New York City's Green Infrastructure: Impacts on Nutrient Cycling and Improvements in Performance

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ABSTRACT

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Urban stormwater runoff from impervious surfaces reduces water quality and ecological diversity in surrounding streams. The problem is exacerbated in older cities with combined sewer systems like New York City, where roughly 30 billion gallons of untreated sewage and stormwater runoff are combined and dumped into the New York harbor annually. Rain gardens and green roofs are designed to naturally manage stormwater, but both performance data and design guidance are limited. In particular, rain gardens are not optimized for nutrient removal, and US green roofs are commonly planted with non-native vegetation, which may not be optimized for water retention.

The first of three studies in this dissertation investigates the overall effect of rain gardens on nutrient removal. Engineers have found there to be tradeoffs between rain garden designs that overall favor greater water retention and those that favor removal of pollutant nutrients, as efficient nutrient removal requires designs that drain slowly, and thus absorb less stormwater. Despite these opposing concerns, this dissertation has found that rain gardens constructed in areas with combined sewer systems should focus on water retention, as the benefits of treating increased amounts of water outweigh admitted downsides, such as the leaching of pollutant nutrients contained in rain garden soil.

The second study investigates how nutrient pollution can be reduced in rain gardens. To do this, it quantifies the rate that the rain garden's soil creates nitrogen pollution, by converting nitrogen from organic to inorganic forms, as inorganic nitrogen is more readily washed out of the soil and into water bodies. Conversely, it also quantifies the amount of nitrogen consumed by plants and also nitrogen emitted in gas form. It then uses the results to construct an overall nitrogen mass balance. The results indicate that the soil used to build rain gardens is in fact too nitrogen rich; inorganic nitrogen supplied by the decomposition of organic nitrogen and by stormwater runoff is far greater than required to maintain vegetative health for rain garden plants. The study concludes that altering rain garden soil specifications could reduce nitrogen pollution.

The third study finds that "industry-standard" green roofs planted with drought-tolerant *Sedum* vegetation might not capture as much stormwater as "next-generation" native systems with irrigation and smart detention. Specifically, the study provides crop coefficients demonstrating reduced evapotranspiration in drought tolerant green roof plants compared to native plants. It also found a native roof's stormwater capture increased with irrigation and the use of a smart runoff detention system, which automatically reduced the volume of water in the cistern that captures roof runoff in advance of a predicted storm.

US government agencies are launching multi-billion dollar greening initiatives that include rain gardens and green roofs designed to manage volumes of stormwater runoff. The research here can assist in quantifying performance and improving green infrastructure designs.

Table of Contents

List of Figures

List of Tables

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Chapter 1: Introduction

1.1 Background

1.1.1 History

The word "sewer" comes from Old English and means seaward (Brown, 2005), because when the first sewer infrastructure was built, household sewage and stormwater from rains were combined in a single pipe and sent untreated directly to nearby waterways. Such a system, however primitive it may seem today, was actually an improvement over conditions in US urban areas as late as the 19th century, where in some cases people used to empty chamber pots directly out of their windows, causing disease and generally unsanitary conditions (Lofrano and Brown, 2010). As a result, while sewers improved the quality of life on the urban watershed, surrounding rivers, lakes, and coasts received the city's untreated fecal and industrial waste, as well as oils, metals, and other pollutants found in stormwater runoff.

In order to reduce water pollution, cities built interceptor pipes along their coastlines in the l9th century to divert sewer pipes to newly constructed wastewater treatment plants (Bloomberg, 2008). As a result of wastewater treatment, water quality improved tremendously nationwide, while contaminated water-related illnesses plummeted, so that today, many US waterbodies are the healthiest that they have been in centuries. During dry weather, most US cities no longer discharge any sewage without treating it first.

However, even small rainstorms still overwhelm these combined sewer systems. During wet weather, the flow in the combined pipes is much greater than the capacity of wastewater treatment plants. In order to prevent wastewater from backing up into homes or damaging wastewater treatment plants, the raw wastewater mix that cannot be treated is discharged directly into waterways. These combined sewer overflows (CSOs) are some of the leading sources of water pollution for 772 US cities (Bloomberg, 2008). Like other older cities, New York City is serviced by a combined sewer system that annually discharges roughly 30 billion gallons of raw CSO into nearby waterways (Bloomberg and Holloway, 2010).

At one time, forests, grasslands, deserts, and swamps absorbed most rain that fell on the American landscape. Most stormwater would either be intercepted in the vegetated canopy or would infiltrate into the ground and recharge the groundwater table, while a smaller portion created creeks, streams, and rivers that flowed to the ocean.

Alas this is no longer the case. Today, waterproof surfaces like asphalt and concrete blanket our landscapes, resulting in much more stormwater runoff. Even in cities without combined sewer overflows, stormwater runoff from impervious surfaces damages water quality and ecological diversity in surrounding streams (Walsh et al., 2005).

Today a concerted effort is being made to reverse some of the damage done to the urban environment. However, it is less a question of returning to a pristine undeveloped state, than of working with current conditions to improve urban ecology, by utilizing recent technological advances. The question then is how do we best retrofit vegetation into the built environment to benefit the local ecosystem and reduce pollution while preserving existing land uses?

1.1.2 Green Infrastructure

In the city imagined by green engineering, stormwater is first absorbed before it reaches the ground by street tree canopies and by roofs with a thin layer of vegetation, known as green roofs. When excess stormwater collects along streetside gutters it is absorbed by small green spaces that absorb rain, known as rain gardens.

Green infrastructure (GI) practices like rain gardens and green roofs use vegetation and soils to manage stormwater at its source, while providing more benefits than single-purpose gray infrastructure (EPA, 2017). Urban vegetation cools cities (Susca et al., 2011), promotes biodiversity by providing food and habitat for native insects and birds (Sandström et al., 2006), and supports psycho-social-spiritual well-being for a wide range of people (Svendsen et al., 2016). In recognition of its benefits, New York City has invested $410 million in capital funding into green infrastructure over the last six years, with another $1 billion budgeted over the next 10 years, and with annual expense costs of an additional $15 million per year (De Blasio and Sapienza, 2017).

1.1.3 Quantifying Performance

While green infrastructure technologies like rain gardens and green roofs are indeed needed to mitigate the environmental problems of urbanization, and specifically to restore pre-development hydrology (Askarizadeh et al., 2015; Walsh et al., 2016), their precise effect on water cycles and urban nutrient levels remains little quantified.

Although rain gardens have been demonstrated to effectively absorb stormwater runoff and treat many pollutants such as metals and oils, their overall effect on nutrient removal is more ambiguous (Collins et al., 2010; Davis et al., 2009; Hatt et al., 2009; Hunt et al., 2006; Li and Davis, 2014). To wit, several studies' findings indicate that rain gardens may actually increase the total concentration of nitrogen and phosphorus pollution in stormwater draining through their soil (Collins et al., 2010; Hunt et al., 2006; Randall and Bradford, 2013; Shetty et al., 2016).

However, it remains unclear what the practical implications are of nitrogen and phosphorus leaching in rain gardens: namely, if rain gardens are simultaneously reducing volumes of nutrient-laden CSOs while also leaching nutrients to the surrounding soil and groundwater, then are they overall adding nutrient pollution or mitigating it? This question needs a definitive answer so that the scientific community may properly advise land managers of New York City, in order to develop recommendations for designing rain gardens to maximally reduce CSOs (De Blasio and Sapienza, 2017).

1.1.4 Improving Performance

And if rain gardens are themselves leaching nutrients, then how can the problem be mitigated; how can nutrient pollution be reduced? How can GI designs be modified to improve environmental performance?

These questions are particularly pertinent in the US, as here green roofs are relatively new and have high potential for improvement. This is due to the fact that they are largely replicated from German specifications, where green roofs have been in use for decades (Macivor et al., 2013;

Mentens et al., 2006), and the German model includes the use of drought-tolerant *Sedum* vegetation native to Europe. However, research shows that plants native to the US better support biodiversity (Lundholm et al., 2010; Vanuytrecht et al., 2014), and may help a green roof retain more stormwater than *Sedum* plants (Aloisio et al., 2016; Li and Babcock, 2014; Nagase and Dunnett, 2012; Whittinghill et al., 2014). Additionally, promising new smart sensors connected to weather forecasting data are capable of increasing stormwater retention further (Kerkez et al., 2016; Roman et al., 2017).

1.2 Research Questions and Dissertation Format

My dissertation considers the following overall question:

How does green infrastructure impact nutrient cycles and how can performance be improved?

Chapter 2 aims to quantify the impact of rain gardens on nutrient cycles by looking at "scaled-up" effects. Specifically, I consider long-term trends and the overall effect of nutrient pollution. I also introduce promising new monitoring methods.

Chapter 3 investigates how nutrient pollution can be reduced in rain gardens. To do this, it quantifies the rate that the rain garden's soil creates nitrogen pollution, by converting nitrogen from organic to inorganic forms, as inorganic nitrogen is more readily washed out of the soil and into water bodies. Conversely, it also quantifies the amount of nitrogen consumed by plants and also nitrogen emitted in gas form. It then uses the results to construct an overall nitrogen mass balance. Finally, it seeks to establish the optimal concentration of nitrogen in the soil used to build rain gardens, considering whether this soil is in fact too nitrogen rich, or rather is too low for rain garden plants' optimal functioning. Generally, this chapter considers how soil specifications can overall reduce nitrogen pollution.

Chapter 4 considers how performance can be improved for green roofs by demonstrating an enhanced green roof design. In particular, Chapter 4 suggests three ways to maximize the stormwater capture performance of the "next-generation" of green roofs: native vegetation, irrigation, and smart detention.

Chapter 5 outlines contributions of this dissertation related to quantifying the performance of rain gardens and green roofs and exploring how performance can be improved.

Chapter 6 proposes avenues of future research that would build off of this dissertation to produce an improved understanding of green infrastructure performance.

After I provide references, I include an Appendix with a study of the water quality performance of rain gardens. The study quantifies how rain gardens' nutrient removal performance is affected by environmental conditions, including temperature, rainfall depth, the dry weather period before a storm, and the watershed size.

Chapter 2: Effect of Rain Gardens on Nutrient Pollution for Combined Sewer Systems

Abstract: This study evaluates the overall impact of rain gardens on nutrient pollution: the long-term ability of rain gardens to filter nutrient pollutants found in stormwater runoff was compared to nutrients removed by reducing sewer overflow volumes to a combined sewer system in New York City (NYC). Long-term nutrient removal was determined by measuring nitrogen and phosphorus levels in water samples during 42 storms at seven different rain gardens located in the Bronx, NYC. The amount of nutrients removed from sewer overflow was quantified by measuring water retention at one of the rain gardens during eight of the 42 storms. The study results indicate the following: first, at the one site where water retention was measured, the averaged retention value was 40%. Second, water quality after filtering at each of the seven sites was fairly consistent, although the position of the rain garden impacted the amount of nutrients filtered through the garden. Third, seasonal trends were more significant than long term trends with regard to the amount of nitrogen leached from the gardens, but the amount of phosphorus leached from the gardens declined over time. Finally, this study found that despite tradeoffs created by rain gardens that themselves leach small amounts of nutrients, the studied rain gardens overall reduce nutrient pollution, as sewer overflow is reduced due to their efficient retention of water.

2.1 Introduction

Rain gardens are widely used to manage stormwater in modern cities (Davis et al., 2009). They are commonly designed to capture a given volume of stormwater runoff (Collins et al., 2010; Davis et al., 2012; Roy-Poirier et al., 2010), but are not optimized to remove nutrient pollutants:

specifically, many studies find increased total nitrogen and total phosphorus concentrations in stormwater that drains through rain garden soil (Collins et al., 2010; Hunt et al., 2006; Randall and Bradford, 2013; Shetty et al., 2016). For this reason, there is a current debate as to whether rain garden designs should predominately favor water quantity or water quality improvements. In particular, the question arises as to whether or not rain garden designs should continue to specify aerobic soils and organic matter soil amendments that increase the quantity of water captured by the rain garden, but also lead to an increase in nutrients leached by the rain garden, which negatively impacts water quality. Answering this question requires better understanding of how the tradeoffs between water retention and nutrient removal might affect overall nutrient pollution.

For example, organic matter is used in rain garden design as a fertilizer, improving plant health (Hunt et al., 2012) and soil drainage (Emerson and Traver, 2008; Turk et al., 2014). However, its presence also causes leaching of the pollutants nitrogen and phosphorus (Clark and Pitt, 2009; Hatt et al., 2009; Hunt et al., 2012, 2006; Li and Davis, 2014; E G I Payne et al., 2014; Randall and Bradford, 2013; Zinger et al., 2013), leading to a tradeoff between improved water retention and nutrient removal. Similarly, the tradeoff between water retention and nutrient removal is also found in the type of soil specified for use in rain gardens. Rapid-draining soils, such as sandy media, allow for higher rates of stormwater runoff to be infiltrated by the garden (Liu et al., 2014). However, the short hydraulic residence time in a fast draining soil does not allow sufficient time for nutrient pollutants to be removed (Collins et al., 2010; Hatt et al., 2009; Hunt et al., 2006; Liu et al., 2014; Peterson et al., 2015). Specifically, fast-draining soils promote aerobic, oxygen-rich environments, which leads to nitrification instead of denitrification, causing increased rates of the pollutant nitrate to be leached out from the garden (Bratieres et al., 2008; Davis et al., 2006; Hunt et al., 2006). Denitrification, which requires a wet, anoxic soil environment, is considered the only permanent way to remove nitrogen from stormwater runoff (Collins et al., 2010). While there may be localized, anoxic conditions within small pockets of a rain garden soil that favor denitrification (Davis et al., 2006; Hunt et al., 2006), in general, the aerobic soils of traditional rain garden designs limit denitrification rates (Lucas and Greenway, 2011).

To address the problem of nitrate leaching from rain gardens, some recent rain garden designs promote denitrification by including internal storage zones (ISZs), which create saturated soil areas with slow infiltration rates (Collins et al., 2010). ISZs tend to be excellent at denitrification, the process through which the pollutant nitrate is converted to nitrogen gas (Gilchrist et al., 2014; Randall and Bradford, 2013; Zinger et al., 2013). However, ISZs themselves often lead to the leaching of other nutrients, particularly ammonium (Gilchrist et al., 2014; Hurley et al., 2017; Li et al., 2014; Randall and Bradford, 2013; Zhang et al., 2011) and phosphorus (Clark and Pitt, 2009; Hurley et al., 2017; Manka et al., 2016; Randall and Bradford, 2013; Zinger et al., 2013). ISZs also reduce the amount of stormwater that the rain garden captures and infiltrates (Liu et al., 2014; Randall and Bradford, 2013; Roy-Poirier et al., 2010), which in turn could decrease overall nutrient removal, depending on the drainage area's setting.

Due to tradeoffs between water quantity removal and water quality concerns, it remains unclear whether rain gardens should be designed for increased water retention or for more efficient nutrient removal. Understanding the answer to this question is a tremendous challenge, because the importance of increasing water retention versus improving the quality of water leached from a rain garden show considerable variation based on microclimatic and site-specific features (Collins et al., 2010; Davis et al., 2012; Lucke and Nichols, 2015; Manka et al., 2016; Roy-Poirier et al., 2010). In addition, another major variable is whether the overall sewer system of a city within which rain gardens are located is a combined or separated system. In combined sewer systems (CSSs), rainwater and sewage both drain into the same pipe system, resulting in combined sewer overflows (CSOs) during heavy rains. CSOs are a leading source of surface waterbody pollution, including nutrient pollution, in many cities (Montalto et al., 2007). In contrast, in separated sewer systems there are two independent pipe networks, one conveying sewage and the other stormwater. In a separated sewer system, only sewage water is treated, leaving the stormwater to drain directly into nearby waterways such as rivers and harbors. As a result, unlike CSSs, separated sewer systems do not overflow during heavy rains. This means that rain gardens installed in areas with separated sewer systems should probably be designed to optimize nutrient removal above water retention. However, in a combined sewer system, it is less clear to what degree rain garden design should focus on nutrient removal or water retention. One goal of this study is to address this question in the context of a CSO shed located in New York City.

A further challenge to understanding the optimal design of a rain garden, is site-specificity of rain gardens performance, which can call for extensive performance monitoring programs in thousands of locations if rain gardens are an important part of a city's stormwater management strategy. The need for widespread performance monitoring is further compounded by the fact that rain garden instrumentation is currently still cost-prohibitive, with per site equipment costs averaging \$41,282, with as much as \$21,837 needed in addition for the installation of monitoring equipment (NYC Parks). Another goal of this study was to develop and test a more cost-efficient method of rain garden performance monitoring.

A final goal of this study was to understand whether a rain garden's performance changes over time, as plants mature and soil composition is exposed to years of seasonal changes. This goal addresses a need for research quantifying how a rain garden's ability to remove pollutants and retain water might be impacted over time, as there is no long-term water quality and quantity performance data available for rain gardens (Collins et al., 2010; Elliott et al., 2011; Koch et al., 2014).

This study quantified the ability of rain gardens to remove nutrient pollutants at seven different rain garden sites located in the Bronx, New York City (NYC), a neighborhood served by a CSS. Nutrient levels before and after stormwater filtered through the rain garden soils were monitored over a three-year period. In addition, water retention at one rain garden site was also quantified for eight storms. The specific objectives of the study were to: (1) to develop non-invasive and cost-effective measurement methods for determining rain garden water retention and water quality, (2) characterize seasonal and long-term trends in rain garden performance, (3) examine performance variability between the different sites, and (4) to compare nutrients leached via infiltration to nutrient pollution resulting from combined sewer overflows in the study area. In the sections that follow, the study sites and study methods are described. The study results are then presented and discussed. The paper ends with a summary of conclusions reached by the study.

2.2 Study Sites, Measurement and Analysis Methods

2.2.1 Rain Garden Sites

The seven target rain gardens were constructed between April 2013 and April 2014 (Table 2.1) in the Bronx, NYC. Each garden contained a 61 cm-layer of engineered sandy loam soil, below which lied a geotextile filter fabric separating the soil from another 61 cm-layer made up of 5 cm-wide diameter crushed stone. The soil surface was topped with 7.6 cm of bark chip mulch, and the rain gardens were planted with native and ornamental plant species including trees, shrubs, grasses, and herbaceous perennials.

Table 2.1 Rain Garden Site Information

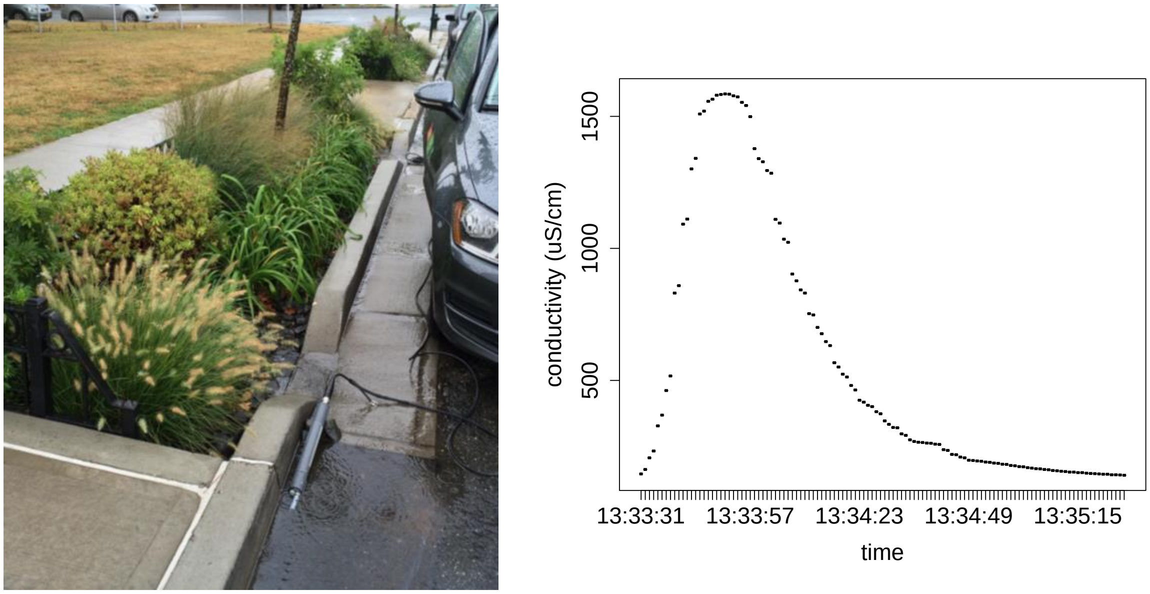
Rain Garden	Construction	Site (m^2)	Land Use
ROWB 26B	4/25/14	9.3	Commercial
ROWB 23	4/25/14	7.0	Residential
ROWB 9A	4/22/13	9.3	Residential
ROWB 9B	4/22/13	9.3	Residential
SGS 21	4/24/13	76.2	Residential
SGS 11	4/24/13	94.8	Residential
SGS 2	5/28/13	115.2	Park

Four of the rain gardens are termed “right-of-way bioswales” (ROWBs) because they are located in the sidewalk and include curb cut inlets and outlets. The remaining three rain gardens, termed “stormwater greenstreets” (SGSs), are “bumped out” into the street so that stormwater may enter more directly rather than forcing stormwater to make a 90 degree turn (see photos in Shetty et al 2016, located in the Appendix of this dissertation). Each garden drains a particular upstream surface area of the urban landscape, termed the drainage area. Estimated drainage areas for each garden are drawn as hatched areas in Figure 2.1.

Figure 2.1 Location of seven rain garden sites in the Soundview neighborhood of the Bronx, NY. The blue hatched area denotes an estimated drainage area that drains into each garden.

2.2.2 Water Retention

Water retention was measured at ROWB 9B during eight storms over a period from 11/19/15 to 2/25/17. In order to determine how much water was retained by ROWB 9B, a salt dilution method was used, where a known amount of salt solution is added to unknown amounts of stormwater runoff, and the degree of dilution downstream of the salt source allows one to calculate the total flow rate (Moore, 2005; Rantz, 1982). This test was performed both upstream and downstream of the rain garden at regular time intervals during each monitored storm in order to find the total amount of water retained by the garden. Measurements were conducted in the gutter stream along the curb. 100mL of $CaCl_2$ salt tracer solution was poured into the curbside gutter stream of stormwater runoff about 15m upstream of a conductivity probe held immediately upstream of the rain garden inlet (Figure 2.2a). As the mass of salt passed by the probe, the conductivity of the runoff would rise and then return to baseline (Figure 2.2b). Then, the measurement was repeated in the gutter immediately downstream of the rain garden. The difference between the flow rate measured upstream and downstream of the rain garden was used to calculate the water retained by the garden.



(**a**) (**b**)

Figure 2.2 Flow measurement to determine stormwater retention by salt dilution (a) conductivity sensor in street gutter; (b) Measured conductivity over time as a mass of salt flowed by the conductivity sensor

Water flow rates were determined for each measurement based on the principle of the conservation of mass:

$$Q = \frac{V_t C_t}{\int_{t1}^{t2} (C - C_b) dt} \tag{2.1}$$

where Q denotes the gutter stream flow rate, V_t and C_t denote the volume and concentration of $CaCl_2$ salt tracer injection solution, C denotes the measured concentration, C_b denotes the baseline concentration of the gutter stream, and t is time.

Measurements were taken every 30 minutes to one hour for the entire duration of the eight storms. The total upstream and downstream volume were found by integrating each discrete flow rate over the time period of the entire storm. The water retention efficiency (WRE) of ROWB 9B was then calculated as the difference between the total upstream and downstream volume divided by the total upstream volume:

$$\text{Water retention efficiency} = \frac{\text{Upstream} - \text{Downstream}}{\text{Upstream}} \tag{2.2}$$

Where upstream and downstream refer to the total volumes found in the gutter stream upstream and downstream of the rain garden by integrating over all flow rates.

2.2.3 Water Quality

At the seven rain gardens described in Section 2.2.1, 595 water samples were collected during 42 storms over a three-year monitoring period (3/30/14 to 6/19/17). Water samples were then tested for concentrations of different forms of both nitrogen and phosphorus. Water quality was sampled via a syringe and chemically analyzed as described in Shetty et al (2016), which is in the Appendix of this dissertation. All seven rain gardens were tested once, during the 42 storms (approximately in the middle of the storm), for phosphorus and nitrogen levels. However, one garden, ROWB 9B, was additionally tested for these nutrient levels in tandem with water quantity measurements (see above), at intervals of about 30 minutes (although sometimes an hour) throughout the storm, in order to capture water quality changes occurring over the course of the storm.

Water samples were collected from each site's inlet (influent), perforated pipe samplers in the soil (infiltrate), and the site's overflow outlet (overflow) (see Appendix). The average concentrations of pollutants in rainwater over the course of each storm, termed event mean concentrations (EMCs), were calculated for influent, infiltrate, and overflow for the eight storms at ROWB 9B for which water retention was quantified. This was done by calculating the total mass of each pollutant by considering the flow rate at the time the water quality samples were collected. The total mass of each pollutant was then divided by the total stormwater volume:

$$\mathrm{EMC} = \frac{\int Q\ C\ dt}{\int Q\ dt} \tag{2.3}$$

2.2.4 Impact of ROWB 9B on Nutrient Pollution

The measured results of this study were then used to estimate the overall effect of an individual rain garden on nutrient pollution, or the total pollution both removed and added by the rain garden. This was accomplished by considering the drainage area of ROWB 9B as a case study area to normalize calculations.

The drainage area of the rain garden, A, was determined experimentally with the following equation averaged for the eight storms where water retention was measured:

$$\text{Upstream} = R_c\ P\ A \tag{2.4}$$

Where Upstream denotes the total volume of stormwater runoff measured just upstream of the rain garden, R_c denotes a runoff coefficient, and P denotes the total precipitation depth. This study selected 0.78 as the runoff coefficient R_c measured by Montalto et al (2007) as part of a hydrologic modeling study for the Gowanus neighborhood of Brooklyn, an area we assume to be representative of NYC. Precipitation was measured with an Onset Hobo U30 weather station that records data from a TR-525i Texas Electronics tipping bucket rain gage on a roof located approximately 1 km from the studied rain gardens.

The total mass of nutrient pollution from CSOs both with and without a rain garden was estimated with the following equation:

$$\textit{Total Nutrient load} = V_{CSO}\, C_{CSO} \tag{2.5}$$

Where V_{CSO} denotes CSO volume and C_{CSO} denotes CSO total nutrient concentration.

To estimate V_{CSO}, a relationship was used between the duration of rainfall leading to a CSO and the minimum cumulative depth of rainfall causing a CSO modeled for Gowanus (Montalto et al., 2007):

$$d = 0.054\, t + 0.232 \tag{2.6}$$

Where d denotes the minimum depth of rainfall in centimeters causing CSOs and t denotes the duration of rainfall in hours leading up to a CSO. For instance, a one hour rainfall with minimum cumulative depth of 0.286 cm (0.054 (1) + .232 = 0.286) would be sufficient to produce a CSO. In order to use this equation with a variety of storm depths and durations, 40 years (March 1977-March 2017) of historical weather data at LaGuardia International Airport, located approximately 5 km from the studied rain gardens, was downloaded from the National Oceanic and Atmospheric Administration’s National Climate Data Center website (www.ncdc.noaa.gov). Hourly precipitation was separated into the total depths of rainfall over the course of a storm using a minimum six-hour dry period to separate out individual storms (Berretta et al., 2014; Carson et al., 2013). The duration of each of the 4,121 storms found over the 40 years was inputted into equation (2.6) as *t*, in order to solve for *d*, the minimum storm depth that would cause a CSO. The volume of CSO for each storm was then calculated with the following equation:

$$V_{CSO} = (d_{storm} - d)\ R_c\ A \tag{2.7}$$

Where d_{storm} denotes the actual storm depths for each of the 4,121 storms.

Since C_{CSO} depends on the degree of dilution of sewage with stormwater, C_{CSO} was solved for each individual storm using equations (2.8) and (2.9), which are respectively a volume and mass balance using figures for NYC's combined sewer system:

$$V_{total} = R_c\ A\ d_{storm} + t\ A\ \frac{TDWF}{TDA} \tag{2.8}$$

$$C_{CSO}\ V_{total} = C_{influent}\ R_c\ A\ d_{storm} + C_{sewage}\ t\ A\ \frac{TDWF}{TDA} \tag{2.9}$$

Where V_{total} denotes total sewer volume, TDWF denotes the total dry weather flow for the year 2015 reported as $1.58 * 10^{12}$ L/year (NYCDEP, 2016), and TDA stands for the total drainage area for all 14 of NYC's wastewater treatment plants, reported as 695 km^2 (NYCDEP, 2017). In this way, the dry weather flow is normalized per area of NYC. C_{sewage} stands for the 60 ppm N concentration reported as typical for raw municipal wastewater (Henze and Comeau, 2008). $C_{influent}$ represents the median nutrient concentration found in this study in stormwater influent and is used to signify the nutrient concentration in stormwater that enters catch basins. This analysis assumes that average assumptions for New York City's sewer system apply to the drainage area of ROWB 9B, and that influent and overflow are not statistically different, as found in Shetty et al (2016).

The reduced load of total nutrient pollution within CSOs owing to rain garden water absorption was then found by first reducing the actual storm depths for each storm by considering the water retention efficiency (WRE) of the rain garden:

$$d_{stormRG} = d_{storm} (1\text{-}WRE) \tag{2.10}$$

before substituting d_{storm} with $d_{stormRG}$ and repeating equations (2.7), (2.8), (2.9), and (2.5). The total mass of nutrients infiltrated into the soil was approximated as the volume of stormwater that goes into the garden multiplied by $C_{infiltrate}$, which denotes the median concentration of nutrients found in this study for stormwater that infiltrates through rain garden soil:

$$\textit{Total Nutrients Infiltrated} = C_{infiltrate}\ R_c\ A\ d_{storm}\ WRE \tag{2.11}$$

Finally, the total nutrient load in CSOs was compared to the reduced nutrient load in CSOs from rain garden water absorption plus the nutrients infiltrated by the rain garden (Figure 2.3).

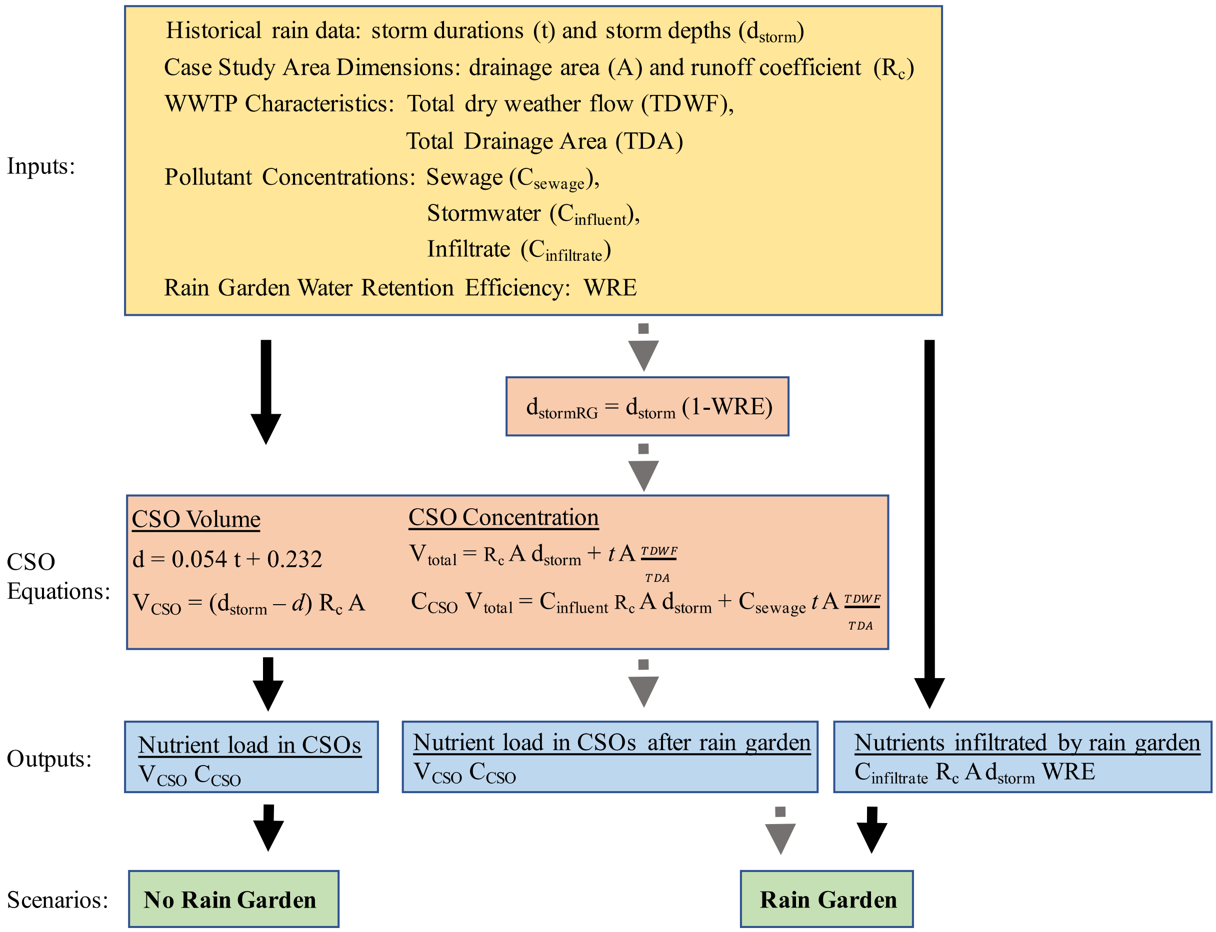


Figure 2.3. Flowchart demonstrating impact of ROWB 9B on nutrient removal. The dotted grey lines denote equations altered by substituting d_{storm} with $d_{stormRG}$.

2.2.5 Statistical Analysis

Statistical analyses were conducted in R v. 3.1.3 (The R Project for Statistical Computing, 2015). Non-parametric Wilcoxon rank-sum tests were used to distinguish statistically significant differences between sites and non-parametric Mann-Kendall trend tests were used to determine changes over time (Helsel and Hirsch, 2002).

2.3 Results

2.3.1 Water Retention

The amount of water retained for eight storms at site ROWB 9B ranged from 20-75% and averaged 40% (Table 2.2). When the measured gutter flow upstream of the site and the cumulative rain depth were used in equation (2.4), the average calculated drainage area was 1258 m^2, which is a 135:1 drainage: site area hydraulic loading ratio.

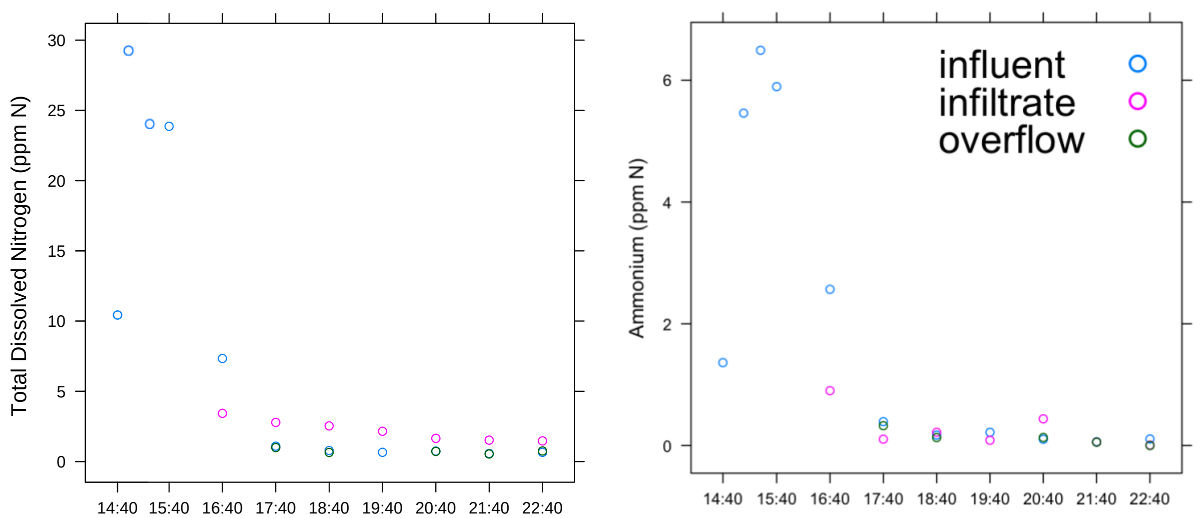
Table 2.2 Percentage of stormwater retention for eight storms monitored at site ROWB 9B

Date	**Total Rain Depth (mm)**	**Retention (%)**	**Drainage Area (m^2)**
11/19/15	**28.0**	**0.44**	**1240**
2/3/16	**20.0**	**0.20**	**1332**
4/7/16	**4.2**	**0.75**	**1242**
7/25/16	**2.6**	**0.46**	**1265**
9/19/16	**2.2**	**0.43**	**1448**
11/29/16	**49.2**	**0.30**	**966**
11/30/16	**12.2**	**0.26**	**1021**
2/25/17	**9.8**	**0.36**	**1552**
Average		**0.40**	**1258**

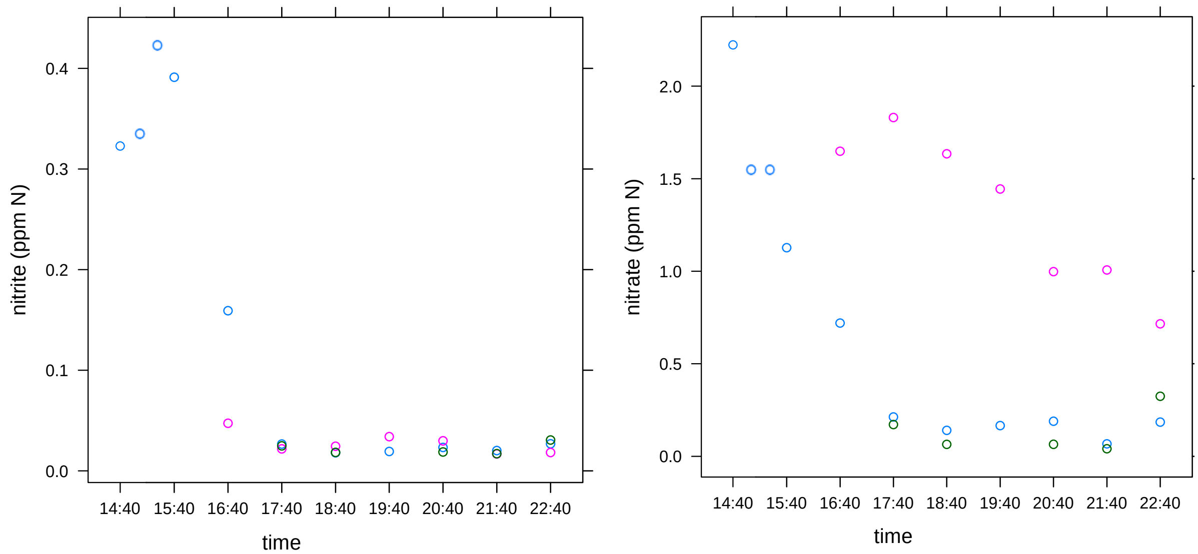
2.3.2 Water Quality

2.3.2.1 First Flush for ROWB 9B

During an individual storm considered representative of all storms measured (Figure 2.4), influent stormwater runoff from the street demonstrated a strong "first flush" effect for each form of dissolved nitrogen, including ammonium, nitrite, nitrate, and total nitrogen. The "first flush" means that there were greater nitrogen concentrations measured at the beginning of the storm, as pollutant concentrations that had accumulated on the street from atmospheric deposition began to be washed into the rain garden. Conversely, reduced nitrogen concentrations were measured toward the end of the storm, as influent pollutants were diluted.



(a) **(b)**



(**c**) (**d**)

Figure 2.4 Influent, infiltrate, and overflow concentrations for site ROWB 9B for the storm on November 19th, 2015. (**a**) total dissolved nitrogen; (**b**) ammonium; (**c**) nitrite; (**d**) nitrate

Infiltrate concentrations, however, for each form of nitrogen were fairly consistent throughout the storm, except for nitrate (Figure 2.4d), which like influent, demonstrated a buildup/wash-off effect, with greater concentrations at the beginning than at the end of the storm.

2.3.2.2 Seasonal and Long-term Variability

In the years following its initial construction, a rain garden undergoes a significant evolution: vegetation matures, soil composition changes, plant roots develop and decay, and soil freezes and thaws during the winter. Nevertheless, this study found no major long term trends for total nitrogen in rain garden infiltrate during the three years of sampling (Figure 2.5). Instead, data indicated a seasonal trend. Phosphorus also had a seasonal trend, but it had a more significant long-term decline (Figure 2.6). Statistically significant differences ($p < 0.05$) are written as letters above the boxplots.

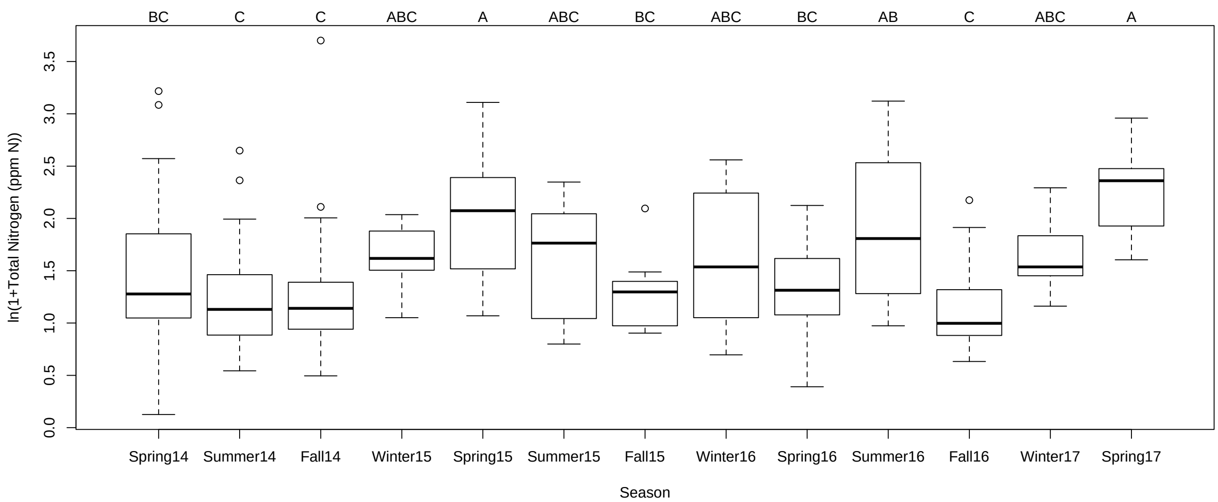


Figure 2.5 Long Term Infiltrate Concentrations of Total Dissolved Nitrogen

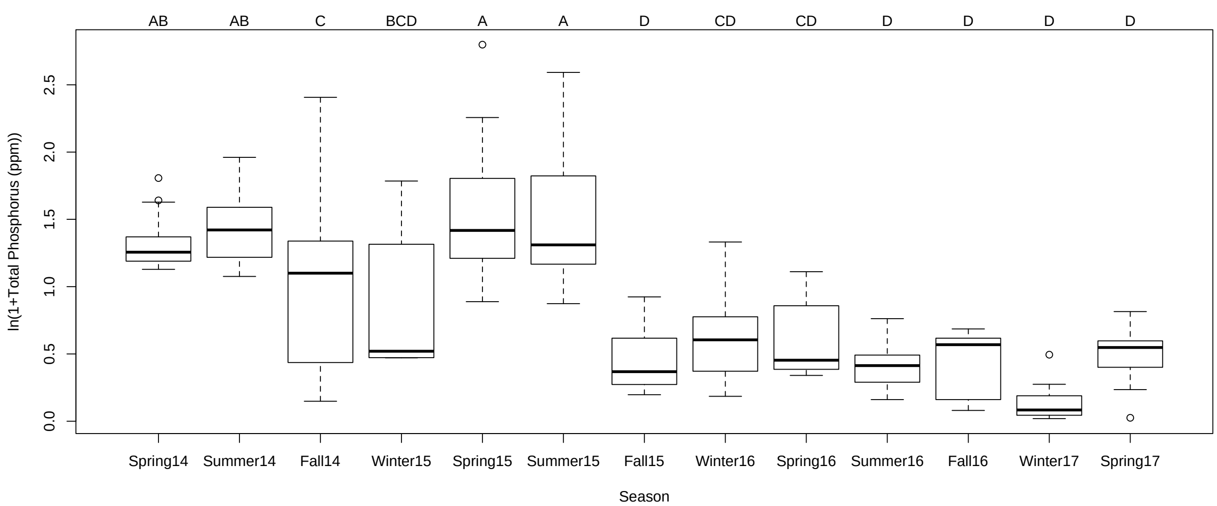


Figure 2.6 Long term Infiltrate Concentrations of Total Dissolved Phosphorus

The decline in total phosphorus over time is also demonstrated by a negative value for Kendall's Tau (-0.38) and a significant p value ($p < 0.001$) in the Mann-Kendall trend test. Total nitrogen did not show a long-term trend, with a Kendall's tau of 0.07 ($p = 0.06$).

2.3.2.3 Site Differences

While the seven sites had fairly comparable infiltrate concentrations of nutrients, there were some statistically significant differences between them. One notable difference is that infiltrate concentrations of total dissolved nitrogen at ROWB 9B had greater concentrations than at ROWB 9A, for example (Figure 2.7).

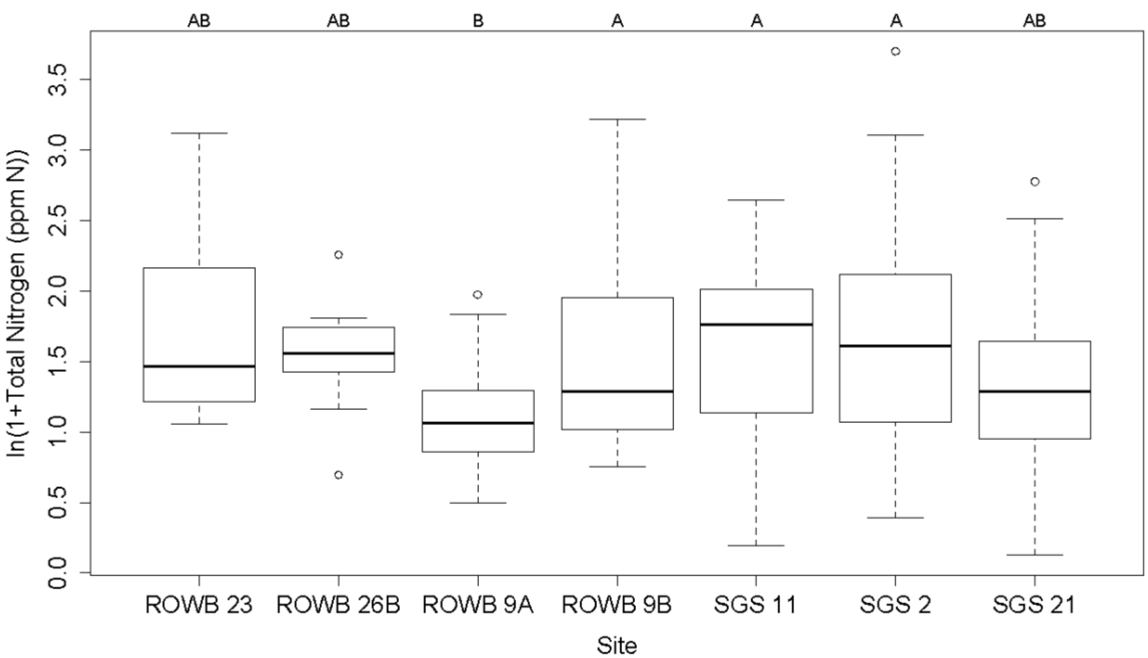


Figure 2.7 Infiltrate Concentrations of Total Dissolved Nitrogen at Seven Rain Gardens

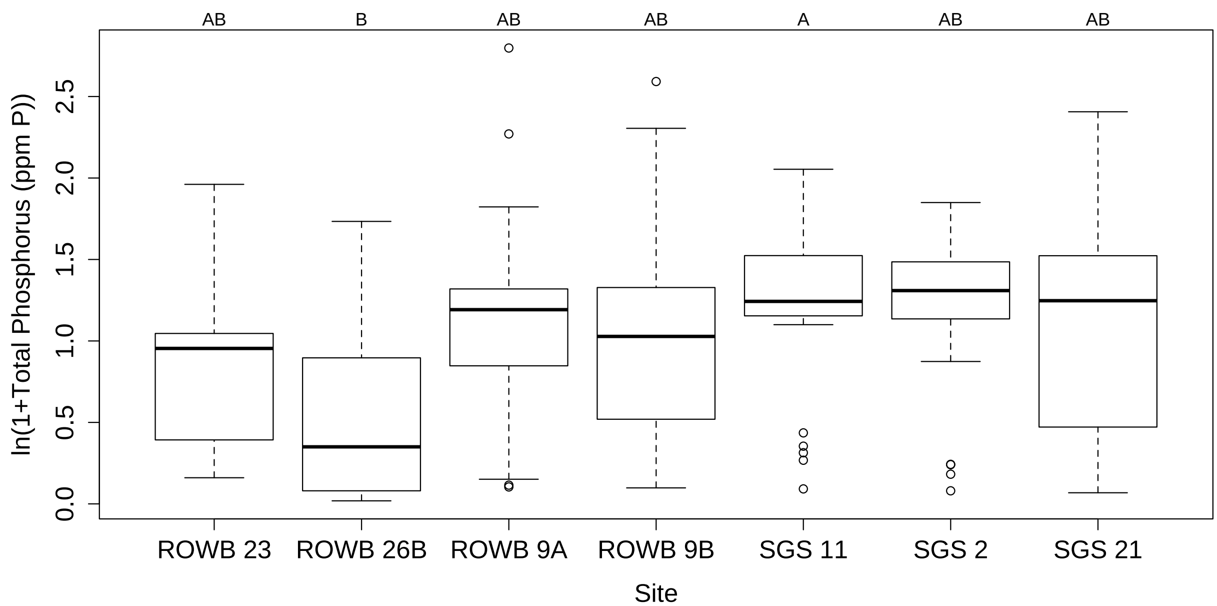


Figure 2.8 Infiltrate Concentrations of Total Dissolved Phosphorus at Seven Rain Gardens

Rates of phosphorus absorbed by the garden were generally comparable, but Site 26B had the lowest infiltrate concentrations (Figure 2.8).

2.3.3 Impact of ROWB 9B on Nutrient Pollution

Stormwater that entered the seven rain gardens for the 42 storms sampled had a median total nitrogen concentration of 1.58 ppm N, while stormwater that infiltrated had a median concentration of 2.85 ppm N (Figure 2.9).

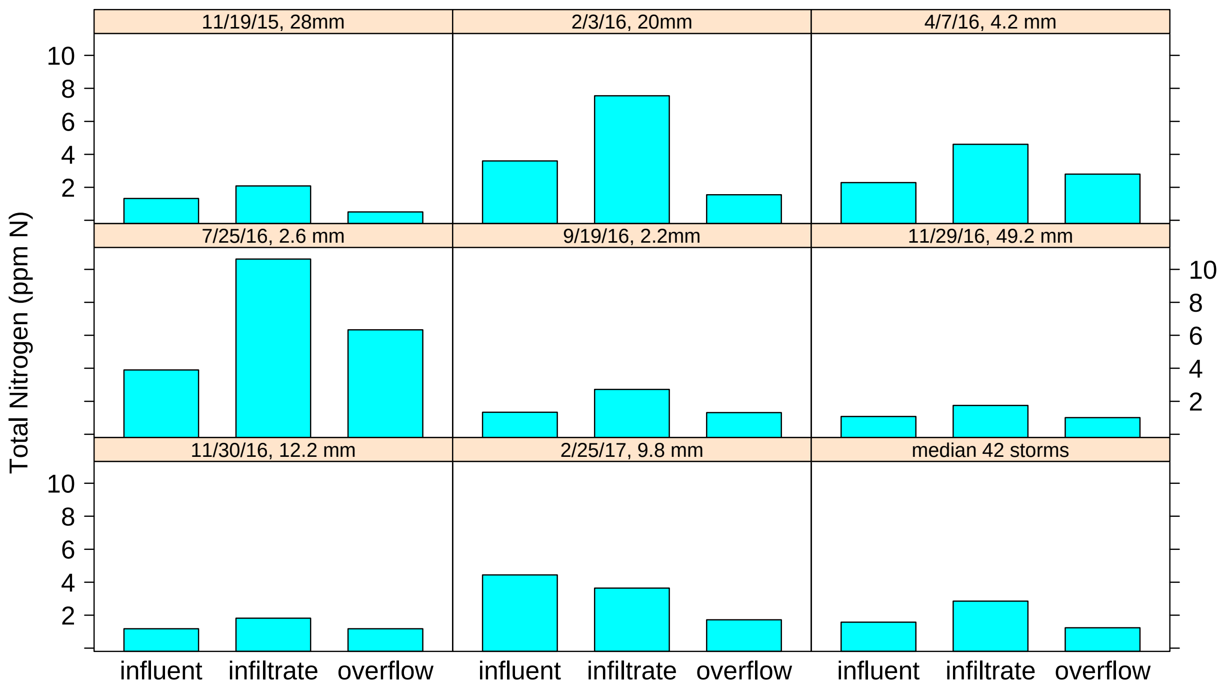


Figure 2.9 Total Nitrogen - Event Mean Concentrations for eight storms at ROWB 9B and median concentrations for 42 storms at all seven rain gardens (bottom right)

For total nitrogen, the frequent measurements taken from ROWB 9B over the course of eight different storms were representative of the averages found at the other seven gardens for all 42 storms measured, as the median total nitrogen concentrations for all seven gardens during all 42 storms were similar to the event mean concentrations found at ROWB 9B for eight storms. In contrast, the median total phosphorus concentrations were greater than the EMCs found at ROWB 9B, as demonstrated in Table 2.3, which also contains the EMCs for all of the different nitrogen and phosphorus species. Therefore, median concentrations of total nitrogen and not total phosphorus were used to estimate the overall impact of ROWB 9B on nutrient removal.

Table 2.3**.** Event Mean Concentrations in ppm for eight storms at ROWB 9B and median concentrations for 42 storms at all seven rain gardens

Date	Stream	Ammonium	Nitrite	Nitrate	Total Nitrogen	Phosphate	Total Phosphorus
11/19/15	influent	0.34	0.03	0.19	1.32	0.22	0.63
	infiltrate	0.27	0.03	1.28	2.08	0.17	0.57
	overflow	0.10	0.01	0.07	0.51	0.20	0.30
2/3/16	influent	0.74	0.04	0.32	3.60	0.07	1.10
	infiltrate	1.29	0.16	0.81	7.55	0.06	0.86
	overflow	0.46	0.02	0.25	1.55	0.03	0.23
4/7/16	influent	0.87	0.01	0.11	2.29	0.07	1.35
	infiltrate	1.06	0.01	0.50	4.61	0.02	1.08
	overflow	1.18	0.01	0.10	2.80	0.06	0.83
7/25/16	influent	1.34	0.07	1.57	3.90	0.56	0.75
	infiltrate	1.38	0.32	6.72	10.62	0.42	0.69
	overflow	1.71	0.11	2.00	6.33	0.92	0.93
9/19/16	influent	0.50	0.01	0.21	1.33	0.17	0.68
	infiltrate	0.29	0.01	1.03	2.72	0.24	0.35
	overflow	0.41	0.01	0.22	1.31	0.16	0.17
11/29/16	influent	0.16	0.01	0.19	1.08	0.14	0.79
	infiltrate	0.12	0.003	0.36	1.75	0.09	0.86
	overflow	0.12	0.005	0.15	1.01	0.08	0.79
11/30/16	influent	0.25	0.01	0.22	1.18	0.07	0.54
	infiltrate	0.11	0.004	0.38	1.82	0.06	0.29
	overflow	0.27	0.01	0.20	1.18	0.07	0.14
2/25/17	influent	0.80	0.01	0.31	4.44	0.26	0.32
	infiltrate	0.54	0.01	2.33	3.64	0.16	0.31
	overflow	0.42	0.01	0.32	1.72	0.23	0.33
42 storm median	influent	0.37	0.02	0.22	1.58	0.20	1.20
	infiltrate	0.40	0.02	0.53	2.85	0.19	2.30
	overflow	0.25	0.01	0.17	1.24	0.13	0.77

As calculated in Section 2.2.4, the drainage area of ROWB 9B with no rain garden contributes a median 5.31 kg N/yr (Figure 2.10) to local waterbodies via combined sewer overflow, including both stormwater runoff and sewage. This amount is reduced by about 50% to a median 2.65 kg N/yr when CSOs are reduced following stormwater absorption by ROWB 9B. The tradeoff of this efficient water retention however is that a median1.35 kg N/yr is infiltrated into the soil, so a total of about 4 kg N/yr could eventually flow toward local waterbodies even with a rain garden.

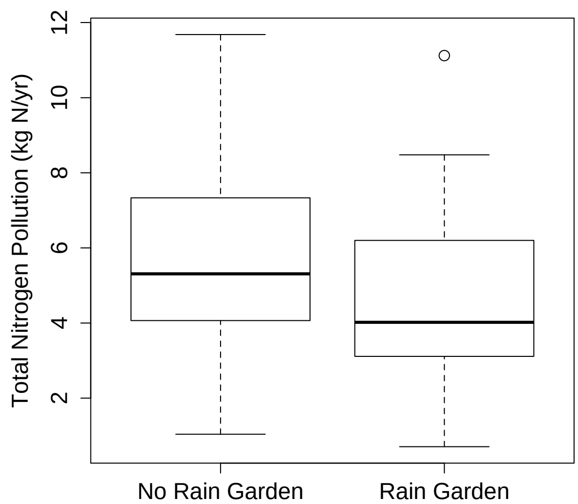


Figure 2.10 Annual total nitrogen pollution with and without a rain garden

2.4 Discussion

When designing rain gardens, engineers have found there to be trade-offs between designs that overall favor greater water retention and those that favor pollutant removal, as efficient pollutant removal requires designs that drain slowly, and thus absorb less stormwater. Despite these opposing concerns, this study has found that rain gardens constructed in areas with combined sewer systems should focus on water retention, as the benefits of treating increased amounts of water outweigh admitted downsides, such as the leaching of pollutant nitrogen contained in rain garden soil.

However, although overall water retention should remain the main focus when designing rain gardens in combined sewer systems, for both combined and separated sewer systems some modifications can nevertheless be made that maximize nutrient removal while not negatively impacting water retention. For example, rates of nutrients infiltrated by the seven gardens were found to vary among each site, while also demonstrating seasonal differences. This means that site designers should consider local conditions when designing sites for nutrient removal, and should account for greater nutrient pollutants during summer months. An example of this would be to use a soil low in phosphorus, as foliage provides ample amounts of phosphorus in the summer and fall months, and so such a modification would not impact overall water retention (as plant growth and thus absorption capacity would be unaffected).

2.4.1 Water Retention

Rain gardens' ability to retain water is substantial, as this study found ROWB 9B to capture 40% of water running through an area 135 times its own size (135:1 hydraulic loading ratio). This demonstrates the ability of even small rain gardens to significantly reduce runoff volumes. Such a high retention is of particular note, as the site studied contains a curb cut inlet that requires stormwater runoff flowing down the curbside gutter to make a 90 degree turn in order to be captured by the site. This study found very little bypass of the inlet, as the pitched concrete apron in the street may create a unique flow profile, slowing down water flow, and thus directing it into the site (Figure 2.11). However, as the "bumpout" sites with more direct inflow were not measured for water quantity, further research is needed to determine whether such a factor negatively or positively impacts retention rates.

Figure 2.11 Efficient inlet design with concrete apron located in the street to slow down and direct water into ROWB 9B

This may indeed have factored into the 40% retention rate observed, as this study found lower water retention than other research (Askarizadeh et al., 2015; Davis et al., 2012, 2009; Lord et al., 2013; Roy-Poirier et al., 2010). However, a more likely factor was the fact that the hydraulic loading ratio for garden ROWB 9B is much greater than the more common rates of 30:1 - 15:1 (Davis et al., 2012; Hunt et al., 2006).

2.4.2 Water Quality

2.4.2.1 First Flush

During the first hours of a heavy rain, all forms of nitrogen within the stormwater influent are found to spike as pollution accumulated on street surfaces is washed into draining stormwater. This phenomenon is referred to as a ‘first flush’. This study additionally found that during the storm, nitrate accumulated in soil due to natural processes of nitrification, as shown in other studies (Hatt et al., 2009; Roy-Poirier et al., 2010; Shetty et al., 2016), is also ‘flushed’ out, as has also been indicated by other studies (Li and Davis, 2014). However, with exceptions made for this spike in infiltrated nitrate, this study found different types of nitrogen to show greater variation in influent compared to infiltrate, in stark contrast to some (Manka et al., 2016) who found very little spike in nitrate infiltration during ‘flush’ periods, as well as little within-site variation from storm to storm.

This study found that first flush pollutants can be quite substantial (Figure 2.4), and rain gardens may benefit from future design amendments to remove these concentrations. In fact, for small rains less than 3 mm per hour in combined sewer areas that usually do not trigger overflows (Montalto et al., 2007), rain gardens may increase overall nutrient pollution because untreated stormwater is directed underground rather than to the wastewater treatment plant. This is because rain gardens’ overall positive impact in curbing sewer overflow is not a factor in light rain where there are no overflows, but the gardens’ own leaching of pollutants continues. Because of such a tradeoff, this study’s findings indicate that rain garden design should consider soil amendments such as granular activated carbon (Li and Davis, 2014) or coconut coir pith as a replacement for compost (Herrera, 2015) that would not impact gardens’ ability to retain water during heavy rains, but would nevertheless reduce the potential for rain gardens to export pollutants.

2.4.2.2 Seasonal and Long Term Variability

This study found that seasonal trends were more significant than long term trends for concentrations of infiltrating total nitrogen. The seasonal trends are likely due to greater influent nutrient concentrations in spring and summer, as found by others (Passeport and Hunt, 2009; Selbig, 2016). The lack of a long-term trend for infiltrating total nitrogen appears to correspond to opposing findings in previous studies, as variables affecting nutrient removal rates are often in conflict; for example, some studies suggested that nutrient removal would improve over the long term, as initial compost and more labile forms of soil organics are washed out of the soil (Chahal et al., 2016; Mullane et al., 2015), or as mature vegetation takes up more nutrients (Collins et al., 2010; Lucas and Greenway, 2008). Conversely, other studies suggested that nutrient removal may decrease as mineralization of soil organic matter increases (Payne et al., 2014). This appears to result in unclear or nonexistent trends, as this study demonstrates, corroborating a similar study that found that a 10-yr old and 2-yr old site had similar nutrient removal efficiencies (Davis et al., 2006).

This study indicated a significant decrease in concentrations of infiltrating phosphorus, indicating that the studied gardens have not lost capacity for phosphorus absorption. Other studies have similarly shown continual phosphorus removal even after seven years in service (Muerdter et al., 2016). However, this finding appears to contradict other studies which suggested that rain garden soil eventually loses capacity for phosphorus absorption (Davis et al., 2010; Lucas and Greenway, 2008).

2.4.2.3 Site Differences

Although median total dissolved nitrogen concentrations over 42 storms corresponded overall to ROWB 9B's mean nutrient concentrations, rates of infiltrate concentrations of total dissolved nitrogen were variable between the seven sites studied. These differences may be due to different construction techniques or from different environmental conditions with distinct land uses and drainage area sizes (Lucke and Nichols, 2015). For example, nitrogen rates are highly variable depending on drainage area size, as well as shading and the pre-existing character of the subsoil where the garden is built, particularly permeability rates. It appears to be due to such factors that ROWB 9B was found with greater nitrogen concentrations than ROWB 9A since it is upstream, and therefore receives the first flush of elevated nitrogen concentrations. Similarly, SGS 11 was found to have greater concentrations than SGS 21 possibly due to its larger estimated drainage area. SGS 2 had the greatest concentrations overall due to its unique inlet with no bypass; the site is designed so that all stormwater even during large storms flows into the site. Such variables should be taken into account when constructing rain gardens, as sites with larger drainage area sizes should particularly be candidates for soil amendments in order to provide more water quality treatment.

Similarly, phosphorus concentrations were reduced at ROWB 26B compared to the other sites. This trend may be due to its commercial rather than residential or park land use (Table 2.1). ROWB 26B may contain the smallest percentage of vegetation in its drainage area, and these vegetated areas could be fertilized with phosphorus. This finding corroborates Passeport and Hunt, who found that park land use more affected concentrations of total phosphorus when compared to other nutrient pollutants in stormwater runoff; they suggested that the increase may be due to the application of fertilizers or plant decomposition (Passeport and Hunt, 2009).

2.4.3 Impact of ROWB 9B on Nutrient Pollution

We found that in a combined sewer system, a rain garden can reduce the amount of nitrogen in combined sewer overflow by about 50%, decreasing nitrogen amounts contributed by its drainage area from 5.31 kg/yr to 2.65 kg/yr. However, as stated previously, the rain garden's impact does have tradeoffs, as it itself contributes 1.35 kg/yr to overall nitrogen pollution, and so as a result overall reduction in nitrogen pollution only amounts to 25%, from 5.31 kg/yr to 4 kg/yr.

This study's finding that rain gardens reduce overall nitrogen pollution may be applicable to rain gardens in other cities with combined sewer systems, as the studied nitrogen concentrations (Table 2.3) were consistent with those found in other studies. Our median influent total nitrogen concentration of 1.58 ppm N agrees very closely with other monitoring studies of stormwater runoff in the United States, such as the 1.62 ppm N found by Li and Davis (2014), the 1.57 ppm N found by Passeport and Hunt (2009), and the 1.47 ppm N averaged from several studies by McNett et al (2011). Our median influent ammonium and nitrate concentrations were also consistent with other research (Collins et al., 2010; Davis et al., 2006; Li and Davis, 2014; Passeport and Hunt, 2009).

However, this study does not assess the impact of rain gardens on overall phosphorus pollution because our measured median total phosphorus and phosphate concentrations were greater than our EMCs for the eight storms where water retention was measured, and also much greater than monitoring studies by others (Davis et al., 2006; McNett et al., 2011; Passeport and Hunt, 2009; Peterson et al., 2015). Environmental factors specific to the local drainage area, such as fertilization during 2014 and 2015 may have influenced the high levels found by this study.

2.4.4 Monitoring Methods

This study offers new cost-effective and non-invasive monitoring methods to quantify both water retention and water quality for rain gardens. Water retention was quantified with an inexpensive conductivity probe and salt, while water quality was sampled with perforated PVC pipes and syringes. In contrast, more common methods of monitoring rain gardens use expensive pressure transducers to measure the depth of water within commercial flumes or weirs, and auto-samplers for water quality (Davis et al., 2012; Dietz, 2016; Hatt et al., 2009; Hunt et al., 2006; Li and Davis, 2014; McNett et al., 2011).

In addition to allowing monitoring at a small fraction of the cost of traditional monitoring methods, the methods in this study are also non-invasive, not affected by inlet geometry, and therefore do not themselves impact the performance of the rain garden. Oppositely, flumes and weirs measure water depth by backing up the water flow, which for rain gardens with curb cut inlets in the street would impede water from entering the garden, and reduce the total water retention performance. An auto-sampler for water quality samples may similarly impact performance by re-routing flow into the sampling device. As a result, our monitoring methods provide results that are more representative of standard designs, rather than traditional monitoring methods which quantify the performance of rain gardens that have been altered by the monitoring equipment itself.

2.4.5 Limitations

On one hand, this study proposed a new model by which a land manager can assess how green infrastructure affects overall urban water and nutrient cycles. As demonstrated, the overall impact is affected by both combined sewer overflow reduction and the infiltrated pollutants that leach out from the garden soil. However, nutrients that infiltrate out from the garden could be further treated by the subsoil beneath the garden (Elliott et al., 2011), which would increase the overall reduction of nutrient pollution. Furthermore, the infiltrate could ultimately flow either towards deep groundwater or towards shallow subsurface urban flows that may be channeled back into the sewer system, which would impact overall nutrient removal in different ways. Future research could more comprehensively explore the fate of infiltrated nutrients, and determine whether they are treated further by the subsoil, and then conveyed back into the sewer system or infiltrate to deep groundwater.

2.5 Conclusions

The goal of the study was to determine the overall effect of rain gardens on nutrient pollution. The results indicate that the seven studied rain gardens consistently leach nutrients for at least three years after construction, and especially during warmer months. In combined sewersheds however, nutrient leaching is counterbalanced by the amount of stormwater retained by the rain gardens, which reduces more nutrient pollution from combined sewer overflows. This study's findings suggest that rain gardens in combined sewersheds, where sewer overflow is a significant problem, should continue to include fast draining soils. For separated sewersheds however, sewer overflow is not a factor, and so alternate designs should focus on reducing nutrient leaching.

Chapter 3: Quantifying Nitrogen Cycling in the Soil, Gas, and Plant phases of Rain Gardens

Abstract: Atmospheric deposition of fossil fuels currently provides an overabundance of nitrogen in urban environments. Despite this, soils used for rain gardens contain organic matter that provides extra nitrogen in the city's nitrogen cycle. Due to these overabundant levels, it is questionable whether we need to add sources of nitrogen to rain garden soil. To test this, we measured levels of two forms of nitrogen, ammonium and nitrate, in rain garden soil, as well as the soil's nitrous oxide gas emissions and the plant biomass and foliar nitrogen. We found an overabundance of nitrogen in shallow depths of soil and mulch, as there all mineralized nitrogen undergoes nitrification, a clear sign of nitrogen saturation. This overabundance is further exaggerated in the summer, when nitrogen cycling rates increase, as seen in higher levels of ammonium in the soil. However, the amount of nitrogen uptaken by the plants and the amount of nitrogen in soil gas emissions remain minimal: levels of nitrous oxide gas emissions from soil were low, though also elevated during summer months, and likewise, levels of nitrogen in plant foliage were low, especially for shrubs. The greatest flux of nitrogen was in stormwater, as 88% of nitrogen in the rain garden is in liquid form. Our data suggest that rain garden plants receive more than sufficient nitrogen nutrition from stormwater runoff, and that future soil specifications should account for these urban sources of nitrogen pollution.

3.1 Introduction

Nitrogen is considered the most problematic pollutant for US coastal waters, having degraded the ecology, fish production, and recreation of two thirds of the American coastline (Howarth and Marino, 2006). Rain gardens, which are a common feature of many cities' green infrastructure

(GI) plans, are vegetated structures designed to reduce water pollution by absorbing stormwater runoff (Clar and Green, 1993). However, recent studies have demonstrated that these GI types frequently contribute to the elevation of nitrogen levels in the environment (Bratieres et al., 2008; Collins et al., 2010; Hunt et al., 2006; Li and Davis, 2014). During wet weather conditions, well-documented concentrations of nitrogen enter rain gardens via runoff, and potentially exit via overflow and/or subsurface infiltration (Davis et al., 2006; McNett et al., 2011; Passeport and Hunt, 2009; Peterson et al., 2015). During dry weather conditions the nitrogen source is soil mineralization, where long term stores of organic nitrogen decompose into soil ammonium. Soil microbes then further transform this ammonium to nitrite and nitrate through complex nitrification processes. Surprisingly, neither mineralization nor nitrification rates have been well-quantified in rain gardens. The overarching goal of this study is to quantify these rates alongside nitrogen gas emissions and plant nitrogen uptake, in order to assess the role that soil, plant, and gaseous phase nitrogen play in the overall nitrogen cycle of rain gardens. We then consider whether rain garden plants could receive sufficient nitrogen nutrition from stormwater inputs alone and, thus, whether adding additional nitrogen to rain garden soil is an unnecessary and detrimental management strategy.

3.1.1 Mineralization and Nitrification in Rain Garden Soils

Mineralization is a commonly used index of soil nitrogen availability (E G I Payne et al., 2014; Schimel and Bennett, 2004): specifically, it indicates the decomposition of organic nitrogen into ammonium and nitrate due to the activity of microbes, and may be the best means to assess soil nitrogen fertility (Robertson et al., 1999). Mineralization processes occur in all soils. In dry soil environments, which are home to nitrifying microbes, nitrification transforms ammonium into nitrite and nitrate. Rain gardens are most frequently made with fast-draining aerobic soils, creating a dry environment hospitable to nitrifying microbes, and thus have high levels of nitrification (Collins et al., 2010; Davis et al., 2006; Lucas and Greenway, 2011; Shetty et al., 2016). As a result of nitrification, Lucas and Greenway (2008) suggest that rain gardens are sources of excess pollution in the form of nitrate (Bratieres et al., 2008; Davis et al., 2006; Hunt et al., 2006). Therefore, it is surprising that mineralization and nitrification rates have not been measured in rain gardens. The measurement of these rates can help quantify the buildup of available nitrogen, and therefore the amount leached out of the garden during rains.

3.1.2 Gas Emissions

Rain gardens might also contribute to nitrogen pollution through microbially-mediated gas emissions that result from nitrification and especially from denitrification processes (Bremner, 1997). Denitrification is encouraged, as it is the conversion of nitrate into exclusively gas forms of nitrogen, removing it from stormwater; this process takes place in wet soil. Since most rain gardens are made with fast-draining aerobic soil, some studies suggest that little denitrification takes place in them (Lucas and Greenway, 2011; Emily G I Payne et al., 2014). Recently conceived are alternative rain garden designs which contain a wet area that serves to encourage denitrification in order to remove nitrogen pollution (Grover et al., 2013; Payne et al., 2017). However, even traditional rain gardens with fast-draining aerobic soils may contain small anaerobic pockets supporting denitrification (Davis et al., 2006; Hunt et al., 2006; E G I Payne et al., 2014; Robertson and Groffman, 2015), especially in soil aggregates of high organic content (Collins et al., 2010; Davis et al., 2010). McPhillips et al (2016) found roadside ditches to be hotspots for denitrification due to periodic saturation during and shortly after storms and resulting elevated nutrient influxes. Increased nutrients due to fertilization (Livesley et al., 2010; McPhillips et al., 2016; Oertel et al., 2016) or N deposition (Templer et al., 2012) have also been associated with increased denitrification.

3.1.3 Plant Nitrogen Intake

Like soil nitrogen gas emissions, uptake of nitrogen by plants is also poorly quantified in rain gardens (E G I Payne et al., 2014; Read et al., 2008) yet potentially significant, with plants considerably increasing nutrient retention, and thus decreasing overall levels of nitrogen pollution (Bratieres et al., 2008; Henderson et al., 2007; Lucas and Greenway, 2011, 2008; Payne et al., 2017). Plant nutrient uptake correlates to the plants' maturity: a substantial rhizosphere community of roots and microbes promotes much greater nutrient uptake in older rather than younger plants (Collins et al., 2010; Lucas and Greenway, 2008). Similarly, sites planted at a high vegetation density will capture more nitrogen than those planted at a lower density (Hunt et al., 2012). Maintenance practices also play an essential role, as fallen vegetation must be collected before it begins to decay and leach nutrients back to the soil (Davis et al., 2006; Peterson et al., 2015).

Plants exhibit a wide variability in nutrient removal between species (Payne et al., 2017; Rycewicz-Borecki et al., 2017; Turk et al., 2016; Zhang et al., 2011), and some plant species have negligible or negative nutrient removal altogether (Bratieres et al., 2008). In fact, only one fourth of plant species was found to remove more nitrogen than bare soil in one study testing the effect of rain garden plants on pollutant removal (Read et al., 2008).

3.1.4 Study Objectives

It is important to better quantify nitrogen cycling in rain gardens in order to ensure that the design and management of these GI types minimizes nitrogen export (Collins et al., 2010; Roy-Poirier et al., 2010). In particular, research that examines the distinctions between mineralization, gas emissions, and plant uptake is needed (Collins et al., 2010; Gilchrist et al., 2014; E G I Payne et al., 2014; Emily G I Payne et al., 2014). This study aimed to meet this need by measuring the levels of nitrogen in soil, gas, and plant forms within seven rain gardens located in New York City (NYC) over a period spanning from June, 2015 to October, 2016. The specific objectives of the study were to: (1) quantify spatiotemporal variability of soil mineralization and nitrification at the study sites, (2) measure nitrogen gas emissions, (3) measure nitrogen plant uptake, and (4) use the results to construct an overall nitrogen mass balance for a rain garden.

3.2 Materials and Methods

3.2.1 Rain Garden Sites

The seven target rain gardens are the same as those from Chapter 2. Soil was sampled at all seven sites, gas at two sites, and plant leaf nitrogen at four sites (Table 3.1).

Table 3.1 Rain Garden sampling summary. Bold X's denote rain gardens where spatial variability was tested, as will be described in section 3.2.2.2

Rain Garden	Soil	Gas	Plant
ROWB 26B	**X**		X
ROWB 23	X		X
ROWB 9A	X		X

ROWB 9B	**X**	X	X
SGS 21	**X**	X	
SGS 11	**X**		
SGS 2	X		

3.2.2 Soil

3.2.2.1 Soil Characterization

Prior to construction of a rain garden, rain garden soil samples were submitted to the Soil Testing Laboratory (STL) at Rutgers University in September 2012 and analyzed for soil texture, organic matter, total carbon, and total nitrogen. In June, 2016, *in situ* soil samples from the studied seven sites were taken with a circular 7.6 cm diameter soil corer and again submitted to the STL at Rutgers for analysis. Two composite samples were taken from each site: a shallow depth sample at 7.6 cm - 22.9 cm depth and a deeper sample at 30.5 cm - 45.7 cm depth. Each composite sample was comprised of three samples spatially distributed across the site, one located at the inlet to the rain garden, one in the middle of the rain garden , and one near the outlet of the rain garden.

3.2.2.2 Soil Decomposition

Soil decomposition was analyzed with an intact soil core method (Hart et al., 1994; Raison et al., 1987). Two 25.4 mm polyvinyl chloride cores that are 50.8 cm in length were sharpened/sanded on the bottom to minimize soil disturbance, modified to test soil at different depths (that is, cut in half lengthwise top to bottom and duct-taped back together so that they may be flipped open), and buried into the soil leaving 5 cm above ground, and a pre-cut 1 cm hole at the top for a rebar to pull the core out (Figure 3.1).

Figure 3.1 Removal of incubated soil core

Each pair of soil cores was located within one meter of the rain garden inlet, with each pair containing two cores that were 15 cm apart. To test initial nitrate and ammonium levels, the first core was removed using the rebar as a handle, while the other core was left in the ground at the site for a seven-day field incubation period. The hole was then backfilled with rain garden soil. The duct tape on the test core was cut and the soil core length measured. The average soil length during the 15-month monitoring period was 36.5 cm. The samples from the top and bottom half of the core were homogenized by hand in separate containers. In the field, 5 g of soil sample was added to 50 mL vials that were pre-filled with 30 mL of a 2M KCl solution; the salts in this solution bind to ammonium and nitrate, allowing quantification of nitrogen levels in the soil: vials were brought back to the lab and shaken at 150 rpm for one hour to mix the soil so that ammonium and nitrate dissolve into the solution, then centrifuged at 4000 rpm for four minutes to filter out the soil with Millex-SV Durapore (PVDF) filters (0.22 µm pore size); the solution was then frozen within 6 hours of sampling. To calculate the mass of soil used in the 5 g sample, soil moisture is taken into account using gravimetric methods (Klute, 1986). After the seven-day field incubation period, the second soil core was retrieved and analyzed in exactly the same manner to compare nitrogen levels.

One pair of soil cores was installed and analyzed at seven rain gardens monthly from June, 2015 to August, 2016, with the exception of January and February 2016 due to freezing conditions. This resulted in the measurement of soil nitrogen content on 26 dates, and the calculation of mineralization and nitrification rates for thirteen incubation periods. We measured spatial variability within a given site at four of the seven sites (Table 3.1), including ROWB 26B and SGS 11 in October 2015 and ROWB 9A and SGS 21 in November 2015. The spatial variability test included five pairs of soil cores rather than just one, with each pair evenly spread down the length of the rain garden from the stormwater drainage inlet to the outlet. Ammonium was analyzed by fluorescence methods (Holmes et al., 1999) and nitrate analyzed by ion chromatography.

3.2.3 Soil Gas Emission

Soil gas emission fluxes were sampled at two of the seven sites with a static chamber method, which is the most common method for analysis of N_2O fluxes (Oertel et al., 2016; Pihlatie et al., 2013). By covering an area of soil with a closed chamber, soil gases are exchanged with the chamber headspace. A PVC cap soil gas chamber was used, 7 cm in height and 20 cm in diameter, with three Swagelok 3 mm tube compression fittings drilled into the top of the chamber (Figure 3.2). Two fittings were attached to septa to allow for syringe withdrawal of gas, while the third was fixed with a 1.2 m long, 3 mm diameter coiled stainless steel tube in order to equalize pressure but prevent significant diffusion of gas out of the chamber. Polyethylene plastic sheeting with a thickness of 4 mm tightly wrapped the chamber and a heavy chain was placed on top of the sheeting to create a seal with the uneven soil. The gas concentration in the chamber was sampled over time, and translated into a flux rate. Gas samples with 15 mL headspace were taken at 2, 5, 10 and 15 min intervals using a 30 mL syringe and stored over-pressurized in pre-evacuated 10 mL vials.

For each test day and site, soil gas emission was calculated by determining a linear slope of concentration of the four time points (McPhillips et al., 2016) and considering the footprint and volume of the chamber and the density of air.

Gas fluxes were measured 16 times at two rain gardens on eight dates between 6/15/15 and 9/21/16. All measurements were conducted during the summer and fall seasons, with the exception of 6/15/15, which was considered summer for statistical analysis. During the sampling dates, the temperature ranged from 8.4 °C to 26.9 °C and the total rain depth over the previous three days before sampling ranged from 0 to 56.1 mm. Measurements were also conducted within one meter of the inlets of each site, an area that may have increased soil gas emissions compared to the rest of the site due to greater nitrogen inputs (Grover et al., 2013).

Figure 3.2 Static gas chamber for soil gas emissions sampling at two NYC rain gardens

3.2.4 Plant Uptake

We measured the level of nitrogen in samples of tissue from ten leaves per plant species, collected among the different plants present of that species, at four different sites, as well as the spatial extent of each plant (plant patch area and foliage area), to estimate total nitrogen content in aboveground foliage.

3.2.4.1 Plant Patch Area

The vegetation in the four sampled rain gardens was arranged in discrete patches of the same species. To calculate the area of each patch, top-down photos were taken in early October 2016, capturing the extent of each patch. A 7.62 cm x 12.7 cm notecard was included in each photo for scale (Figure 3.3). Using the program ImageJ, the perimeter of each plant patch was traced to measure area.

Figure 3.3 Top-down photo of a plant patch (Symphyotrichum novae-angliae)

3.2.4.2 Foliage area

The LAI-2000 (Li-COR Inc.) measures the canopy volume, or the area covered by leaves. One above-canopy reading and two below canopy readings for each plant patch were obtained in early November 2016. The LAI-2000 computes gap fraction, the average transmittance of light passing through a known distance of canopy. We estimated foliage area with the following equation: plant patch area * canopy distance at 90 degrees * 1-gap fraction.

3.2.4.3 Plant tissue nitrogen

We sampled ten leaves collected evenly from different patches of each plant species at each rain garden on October 5th, 2016. On this date, ROWB 9A and ROWB 9B were planted identically with *Hemerocallis* ‘Lady Florence’, *Liriope muscari, Panicum virgatum, Pennisetum* *alopecuroides, Quercus palustris, Rosa* 'Radrazz' Knockout, and *Symphyotrichum novae-angliae*, ROWB 26B was planted with *Amelanchier canadensis*, *Aronia melanocarpa*, *Nepeta* x faassennii 'Walker's Low', *Panicum virgatum*, and *Spiraea tomentosa*, and ROWB 23 was planted with *Aronia melanocarpa*, *Echinacea purpurea*, *Eupatorium* sp., *Nepeta* x faassennii 'Walker's Low', and *Panicum virgatum*. Each collected leaf was photographed in the field immediately after collection with a 7.62 cm x 12.7 cm card for scale. Using ImageJ, the perimeter of each leaf was traced to calculate the area. After air drying in the laboratory, all leaves were weighed, crushed, ground, and analyzed for carbon and nitrogen using a FlashEA® 1112 Carbon and Nitrogen analyzer (Thermo Scientific). We quantified carbon and nitrogen ratios (C:N), nitrogen per unit mass, nitrogen per unit area, and nitrogen per leaf. For statistical analysis, plant species were grouped into grasses, perennials, and woody species.

3.2.5 Statistical Analysis

We conducted statistical analyses in R v. 3.1.3 (The R Project for Statistical Computing, 2015). We exclusively used non-parametric statistics, including Wilcoxon rank-sum tests to distinguish if different sites, sampling dates, and soil depths were statistically different from one another, and Spearman rank correlation coefficients to determine correlations with environmental factors (Helsel and Hirsch, 2002).

3.3 Results

3.3.1 Soil

3.3.1.1 Soil Characterization

The one pre-construction (pre-con) sample had more total carbon, and less total nitrogen than the average in-situ soil samples at both the shallow (7.6cm-22.9cm) and deep (30.5cm - 45.7cm) depths (Table 3.2). The pre-con sample also had more organic matter and a finer soil texture.

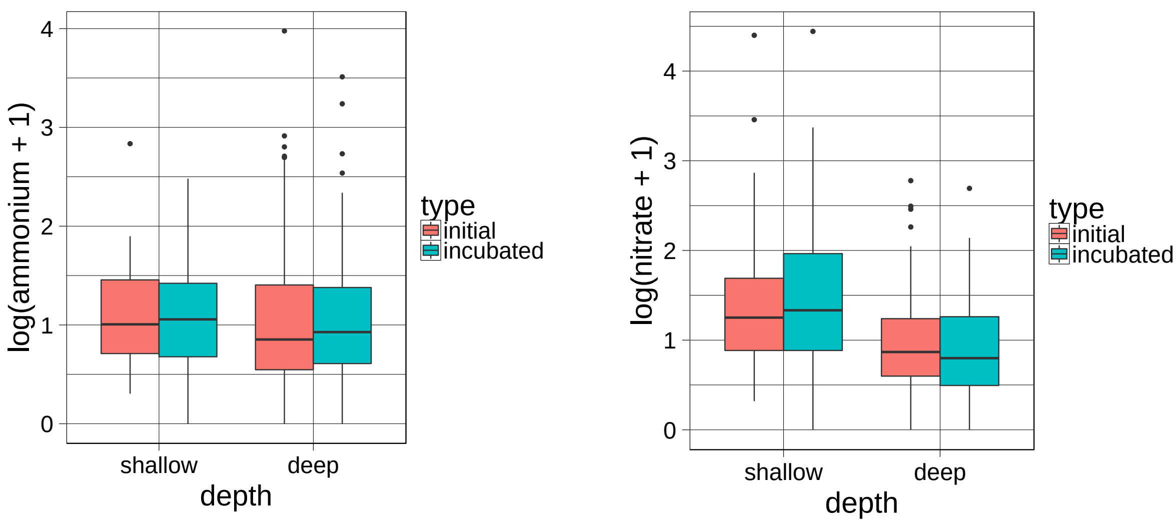
Table 3.2**.** Soil macronutrient and texture analysis means ± standard error

Soil	Total N (%)	Total C (%)	C:N ratio	Organic (%)	Sand (%)	Silt (%)	Clay (%)
Pre-con	0.10	3.3	33	5.7	71	22	7
Shallow	0.15 ± 0.02	2.9 ± 0.5	19.4 ± 0.4	5.0 ± 0.8	80.1 ± 1.2	13.4 ± 1.3	6.5 ± 0.4
Deep	0.13 ± 0.01	2.4 ± 0.3	18.7 ± 1.2	4.1 ± 0.5	78.6 ± 1.9	13.5 ± 1.5	7.9 ± 0.6

3.3.1.2 Soil Decomposition

Non-parametric two-sided Wilcoxon rank-sum hypothesis tests were performed to identify statistically significant differences among soil nitrogen levels for the different sites, depths, and incubation dates. The seven different sites produced statistically similar results, although the site named ROWB 23 had the most soil inorganic nitrogen in initial soil extracts, with statistically greater concentrations of soil ammonium than SGS 21 ($p=0.019$) and SGS 11 ($p=0.047$). ROWB 23 also had greater soil nitrate than SGS 21 ($p=0.006$), SGS 2 ($p=0.011$), and ROWB 9B ($p=0.014$). Initial inorganic nitrogen levels among the other sites were statistically similar, as were test results following incubation, as seen in similar mineralization and nitrification rates.

Soil depth was found to influence nitrogen levels more than placement within the garden. The five soil cores evenly spaced down the length of the four sites (ROWB 26B, ROWB 9B, SGS 11, SGS 21) were used to test the soil's spatial placement variability; from inlet to outlet, samples displayed statistically similar levels of inorganic nitrogen, mineralization, and nitrification. On the other hand, the shallow halves of soil cores for all sites had significantly greater soil initial nitrate ($p=0.008$), mineralization ($p=0.044$) and nitrification ($p=0.043$) (Figure 3.4) compared to the deep halves, but had no difference in soil ammonium ($p=0.131$).



(**a**) (**b**)

Figure 3.4 Soil Extractable N (log (1+ ppm N g^{-1} dry soil)) (**a**) Ammonium; (**b**) Nitrate

Soil inorganic ammonium displayed a seasonal trend (Figure 3.5), with greater soil ammonium concentrations in the summer than winter. June 2015 demonstrated statistically greater initial ammonium concentrations than every other date, with the exception of June 2016. The other dates were not statistically different. Initial soil nitrate concentrations also appeared greater in the summer (Figure 3.6), but the differences were not statistically significant. After a week incubation, soil samples displayed highly differing rates of nitrification, though no significant differences in mineralization rate.

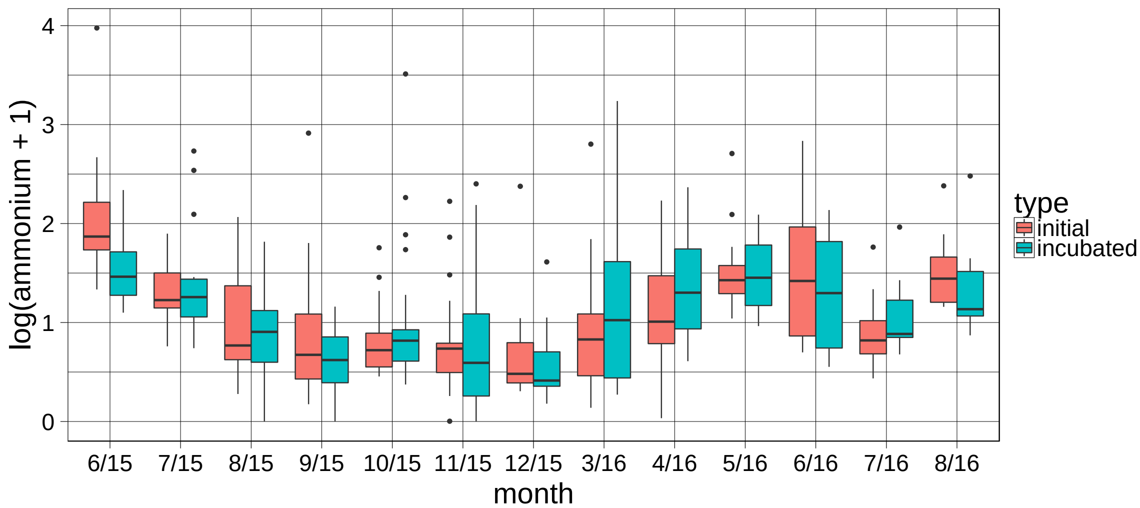


Figure 3.5 Soil extractable ammonium and one-week incubations at seven rain gardens sampled at two depths from June, 2015 to August, 2016. Units are log (1+ ppm N g^{-1} dry soil)

Nitrate levels were highly affected by precipitation, as periodic rains tend to wash away nitrate from the soil. For example, June 2016 demonstrated a large drop in soil nitrate (Figure 3.6), as its 7-day incubation period with most rain (Table 3.3).

Table 3.3 Rain amounts for each 7-day incubation period

Period	Jun-15	Jul-15	Aug-15	Sep-15	Oct-15	Nov-15	Dec-15	Mar-16	Apr-16	May-16	Jun-16	Jul-16	Aug-16
Rain (mm)	24.9	4.6	2.8	0.0	57.7	56.1	0.0	9.9	4.6	66.5	73.2	63.5	0.3

In September 2015 and December 2015, the only months when rain did not occur during the incubation period (Table 3.3), there was a clear increase between initial and incubated soil levels for nitrate (Figure 3.6), and a slight corresponding drop in soil ammonium (Figure 3.5). Ammonium levels however showed mostly seasonal variability.

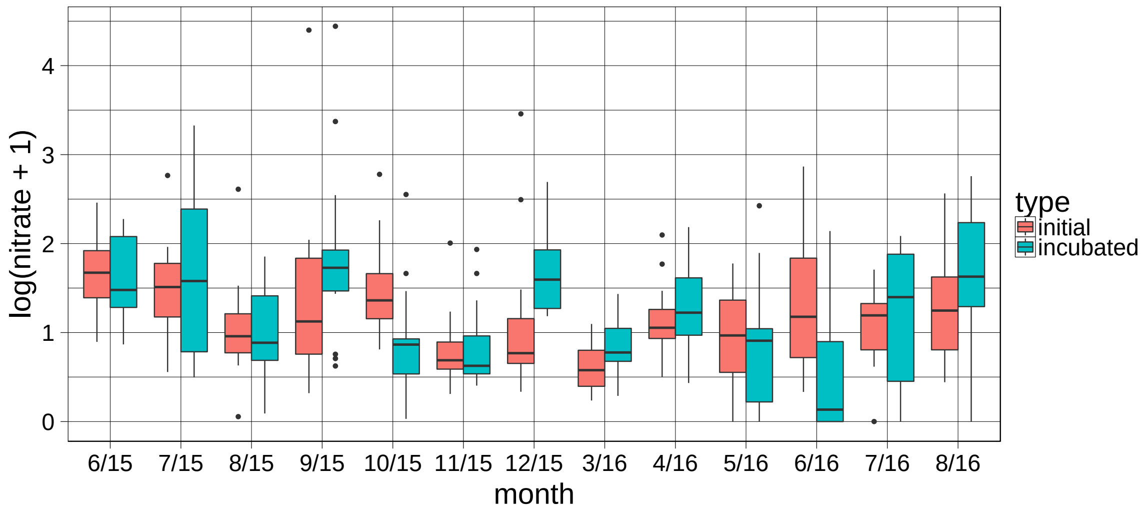


Figure 3.6 Soil extractable nitrate and one-week incubations at seven rain gardens sampled at two depths from June, 2015 to August, 2016. Units are log (1+ ppm N g^{-1} dry soil)

Soil mineralization and nitrification follow a "one to one" relationship (Figure 3.7). This ratio implies that all organic matter is eventually transformed into nitrate, as ammonium resulting from mineralization was in turn quickly nitrified and stored as nitrate. Shallow samples generally had greater nitrification than deeper samples.

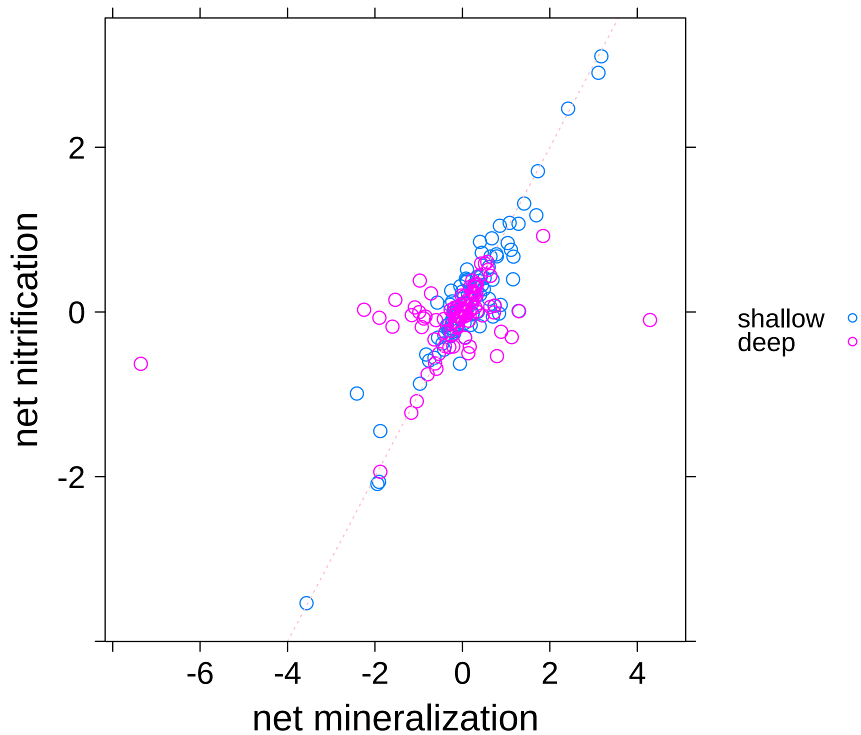


Figure 3.7 Relationship between net nitrogen mineralization and net nitrification for shallow and deep halves of soil cores. Units are μg N per day per gram dry soil.

The average September and December 2015 mineralization rates were 0.11 μg N per day per gram dry soil. Using the generally accepted soil bulk density value of 1.5 g/cm^3 (Grover et al., 2013), this translates to 32 μg N m^{-2} h^{-1}.

3.3.2 Soil Gas Emission

Soil N_2O gas emissions overall averaged 35 μg N m^{-2} h^{-1}, with a maximum of 149 μg N m^{-2} h^{-1}, and one negative flux of -78 μg N m^{-2} h^{-1}. As seen in Figure 3.8, the two sites were not statistically different from each other (p=0.959) but the six measured summer fluxes were statistically greater than the ten fall fluxes (p = 0.042). There were neither significant correlations with 3-day antecedent rain (p=0.515) nor with temperature (p=0.081).

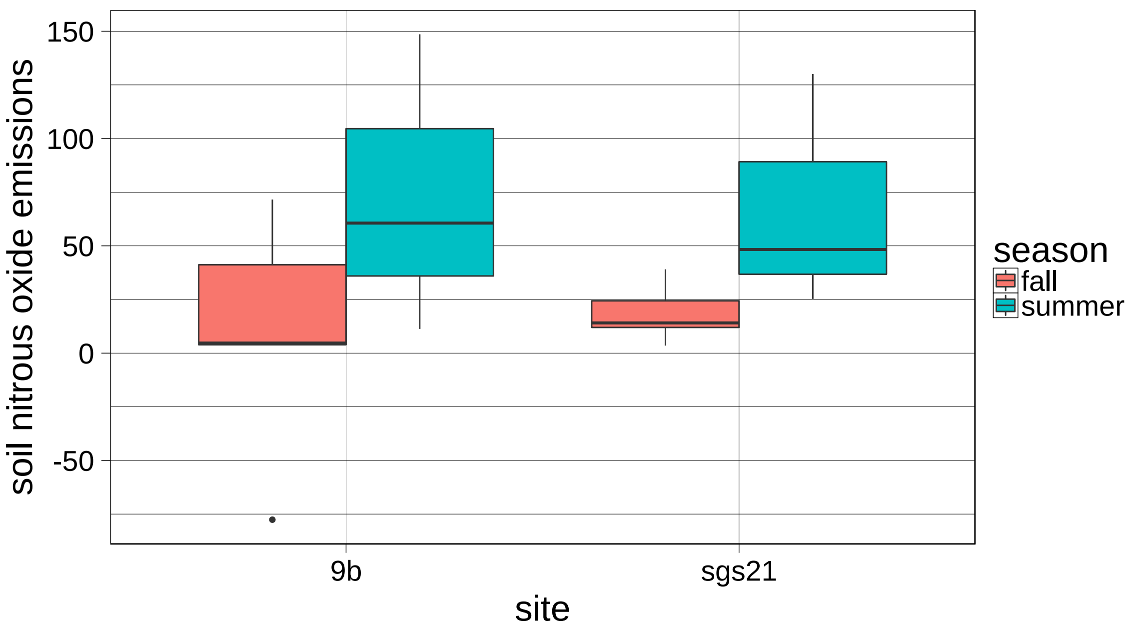


Figure 3.8 Soil Nitrous Oxide Gas Emissions (ug/m^2*h^{-1})

3.3.3 Plant Uptake

The woody plant species, including *R.* 'Radrazz' Knockout, *S. tomentosa, A. canadensis, A. melanocarpa,* and *Q. palustris*, had lower foliar nitrogen per rain garden area (Figure 3.9) than the grasses (p=0.0027), which include *P. virgatum*, *P. alopecuroide*, and *L. muscari*, and the perennials (p= 0.0003), which include *E. purpurea*, *H.* 'Lady Florence', *S. novae-angliae,* and *N. faassennii*. The grasses and perennials had statistically similar foliar nitrogen per area (p=0.3711).

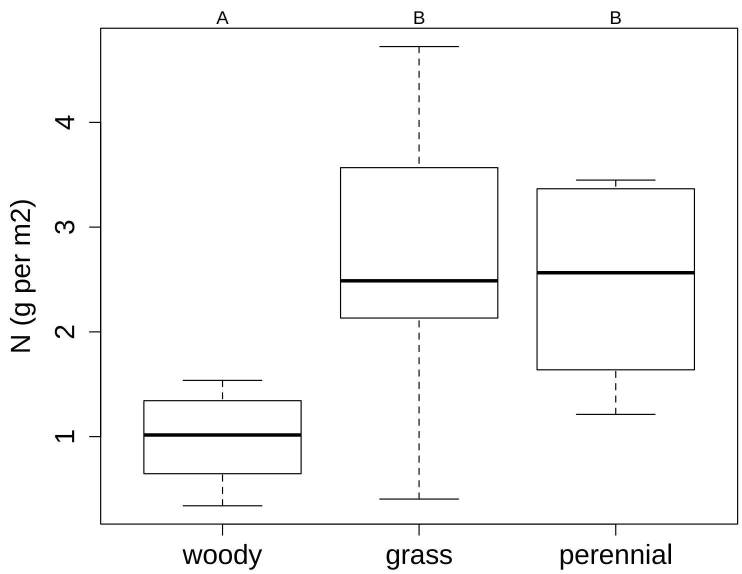


Figure 3.9 Foliar nitrogen per area

The woody plants also had lower nitrogen on a per leaf area basis than the grasses ($p<0.0001$) and the perennials ($p<0.0001$). Woody leaves had an average C:N ratio of 28.1, which was greater than the grasses average ratio of 20.9 ($p=0.0003$), and the perennials average ratio of 13.6 ($p<0.0001$).

3.4 Discussion

3.4.1 Soil

3.4.1.1 Soil Characterization

The increase in nitrogen and phosphorus from the one initial pre-construction sample to the *in-situ* samples at the shallow and deep depths (Table 3.2) could have resulted from mixing from the plant root-balls. Nursery-grown plants used for rain gardens are typically grown in potting containers with nutrient-rich garden soil, and are transferred from these pots to the soil during planting. The nutrients from the fertile garden soil may not be considered by rain garden site designers and engineers, who generally focus on specifications for the rain garden soil alone. The increase could also have resulted from captured nutrients from stormwater runoff.

The C:N ratios of the in-situ samples may be less than optimal for nutrient removal. They are below 25:1 (Table 3.2), which is considered the critical C:N ratio above which microbes import nitrogen to meet their growth requirements, and below which there may be net nitrogen release from decomposing organic matter (Chapin et al., 2002; Robertson and Groffman, 2015). Soils with greater C:N ratios can remove more nitrogen from stormwater by promoting immobilization over mineralization (Bernot and Dodds, 2005; Liu et al., 2014; E G I Payne et al., 2014).

3.4.1.2 Mineralization and Nitrification

Sampling location produced little variability in soil nitrogen concentrations, as there were few differences between different sites and no trends from the inlet to the outlet. However, shallower soil samples supported greater soil nitrate, mineralization rates, and nitrification rates compared to deeper soil, which might be expected for the following reasons.

Shallow soil is topped with 5.1-7.6 cm of bark chip mulch. Mulch layers encourage soil decomposition, as they remove organic nitrogen from stormwater runoff through sorption processes (Davis et al., 2006). Mulch may support substantial microbe populations to degrade captured organic nitrogen from stormwater runoff, so the captured organic nitrogen and ammonium may sorb to mulch, and then mineralize and nitrify between rain events (Davis et al., 2006).

Decomposing plant matter also increases nitrogen levels in shallow soil, as does stormwater influent. In most soils, the quantity and quality of detrital inputs are the main factors that control rates and patterns of mineralization and immobilization (Robertson and Groffman, 2015). In the long term, fertilization can increase mineralization of nitrogen four-fold (Raison et al., 1987). The increased nitrogen from decomposing plant matter or from stormwater runoff could be increasing the soil nitrate and the mineralization and nitrification rates.

The difference between shallow and deeper soils may also be due to a more conducive environment for nitrogen decomposition on the top of the soil. Surface soils in undisturbed areas tend to contain faster cycling carbon, whereas subsoils can be dominated by ancient (>2000 year old) carbon that decomposes very slowly, even controlling for soil composition, indicating that the stability is due to soil depth, not chemical composition (Fontaine et al., 2007). Shallow soils have greater microbial biomass and more wildly fluctuating temperature changes, which induces faster decomposition than that found in deeper soil (Sanaullah et al., 2011). Likewise, moisture levels and temperature are more stable in the subsoil (Dungait et al., 2012), and deeper soils tend to support less carbon and nitrogen (Li and Davis, 2014). There are also increased oxygen levels closer to the surface, supporting nitrification which is an aerobic process. Other rain garden monitoring studies found most total nitrogen (Zhang et al., 2011) and nitrate (Elliott et al., 2011) to be produced in shallow soils, suggesting that nitrification mainly occurs in the upper soil layers (Elliott et al., 2011). Furthermore, aerobic metabolism of the organic nitrogen at shallow depths may result in greater ammonium and nitrate through enhanced ammonification and nitrification (Davis et al., 2006).

Nitrogen cycling breaks down organic matter into nitrate and ammonium; nitrate is easily washed away by rainwater, whereas ammonium tends to stick to the soil. For this reason, measuring ammonium levels may give us a more accurate picture of the nitrogen cycling level overall, although both nitrate and ammonium are considered indices of nitrogen cycling (Davidson et al., 2000). We found rates of ammonium to increase in the summer (Figure 3.5), indicating that warmer weather encourages the mineralization process (Auyeung et al., 2013; Nadelhoffer et al., 1984). A similar trend is visible in grassland ecosystems, which show increased ammonium (and nitrate) in the summer and decreased concentrations in the fall (Kastovska et al., 2015). Conversely, dry weather favors the nitrification process; data from the dry periods of September 2015 and December 2015 show that after a week's incubation, the soil has decreased ammonium levels and increased nitrate levels. Still, the increase in nitrate outweighs the reduction in ammonium, indicating continual production of ammonium through ongoing decomposition of organic nitrogen.

Nitrate accumulated during the dry incubation periods, and was washed away during rainy incubations. This illustrates nitrate's "leaky" nature as a highly mobile anion that does not bind to soil particles and so is readily washed out (Davis et al., 2006; Robertson and Groffman, 2015), potentially contaminating groundwater (Collins et al., 2010). Nitrate is the most common drinking water pollutant (Groffman et al., 2002) and the most difficult form of nitrogen to address (Davis et al., 2006).

Compared to ammonium levels, levels of nitrate fluctuated greatly through high rates of accumulation and leaching as seen in Figure 3.7. The bottom left of Figure 3.7, shows a 1:1 relationship between mineralization and nitrification rates, indicating only nitrate levels to be decreasing, (as nitrate is washed out by rain water), whereas ammonium is stable. Ecosystems tend to lose nitrogen by both denitrification and leaching after nitrification but before uptake by plants (Robertson and Groffman, 2015).

The tight correlation between net mineralization and net nitrification suggests a nitrogen saturated environment. Our research shows that in urban rain gardens, nitrogen accumulates as nitrate, rarely as ammonium. Net nitrification is typically 100% of net mineralization in nitrogen-saturated tropical ecosystems (Vitousek and Matson 1988) or agricultural systems (Schimel and Bennett, 2004), while only a small fraction of net mineralization in temperate ecosystems (McNulty et al 1990), because nitrifying microbes thrive in nitrogen-saturated environments, but compete poorly for ammonium against plants and heterotrophic microbes. Additionally, in nitrogen-saturated environments, nitrifiers live in close association with mineralizers, making nitrate the dominant nitrogen form moving through the soil, with plants shifting to relying on nitrate for nitrogen (Schimel and Bennett, 2004).

Our data suggest that the soil is supplying much more nitrogen than the plants actually need. Gross rates of nitrification exceed rates of nitrate uptake by plants and microorganisms, resulting in the accumulation of nitrate; this indicates that excess nitrogen is cycling relative to the ability of plants and microbes to assimilate it (Davidson et al., 2000).

Li and Davis did not account for mineralization of in-situ soil organic nitrogen in their pioneering nitrogen mass balance of rain gardens (Li and Davis, 2014); while they noted that rain gardens increase nitrogen levels, they assumed that the source of this increase was organic material deposited via stormwater runoff. Were this to be the case, levels of nitrogen would be higher close to the drainage inlet, where organic matter would be collected. However, since we found no differences in nitrogen levels related to proximity to the inlet or the outlet, we believe that we are measuring mostly the turnover of soil organic matter rather than organics from stormwater runoff. Therefore, conceiving of rain garden nitrogen dynamics by solely considering the levels of nitrogen in stormwater before and after passing through the rain garden, as conducted by previous researchers (Collins et al., 2010; Li and Davis, 2014) is an inadequate model for quantifying nitrogen cycling.

3.4.2 Soil Gas Emission

We found greater nitrous oxide emissions during summer than during fall. This increase may be due to the connection between warmer temperatures and high rates of denitrification, which produces nitrous oxide gas (Hatt et al., 2009; Livesley et al., 2010). High levels of ammonium in the soil leads to increased denitrification (Groffman et al., 2002), so the summer's increase in ammonium (Figure 3.5) possibly further increased summer gas emissions of nitrous oxide.

Overall, the levels of nitrous oxide emitted by our rain gardens were similar to rates found in similar studies around the world: our mean reported N_2O flux of 35 μg N m^{-2} h^{-1} was in the same range as the mean N_2O fluxes of 13.8 and 65.6 μg N m^{-2} h^{-1} found in two rain garden designs that treat runoff from a parking lot in Melbourne, Australia (Grover et al., 2013). Our fluxes were also similar to the 14.0 μg N m^{-2} h^{-1} found in garden areas in Melbourne and the 27.9 μg N m^{-2} h^{-1} reported for irrigated lawns in Melbourne (Livesley et al., 2010). Our fluxes were greater than the 7.2 and 10 μg N m^{-2} h^{-1} mean N_2O emissions reported for urban lawns in California and Colorado respectively (Kaye et al., 2004; Townsend-Small and Czimczik, 2010), and the 4.6 and 3.0 μg N m^{-2} h^{-1} found in New York respectively at stormwater detention basins (McPhillips and Walter, 2015) and grassed roadside ditches (McPhillips et al., 2016). Our peak emissions of 148.6 μg N m^{-2} h^{-1} was lower than the peak of 1100 μg N m^{-2} h^{-1} found by Grover et al or the 720 μg N m^{-2} h^{-1} found by Townsend, but greater than the 50-60 μg N m^{-2} h^{-1} peak emissions found in the other studies (Livesley et al., 2010; McPhillips et al., 2016), and about the same range as other studies (Kaye et al., 2004; McPhillips and Walter, 2015).

3.4.3 Plant Uptake

Compared to perennial plants and grasses, the leaves of woody plant species have lower nitrogen per site area, lower nitrogen per leaf area, and greater C:N ratios, showing a low rate of nitrogen uptake. While Collins et al (2010) suggested that trees and shrubs may remove more nitrogen in the long run due to deeper rooting systems and greater biomass, Turk et al (2016) found that herbaceous perennials outperformed trees and shrubs on nitrogen removal per area. Plants with the best nutrient removal grow rapidly in nutrient-rich environments and have to ability to store excess nutrients (Zhang et al., 2011). Perhaps the grasses and perennials are growing more rapidly, while the shrubs may be storing excess nutrients in woody biomass (Emily G I Payne et al., 2014) not sampled in this study.

We found that foliar nitrogen ranged between 1.0 g/m^2 for woody species to 3.2 g/m^2 for grasses. Our values were much lower than those estimated by others for total plant uptake. Wetland plants may uptake 51 g N $m^{-2}y^{-1}$ (Davis et al., 2006), and Lucas and Greenway found that nitrogen uptake ranged from 51-65 g N $m^{-2}y^{-1}$ (2011).

3.4.4 Overall Nitrogen Mass Balance

We found that soil decomposition amounts to 32 g N $m^{-2}y^{-1}$ per year, gas losses are 3 g N $m^{-2}y^{-1}$, and plant uptake is 16 g N $m^{-2}y^{-1}$. We compared these measurements to nitrogen concentrations found in stormwater runoff as reported for five of these seven sites in Shetty et al (2016), to create an overall nitrogen mass balance. The median nitrogen concentration of stormwater influent was 1.6 ppm N, while infiltrate had 2.7 ppm N. In Figure 3.10, we assume an average annual rainfall of 1.29 m (Carson et al., 2013) and estimate the amount of nitrogen in stormwater runoff by pairing the reported stormwater concentrations to the 40% median water retention and the drainage area: site area ratio of 135 we found at ROWB 9B as described in Chapter 2.

We compared the amount of nitrogen in stormwater runoff to our measured soil, gas, and plant phase fluxes by approximating with the following assumptions. Soil decomposition was estimated with data from September and December months only, while soil decomposition may vary seasonally. Although we only sampled plant leaf tissues above ground and did not sample roots, Zhang et al found that plants stored a similar amount of nitrogen above and below ground in terms of both concentration and accumulation (Zhang et al., 2011). We therefore doubled our values in Figure 3.10 to account for belowground plant nitrogen uptake. Furthermore, plants may directly recapture nutrients from their leaves before they fall during autumn senescence, and the soil may indirectly recapture leaf nutrients after the leaves fall off and decay before they are collected by maintenance staff. We approximate that leaf tissues are promptly removed by maintenance staff in our estimates by assuming that foliar nitrogen is removed from the system. Similarly, we only sampled soil nitrous oxide gas emissions, while soil nitrogen gas emission also includes nitric oxide and nitrogen gas. Nitrous and nitric oxide emissions may be a similar order of magnitude, and equal when the soil is at field capacity (Davidson et al., 2000). We similarly assume that nitrogen gas was a similar magnitude and tripled our nitrous oxide emission values in Figure 3.10 to account for all three forms of nitrogen gases.

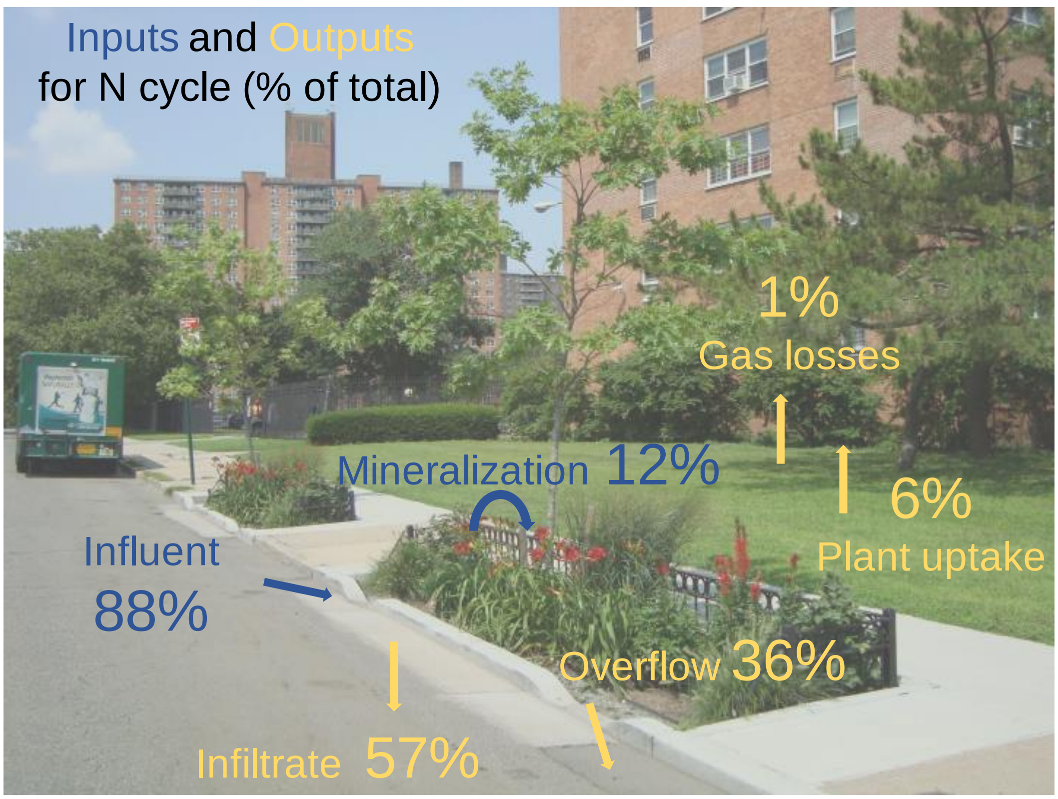


Figure 3.10 Approximate overall nitrogen mass balance with site ROWB 9A in the foreground and site ROWB 9B in the background.

Roughly 88% of nitrogen inputs enter a rain garden each year via stormwater runoff influent (Figure 3.10), while 12% is made available via local soil decomposition. 6% of these inputs are assimilated into leaf tissue, 1% is lost via soil gas emissions, 36% overflows back to the sewer system, and 57% infiltrates into the ground.

While we acknowledge that our simplifying assumptions drastically limit the precision of our overall estimate, even to approximately quantify each nitrogen flux within rain gardens elucidates the following patterns.

The rain garden nitrogen cycle is dominated by liquid and soil phase fluxes, with relatively much smaller fluxes in the gas and plant phases. In contrast to natural ecosystems, where external inputs of nitrogen are about 10% of the amount of nitrogen that annually cycles (Chapin et al., 2002), stormwater inputs of nitrogen in the studied urban rain gardens constitute about 88% of the nitrogen that annually cycles. We found that gas emissions were a small portion of the overall mass balance, which was similar to Payne et al, who found that N_2O emissions were less than 1.5 % of the incoming nitrogen amount (Payne et al., 2017). We also found that plant uptake was a smaller component of the nitrogen mass balance compared to others (Davis et al., 2006; Lucas and Greenway, 2011; Zhang et al., 2011).

Our data indicate that stormwater runoff contains significant amounts of nitrogen, and far more than needed for plant uptake. Detailed design guidance is not available for rain gardens (Davis et al., 2009), and high organic matter contents are frequently specified for rain garden soil, intended to aid plant growth rather than improve water quality (Hunt et al., 2012). However, our data suggest that stormwater inputs of nitrogen are more than enough to promote healthy vegetation. Nutrient-poor soil media and soil carbon enhancements such as coconut coir pith as a replacement for compost (Herrera, 2015) might enhance long-term net immobilization in order to reduce nutrient leaching.

3.4.5 Limitations

The act of taking net mineralization measurements has the side effect of cutting soil roots with the sharpened edge of the soil corer, eliminating plant uptake of nitrogen sources, which alters the nitrogen system, enabling higher microbe uptake and lower net mineralization (Schimel and Bennett, 2004). In addition, the soil core may sever roots that nevertheless may continue to take up nutrients within the core, again resulting in underestimates of net mineralization (Raison et al., 1987; Robertson et al., 1999). On the other hand, the incubated core may have slightly greater soil moisture than the rest of the site (Hart et al., 1994; Schimel and Bennett, 2004), potentially leading to overestimates of the decomposition rate (Sanaullah et al., 2011). Ultimately, no method for assessing soil nitrogen provides an unequivocal estimate (Hart et al., 1994). Our intact soil core method may reduce soil disturbance compared to other field methods of quantifying soil mineralization, which tend to be more reliable than laboratory incubations (Raison et al., 1987).

Our mineralization data were also limited by periodic rains washing away accumulated nitrate, preventing a clear measurement of inorganic nitrogen changes for all incubation periods with the exception of September and December 2015. However, this condition allowed us to contrast the fluctuating changes of soil nitrate due to leaching with the more gradual seasonal changes of soil ammonium. Future research could quantify such washouts by sealing off the top and bottom of incubated soil cores with ion exchange resin bags, which prevent loss of ammonium and nitrate (Hart et al., 1994).

3.5 Conclusions

We quantify soil mineralization in rain gardens and find evidence that rain garden soil is nitrogen saturated. Our data suggest that plant nitrogen needs are minimal compared to sources of nitrogen in stormwater, and rain garden soil could better account for the overabundance of nutrients within stormwater, in order to reduce nutrient leaching. Due to this finding we would advocate for more carbon based organic matter used in soil mixes in order to reduce nitrogen pollution. Excess nitrogen is not only linked with increased nitrogen pollution, but even with declines in forest productivity and increased tree mortality (Chapin et al., 2002). When planted in a nutrient poor soil, native plants may also better outcompete weeds, potentially reducing maintenance (Levin and Mehring, 2015). Our recommendations may support healthier plants with reduced maintenance and nutrient pollution.

Chapter 4: Comparing Two Sedum Green Roofs to a "Next Generation" Native System with Irrigation and Smart Detention

Abstract: The goal of this study is to measure the effects of a "next generation" green roof on water retention: we compare two "industry-standard" green roofs planted with *Sedum*, a drought-tolerant succulent plant, to a native green roof actively irrigated with smart sensors linked to a runoff detention system. We evaluate four years of climate and runoff data from each roof to determine water retention for different storm sizes and seasons. We model long-term water retention by developing empirical relationships between rain and runoff at each roof, and then applying each relationship to historical rainfall data. We then use the empirical relationships, along with local climate data, to model evapotranspiration with seasonal crop coefficients for native and succulent vegetation at the three roofs. We find that despite being irrigated, the native green roof has greater water retention (63%) than the two unirrigated green roofs planted with drought-tolerant *Sedum* vegetation (53%, 42%). The native roof also has higher crop coefficients (1.12, 1.13) than the *Sedum* roofs (0.56, 0.57), indicating that the native plants can transpire more stormwater for given climate and soil moisture conditions. We also find the native roof's 63% stormwater capture increased to 71% with the use of a smart runoff detention system. Our study indicates that native green roofs that are actively controlled with smart sensors and detention systems will capture more stormwater than industry-standard green roofs, which are planted with drought-tolerant succulent vegetation and are passively controlled.

4.1 Introduction

Urban stormwater runoff from impervious surfaces reduces water quality and ecological diversity in surrounding streams (Walsh et al., 2005). As rooftops represent about half of the impervious surfaces in some cities (Vanuytrecht et al., 2014), the implementation of stormwater management techniques on roofs can help improve the health of urban water bodies. Roofs with a thin layer of vegetation, known as green roofs, can annually retain 30-86% of rain that falls on them (Li and Babcock, 2014). Because the USA has no national standards for green roofs, green roof materials and configurations vary widely across the country (Carson et al., 2013). Nonetheless, most “industry-standard” green roofs in temperate North America are unirrigated and planted with drought-tolerant succulent vegetation. In this paper, we explore how water retention in industry-standard green roofs compares to a “next generation” native green roof system that incorporates sensor controlled irrigation and “smart” on-site stormwater detention.

4.1.1. Industry Standard (Sedum) and Native Green Roofs

The conditions on green roofs differ from those in native habitats on the ground: roofs are colder in the winter, hotter in the summer, and prone to rapid substrate drying (Butler et al., 2012). Therefore, green roofs are typically planted with desert vegetation such as *Sedum* (Heim et al., 2017; Lundholm et al., 2010), which are drought-tolerant and capable of surviving harsh rooftop conditions (Heim et al., 2017; Poë et al., 2015; Vanuytrecht et al., 2014). Since most *Sedum* species are native to Europe or Asia, however, there is an increasing push to identify native North American species capable of withstanding a roof’s harsh climate (Heim et al., 2017). The Sustainable Sites Initiative certification program created by the United States Green

Building Council, for example, awards points for green roof projects that include native plants (Butler et al., 2012).

Native plants not only enhance biodiversity (Lundholm et al., 2010; Vanuytrecht et al., 2014); some studies indicate that they might also help a green roof retain more stormwater than *Sedum* plants (Aloisio et al., 2016; Li and Babcock, 2014; Nagase and Dunnett, 2012; Whittinghill et al., 2014). Succulent plants such as *Sedum* use crassulacean acid metabolism photosynthesis, where plants transpire at night and close stomata during the day in order to reduce water loss. In contrast, native species transpire more water in between rain events, potentially creating more space in a green roof's substrate or soil-layer for rainwater storage (Nagase and Dunnett, 2012).

While some studies point to increased stormwater retention for native green roof species, others found no significant differences in hydrologic performance between *Sedum* and native plants (Graceson et al., 2013; Heim et al., 2017; Soulis et al., 2017; Stovin et al., 2015). In fact, some researchers reported that *Sedum* had greater evapotranspiration (ET) and helped retain more stormwater runoff than a native treatment (Stovin et al., 2015). Perhaps due to this uncertainty, engineers remain hesitant to promote native plants, in contrast to the general trend among architects, landscape architects, and biologists to increase their usage (Butler et al., 2012). Further research to understand how vegetation affects green roof stormwater capture is needed to reduce uncertainty on this topic (Nagase and Dunnett, 2012; Poë et al., 2015; Schroll et al., 2011; Stovin et al., 2015). In particular, crop coefficients for actual evapotranspiration (AET) models need to be refined for different green roof vegetation options to help quantify the effect of different vegetation on ET rates, and thus green roof hydrologic performance (Hardin et al., 2012; Schneider et al., 2011; Voyde, 2011).

4.1.2. Irrigation Systems in Green Roofs

In temperate climates, plants can only survive summer rooftop conditions if they are drought-tolerant or are provided irrigation during dry periods (Van Mechelen et al., 2015). Since moist soils retain less water, it is commonly believed that irrigation reduces green roof stormwater retention (Schroll et al., 2011; Van Mechelen et al., 2015; Volder and Dvorak, 2014; Whittinghill et al., 2014) and that unirrigated shallow roof systems provide greater stormwater benefits than irrigated green roofs (Volder and Dvorak, 2014).

4.1.2.1. Benefits of Irrigation:

Irrigation has been found to improve plant survival and increase the range of plant species capable of surviving harsh rooftop conditions (Macivor et al., 2013; Van Mechelen et al., 2015). Irrigation also increases cooling and evapotranspiration (ET) (Hardin et al., 2012; Van Mechelen et al., 2015). As modeled by Soil Moisture Extraction Functions (SMEFs) (Zhao et al., 2013), the relationship between actual ET and potential ET (PET), which is defined as ET/PET, instantly decreases when soil moisture is less than maximum (Hardin et al., 2012).

Draining green roofs to cisterns, that in turn provide water to irrigate the roof during dry periods, has increased total evapotranspiration compared to roofs without irrigation (Carson, 2014; Hardin et al., 2012). Hardin et al (2012) experimentally found that the water retention performance for a green roof in Florida increased from 43% annual retention to 83% annual retention with a cistern sized to store a 127 mm (5 in.) storm.

4.1.2.2. Drawbacks of Irrigation:

Some consider unirrigated green roofs more sustainable than irrigated systems due to reduced maintenance and water use (Van Mechelen et al., 2015). While green roofs that drain to on-site cisterns can irrigate with re-used stormwater runoff, they still typically require a supplemental potable water source for periods when there is insufficient water in the cistern to maintain vegetative health (Hardin et al., 2012).

Since moist soils cannot retain as much water as dry soils, it is not surprising that many studies have found that irrigation reduces green roof water retention (Schroll et al., 2011; Van Mechelen et al., 2015; Volder and Dvorak, 2014; Whittinghill et al., 2014). Li and Babcock (2014) submit that since *Sedum* is drought-tolerant, the lower moisture content of unirrigated green roof soil planted with *Sedum* will absorb more rain and compensate for lower plant performance. However, studies have not considered if irrigation may compensate for this reduction in water retention by promoting more robust vegetation with enhanced evapotranspiration and canopy interception.

4.1.3. Smart Detention

Rainwater harvesting (RWH) (Montalto et al., 2007) describes collecting rooftop stormwater runoff in tanks and then reusing the water. If water stored in a RWH system is not consumed or emptied in between rain events, the system will not be effective for stormwater management.

This is because it remains full from prior storms, and is thus likely to overflow during the following rain event (Debusk et al., 2012; Shannak et al., 2014).

Real-time control sensors, such as those manufactured by OptiRTC and connected to weather forecasting data (Debusk et al., 2012; Goodman and Quigley, 2015; Kerkez et al., 2016), allow capture of more stormwater than traditional RWH systems via controlled releases that lower RWH tank water levels before rains begin. By ensuring that tanks contain sufficient capacity during storms, real-time control sensors can guarantee the wet-weather performance of RWH systems. However, further research is required in order to determine how often and how much a tank should be drained (Tsang and Jim, 2016; Van Mechelen et al., 2015) and to quantify variability in the resulting stormwater benefits (Kerkez et al., 2016).

4.1.4. Study Objectives

Our overall goal is to investigate factors that show potential to maximize the stormwater benefits of green roofs. We do so by analyzing four years of rainfall and runoff data from three full-scale extensive (depth < 150 mm) green roofs, including two non-irrigated *Sedum* roofs and one irrigated native roof with a smart detention system. We compare water retention seasonally for different storm sizes and we develop an empirical relationship between rain and runoff to model long-term water retention. We then consider how water performance from each roof was affected by vegetation, irrigation, and smart detention. Finally, we distinguish contributions to green roof stormwater retention in natives and in *Sedum* by modeling ET with seasonal crop coefficients using climate data and soil moisture sensors. We focus on extensive rather than intensive (depth > 150 mm) green roofs in this study, since extensive systems are lightweight, cheaper, lower maintenance, and will likely be the majority of new green roofs that are constructed (Volder and Dvorak, 2014). Our specific objectives are:

1. To compare water retention between an irrigated green roof planted with native vegetation and two unirrigated green roofs with drought-tolerant vegetation;
2. To provide seasonal crop coefficients that distinguish evapotranspiration in native vegetation from drought-tolerant vegetation, and
3. To quantify the effect of real-time-control software on long-term stormwater capture.

4.2. Materials and Methods

4.2.1. Green Roof Sites and Instrumentation

The Ranaqua roof is 638 m^2 and located above a New York City (NYC) Department of Parks & Recreation (NYC Parks) auto garage in the Bronx, NYC (Figure 4.1). In October 2012, NYC Parks installed a variation of American Hydrotech Inc. Garden Roof Assembly on three quadrants (quads) of the roof, each draining a separate drainage area, while one quad remained bare as a control. The three vegetated quads have a profile of 127 mm of soil/ substrate underlain by a 0.25-mm non-woven polymeric filter fabric, a 15-mm drainage layer made of polyethylene cups, and a 10-mm root barrier. The growing substrate on quad 1 is an American Hydrotech soil blend known as LiteTop with 3-6% organic content and 32% maximum media water retention. Quad 4, alternately, has Norlite Expanded Shale aggregate and We Care Compost, with 5-8% organic content and 38% maximum media water retention, but aside from the soil was designed identical to quad 1. Quads 2 and 3 were not analyzed in this study as will be explained in section 4.2.2.2.

Figure 4.1 Ranaqua green roof is located above a NYC Parks auto garage in Bronx, NYC

Non-vegetated walkways that are 61 cm wide surround each quad and each rooftop HVAC unit with Delaware river gravel. Aluminum barriers were installed with roofing cement along high points to ensure that no stormwater runoff flows between quads. Prior to construction, a topographical survey found roof slopes toward each drain to be 1.04%.

Stormwater runoff from each quad is drained into individual 1892 L water tanks located beneath each quad in the auto garage. The un-vegetated "control" quad (quad 3) drains into two such tanks (Figure 4.2). An HRXL-Max- Sonar-WR #MB7360 MaxBotix Inc. acoustical sensor records the water level in each tank, allowing determination of stormwater runoff during storms based on increases in the water level. Overflow from each tank is measured with a V-notch weir.

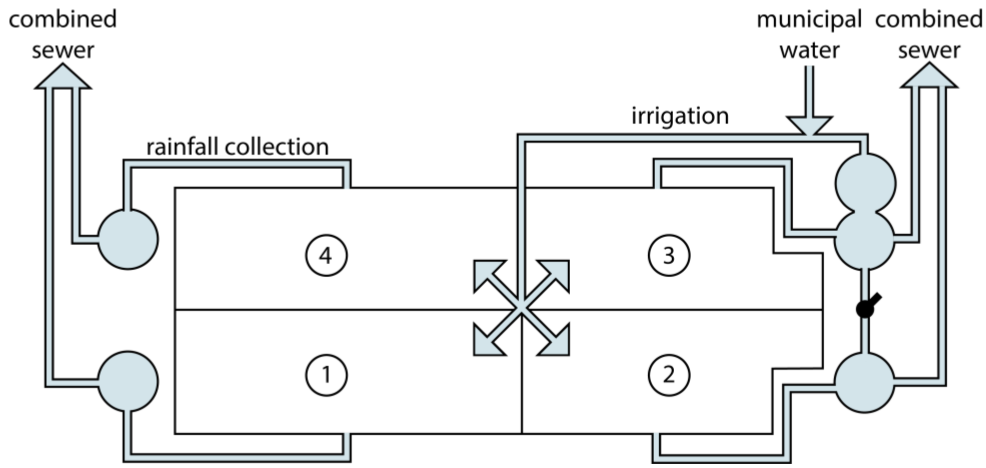


Figure 4.2 Each quadrant (quad) drains to an individual cistern located beneath the roof. Quad 3 has a double cistern, cistern 3. A valve is open, hydraulically connecting cisterns 2 and 3; cisterns 2 and 3 irrigate the entire green roof during warmer months (roughly June to October). When they are empty, irrigation is switched to municipal potable water

Climate data is measured with an Onset Hobo U30 weather station located on the un-vegetated quad. The weather station records data from a TR-525i Texas Electronics tipping bucket rain gage, an S-THB- M002 Onset temperature and relative humidity sensor, and three 10HS Decagon Devices soil moisture sensors, one buried in each vegetated quad.

The performance of this green roof at Ranaqua was compared with the performance of two green roofs planted with *Sedum* vegetation and located in Manhattan: one termed 118th, which has 32 mm of substrate and was built in 2007, and another, termed USPS, that has 100 mm of substrate and was built in 2009. Both green roofs contain Onset Hobo U30 weather stations that record rainfall from tipping bucket rain gages and roof runoff from custom designed weir devices. The custom weirs were designed to fit above existing roof drains and include a V-notch weir and an ultrasonic sensor. 118th has a CS615 Campbell Scientific soil moisture sensor and USPS has a EC5 SMC-005 Onset soil moisture sensor. Carson et al (2013) contains a full description of the 118th and USPS green roofs, instrumentation set-up, calibration and monitoring protocols.

4.2.2. Experimental Site (Ranaqua)

4.2.2.1. Vegetation

Each vegetated quad at Ranaqua was divided further into three zones based on plant irrigation requirements (Figure 4.3): a wet zone planted with *Symphyotrichum novae-angliae, Verbena hastata, Scirpus cyperinus, Carex vulpinoidea Michx., Dichanthelium clandestinum, Elymus virginicus, Eupatorum maculatum, Eupatorium perfoliatum, Euthamia graminifolia, Helenium autumnale, Juncus tenuis, Monarda fitulosa, Eragrostis spectabilis, Rubus Flagellaris, Parthenocissus quinquefolia*, a medium zone planted with *Eupatorium serotinum, Pycnanthemum tenuifolium, Pycnanthemum virginianum, Solidago juncea, Symphyotrichum laeve, Symphyotrichum pilosum,* and a dry zone planted with *Andropogon virginicus, Danthonia spicata, Euthamia tenuifolia, Panicum virgatum, Solidago nemoralis, Sorghastrum nutans, Tridens flavus, Schizachyrium scoparium.*

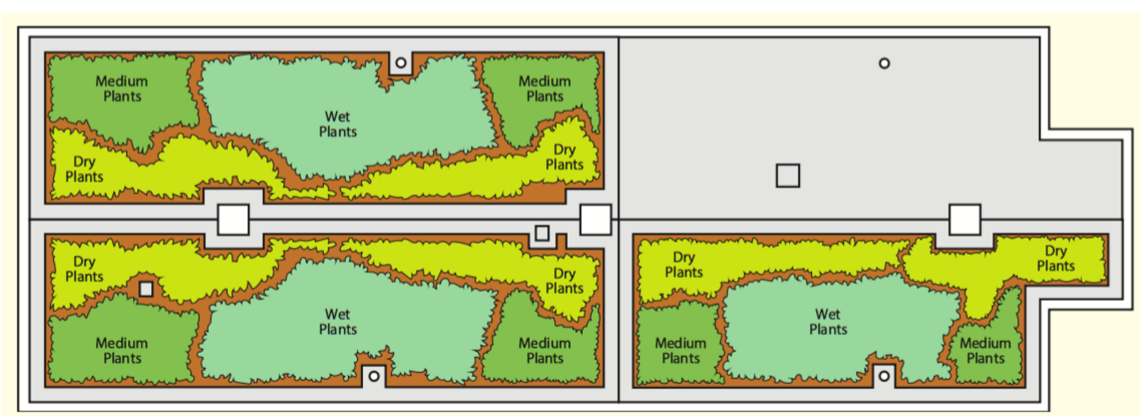


Figure 4.3 Location of the wet, medium, and dry planting zones for irrigation. Drawing provided by NYC Parks.

These plants were selected from native communities expected to resemble the windy and limited-soil depth conditions found on green roofs. Many of them are native to either the Rocky summit grassland community found on mountains throughout New York State, or the Hempstead Plains grassland community found on Long Island (Edinger et al., 2014).

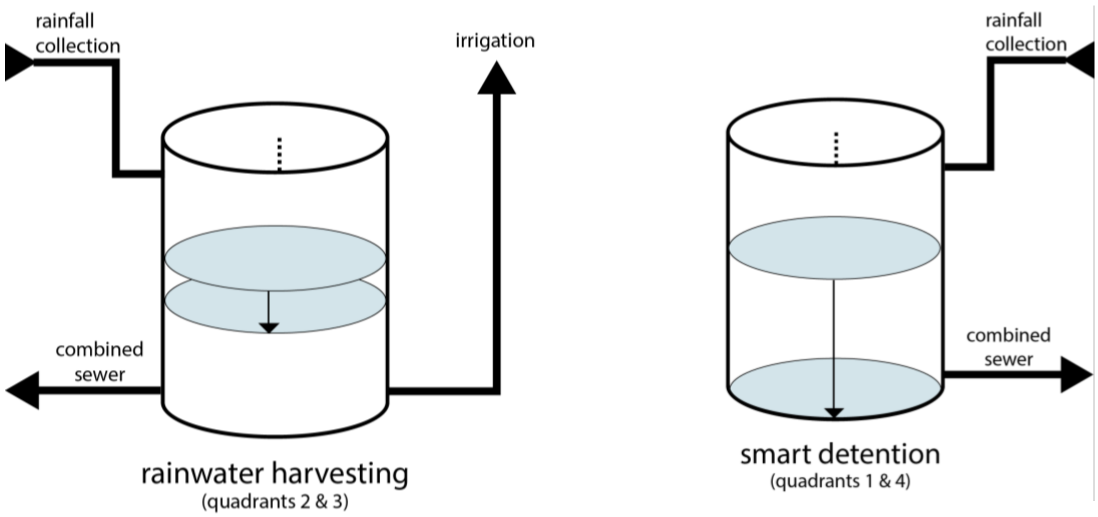
4.2.2.2. Irrigation

Irrigation was installed in September 2014. Irrigation was applied almost daily during summer months at a maximum of 19 mm/week in the wet zone, 13 mm/week in the medium zone, and 8 mm/week in the dry zone. Each zone has an NLCSMS100 Netafim soil moisture sensor programmed to irrigate when volumetric moisture content drops beneath 14%, which was the Allowable Depletion (George et al., 2000) selected for the roof. The 10HS Decagon Devices soil moisture sensors described in Section 4.2.1, rather than the Netafim soil moisture sensors, were used for analysis because they were located approximately at the boundary between the wet, medium, and dry zones and consequently may be more representative of each entire quad.

All irrigation water is collected from the two tanks that drain the un-vegetated quad (3), and when these tanks are empty, irrigation switches to municipal potable water (Figure 4.2). In order to reduce potable water use, the building management opened a valve that hydraulically connects tanks 2 and 3. Since the tanks consequently have the same water level height, we cannot isolate stormwater runoff from quads 2 and 3 and, as a result, we do not investigate the green roof performance of quads 2 and 3 in this study. We also cannot quantify the substantial (Hardin et al., 2012) stormwater benefits caused by irrigating with captured rainwater from these quads. In other words, we cannot factor irrigated water into runoff measurements because we cannot quantify runoff from quads 2 and 3. We therefore analyze water performance for quads 1 and 4 by treating them as irrigated green roofs, without distinguishing whether irrigation water comes from potable sources or from captured and re-used stormwater runoff.

4.2.2.3. Smart Detention

To further improve stormwater benefits beyond irrigation, the tanks are programmed with smart control sensors from Opti RTC that communicate with local weather forecasting data in order to drain tank volumes before expected storms (Kerkez et al., 2016). From March 2013 to September 2014, the tank controller was programmed to estimate the predicted volume of stormwater runoff, and drain just that predicted amount from the tanks, while leaving the remaining amount available for irrigation (Figure 4.4a). In September 2014, when the irrigation system was installed, tanks 1 and 4 were re-programmed to drain empty 24 hours after a rain ends (Figure 4.4b). This arrangement reduces combined sewer overflows by reducing the peak flow when the sewer system is at capacity, while keeping the tanks empty as long as possible. In July 2015, Opti controllers underwent a major upgrade to “OptiNimbus 1.0” which included improvements in rain forecasting.



(**a**) (**b**)

Figure 4.4 (**a**) Cisterns 2 and 3 follow "rainwater harvesting" logic - If there is greater than 60% forecast of rain, tanks 2 and 3 drain just the predicted volume, while saving remaining volume for irrigation; (**b**) Cisterns 1 and 4 follow "smart detention" logic – they drain empty 24 hours after a rain event.

In this study, runoff from the Ranaqua green roof for each storm is calculated based on the increase in cistern height as well as any overflow, as these flows represent water that was not absorbed by the green roof:

$$\text{Ranaqua runoff} = \text{increase in cistern height} + \text{overflow from cistern} \tag{4.1}$$

Ranaqua's smart detention performance (termed Ranaqua SD) for each storm is calculated based on the overflow alone, since water that only increases the tank height does not leave the facility and enter the sewer system during wet-weather:

$$\text{Ranaqua SD runoff} = \text{Ranaqua runoff} - \text{increase in cistern height} \tag{4.2}$$

Ranaqua SD runoff is zero unless the tank overflows during the storm.

4.2.3. Analysis

4.2.3.1. Stormwater Capture

During a four-year monitoring period between March 1st, 2013, and March 1st, 2017, we found water performance data for 427 storms at the Ranaqua green roof. The depth of rain was determined by discretizing the continuous rain gage data into storms using a minimum six-hour dry period to separate storms (Berretta et al., 2014; Carson et al., 2013; Fassman-Beck et al., 2013). We then eliminated storms for which the rain gage time stamp did not match the time when cistern depth changed (10 storms), where blank cistern value readings indicated sensor errors (2 storms), where the tanks were already full at the beginning of the storm (103 storms), where the runoff depth exceeded the rainfall depth (48 storms), and where rainfall depth exceeded 100 mm (2 storms). This quality control resulted in 262 storms at the Ranaqua green roof suitable for analysis during the four-year period. The USPS green roof was monitored between June 17th, 2011 and May 7th, 2016 and the 118th green roof was monitored between June 29th, 2011 and May 8th, 2016. Quality control of storms suitable for analysis for these roofs is found in Carson et al (2013). A summary of the resulting storms used for analysis is provided in Table 4.1.

Table 4.1 Number of storm events in each size category and season

	Spring	**Summer**	**Fall**	**Winter**	**Total**
118th	**101**	**89**	**82**	**61**	**333**
< 2 mm	39	32	31	20	122

2-10 mm	32	31	23	18	104
10 - 25 mm	14	22	9	14	59
> 25 mm	16	4	19	9	48
USPS	**101**	**55**	**73**	**85**	**314**
< 2 mm	40	16	27	38	121
2-10 mm	36	19	20	20	95
10 - 25 mm	16	13	13	22	64
> 25 mm	9	7	13	5	34
Ranaqua	**81**	**61**	**58**	**62**	**262**
< 2 mm	30	21	20	20	91
2-10 mm	25	20	18	22	85
10 - 25 mm	14	12	9	9	44
> 25 mm	12	8	11	11	42

We developed Characteristic Runoff Equations (CREs) for each green roof (Carson et al., 2013; Mentens et al., 2006) by fitting quadratic empirical relationships between the measured rain and runoff depths. For the CRE regression analysis, we additionally removed storms that produced no runoff (Carson et al., 2013), which resulted in 191 storms for Ranaqua, 188 storms for 118th, and 159 for USPS. This step was necessary to avoid the prediction of negative runoffs for small rainfall events. To determine long-term water retention for each roof, we applied the CREs to 40 years of historical climate data (March 1977- March 2017) from the weather station at LaGuardia International airport downloaded from the National Oceanic and Atmospheric Administration's National Climate Data Center website (www.ncdc.noaa.gov). For each of the

4,120 storms measured at this weather station over the 40 years, we input the measured rain data into the CRE and calculated an estimate for runoff. Runoff was set to zero for rainfall below the x-intercept values of the CREs (Carson et al., 2013). With the measured rain and these modeled runoff results, we then determined percent retention with the following equation:

$$\text{Percent retention} = \frac{\text{rain} - \text{modeled runoff}}{\text{rain}} \tag{4.3}$$

We report percent retention both annually, with cumulative rain and modeled runoff summed for each of the 40 years, and also per event, using measured runoff for different storm sizes grouped by season during the four-year monitoring period.

4.2.3.2. Evapotranspiration

The Hargreaves-Semami equation (Hargreaves and Samani, 1985) was used to model PET. While other potential evapotranspiration models such as Penman-Monteith (Allen et al., 1998) require relative humidity, solar radiation, and wind data, Hargreaves only requires temperature data and the latitude of the site. Despite its limited inputs, it has been found in green roof studies to perform as well or better than more complex models (Berretta et al., 2014; Carson, 2014; Stovin et al., 2013). Potential evapotranspiration using Hargreaves-Semami was calculated using a daily time step (Digiovanni et al., 2013) in the open-source software R using the evapotranspiration package.

Crop coefficients (K_c) were derived by modeling actual ET (AET) with a soil moisture extraction function (Zhao et al., 2013):

$$AET = K_c * PET * \theta / \theta_{max} = Rain + Irrigation - Runoff \quad (4.4)$$

where θ denotes each soil moisture value and θ_{max} denotes maximum water storage for the soil/ substrate type. While there are several forms of the soil moisture extraction function, the form in Equation (4.4) has been used in other green roof ET studies (Hakimdavar et al., 2016; Poë et al., 2015) and only requires the maximum water storage for the soil, a specification typically reported by green roof soil suppliers (Hakimdavar et al., 2016). Soil moisture readings were normalized using the following equation:

$$\theta_{normalized} = (\theta - min) * \theta_{max} / (max - min) \quad (4.5)$$

where max and min denote the maximum and minimum values for each soil moisture sensor recorded during the entire monitoring period.

Crop coefficients for all three roofs were derived using equation (4.4) over a 24-month period from November 18th, 2013 to December 16th, 2015, after removing the month of July, 2015 due to equipment failure. Data from one soil moisture sensor from each quad and roof were averaged for each day and paired with daily PET in order to model daily AET. The total rain, runoff, and irrigation over the entire period were considered when fitting the seasonal and overall crop coefficients.

4.3. Results

4.3.1. Rainfall Retention

Despite containing slightly different soils, Ranaqua quads 1 and 4 had virtually identical runoff for 262 storms over the four-year monitoring period (Figure 4.5).

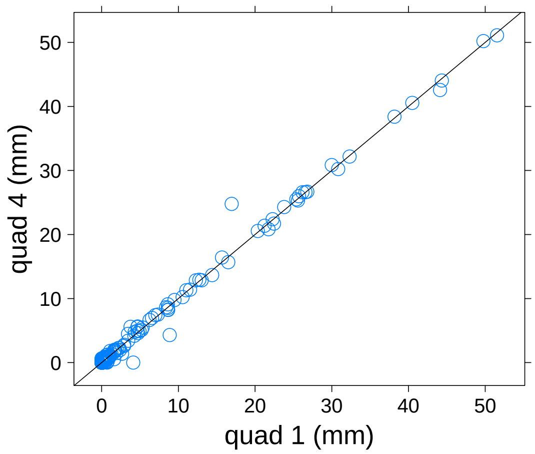
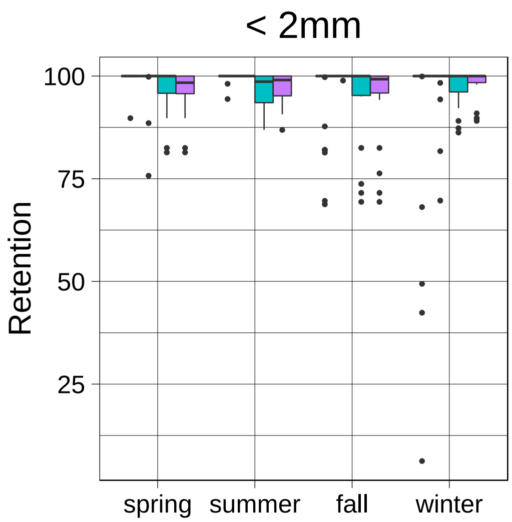


Figure 4.5 Stormwater runoff depths for 262 storms from two quadrants at the Ranaqua green roof with slightly different soil types. The black line represents 1:1.

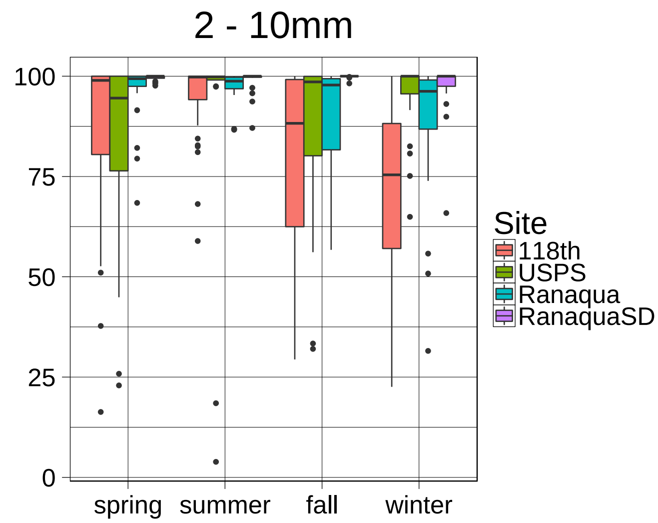
Due to the similarity between the quads, Ranaqua green roof refers to quad 1 data alone for the remainder of this paper.

4.3.1.1. Seasonal Performance

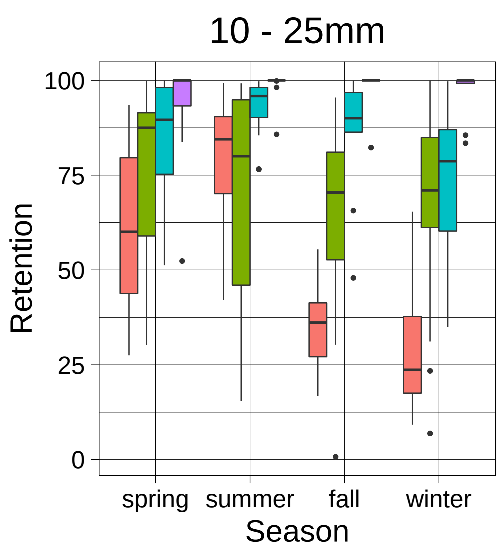
Water retention calculated per event for each roof is plotted seasonally for four ranges of rainfall depths (Figure 4.6). Larger rain depths caused lower retention rates. In the smallest events (Figure 4.6a), retention rates were nearly 100% for all roofs, although Ranaqua had slightly lower retention than 118th and USPS.



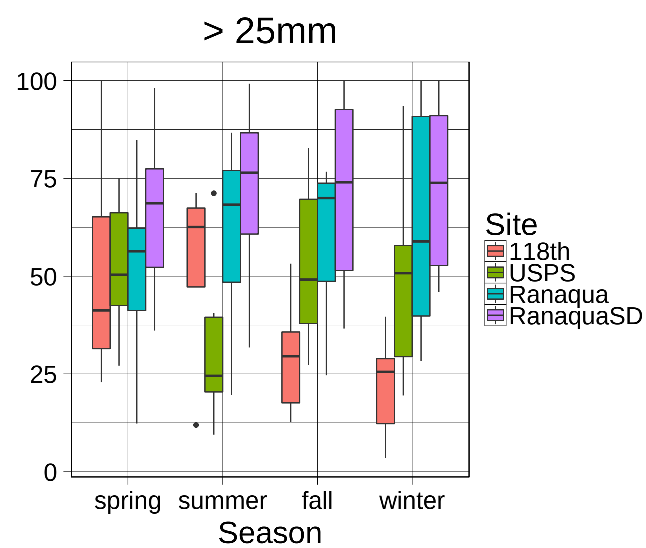
(**a**)



(**b**)



(**c**)



(**d**)

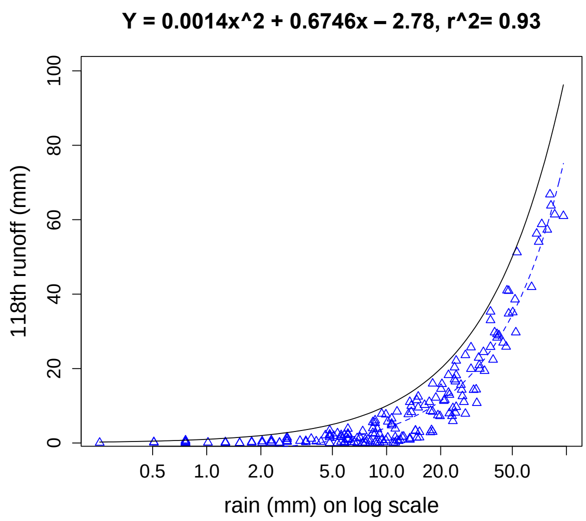
Figure 4.6 Seasonal box plots for different rain depths. (**a**) < 2mm; (**b**) 2-10mm; (**c**) 10-25mm; (**d**) > 25mm

In the 2-10mm group, retention was reduced for 118th, particularly for fall and winter (Figure 4.6b). The strong seasonal trend for 118th continued for the larger storms, as presented in Figure 4.6c and Figure 4.6d. USPS never demonstrated a strong seasonal trend for any storm size.

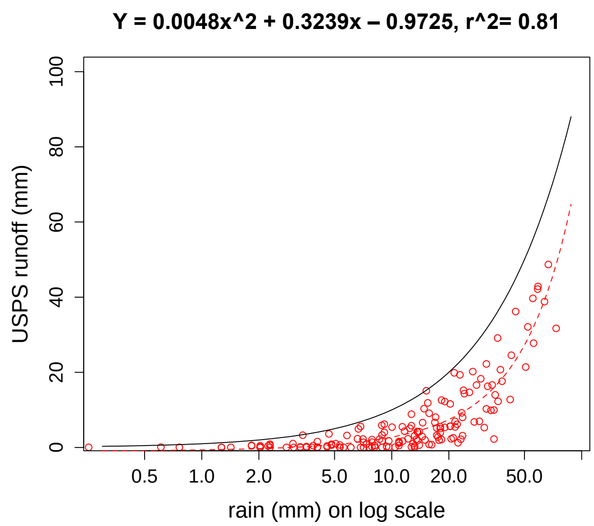
Ranaqua had a seasonal trend and had greater retention than USPS or 118th for storms greater than 10 mm. Smart detention improved overall performance, and the difference between RanaquaSD and Ranaqua was greatest for storm sizes in the 10-25mm range.

4.3.1.2. Long-term Water Retention

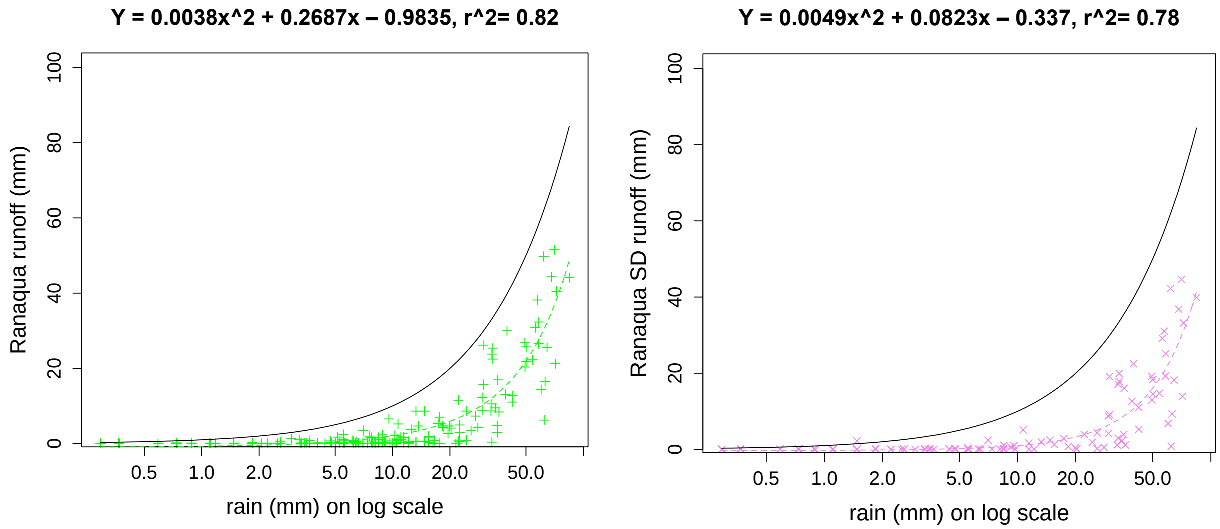
Our CREs for 118th, USPS, Ranaqua, and Ranaqua SD are each presented above corresponding plots of rainfall versus runoff depths in Figure 4.7 for all storms with non-zero runoff, where y denotes runoff depth and x denotes rainfall depth. The CREs are also drawn as dotted lines, while the black lines denote 1:1, representing hypothetical roofs where all rain becomes runoff. 118th appears to attenuate runoff the least, compared to USPS and Ranaqua.



(a)



(b)



(**c**) (**d**)

Figure 4.7 Rainfall versus runoff depths for all storms with non-zero runoff. Characteristic Runoff Equations (CREs) are written above and drawn as dotted lines for (**a**) 118th; (**b**) USPS; (**c**) Ranaqua; (**d**) Ranaqua SD. The CREs are applicable for rain greater than the x-intercepts, which are 4.1mm for 118th, 2.9 mm for USPS, 3.5 mm for Ranaqua, and 3.4 mm for Ranaqua SD. The black solid lines denote 1:1.

While improvements in modeling runoff could be made by considering other factors such as the antecedent dry weather period before storms or potential evapotranspiration (Elliott et al., 2016), our CREs explained 78-93% of all variation using rainfall depth alone. Each CRE was used to model annual retention with forty years of measured rainfall. The mean simulated annual retention from 1977-2017 is plotted in Figure 4.8.

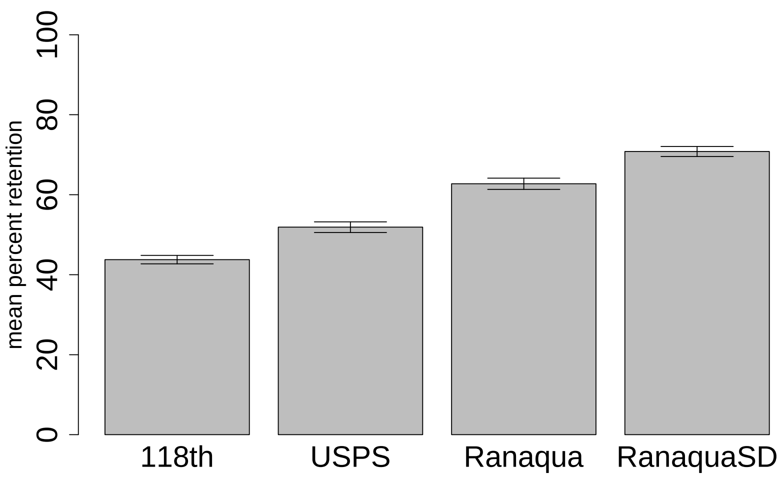


Figure 4.8 Modeled mean annual retention for each roof. The error bars denote standard error of the mean.

As presented in Figure 4.8, the 118th green roof retains the least amount of stormwater annually (42%), followed by USPS (53%), Ranaqua (63%), and Ranaqua with Smart detention (71%). The smart detention software of Opti RTC increased stormwater capture from 63% to 71%.

4.3.2. Evapotranspiration

In Table 4.2, we present crop coefficients for the Ranaqua green roof, containing native grasses and forbs, and for the green roofs at USPS and 118th, which are planted with *Sedum* vegetation. For each roof, we provide two forms of the crop coefficient: the crop coefficient which fits the SMEF in equation (4.4), as well as the crop coefficient as ET divided by PET, a simpler form reported in some studies (Elliott et al., 2016; Harper et al., 2013). We include both Ranaqua quads 1 and 4 in Table 4.2 in order to represent variability from using different soil moisture sensors. Crop coefficients for Ranaqua were found to exceed those of the other roofs, particularly during the growing season.

Table 4.2 Two common forms of seasonal crop coefficients for each roof

	Ranaqua Quad 1		Ranaqua Quad 4		USPS		118th	
Season	SMEF	ET/PET	SMEF	ET/PET	SMEF	ET/PET	SMEF	ET/PET
Overall	1.93	1.12	3.32	1.13	1.63	0.57	1.84	0.56
Spring	1.10	0.67	1.79	0.69	1.01	0.29	1.92	0.58
Summer	1.95	0.89	5.17	0.90	0.95	0.32	1.74	0.38
Fall	2.84	1.87	4.40	1.87	2.33	1.20	2.05	0.81
Winter	2.57	2.02	3.09	2.01	3.47	1.17	1.68	0.78

The overall crop coefficients were then used to model AET with measured soil moisture and PET over the same 2-year period from which the crop coefficients were derived. Figure 4.9 presents these modeled estimates of monthly AET for each roof, along with monthly PET. Due to a similar trend between the quads, quad 4 is omitted and Ranaqua denotes quad 1.

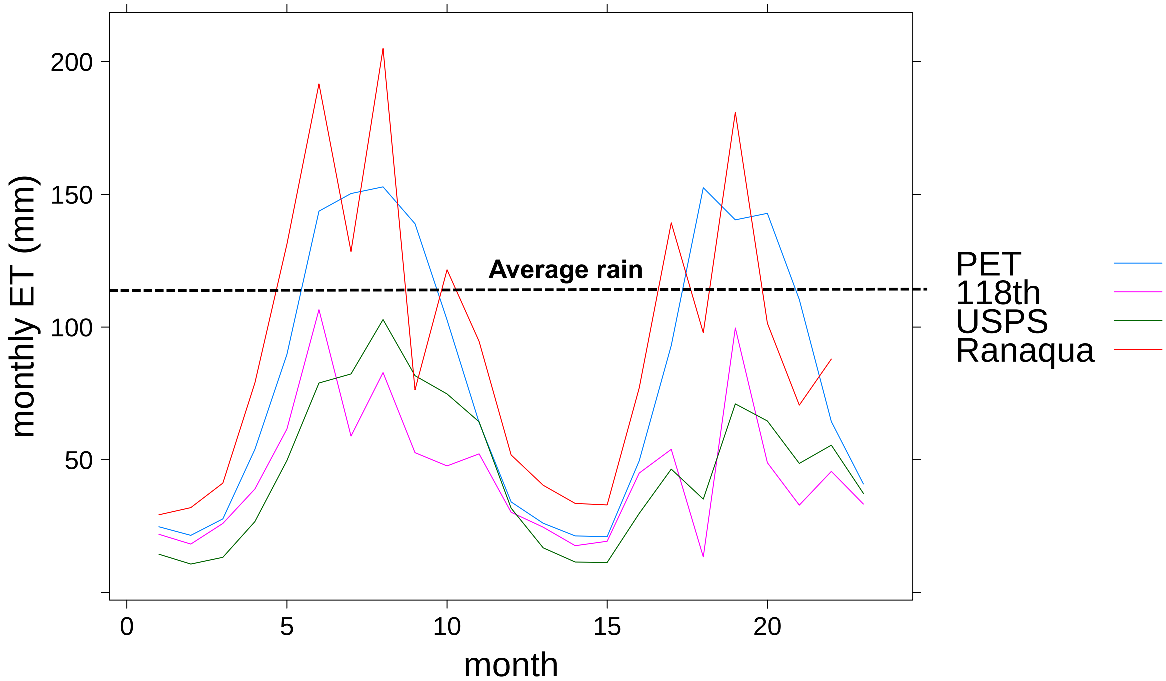


Figure 4.9 Monthly potential evapotranspiration (PET) and monthly evapotranspiration (ET) from each green roof from November 2013 to December 2015, omitting July, 2015. 40-yrs average rain from La Guardia's weather station is drawn as constant horizontal dotted line, rather than monthly, in order to reduce noise from rain variability and more clearly demonstrate when PET and ET surpass average rain.

Since monthly rain contains significant variability, the average monthly rain over 40 years from 1977-2017 at LaGuardia's weather station is presented as a constant dotted line. While this simplification obscures seasonal differences in rain, the New York City climate contains little variation in rain between seasons (Carson et al., 2013). As Figure 4.9 indicates, the Ranaqua green roof had much greater rates of ET than 118th and USPS. Ranaqua's ET even frequently exceeded PET, while ET at USPS and 118th remained below PET.

4.4. Discussion

Our study indicates that industry standard green roofs might not capture as much stormwater as “next generation” native systems with irrigation and smart detention. We provide crop coefficients demonstrating reduced evapotranspiration in green roof plants with CAM photosynthesis (0.56, 0.57) compared to native plants (1.12, 1.13). We also quantify the performance of real-time control cistern irrigation, and found that smart irrigation increased retention/detention from 63 to 71%.

Ranaqua overall had the greatest retention compared to the other more traditional green roofs (Figure 4.8), challenging the extent to which irrigation reduces retention, as suggested by others (Schroll et al., 2011; Van Mechelen et al., 2015; Volder and Dvorak, 2014; Whittinghill et al., 2014). We attribute Ranaqua’s greater performance to canopy interception and evapotranspiration provided by the robust vegetation. Irrigation likely contributed to healthier vegetation with substantial plant biomass. Increased performance due to more massive vegetation likely overrode any reduction in water retention due to diminished soil storage volume on account of irrigation. This finding directly contradicts Li and Babcock (2014), who suggested that the lower soil moisture content of *Sedum* soil would compensate for decreased water absorption from the vegetation.

The only area where 118th and USPS outperformed Ranaqua was small storms (Figure 4.6a). Since Ranaqua has the lowest performance during the summer especially for small storms, irrigation may have reduced storage capacity, causing some runoff even during small events.

Warmer summer temperatures are generally associated with greater rates of green roof water retention (Graceson et al., 2013; Mentens et al., 2006; Mobilia et al., 2015; Poë et al., 2015; Vanuytrecht et al., 2014). Elliott et al (2016) found that 118th displayed a much greater seasonal trend when compared to USPS. The soil layer on 118th is only 32 mm deep, while the soil on USPS is 100 mm deep. Greater seasonal variation is associated with thin soil, longer antecedent dry weather periods, and large storms (Elliott et al., 2016; Schroll et al., 2011). Despite its thicker soil (127 mm), Ranaqua also followed a seasonal trend, perhaps due to its native grasses, which have been found to display a greater seasonal effect than *Sedum* on water retention (Soulis et al., 2017).

However, Ranaqua has higher retention in the fall than spring, while 118th has greater retention in the spring than fall. We believe that this trend is caused by the annual behavior of native grasses in the growing season. During the winter, the dead stems of native grasses are cut and removed (Figure 4.10a). The removal of absorptive plant material may reduce canopy interception during the winter and spring until the next growing season.

(**a**) (**b**)

Figure 4.10 Ranaqua green roof during the (**a**) winter; (**b**) summer

Although stormwater performance is highly impacted by climate conditions (Mobilia et al., 2017), seasonal ET remains poorly understood (Digiovanni et al., 2013; Li and Babcock, 2014). One study found that *Sedum* had greater crop coefficients in the spring and fall than in the summer (Schneider et al., 2011).

We found greater crop coefficients for Ranaqua than for the two *Sedum* roofs (Table 4.2), which highlights the ability of native vegetation to assist in capturing more stormwater for a given climate and soil moisture condition. Crop coefficients for USPS and 118th were in the general range found by other studies modeling green roofs planted with *Sedum* vegetation (Harper et al., 2013; Schneider et al., 2011). Notably, ET/PET was 1.12 for quad 1 and 1.13 for quad 4 at Ranaqua, which is greater than prior studies of green roof vegetation (Voyde, 2011). The greater crop coefficients at Ranaqua may represent the ability of native vegetation to improve water retention through both ET and canopy interception. It is possible that these benefits are amplified when native plants are irrigated sufficiently to develop substantial canopy biomass.

ET was greater at Ranaqua than for the *Sedum* roofs (Figure 4.9). Ranaqua's ET even exceeded PET, while ET at the *Sedum* roofs remained below PET. We found that PET exceeds rainfall during the summers in temperate climates, as suggested by Carson (2014). With higher crop coefficients and with cistern irrigation, Ranaqua takes advantage of the greater PET over the summer, with ET even more than the average rainfall. 118th and USPS do not provide as much ET due to the lack of a cistern to retain and ET additional stormwater via irrigation, and due to their *Sedum* vegetation, which transpires slower and does not grow as large a canopy to intercept rain as native vegetation.

However, Figure 4.9 also indicates that PET remains much lower than the average rain during the fall, winter, and spring. Without the evaporative demand associated with PET, water retention for green roofs in temperate climates will be limited during these cooler seasons.

Our study is advantaged by including four years of water quantity data from three full-scale green roofs. Many monitoring studies contain too short a monitoring period, potentially producing contradictory results (Czemiel Berndtsson, 2010), with few studies even lasting a whole year (Soulis et al., 2017). Variation in water retention among sites reported in the literature may in fact be largely due to different durations of monitoring, rather than actual performance (Carson et al., 2013; Fassman-Beck et al., 2013).

Our study was limited however by the different characteristics of each roof, such as the varying soil depths, because we were unable to directly compare the effect of vegetation and irrigation on water retention. To an extent, the trend in Figure 4.8 is expected, because 118th has the thinnest soil depth (32 mm), followed by USPS (100 mm), and Ranaqua (127 mm). Several studies have proven that the depth of soil media increases water retention (Mobilia et al., 2015; Montalto et al., 2007; Monterusso et al., 2004) particularly during the summer (Mentens et al., 2006). However, the relationship between soil depth and water retention is not straightforward

(Graceson et al., 2013; VanWoert et al., 2005). In fact, Fassman-Beck et al (2013) found that four different green roof soil depths were equally effective in reducing stormwater runoff. While Ranaqua has deeper soil than USPS and 118th, the increase in mean retention appears greater than that solely to be caused by the increase in soil depth. Stovin found that retention was more influenced by higher crop coefficients than by increasing soil depth in a hydrologic modeling study (Stovin et al., 2013). Ranaqua's greater retention may be due to both higher crop coefficients and soil depth. Future research could more directly test the effect of irrigation on water retention by comparing irrigated and unirrigated roofs that are otherwise identical.

4.5 Conclusions

Most extensive green roofs in temperate North America are passively drained and planted with drought-tolerant vegetation. Our data suggest that actively irrigated green roofs planted with native vegetation may reduce more stormwater runoff than more common green roofs planted with *Sedum*. Our reported crop coefficients demonstrate that natives are beneficial for stormwater management, and not just for biodiversity. Our work may justify increased construction of "next generation" green roofs with native vegetation and smart irrigation in order to maximize water retention.

Chapter 5: Contributions

The goal of this dissertation is to better quantify the performance of New York City's green infrastructure and investigate how it can be improved. My dissertation achieves this goal by contributing a tremendous amount of green infrastructure performance data. Chapter 2 contains a massive dataset with 595 water samples collected during 42 different rains analyzed for nitrogen and phosphorus over three-year monitoring period. Chapter 3 includes data from 423 soil samples, 64 gas samples, and 160 leaf samples collected on 32 dates over a monitoring period of 15 months. Chapter 4 contributes performance data for more than 400 storms from each of three full-scale green roofs over four years. Analysis of these voluminous data has produced several contributions, as will be discussed in the following subsections that correspond to individual chapters within this dissertation.

5.1 Chapter 2: Effects of Rain Gardens on Nutrient Pollution: Long-Term Trends and Overall Significance

Chapter 2 contributes a new model by which a land manager can quantify the overall effect of rain gardens on nutrient pollution. The study was also conducted with promising low-cost and non-invasive monitoring methods and reveals long-term, seasonal, and site-specific trends.

5.1.1 New Model - Environmental Indicator of Overall Performance

Prior research has found rain gardens to be sources of nutrients (Collins et al., 2010; Hunt et al., 2006; Randall and Bradford, 2013; Shetty et al., 2016), yet does not investigate the overall significance of nutrient leaching to the city writ large. Chapter 2 provides a new model by which a land manager can quantify the overall effect of rain gardens on nutrient pollution: it compares the nutrient mass mitigated due to CSO reduction to the nutrient mass added by the rain gardens themselves. By clarifying the ultimate implications of the tradeoff between water retention and pollutant removal, this model provides the missing link between scientific performance data and overall rain garden performance, and may be of great interest to land managers and practitioners.

Currently, in combined sewer systems, New York City's Department of Environmental Protection (DEP) is required to mitigate CSOs by constructing green infrastructure. As a result, NYC DEP is designing and constructing more than one thousand rain gardens per year specifically to reduce CSO volumes (De Blasio and Sapienza, 2017). However, recent Municipal Separate Storm Sewer System (MS4) regulations also now require green infrastructure designs in separated sewersheds as well (De Blasio and Sapienza, 2016). The research in chapter 2 suggests that DEP should consider an alternate design for these areas, which represent 35% percent of NYC's sewered land area (Bloomberg, 2008). Rain garden design is moving away from "one size fits all" design specifications, and is increasingly targeted toward regulatory needs and local conditions (Hunt et al., 2012), and Chapter 2 provides a model demonstrating that the type of sewer system in which the practice is located can impact overall design and performance.

5.1.2 Novel Monitoring Methods

Chapter 2 also contributes low-cost and non-invasive monitoring methods.

Current rain garden monitoring most commonly involves flume and weirs to quantify water retention, and auto-samplers to collect samples for water quality testing (Davis et al., 2012; Dietz, 2016; Hatt et al., 2009; Hunt et al., 2006; Li and Davis, 2014; McNett et al., 2011).

However, the use of weirs and flumes has significant drawbacks, as they constrict inlet widths, reducing inflow, so that the equipment monitors an altered garden design, and does not provide useful information regarding standard designs that have not been impacted by the monitoring equipment. In this way, prior monitoring of rain gardens illustrates the Heisenberg uncertainty principle: observing a system can itself alter it. Chapter 2 details a monitoring system that avoids this issue, presenting the non-invasive sampling methods we developed that do not themselves impact rain garden performance.

This new system also has the advantage of lowering costs significantly, as common monitoring protocols may also cost approximately $50k per site in instrumentation alone (NYC Parks). Chapter 2 however demonstrates a low-cost method with equipment more than an order of magnitude cheaper than prevailing methods, roughly $200-$400. The necessity of our inexpensive methods is highlighted by the demand for more widely scalable methods: in contrast to centralized wastewater infrastructure, decentralized green infrastructure is distributed throughout different microclimates in thousands of different locations, each of which may demonstrate site-specific variability (Collins et al., 2010; Davis et al., 2012; Lucke and Nichols, 2015; Manka et al., 2016; Roy-Poirier et al., 2010). Our inexpensive monitoring methods therefore provide a more viable means of monitoring a large number of installations.

While our monitoring methods are inexpensive in equipment costs, they are admittedly labor-intensive, and typically involve standing in the rain for long hours, potentially at night and on weekends, and often with little prior notice. However, our methods could be used to teach field methods to students in engineering or environmental science. As our sampling methods are straightforward to learn with simple training, data collection could be paired with citizen science programs (Farnham et al., 2017) or taught to undergraduate or high school students for school summer internships.

Labor-intensive methods also have an additional benefit – they force qualitative observation of the study sites. Standing in the rain and observing how rain gardens absorb stormwater on over 42 occasions is instructive in and of itself. For instance, while I stood by the adjacent rain gardens termed ROWB 9A and ROWB 9B, I noticed that when the rain intensity diminished, the upstream garden, 9B, while retaining a significant volume of water, still demonstrated some overflow. However, downstream 9A, in turn benefitting from 9B's retention, would often stop overflowing altogether. These first-hand observations concretely demonstrate rain gardens' impressive effectiveness: despite their tiny size (each measuring about 9 m^2), water falling on the entire drainage area of an estimated 950 m^2 was not entering the sewer system.

Simple observation also allowed our research team to develop conceptual models of rain garden water absorption processes. I observed that during intense rains, the gardens quickly pond with stormwater above the soil surface, but also drain quickly. However, when rain is of low intensity, the garden surface does not pond immediately, but rather the soil appears to saturate slowly from the bottom of the soil up to the top, and will rather pond toward the end of the storm. Such observations allowed my team to develop a more concrete, conceptual model of the logistics of rain garden functioning.

In contrast, other rain gardens could benefit from more qualitative observation: a rain garden developed by another research team in Queens, termed Nashville, provides ambiguous results despite being heavily instrumented with roughly $50k in monitoring equipment; this is in my opinion due precisely to the lack of onsite observation by research team members. For example, when inflow was less than expected, the researchers hypothesized that the inlet at Nashville was clogged with leaves and trash, without verifying whether this was actually the case. Conversely, when inflow was greater than expected, researchers hypothesized that additional stormwater approached the site due to clogged catch basins upstream producing a larger drainage area (Montalto et al., 2013). Such attempts to explain data absent first-hand observation are rife with opportunities for error and misinterpretation.

5.1.3 New Long-term and seasonal trends

It is an open question how pollutant removal may change over time, as stormwater pollutants accumulate and biodegrade in the rain garden, and as plants mature (Collins et al., 2010; Elliott et al., 2011; Koch et al., 2014). This dissertation finds evidence that long-term changes in rain garden effectiveness are very limited, as nitrogen concentrations demonstrated no long-term changes over time. On the other hand, we did observe a long-term decrease in the concentrations of infiltrated phosphorus over time.

Conversely, both nitrogen and phosphorus demonstrated noticeable seasonal trends, with increased concentrations of both found in rain garden soil during summer months.

5.2 Chapter 3: Quantifying Nitrogen Cycling in the Soil, Gas, and Plant phases of Rain Gardens

Prior research has frequently cited rain gardens as sources of nitrogen (Collins et al., 2010; Hunt et al., 2006; Randall and Bradford, 2013; Shetty et al., 2016), yet views nitrogen cycling as a black box, virtually unknown (Emily G I Payne et al., 2014), without investigating the root cause of the issue: the decomposition of soil organic nitrogen into simpler inorganic forms that are available to plants and readily washed out of the garden. Chapter 3 provides measurements of mineralization and nitrification to quantify the rate that the soil supplies inorganic nitrogen to the rain garden. These measurements may improve efforts to reduce mineralization and nitrification rates, and thereby mitigate nitrogen pollution. Chapter 3 also found evidence of nitrogen saturation, especially in the summer and in shallow depths of soil.

Chapter 3 then investigates if nitrogen pollution can be reduced by addressing: 1- Are plants receiving enough nitrogen? And 2 - Is rain garden soil too nitrogen rich? Chapter 3 answers these questions by providing a nitrogen mass balance. Although approximate, the nitrogen mass balance demonstrates that stormwater provides sufficient nitrogen to meet plant needs, and that rain gardens do not need additional fertilizer from organic matter in the soil. Chapter 3 concludes with soil recommendations that could result in reduced nitrogen export.

Chapter 3 provides an urban nitrogen cycling mass balance: in contrast to natural ecosystems, where external inputs of nitrogen are about 10% of the amount of nitrogen that annually cycles (Chapin et al., 2002), stormwater inputs of nitrogen in the studied urban rain gardens constitute about 88% of the nitrogen that annually cycles. Chapter 3 therefore demonstrates that the nitrogen cycle is open in urban areas but comparatively closed in natural areas. Due to this improvement in understanding, urban land managers may design and maintain rain gardens while accounting for the considerable inputs of nitrogen that enter via stormwater runoff. One major alteration suggested by our research's results would be the alteration of rain gardens' design, so that from the outset they contain less organic nitrogen in their soil upon construction; the other major alteration suggested is that they be maintained without additional fertilizer.

As a result, chapter 3 suggests that nutrient-poor soils with carbon amendments may reduce nitrogen leaching. They may also provide healthier plants: as plants have evolved to be nutrient limited, too much nitrogen can actually harm them (Boersma and Elser, 2006). Excess nutrients tend to stimulate plant growth aboveground rather than belowground, with detrimental long-term impacts to plant health. Nutrient-poor soils may also reduce gardening maintenance, as nutrient-rich soil attracts weeds (Levin and Mehring, 2015).

Chapter 3 also provides measurements of soil gas nitrogen fluxes and foliar nitrogen, both ill-quantified aspects of the rain garden nitrogen budget (Grover et al., 2013; E G I Payne et al., 2014; Read et al., 2008). Chapter 3 demonstrated that soil gas emissions were low, with increased emissions during the summer. Chapter 3 also demonstrated differences in the amount of nitrogen taken up by plants: perennials and grasses appeared to take up greater amounts of nitrogen than shrubs. Overall, the study found that plant nitrogen needs are much lower than expected: the amount of nitrogen that flows in stormwater runoff is much greater than the plants actually need. By comparing the amount of nitrogen required by plants to the amount that enters, Chapter 3 contributes to the scientific knowledge of rain gardens.

5.3 Chapter 4: Comparing Two Sedum Green Roofs to a "Next Generation" Native System with Irrigation and Smart Detention

Chapter 4 contributes data that may increase the amount of stormwater absorbed by the next generation of green roofs.

5.3.1 Native vegetation

Few studies have quantified differences in green roof water retention between Sedum and native plants (Poë et al., 2015; Stovin et al., 2015). Chapter 4 however contributes crop coefficients that quantitatively demonstrate that the native plants at the Ranaqua green roof transpire at twice the rate of the two *Sedum* green roofs for identical climate and soil moisture conditions, implying that they are capturing significantly more water than the *Sedum* green roofs.

As a result, chapter 4 contradicts research suggesting that vegetation has a minimal impact on runoff reduction (Czemiel Berndtsson, 2010; VanWoert et al., 2005), as native vegetation might actually dramatically increase stormwater capture when plants are irrigated sufficiently.

5.3.2 Irrigation

Another contribution of Chapter 4 is to challenge the extent to which irrigation reduces retention, as suggested by others (Schroll et al., 2011; Van Mechelen et al., 2015; Volder and Dvorak, 2014; Whittinghill et al., 2014). Irrigation at Ranaqua likely contributed to healthier and more massive vegetation, capable of greater evapotranspiration and canopy interception, thus increasing retention rates.

5.3.3 Smart Detention

Most green roofs have no system in place to regulate water that has already infiltrated through the roof garden's soil, meaning rainwater passes through the rain garden, some of it being absorbed, while the rest drains from the roof's drain into the sewers. Other roof gardens are built over tanks that collect stormwater that has infiltrated through green roof soil; however almost all of these roof gardens have no measures in place to empty tanks in between rains, meaning tanks often overflow from subsequent storms, with excess rainwater flowing into the sewers during heavy rains, contributing to combined sewer overflows.

Our research team however tested the effects of using smart detention to regulate irrigation; this system is calibrated to pump water out of the tank prior to storms, so that excess stormwater will flow into sewers before a storm begins, thus reducing combined sewer overflows. As a result, we found that smart detention can increase long-term stormwater capture from 63% to 71%.

Chapter 6: Future Research

My dissertation proposes several avenues for future research, as presented in the following subsections, which refer to chapters herein.

6.1 Chapter 2: Effects of Rain Gardens on Nutrient Pollution: Long-Term Trends and Overall Significance

6.1.1 Continuous sensors

Given that my salt dilution methods to monitor water retention at rain gardens are quite labor intensive, future research could investigate measures to mitigate the amount of time required for monitoring, including sensors that could be left *in-situ*: these could include shallow wells that measure water depth, piezometers to measure pressure, or soil moisture sensors that measure water content in the soil. These sensors could be calibrated against the more labor intensive methods employed in Chapter 2. This arrangement would produce performance data for many more storms, as opposed to the handful I monitored with labor-intensive methods.

6.1.2 Determine Fate of Infiltrated Water

Chapter 2 demonstrates that the amount of nitrogen and phosphorus that rain gardens reduce from combined sewer overflows is greater than the amount leached by the gardens themselves. A major limitation inherent in this analysis is that we have not considered the fate of the infiltrated stormwater. In urban areas, infiltrated stormwater is frequently routed back into the sewer system. However, when rain gardens are constructed above a layer of subsoil, studies have found that infiltrated stormwater is also treated by that subsoil, further increasing rain gardens' removal of polluting nutrients like nitrate (Elliott et al., 2011).

As for now however it remains unclear to what extent such underlying subsoils contribute to rain garden's effectiveness, and under what precise conditions their impact is greatest. Future research could use tracer methods to consider how the fate of infiltrated nutrients is impacted by local construction history and subsoil conditions, and how this impacts the overall degree of nutrient pollution removal by rain gardens, in other words the results presented in Chapter 2.

6.1.3 First flush equivalent to model nitrate buildup and washout

Chapter 2 demonstrated that like influent, which demonstrates a first flush phenomenon due to deposition and accumulation of pollutants on street surfaces, nitrate also builds up in between storms and is washed out of the soil as infiltrate. If initial infiltrate nitrate concentrations are compared with respect to the duration of time between storms, meaning the length of time nitrate was given to accumulate in the soil, one may determine the rate at which nitrate builds up.

The infiltrate nitrate concentration could also be plotted against cumulative rain during individual storms in order to determine the rate at which nitrate washes out of soil during rains, giving the relationship between rain volume and nitrate washout.

6.1.4 Fast Draining Internal Storage Zones (ISZs)

In Chapter 2, I explained a tradeoff regarding rain garden construction, between the use of fast-draining or slow-draining designs: fast-draining rain gardens absorb more water, yet removal of nitrogen requires internal storage zones with slower infiltration rates. Future research could attempt to design rain gardens with both goals in mind. Rain gardens outfitted with smart sensors connected to weather forecasting data, such as those manufactured by OptiRTC as described for the Ranaqua green roof in Chapter 4, could store and denitrify stormwater for small storms, yet drain the water in advance of an upcoming storm. In this way, rain gardens could denitrify while also capturing as much stormwater as possible.

6.2 Chapter 3: Quantifying Nitrogen Cycling in the Soil, Gas, and Plant phases of Rain Gardens

6.2.1 Balanced Nutrition

Future research should explore our suggestion that stormwater provides enough nitrogen for plant health. The optimal blend for rain garden soil is not nutrient rich, but contains sufficient nutrients for plant growth (Liu et al., 2014). Studies should investigate whether a nitrogen-poor soil composition would be better suited to rain gardens, and study the needs of soil with overabundant nitrogen, in order to provide balanced nutrition. Studies should simultaneously test how to mitigate pollutants from stormwater runoff that may damage plant health such as oils, salts, and metals.

6.2.2 Plant Availability

Future studies should also test the degree to which plants can absorb the nitrogen in stormwater given that it comes in rapid waves. Some studies have suggested that biological processes involving microbes and plants may be too slow to significantly affect nutrient levels of rapidly infiltrating stormwater during actual storms but rather alter the nitrogen balance between storms by transforming trapped pollutants (Davis et al., 2010, 2006). This would mean that plants would be unable to absorb the nitrogen from stormwater runoff in the same way they absorb nitrogen contained in fertilizers. However, other studies have suggested quite rapid processing (Emily G I Payne et al., 2014), with vegetated systems absorbing more nutrients than non-vegetated systems even during rapid first flushes at the beginning of storms (Henderson et al., 2007; Lucas and Greenway, 2008). This enhanced nutrient removal may be due to microbe activity (Henderson et al., 2007; Read et al., 2008).

Specifically, the microbe activity refers to a process called microbial immobilization. During these rapid influxes of nitrogen from stormwater runoff, microbes quickly absorb nitrogen, thus 'immobilizing' it, meaning nitrogen pollution does not leach out of the garden as it otherwise would. This process occurs much more rapidly than plant uptake (Lucas and Greenway, 2011; E G I Payne et al., 2014), with significant amounts of nitrogen immobilized within the first 15 minutes (Lucas and Greenway, 2011). It is for this reason that rain gardens may be able to process rapid influxes of nitrogen from stormwater as well as they process stable levels from sources such as organic matter (i.e. plant detritus and fertilizers).

This process may be aided in fact by mature plants' root systems. Especially well-established plants' rhizospheres, or the area surrounding its root system, stimulate enhanced microbe activity (Davis et al., 2006; Lucas and Greenway, 2008; Muerdter et al., 2016). Therefore, plant root systems could be supporting the growth of microbes which trap incoming stormwater nutrients, and then release it back for the plants. Future research that more explicitly tests plant-microbe interactions could result in major effects on plant selection and nutrient removal (E G I Payne et al., 2014).

6.2.3 Comprehensive Gas measurements

We did not measure N_2; however, to fully understand the nitrogen cycling through a rain garden, the levels of N_2 coming out should be measured, as there is evidence that they may be significant (Morse et al., 2015). Higher levels of N_2 emitted from the garden would imply that nitrogen pollution is being converted by the garden into harmless N_2 gas. This amount should be quantified. Similarly, future research should also quantify soil O_2, an important control on denitrification rates, likewise in order to better understand the nitrogen cycle in rain gardens, as today our knowledge remains incomplete (Burgin and Groffman, 2012).

6.2.4 Future Soil Specifications

Future research could test if nutrient-poor soil specifications could produce rain gardens that actually reduce nitrogen pollution as theorized, and also reduce maintenance from weeds.

6.3 Chapter 4: Comparing Two Sedum Green Roofs to a "Next Generation" Native System with Irrigation and Smart Detention

6.3.1 Crop coefficients

Future research should generate crop coefficients for more types of green roof vegetation. Chapter 4 provides crop coefficients that quantify transpiration for both native and *Sedum* green roof vegetation, and were calculated in a standardized way that may facilitate comparison with future research. Although parameterization for crop coefficients is completely contingent upon the form of the soil moisture extraction function (SMEF) and assumptions used (Carson, 2014), Chapter 4 uses the open-source R package Evapotranspiration, which may enhance consistency in presented crop coefficients and comparison among different datasets by allowing researchers to use the same computer code and assumptions when calculating PET (Guo et al., 2016). Chapter 4 additionally used a simpler form of the soil moisture extraction function (Hakimdavar et al., 2016) and simpler PET models such as Hargreaves (Hargreaves and Samani, 1985) to calculate crop coefficients. The use of these simpler models will further reproducibility and comparison with future research quantifying the effect of more types of green roof vegetation, such as trees and shrubs found on intensive green roofs. While Chapter 4 found native grasses to have crop coefficients that were greater than previous studies (Voyde, 2011), future studies could quantify whether trees or shrubs may have even greater crop coefficients, which would increase the amount of water transpired on green roofs.

6.3.2 Irrigation

Chapter 4 provides data indicating that irrigation does not reduce water retention, but can actually increase water retention by promoting more lush vegetation with substantial biomass for enhanced canopy interception and evapotranspiration. However, our study was limited because the different roofs have different characteristics such as different soil depths, which also affect water retention (Mentens et al., 2006; Mobilia et al., 2015; Montalto et al., 2007; Monterusso et al., 2004). Future research could test the effect of irrigation on water retention for two roofs that are otherwise identical. In fact, NYC Parks plans to test this next year by shutting off the irrigation for one of the green roof quads at Ranaqua.

Future research could also explore methods of promoting evapotranspiration via irrigation at Ranaqua. Irrigation has been found to increase evapotranspiration (Hardin et al., 2012; Van

Mechelen et al., 2015), which is correlated with stormwater capture, cooling and carbon sequestration (Digiovanni et al., 2013). Ranaqua's green roof is currently irrigated when volumetric soil moisture content drops below 14%, which was the Allowable Depletion selected for the roof (George et al., 2000). Future research could test the effect of increasing the threshold from 14% to say, 20%, which would increase the amount irrigated and thereby may provide enhanced stormwater capture, cooling, and carbon sequestration, or inversely determine the limit at which increased irrigation ceases to increase water retention.

6.3.3 Smart detention

Future research could maximize green roof water retention rates beyond the 71% reported in Chapter 4 by not only promoting robust vegetation with substantial biomass, as Chapter 4 demonstrates, but also by optimizing irrigation and smart detention.

While Chapter 4 found that smart detention can increase long-term stormwater capture from 63% to 71%, the study did not comprehensively investigate how smart irrigation and detention can be optimized. There are very few studies examining how often to irrigate and how often to drain cisterns intended to store green roof runoff for irrigation between storms (Tsang and Jim, 2016; Van Mechelen et al., 2015), and future research should respond to this need.

Currently, two quadrants at Ranaqua drain just the predicted volume when there is greater than 60% forecast of a rain of greater than 13 mm, saving the remaining volume for irrigation, while the other two quadrants drain completely empty 24 hours after a rain event. This latter system of more frequent emptying of cisterns is associated with reduced runoff, while the former system of more conservative detention using less tank water produces more runoff, since cisterns are more likely to overflow during the following rain (Shannak et al., 2014). Future research could decrease the 60% and 13 mm minimums mentioned above to lower thresholds, which may ensure that the tank remains empty for more potential storms.

6.4 Concluding Remarks

Future research is needed to quantify and improve green infrastructure performance. Although several years of research have improved the understanding of GI performance (Davis et al., 2009), current GI design practices are still highly empirical, meaning dependent on trial and error and lacking a complete understanding of all the factors at work (Davis et al., 2012). Green infrastructure research is perhaps at an early stage, similar to wastewater research in the early 20th century, when wastewater was aerated with oxygen without any scientific understanding of how the process promotes microbes that break down organic pollutants (Lofrano and Brown, 2010).

Similarly, land managers at NYC Parks rely on their own experience to select vegetation that tends to survive in the harsh living conditions of NYC streets and rooftops, without developing a detailed and systematic understanding of how nutrient pollutants in urban stormwater could affect specifications for rain garden soil, or how evapotranspiration in *Sedum* compares to native grasses.

In the future, green infrastructure research may one day approach the complexity of wastewater treatment research, where scientific understanding of microbial processes allows detailed optimization of wastewater treatment. My hope is that this dissertation will assist us along the path to better performing green infrastructure.

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Appendix: Bioretention Infrastructure to Manage Nutrient Runoff from Coastal Cities

ABSTRACT: Although bioretentions have been proven to reduce stormwater runoff volumes and remove many pollutants, reports are variable as to the nutrient removal performance of this green infrastructure technology, pointing to the need for further investigation. In this work, the nutrient removal performance of six bioretentions in New York City (NYC)'s Bronx River Sewershed was quantified by identifying differences among dissolved nutrient concentrations in influent, infiltrate, and overflow liquid samples during 26 storms. Results indicate that the studied bioretentions are sources, rather than sinks, of nitrogen and phosphorus. Statistical analyses revealed higher levels of measured water quality indicators in the infiltrate of the bioretentions as temperature and the dry weather period before a storm increase, and lower levels of most indicators as the rainfall depth before sampling and the loading ratio of the bioretention increase. In this paper, the monitoring methods developed for evaluating the nutrient removal performance of the bioretentions are described, the data collected on bioretention nutrient removal performance is reported, and the statistical correlations between measured water quality indicators, local environmental conditions and the bioretention site properties are discussed.

INTRODUCTION

The US Environmental Protection Agency (EPA) has identified nutrient pollution of US water bodies, including coastal waters, as one of the nation's top environmental concerns (US EPA 2009). Stormwater runoff is a significant export path for nutrients, including nitrogen, that have accumulated in urban environments (Collins et al. 2010). Although bioretentions have proven ability to reduce the quantity of stormwater runoff, many current bioretention designs are not optimized for water quality goals, especially nutrient removal.

The reported removal of nutrients by bioretentions has been variable. For nitrogen, reported removal performances range from 60% removal to the actual export of nitrogen from bioretentions (Chen et al., 2013). Nitrogen export from bioretentions is believed to be associated with the fact that these green infrastructure installations are typically designed to drain rapidly, which favors aerobic nitrifying conditions (Figure 1). Because denitrification is negligible under aerobic conditions, nitrate can accumulate within bioretentions during dry-weather periods, and then be released as infiltrate during the next rain event (Roy-Poirier et al. 2010).

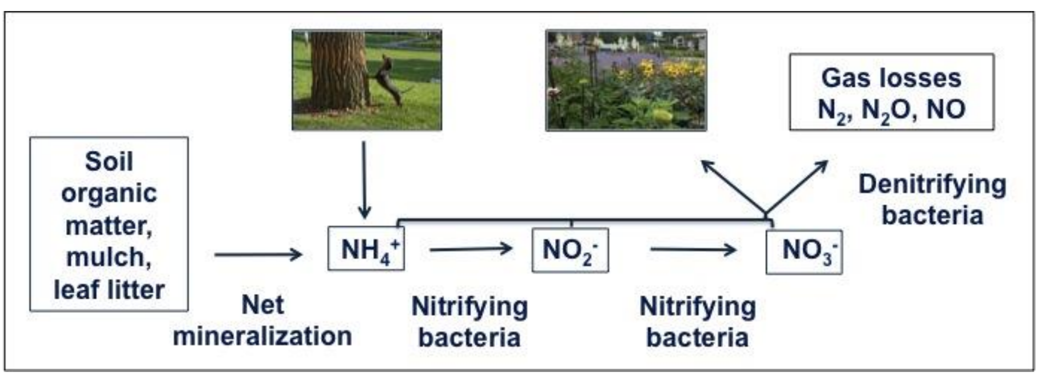

FIG. 1. Simplified Bioretention Nitrogen Cycle

The goal of this work is to provide further insight into bioretention nutrient removal performance through provision of additional measurement data coupled with data analysis. To achieve this goal, a series of low-cost and easily scalable water quality samplers were installed at six streetside bioretentions located within New York City's Bronx River sewershed area, a city identified priority sewershed where hundreds of new green infrastructure installations are being constructed to reduce pollutant loading to Long Island Sound. Data obtained from 26 storms at the six bioretentions were analyzed for water quality (total nitrogen, ammonium, nitrite, nitrate, dissolved nitrous oxide, orthophosphate, total phosphorus, sulfate, chloride, pH, conductivity). Wilcoxon rank-sum tests were then performed to explore how bioretentions affect the urban nitrogen cycle, by identifying statistically significant differences between bioswale nitrogen influent, infiltrate, and overflow. Finally, Spearman rank correlation coefficients were used to link bioretention nutrient removal performance to the site properties and local environmental conditions

In the following sections of the paper the field-monitoring and measurement methods developed for assessing bioretention nutrient removal performance are reported, together with the associated performance data. Then, the results of the Wilcoxon rank-sum tests and the Spearman rank correlation coefficients are presented and discussed. Finally, some conclusions are drawn about the nutrient cycling processes within the studied bioretentions.

METHODS

Six test bioretention sites, constructed between September 2013 and March 2014, were selected for the study. These sites included two "right-of-way bioswales" (ROWBs) and four "stormwater greenstreets" (SGS). ROWBs follow a standardized tree-pit-like design (Figure 2, a), while SGS are larger, with geometry "bumped" into the street to allow for more direct inflow (Figure 2, b). Site planting area for each site varies from 8.4 m^2 to 124.3 m^2, soil depth varies from 45.7 cm - 91.44 cm, and the impervious contributing watershed area ranges from 2312 m^2 to 9995 m^2. All six sites are located within the Bronx River sewershed, in the dense, and primary residential, neighborhood of Soundview (Figure 3). Urban features of the neighborhood include a large park, three major highways, and several high-rise residential housing complexes.

FIG. 2. Bioretention study sites: Right-of-Way Bioswale (a) and Stormwater Greenstreet (b)

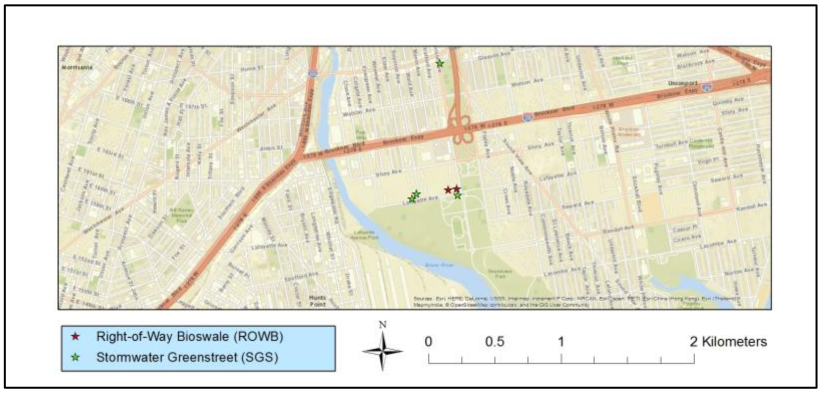

FIG. 3. Map of six research sites (Two ROWB, Four SGS)

Stormwater that infiltrated into each bioretention (termed bioretention infiltrate) was sampled with 12.7mm diameter perforated polyvinyl chloride PVC pipe samplers, which are 50.8 cm in length and perforated with 8 mm diameter holes every 25.4 mm (Figure 4). The samplers' perforated length of 30.4 cm was positioned such that each sampler had 10.2 cm without perforations at each of its ends, so that water would collect at the bottom end and so that water would not directly enter the sampler at the top end until infiltrating through at least 10.2cm of soil. Prior to installation, samplers were capped and wrapped in polypropylene geotextile filter fabric. Samplers were then hammered into the bioretention soil to their full depth of 50.8 cm at each site. During each stormwater sampling event, infiltrate samples were extracted from the water quality samplers using polypropylene syringes fitted with Luer Lok connections to 3mm ID Tygon tubing (Figure 4).

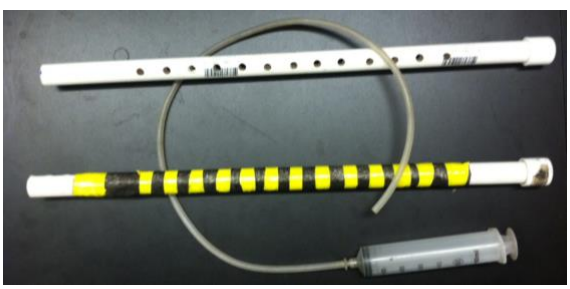

FIG. 4. Water quality samplers and syringe

In addition to the collection of the stormwater infiltrate samples, the stormwater entering the bioretentions from the street runoff (termed influent) was also collected via syringe. During intense storms beyond the capacity of the bioretentions, the syringing method was also utilized to collect bioretention overflow.

Data on the nutrient removal performance of the bioswales was collected for 302 samples during 26 storms during the measurement period from March 2014 to August 2015. Collected samples were analyzed in the field by probe for pH (Fisher Scientific Accumet portable APG2 pH/mv meter) and electrical conductivity (Cole Parmer conductivity meter). Samples were then immediately filtered with Millex-SV Durapore (PVDF) filters (0.22 µm pore size), stored in 50 mL polypropylene tubes, and kept on ice until they were frozen, which occurred within 180 minutes of sampling.

The chemical analyses of samples were conducted within 60 days of sampling. Ammonium

(NH_4^+) was analyzed by fluorescence methods (Holmes et al. 1999), Nitrate (NO_3^-), nitrite (NO_2^-), orthophosphate (PO_4^{3-}), sulfate (SO_4^{2-}), and chloride (Cl^-) were analyzed by ion chromatography, total nitrogen (TN) and total phosphorus (TP) were analyzed by the persulfate digestion method and ion chromatography (Patton and Kryskalla, 2003). Dissolved nitrous oxide (N_2O) was measured by polarographic techniques (Unisense N_2O-R probe).

RESULTS / DISCUSSION

To determine whether the chemical composition of the bioretention infiltrate or overflow was different than that of the influent, non-parametric two-sided Wilcoxon rank-sum hypothesis tests (Helsel and Hirsch, 2002) were performed for all measured water quality parameters. Figure 5 summarizes the results of the dissolved nitrogen measurements at the six bioretentions, presented as box-plots, with statistically significant differences from the hypothesis tests ($p < 0.05$) written as letters above the plots. As seen in Figure 5a, the A listed above both influent and overflow indicates no significant differences in total nitrogen between them. B listed above infiltrate indicates greater concentrations of total nitrogen than the influent or overflow, pointing to the fact that the studied bioretentions could be sources of nitrogen loading in urban watersheds. Other than stormwater runoff into bioretentions, nitrogen inputs into these installations which could be responsible for the leaching of total nitrogen can comprise organic forms of nitrogen, including net mineralization (microbial decomposition minus uptake of soil organic matter, mulch or leaf litter) and animal waste, while nitrogen outputs can include plant uptake and gas losses.

Out of the different nitrogen species considered, NH_4^+ is the only nitrogen species that did not show a significant difference between influent and infiltrate (Figure 5b). This suggests that nitrogen inputs to bioretentions undergo NH_4^+ oxidation as organic nitrogen is transformed to NO_3^- in between rain events. The accumulation and subsequent leaching of NO_3^- as bioretention infiltrate is seen in Figure 5d.

NO_2^- was the only nitrogen species for which influent was greater than infiltrate or overflow (Figure 5c). The NO_2^- might be produced from chemical reactions between rain and nitrogen oxide air pollution. Because street runoff has a limited residence time between rainfall and bioretention inflow, oxidation of NO_2^- to NO_3^- might not have time to occur. Unlike for street runoff, however, NO_2^- might have sufficient residence time in a bioretention soil to oxidize to NO_3^-. In addition, bioretention soil may have a greater density of nitrite-oxidizing microbes when compared to street surfaces.

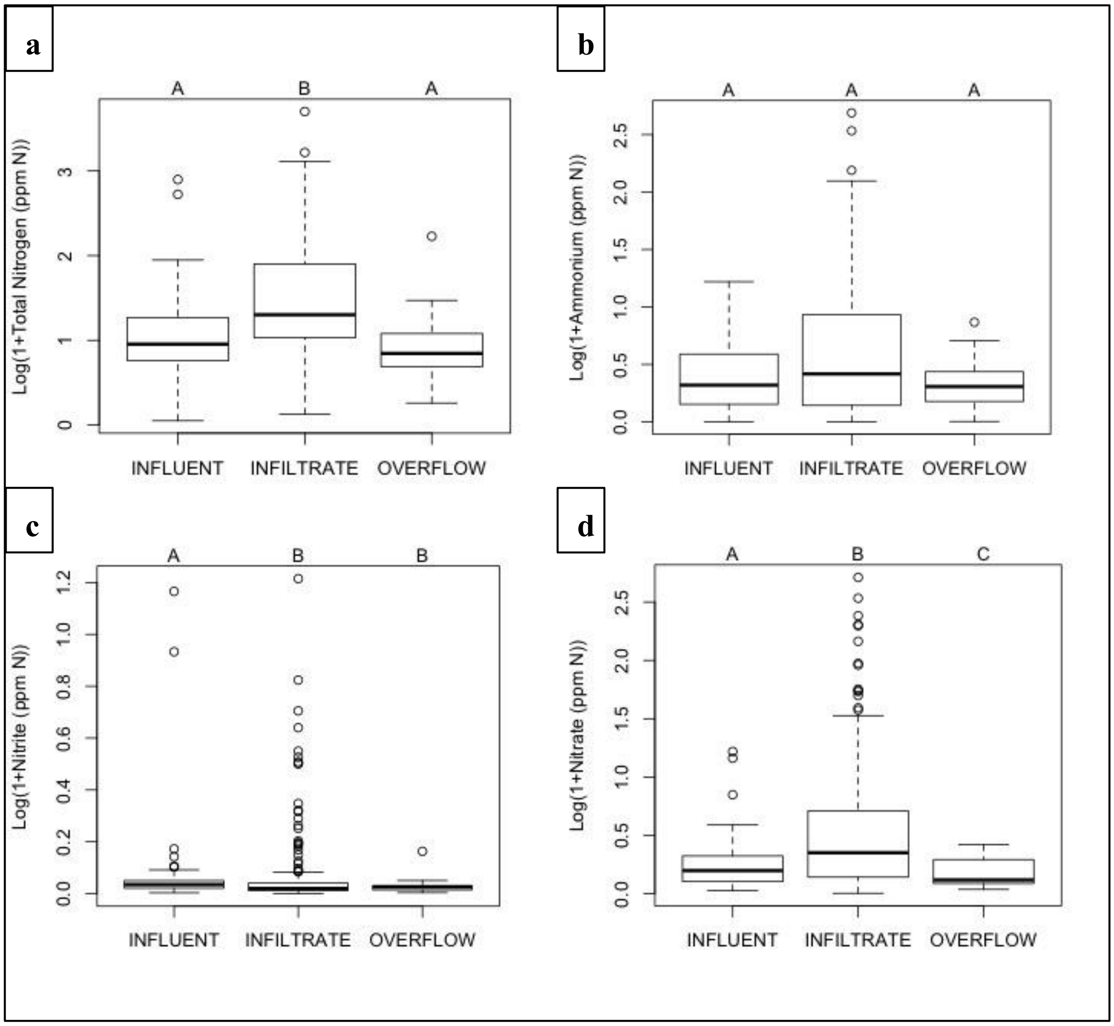

FIG. 5. Nitrogen concentrations in influent (n=74), infiltrate (n=199), and overflow (n=29). Measurements in ppm N are plotted on a log scale to accentuate differences.

Overflow had lower concentrations of both NO_2^- and NO_3^- when compared to influent (Figure 5c, 5d). The reduced concentrations may be attributed to denitrification occurring due to the temporary saturated soil conditions and ponding during the storm events when bioretentions fill to capacity and overflow.

The *p*-values for all of the Wilcoxon rank-sum tests are presented in Table 1, with significant relationships ($p < 0.05$) highlighted in bold. If the median concentration of influent was greater than infiltrate or overflow, + is listed by the *p*-value, and if the median concentration of influent was lower, - is listed by the *p*-value. Dissolved N_2O concentrations, analyzed on 29 unfiltered samples for two storms, were consistently below the 0.001 mg/L detection limit of the probe, so were omitted from the table.

Table 1. Wilcoxon rank-sum test *p*-values

	Influent vs Infiltrate		Influent vs Overflow	
pH	0.21	-	0.11	-
Electrical Conductivity (EC)	**4.02E-03**	**-**	**0.03**	+
Nitrite (NO_2^-)	**1.49E-03**	+	**0.02**	+
Nitrate (NO_3^-)	**1.63E-04**	**-**	**0.04**	+
Ammonium (NH_4^+)	0.05	-	0.58	+
Total Nitrogen (TN)	**7.13E-09**	**-**	0.16	-
Orthophosphate (PO_4^{3-})	**0.04**	+	0.61	+
Total phosphorus (TP)	**0.01**	**-**	0.77	+
Chloride (Cl^-)	**1.27E-04**	**-**	0.17	+
Sulfate (SO_4^{2-})	**4.93E-09**	**-**	**0.01**	+

For most water quality parameters tested, infiltrate was statistically greater than influent. PO_4^{3-}, however, had a lower concentration in the infiltrate, suggesting that PO_4^{3-} in influent could be binding to bioretention soil. However, since TP had a greater concentration in the infiltrate, microbes could be assimilating the PO_4^{3-} in between storms, and then decomposing and releasing it as organic phosphorus. In addition, soil organic matter may contribute to phosphorus leaching (Hunt et al. 2012).

Cl^-, EC, and SO_4^{2-} were found to be elevated in the infiltrate. Perhaps de-icing salts applied during the winter are accumulating in the soil, and washed out over the year.

Overflow was generally not significantly different from influent, except for NO_3^- and nitrite NO_2^- as described above, and for SO_4^{2-} and EC, which could be reflective of salts settling out of influent along the site's flow path. pH was not significantly different between influent, infiltrate, and overflow, although influent was slightly more acidic, with soil and aboveground vegetation possibly buffering the pH as stormwater flows through the bioretention.

Non-parametric spearman rank correlation coefficients (Helsel and Hirsch, 2002) were created for bioretention infiltrate (Table 2 and Figure 6) and for street runoff influent (Table 3) in order to identify predictors of nutrient removal performance. Cumulative rain depth at the time of sample collection, the antecedent dry hours prior to rainfall, and the average temperature of the day were extracted from LaGuardia Airport's weather station, located approximately 5 km from the bioretention sites. In order to determine the onset of a dry period, prior rainfall was discretized into storms using a standard 6 h dry weather period between individual storms and a 2.5mm minimum rainfall depth (Carson et al. 2013). A loading ratio was defined to represent watershed area divided by bioretention area. Significant relationships ($p < 0.05$) are presented in bold in Table 2 and are represented with dotted lines in Figure 6. The *p*-value denotes the likelihood that the predictor and water quality metric correlate by chance.

Table 2. Spearman Rank Correlation Coefficients (Spearman's rho) between Bioretention Infiltrate and Environmental Factors (n=199)

	Rain		Dry Period		Temperature		Loading Ratio	
	rho	*p*-value	rho	*p*-value	rho	*p*-value	rho	*p*-value
pH	-0.04	0.73	**-0.30**	**0.01**	**-0.56**	**8.19E-07**	-0.07	0.58
Electrical Conductivity (EC)	**0.40**	**1.26E-04**	0.09	0.42	0.08	0.48	**-0.37**	**5.16E-04**
Nitrite (NO_2^-)	**0.15**	**0.03**	0.08	0.25	-0.06	0.38	-0.06	0.40
Nitrate (NO_3^-)	**-0.36**	**2.04E-07**	0.12	0.10	0.08	0.29	0.09	0.22
Ammonium (NH_4^+)	**0.22**	**1.90E-03**	-0.13	0.07	-0.11	0.13	-0.12	0.09
Total Nitrogen (TN)	0.05	0.45	0.10	0.18	0.11	0.12	**-0.21**	**2.92E-03**
Orthophosphate (PO_4^{3-})	**-0.26**	**1.74E-04**	0.02	0.75	**0.28**	**4.63E-05**	-0.04	0.53
Total phosphorus (TP)	0.00	0.96	-0.12	0.11	**0.28**	**5.97E-05**	-0.08	0.27
Chloride (Cl^-)	**0.17**	**0.02**	-0.10	0.16	**-0.39**	**8.10E-09**	-0.13	0.07
Sulfate (SO_4^{2-})	0.13	0.07	0.11	0.11	**-0.32**	**5.55E-06**	**-0.26**	**2.75E-04**

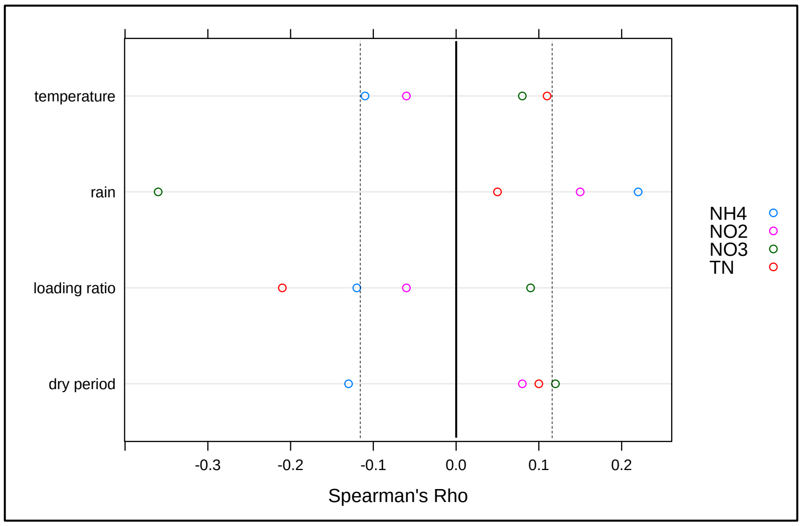

FIG. 6. Spearman rank correlation coefficients for predictors of nitrogen concentrations in bioretention infiltrate

Despite insignificant *p*-values for nitrogen species, temperature appeared to accelerate both nitrogen and phosphorus cycling. Temperature had negative correlations with both NO_2^- and NH_4^+, and a positive correlation with NO_3^-, indicating that greater temperatures may stimulate both NO_2^- oxidizing and NH_4^+ oxidizing microbes. TN may be positively correlated with temperature because it may heighten microbe activity, accelerating decomposition of more recalcitrant forms of organic nitrogen into those that may be more readily leached by infiltrating stormwater runoff. Temperature had a positive relationship with PO_4^{3-} and TP, perhaps because organic phosphorus decomposition is increased at greater temperatures. Temperature had strong negative correlations with Cl^-, pH, and SO_4^{2-}, which might suggest the influence of de-icing agent accumulation in the winter and washout during warmer weather.

One of the strongest correlations overall was a negative relationship between the cumulative depth of rain at the time of sample collection and NO_3^-, suggesting that washout of the accumulated nitrate is highly influenced by the depth of rain, and that samples collected toward the end of a storm will contain lower concentrations of nitrate than samples collected toward the beginning. This result mirrors findings by Li and Davis (2014), who found infiltrate NO_3^- to exhibit high variability, in comparison to other nitrogen species, during an individual storm, with NO_3^- measurements showing an initial spike before decreasing over time. Conversely, rain had a positive relationship with NO_2^-, NH_4^+, and TN, indicating the possibility of priming effects, such as a sudden increase in soil organic decomposition, which can be triggered during rain events (Brown et al. 2013). The positive relationship between rain and Cl^- and EC may suggest that more intense rain tends to wash out tightly bound salts that accumulated during the winter. The negative relationship with PO_4^{3-}may indicate dilution.

Since the loading ratio is related to the quantity of water a site will receive per area, loading ratio mainly had negative correlations with the concentrations of the water quality parameters, demonstrating the influence of dilution. NO_3^-, however, had an insignificant positive correlation, perhaps because a larger loading ratio means a larger watershed collecting a larger nitrogen load from atmospheric deposition, a portion of which accumulates and oxidizes within bioretention media to NO_3^-.

Dry period had a positive correlation with NO_3^- and a negative correlation with NH_4^+, further demonstrating that NH_4^+ appears to be nitrifying to NO_3^-, which then accumulates within the bioretentions in between rain events. pH had a negative relationship with dry period, perhaps because influent stormwater runoff contains salt residues or concrete particles that may temporarily increase the pH.

Table 3. Spearman Rank Correlation Coefficients between Street Runoff Influent and Environmental Factors (n=74)

	Rain		Dry.Period		Temperature		Watershed	
	rho	*p*-value	rho	*p*-value	rho	*p*-value	rho	*p*-value
pH	**0.38**	**0.01**	**0.31**	**0.03**	0.24	0.08	-0.13	0.34
Electrical Conductivity (EC)	0.07	0.60	-0.24	0.08	**-0.32**	**0.02**	0.08	0.58
Nitrite (NO_2^-)	**-0.56**	**2.66E-07**	**0.46**	**3.36E-05**	**0.38**	**7.21E-04**	0.00	1.00
Nitrate (NO_3^-)	**-0.44**	**9.74E-05**	**0.36**	**1.47E-03**	**0.38**	**9.28E-04**	-0.02	0.86
Ammonium (NH_4^+)	**-0.28**	**0.01**	0.15	0.21	0.17	0.15	0.01	0.91
Total Nitrogen (TN)	**-0.25**	**0.03**	**0.35**	**1.93E-03**	**0.47**	**1.90E-05**	-0.20	0.09
Orthophosphate (PO_4^{3-})	**-0.28**	**0.02**	**0.30**	**0.01**	**0.41**	**3.31E-04**	-0.04	0.72
Total phosphorus (TP)	-0.04	0.74	**0.49**	**1.07E-05**	**0.49**	**1.12E-05**	0.03	0.81
Chloride (Cl^-)	-0.19	0.10	**0.36**	**1.82E-03**	**0.29**	**0.01**	-0.04	0.71
Sulfate (SO_4^{2-})	**-0.21**	**0.08**	**0.39**	**5.11E-04**	**0.31**	**0.01**	-0.11	0.34

In terms of predictors for the water quality of the bioretention influent generated by street stormwater runoff, correlations were more consistent across water quality parameters than for infiltrate. Rain depth was correlated with reduced concentrations of water quality parameters, again demonstrating the influence of dilution. In contrast, temperature and antecedent dry hours were associated with positive correlation coefficients. Warmer temperatures may be related to increased vegetative nutrient inputs from pollen or leaf litter. Dry period also appeared to have a strong correlation with increased concentrations of the nutrient water quality parameters, most likely because a longer dry period leads to more atmospheric deposition of nutrients on New York City street surfaces.

CONCLUSIONS

This paper presents monitoring data and analysis from a study to explore bioretention nutrient performance at six streetside bioretentions located in New York City. The results of the study show higher concentrations of TN and NO_3^-, but not NH_4^+ or NO_2^-, in bioretention infiltrate than bioretention inflow. This finding points to the fact that that both ammonium oxidation and nitrite oxidation occur in urban bioretention installations, suggesting that nitrifying bacteria most likely dominate within the bioretention soils. The study results also indicate an accumulation of nitrate in the bioretentions between rainfall events, which is attributed to the local addition of nitrogen via soil decomposition and animal waste. This dry-weather accumulation of nitrate further supports the deficiency of denitrification within the bioretentions, which is considered the only permanent removal mechanism for bioretention nitrogen (Collins et al. 2010).

Influent concentrations of TN and TP were 1.6 ppm N and 2.1 ppm P, while infiltrate concentrations were 2.7 ppm N and 2.5 ppm P. The poor nutrient removal rates found in the studied bioretentions points to a need for enhanced bioretention designs that could support cities in their goals to reduce nutrient pollution sources for watershed health. Further studies should quantify soil mineralization inputs, gas losses, and plant uptake of bioretention nutrients, and explore the performance of alternative bioretention designs that promote denitrification with small pockets of saturated soil (Roy-Poirier et al. 2010) and reduce leaching with more tightly controlled soil organic compositions (Roseen and Stone 2013).

While ideally bioretentions should not leach nutrients, export of nitrogen and phosphorus from the studied bioretentions is only of similar order of magnitude as urban lawn leachate (Sharma et al. 1996). Considering that NYC has thousands of acres of lawn on private property and within parks, the bioretentions may not be a significant contributor of nitrogen. In fact, by reducing nutrient-rich combined sewer overflows through infiltrating stormwater runoff, the hydrologic performance of the bioretentions may have a greater impact than water quality performance on the urban nitrogen cycle.