-01-10	6.6	0.149	0		426	0.6	1.07	370
02-Jun-10	33.0	0.178	2	11	30	2.8	3.44	575
03-Jun-10	14.8	0.687	4	35	10	1.6	8.46	105
04-Jun-10*	1.8	--	--	--	31	0.4	0.50	215
06-Jun-10	22.2	0.495	2	0	21	0.8	1.93	690
09-Jun-10*	3.0	--	--	--	69	0.4	0.77	235
09-Jun-10*	2.8	--	--	--	10	0.6	2.58	65
12-Jun-10	23.6	0.097	2	0	54	6.4	3.45	410
16-Jun-10	17.8	0.579	4	0	64	2.8	14.2	75
22-Jun-10	19.4	0.389	4	0	61	5.0	2.74	425
24-Jun-10	31.2	0.924	6	0	36	4.2	8.14	230
26-Jun-10	11.8	0.310	1	46	48	0.8	1.46	485
27-Jun-10	24.2	0.440	4	0	26	2.0	3.68	395
09-Jul-10*	11.4	--	--	--	268	0.6	1.63	420
11-Jul-10*	15.6	--	--	--	52	7.0	3.17	295
15-Jul-10*	1.0	--	--	--	97	0.2	0.33	180
19-Jul-10*	5.2	--	--	--	71	3.6	2.60	120
23-Jul-10	29.4	0.830	4	79	95	3.6	3.56	495
24-Jul-10	12.2	0.185	1	99	27	2.2	2.19	335
28-Jul-10*	8.6	--	--	--	85	3.2	0.97	530
02-Aug-10*	13.8	--	--	--	117	8.0	82.8	10
08-Aug-10*	3.8	--	--	--	81	1.8	0.63	360
09-Aug-10*	10.8	--	--	--	26	1.8	7.20	90
15-Aug-10	17.0	2	2	0	137	3.8	1.62	630
19-Aug-10*	5.8	--	--	--	88	3.8	34.8	10
22-Aug-10	27.8	0.139	1	709	50	1.8	1.01	1645

Note: Italicized events had larger drainage area.* denotes no runoff.

Table B3: Runoff stage depth values within the rain barrel.

Storm Event	Peak Flow (L/min)	Highest Level (m)	Average Stage (m)	Approximate contact time with media (min)
11-Jul-09	12	0.916	0.359	270
23-Jul-09	0.5	0.163	0.108	228
25-Jul-09	0.67	0.22	0.132	327
26-Jul-09	0.33	0.12	0.089	167
26-Jul-09	0.67	0.184	0.123	374
26-Jul-09	0.2	0.102	0.084	185
29-Jul-09	0.4	0.14	0.095	217
9-Aug-09	2.0	0.355	0.224	302
9-Aug-09	2.0	0.327	0.182	352
11-Aug-09	2.0	0.329	0.128	681
20-Aug-09	16	0.922	0.4	398
29-Aug-09	0.33	HOBO logger malfunction		
28-Sep-09	1.0	0.177	0.105	200
29-Sep-09	0.67	0.184	0.159	144
29-Sep-09	0.33	0.192	0.163	155
2-Oct-09	1.0	0.269	0.148	331
4-Oct-09	1.0	0.319	0.231	423
6-Oct-09	2.0	0.452	0.336	496
9-Oct-09	1.0	0.392	0.278	1020
23-Oct-09	1.0	0.189	0.149	303
28-Oct-09	0.10	0.153	0.144	103
31-Oct-09	0.50	0.2	0.166	296
7-May-10	0.67	0.413	0.229	1179
13-May-10	2.0	0.32	0.171	486
2-Jun-10	2.0	HOBO logger malfunction		
3-Jun-10	4.0			
6-Jun-10	2.0			
12-Jun-10	2.0	0.575	0.257	1843
16-Jun-10	4.0	0.422	0.313	268
22-Jun-10	4.0	0.463	0.183	1438
24-Jun-10	6.0	0.632	0.360	896
26-Jun-10	1.0	0.35	0.148	1604
27-Jun-10	4.0	0.548	0.326	1504
24-Jul-10	1.0	HOBO logger malfunction		
15-Aug-10	2.0			
22-Aug-10	1.0			

Appendix C: Plant List

Table C1: Greenroof vegetation

Plant Species	**Common name**
Allium schoenoprasum var. sibiricum	wild chives
Allium cernuum	nodding wild onion
Penstemon hirsutus	hairy beardtongue
Saxifraga virginiana	Virginia saxifrage
Packera paupercula	prairie ragwort
Minuartia stricta	rock sandwort
Geum triflorum	prairie smoke
Ranunculus fascicularis	early buttercup
Carex eburnea	ebony sedge
Panicum acuminatum	hairy panic grass
Opuntia humifusa	prickly pear
Scutellaria parvula	dwarf skullcap
Verbena simplex	slender vervain
Isanthus brachiatus	false pennyroyal
Solidago nemoralis	grey goldenrod